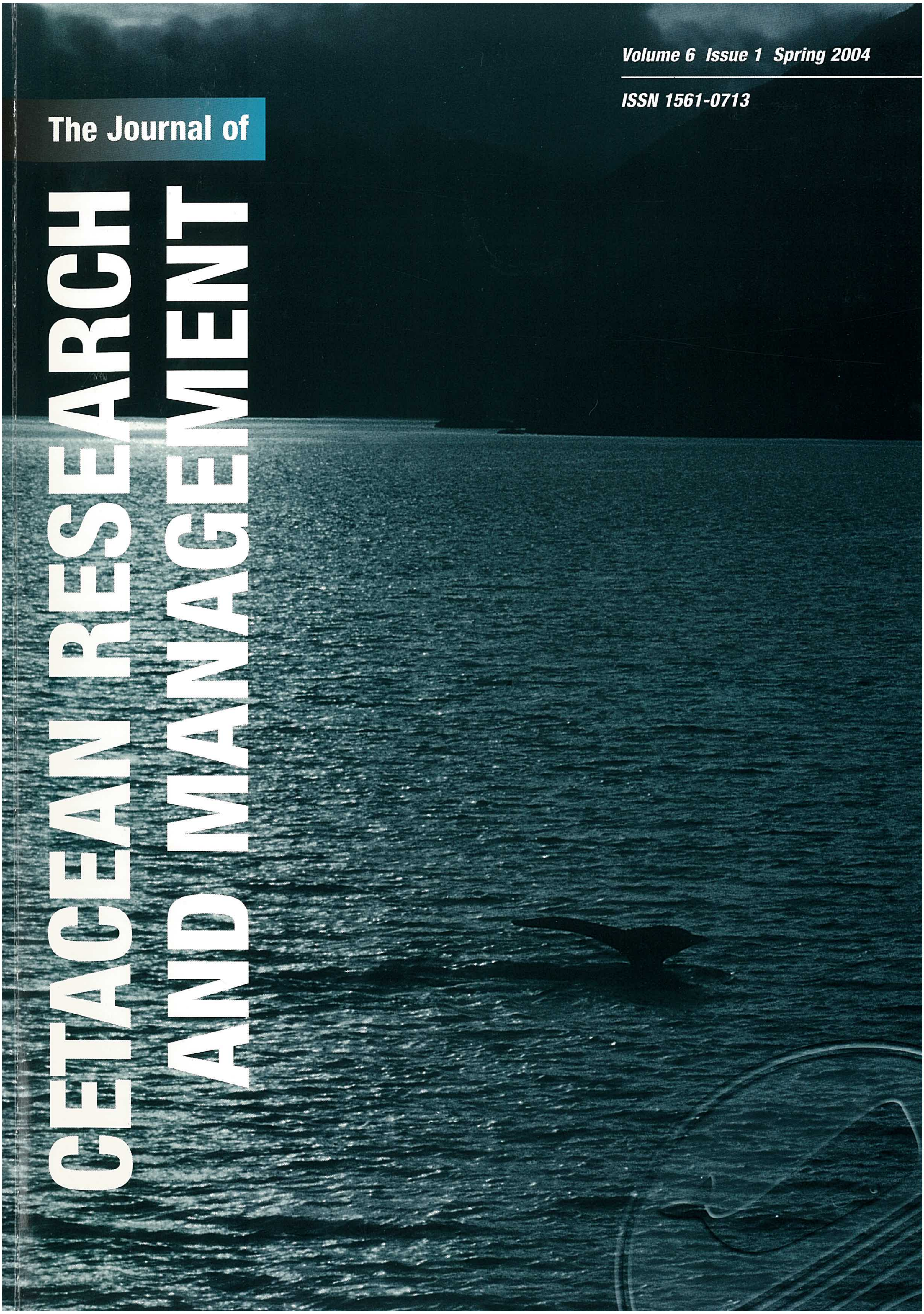
Volume 6 Issue 1 Spring 2004

ISSN 1561-0713

The Journal of CETACEAN RESEARCH AND MANAGEMENT



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Editorial

Welcome to this the first issue of the sixth volume of the Journal of Cetacean Research and Management. This volume contains thirteen papers covering a wide range of management issues. One of the most important threats to cetacean populations is their incidental capture in fisheries. There are two papers looking at this matter for the harbour porpoises in the North Atlantic, one (Neimanis *et al*) examines entrapment in herring weirs in Canadian waters, whilst the other (Vinther and Larsen) provides updated estimates of the bycatch in Danish bottom-set gillnets.

An important contribution the *Journal* makes is to publish papers on species and areas for which there is relatively little information. Such baseline data are important in determining what, if any, management and conservation actions are necessary. In this issue, information is provided on humpback whales and sperm whales from the Mediterranean Sea (Frantzis *et al.*, Druout *et al.*), Irrawaddy dolphins from the Philippines (Smith *et al.*) and humpback whales from northeastern Brazil (Zerbini *et al.*), as well as the first abundance estimate of Atlantic white-sided dolphins from northwest Scotland (MacLeod).

Although such data need to be interpreted carefully, strandings information can provide valuable information on the biology of species and provide insights into when conservation and management actions are necessary. The value of such data increases greatly with increased periods of time. This issue contains two papers that analyse lengthy time series of strandings data from the UK and Ireland (MacLeod *et al.*) and from the northwestern USA (Norman *et al.*). Another important and sometimes underutilised source of information on whale biology, distribution and status comes from whaling logbooks. This issue contains a fascinating paper on the distribution of the endangered North Pacific right whale based on such records from the 19th and 20th centuries (Clapham *et al.*).

There are a number of major research techniques that, whilst they provide important information, have the potential to adversely affect the animals that we are interested in. It is essential that we continue to review these approaches to ensure that the benefits gained are not outweighed by the damage that may be caused and so that we can constantly improve field techniques to minimise any risk. This issue contains a valuable review of capture-release, handling and tagging of odontocetes (Norman *et al.*).

Similarly, uncontrolled tourist activities can have an adverse effect on cetaceans. This is a difficult issue to address as it often requires extrapolating short-term reactions to long-term effects, but given the expansion of such activities around the world it is extremely important. Two papers in this volume address such issues, one (Scheidat *et al.*) examines the response of humpback whales to whalewatching boats off Ecuador, whilst the other (Samuels and Bejder) looks at the effects of interactions between humans and bottlenose dolphins off Florida.

G.P. Donovan
Editor

Distribution of North Pacific right whales (*Eubalaena japonica)* as shown by 19th and 20th century whaling catch and sighting records

PHILLIP J. CLAPHAM*, CAROLINE GOOD+, SARA E. QUINN*, RANDALL R. REEVES#, JAMES E. SCARFF** AND ROBERT L. BROWNELL, JR.++

Contact e-mail: phillip.clapham@noaa.gov

ABSTRACT

North Pacific right whales (*Eubalaena japonica*) were extensively exploited in the 19th century, and their recovery was further retarded (severely so in the eastern population) by illegal Soviet catches in the 20th century, primarily in the 1960s. Monthly plots of right whale sightings and catches from both the 19th and 20th centuries are provided, using data summarised by Scarff (1991, from the whale charts of Matthew Fontaine Maury) and Brownell *et al.* (2001), respectively. Right whales had an extensive offshore distribution in the 19th century, and were common in areas (such as the Gulf of Alaska and Sea of Japan) where few or no right whales occur today. Seasonal movements of right whales are apparent in the data, although to some extent these reflect survey and whaling effort. That said, these seasonal movements indicate a general northward migration in spring from lower latitudes, and major concentrations above 40°N in summer. Sightings diminished and occurred further south in autumn, and few animals were recorded anywhere in winter. These north-south migratory movements support the hypothesis of two largely discrete populations of right whales in the eastern and western North Pacific. Overall, these analyses confirm that the size and range of the right whale population is now considerably diminished in the North Pacific relative to the situation during the peak period of whaling for this species in the 19th century. For management purposes, new surveys are urgently required to establish the present distribution of this species; existing data suggest that the Bering Sea, the Gulf of Alaska, the Okhotsk Sea, the Kuril Islands and the coast of Kamchatka are the areas with the greatest likelihood of finding right whales today.

KEYWORDS: NORTH PACIFIC RIGHT WHALE; NORTH PACIFIC; DISTRIBUTION; WHALING – HISTORICAL; WHALING – MODERN; WHALING – ILLEGAL; MIGRATION; CALVING

INTRODUCTION

North Pacific right whales (*Eubalaena japonica*) were intensively hunted from 1835 (Scarff, 1986; 2001). The species was depleted throughout its range by 1900, when it had ceased to be a principal target of commercial whaling (Scarff, 2001). Sporadic catches of right whales for commercial and scientific purposes were reported in the early 20th century, and it is now known that Soviet whalers illegally killed at least 508 right whales in the North Pacific from the 1950s to the early 1970s (Yablokov, 1994; Doroshenko, 2000). These catches have retarded the recovery of both the eastern and western North Pacific populations; in particular, the catches had a devastating impact on the former (Brownell *et al.*, 2001).

Brownell *et al.* (2001) reviewed all known 20th century sightings, strandings and catches of North Pacific right whales. Earlier, Scarff (1986; 1991) examined the distribution of 19th century right whale catches using the whale charts compiled by Matthew Fontaine Maury (1851; 1852 *et seq.*, 1853). To date, right whale locations contained in these two large datasets have not been plotted together. This paper provides monthly plots of North Pacific right whale sightings and catches in order to investigate seasonal movements, and to compare the distribution of this species in the 19th and 20th centuries.

METHODS AND MATERIALS

Data sources

Nineteenth century

Data on 19th century catches and sightings of right whales from whaling vessels were taken from Scarff's (1991, table 3) summary of Maury's whale charts (Maury, 1852). Few copies of the Maury whale charts are available in public libraries. A portion of one of these charts can be seen in Scarff (1986, fig. 2).

It is important to recognise, however, that the Maury data for the North Pacific have not been validated by direct examination of his primary sources (whaler logbooks). A recent study for the North Atlantic comparing data found in the logbooks with what was depicted on Maury's 1852 whale chart has revealed major discrepancies (Reeves, T.D. Smith and E. Josephson, pers. comm.).

A general description of Maury's whale charts can be found in Bannister and Mitchell (1980) and in Scarff (1991). The data in Maury's whale charts were shown in 5-degree squares. For each of the 12 calendar months, Maury's (1852 *et seq.*) whale chart displays the data as coloured histograms reflecting the number of days on which (a) whale ships were in the square; (b) right whales were seen; and (c) sperm whales were seen. Scarff (1991) shows numerically the following data: (a) and (b)/(a), the latter figure described as the percentage of days on which right whales were seen,

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* *Large Whale Biology Program, Northeast Fisheries Science Center, 166 Water Street, Woods Hole, MA 02543, USA.*
+*Office of Protected Resources, NOAA Fisheries, 135 East-West Highway, Silver Spring, MD 20910, USA.*
Okapi Wildlife Associates, 27 Chandler Lane, Hudson, Quebec J0P 1H0, Canada.
** *1807 M.L. King Way #A, Berkeley, CA 94709, USA.*
++ *Southwest Fisheries Science Center, 1352 Lighthouse Avenue, Pacific Grove, CA 93950, USA.*

which represents a crude index of abundance adjusted for effort. Thus, the Maury charts provide a useful overview of where North Pacific right whales were —and were not — found in the first half of the 19th century.

Twentieth century

Records of 20th century right whale sightings and catches were taken from the comprehensive review by Brownell *et al.* (2001). This dataset includes sightings of 1,965 animals, as well as 741 catches, from 1900 to 2000; these are summarised in Table 1. Thirteen strandings of right whales (12 from the western and one from the eastern North Pacific) were ignored for the purpose of the present analysis.

Records of right whales from the plots of catch positions given by C.H. Townsend (1935) were not included. There is likely to be considerable overlap between the Maury and Townsend data, but the extent of this problem has yet to be systematically investigated.

Table 1

Total numbers of sightings, catches and strandings/entanglements of right whales recorded in the North Pacific between 1900 and 2000 (from Brownell *et al.*, 2001). For the purpose of that paper, the boundary between the western and eastern regions was set at the 180° line of longitude.

Region	Sightings	Catches	Strandings or entanglements	Total
Western North Pacific	988	330	12	1,330
Eastern North Pacific	693	411	1	1,105
Area not specified	284	-	-	284
Total	1965	741	13	2,719

Plots

Monthly plots of the right whale data were created using ArcView Geographic Information System software. The data from Scarff (1991) were plotted by 5-degree square as they appear in his table 3, but corrected for errors. Negative data (i.e. cases in which search effort was made by whalers but no right whales were seen) were also plotted. A few records of whales involving locations at latitudes above 60°N were presumed to be of bowhead whales (*Balaena mysticetus*) and were ignored. While Scarff (1991) considers the 50 days of sightings made in the period May to August between 60° and 65°N and 165° to 170°W to have been of right whales, a more conservative approach is taken in this analysis and these are excluded from the plots.

For the 20th century data from Brownell *et al.* (2001), each record was plotted with a precise position, if available. In cases in which locations were reported to within a 5 or 10-degree square, the midpoint of that square was used. In most cases, the exact number of right whales sighted or killed was available, but in some instances only a range was given; in these cases, the midpoint of the range was plotted. In some instances, location information was insufficient to allow a record to be plotted, and such records were therefore excluded.

The 20th century dataset already included all locations of right whales reported by Japanese sighting surveys. However, additional data on the effort involved in most of those surveys (those from 1964 to 1990) were taken from the maps provided by Miyashita *et al.* (1995) in order to show areas where search effort existed but no sightings of right whales were made.

RESULTS

Figs 1 to 12 show the reported locations of right whales for the months of January to December, respectively. Fig. 13 shows additional 20th century records for which information was available for location but not month of sighting. A general narrative summary of distribution by month (as well as of areas in which there was search effort but no right whale sightings) is given in Table 2.

DISCUSSION

Overview

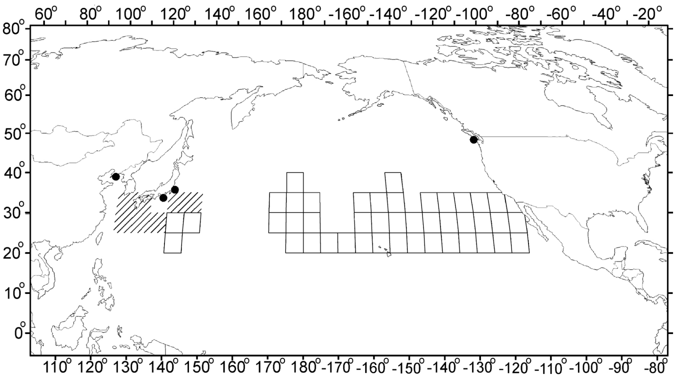
From the data presented here, several observations can be made. First, the historic distribution of right whales as shown by the Maury data was often different from that seen in the 20th century plots. For example, the historic data show that

Table 2

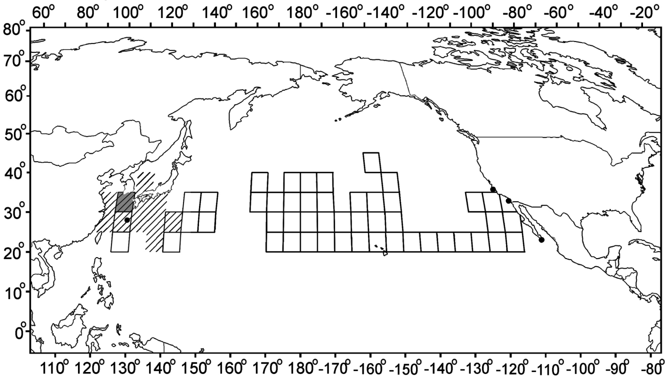
Summary of right whale distribution, by month, as shown in Figs 1 to 12. A distinction is made between data from Maury's charts (termed here 'historic', i.e. 19th century) and data from the 20th century (termed here 'modern').

Month	Sightings	Search effort but no sightings
Jan	Very few sightings, all modern: Japan, Yellow Sea, British Columbia.	Broad band 20-35°N
Feb	Very few sightings, mostly modern: southern Japan, California, Baja.	Broad band 20-40°N
Mar	Many more sightings, mostly historic: Japan, Korea, offshore 30-40°N, Gulf of Alaska and south of Aleutians.	Broad band 20-30°N (and south of this)
Apr	Many sightings, mostly historic: Japan, Sea of Japan, northern Kurils, broad band 35°N to Aleutians and Gulf of Alaska, some SE Bering Sea; a few modern sightings off west coast of USA.	Broad band 20-35°N, central Aleutians
May	Many sightings, mostly historic: Sea of Japan, northern Kurils, Sakhalin Island; many sightings in broad band north of 40°N, many in eastern Bering Sea and Gulf of Alaska.	Broad band 20-35°N
Jun	Many sightings: historic records in Okhotsk Sea, but most in broad band north of 40°N and throughout Bering Sea and Gulf of Alaska; many modern sightings in SE Bering Sea.	Almost complete band 20-40°N
Jul	As June, plus more modern sightings in western Gulf of Alaska.	As June
Aug	As June.	As June
Sep	Many sightings: most modern records from Okhotsk Sea, with diminishing numbers in eastern Aleutians and SE Bering Sea; many historic sightings in Gulf of Alaska and in broad band from 40° to 60°N (but diminishing relative to summer).	Broad band 20-40°N
Oct	Sightings diminished further; significant historic numbers south of 40°N, also in NE Pacific and Okhotsk Sea; very few in eastern North Pacific, and very few modern sightings anywhere.	Much effort and no sightings 20-30°N, many zeroes 30-40°N
Nov	With one exception, the only sightings are a handful of historics: Okhotsk Sea and North Pacific far offshore in mid latitudes.	Broad band 0-40°N, also NE Pacific
Dec	Historic sightings in only two squares, 30-35°N/145-155°W; only one modern sighting.	Eastern North Pacific 20-35°N

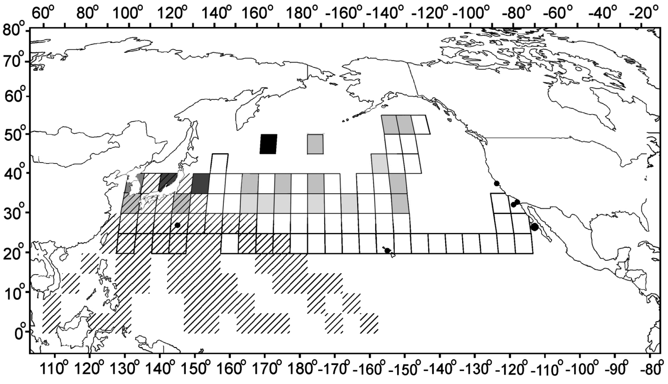
January



February



March



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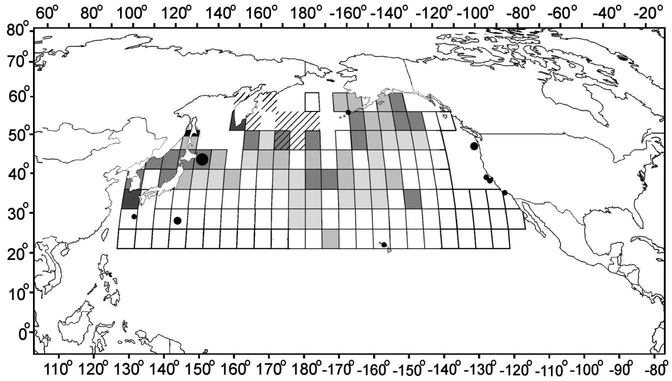
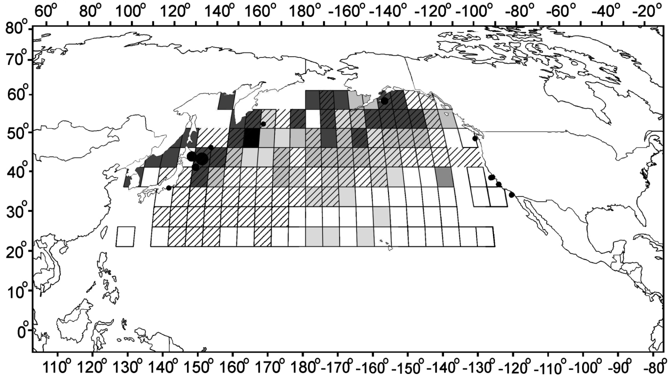
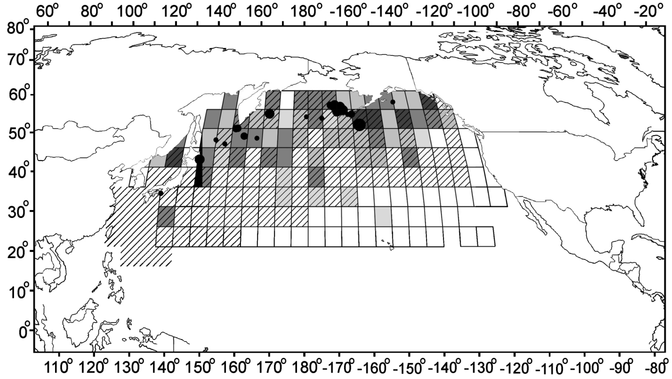


May



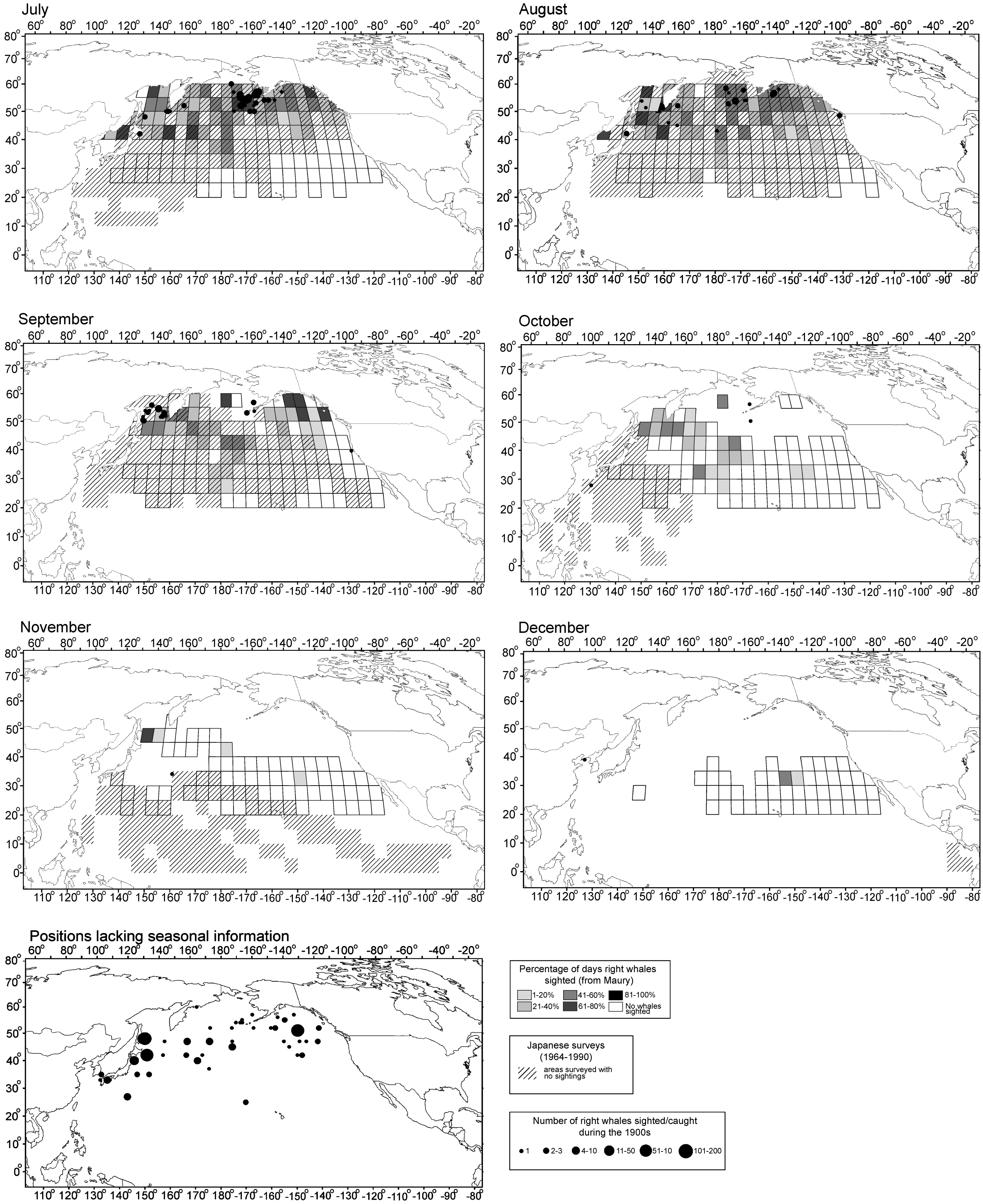
June



Figs 1-12. Reported distribution of North Pacific right whales by month from January to December, respectively. Sources are 19th century whaling records (Scarff, 1991, from Maury, 1852), and 20th century sighting and whaling catch data (Brownell *et al.*, 2001). Note that because of the different nature of the source information, 19th and 20th century records are represented by different measures (percentage of search days on which right whales were recorded, and number of right whales recorded, respectively). All 20th century sightings are indicated by circles of varying size depending on number of whales seen (see key). Nineteenth century records were summarised by 5-degree square and plotted using different degrees of shading (see key). Outlined squares with no shading in them indicate squares for which there was known 19th century search effort but in which no right whale sightings were recorded. Hatching indicates squares that were surveyed by Japanese sighting surveys from 1964-1990, but where no right whale sightings were made (Miyashita *et al.*, 1995). Blank areas with no border or incomplete borders indicate no sightings and no search effort of any kind in either the Maury data or the Japanese sighting surveys. Fig. 13. Reported distribution of North Pacific right whales in the 20th century from records for which there was information on location but not the month of sighting. Data from Brownell *et al.* (2001).

virtually the entire Gulf of Alaska seems to have been used as a summering ground, together with adjacent waters off the coasts of British Columbia, the Alaska Peninsula and the southeastern Bering Sea. Similarly, right whales were once abundant in the Sea of Japan, an area from which they are largely absent today.

Overall, the geographic distribution of the species in the North Pacific was considerably greater in the 19th century than it appears to be in modern times (Scarff, 1991). This is not surprising since most of the species was removed by whaling. The remaining animals appear to constitute two relatively discrete remnant populations (Brownell *et al.*, 2001). The size of the western population, much of which is believed to summer in the Okhotsk Sea, is not clear. Data from Japanese minke whale sighting surveys in the Okhotsk Sea in 1989, 1990 and 1992 were used to calculate an estimate of abundance of 922 (CV = 0.433; IWC, 2001, p.22), although both positive and negative potential biases were identified in the survey methodology. Noting the wide confidence intervals associated with this survey, we do not believe that an abundance in the high hundreds is consistent with other sighting data on this population (see Brownell *et al.*, 2001), and suggest that the western population's size is likely to be smaller than this.



Figs 1-13. *Continued.*

Following catches of 372 right whales by Soviet pelagic whaling operations (primarily in the 1960s; Doroshenko, 2000), the eastern North Pacific population is believed to contain fewer (perhaps far fewer) whales than the western population (LeDuc *et al.*, 2001). Recent surveys, photo-identification and genetic studies suggest that this population may number in the tens of animals.

During much of the year, the historic distribution had a large offshore component in deep water far from the coast. By contrast, most of the 20th century sightings were relatively close to land, notably off the Aleutian Islands and in the Bering and Okhotsk Seas. Given the extensive coverage of Japanese sighting surveys, this recent absence of right whales in deep water cannot be wholly attributed to lack of effort (see for example May, Fig. 5). Scarff (1991) noted this offshore component of the Maury data and suggested that North Pacific right whales may have wintered and calved far offshore, rather than in the coastal habitats which many North Atlantic right whales (*E. glacialis*) and southern right whales (*E. australis*) are known to inhabit during winter (Kraus *et al.*, 1986; Best, 1990; Payne *et al.*, 1990). This is discussed further below.

Seasonal distribution

Overall movements

Seasonal movements of right whales are apparent in the data presented here. To some extent these may reflect survey and whaling effort rather than real migratory movement, and the apparent movements summarised here must be considered with that caveat in mind.

There were very few sightings of right whales anywhere in January and February despite a certain amount of historical effort, notably in offshore areas (Figs 1 and 2). There are also few recent sightings of right whales in coastal waters at this time, despite considerable whalewatching effort in some locations (e.g. California and Baja California). In March, most whale sightings appeared in the Maury data in a latitudinal band between 30 and 40°N, reflecting that in March over 90% of the whaling effort occurred south of 40°N. Other (though fewer) sightings occur further to the north; in the Maury data, the area north of 45°N had consistently high encounter rates even at this time of year, albeit with small sample sizes.

By April, right whales were widely distributed from 35°N and had penetrated the Bering Sea; by May and June there were large numbers of sightings in both the Bering Sea and the Gulf of Alaska. This pattern continued in July and August, primarily north of 40°N, in both the Maury and the 20th century datasets. In September, the beginnings of a southward movement were evident; current data suggest that the last right whales leave the southeastern Bering Sea in October. By October, right whales were primarily seen in mid latitudes (30-50°N), and they largely disappear from the records in November and December.

The movements that can be inferred from the combined Maury and 20th century data are similar to those evident in the seasonal distribution of North Pacific right whales shown by Townsend (1935). Townsend's maps show right whales abundant in the Gulf of Alaska and adjacent areas (the Northwest Ground) between May and August, with smaller numbers in April and September (presumably reflecting whales moving into and out of the area). Right whales were also abundant both south and north of the Aleutians in May and June, as well as in the Bering Sea in summer and early autumn. Townsend's maps show concentrations of whales from the eastern coast of Kamchatka to the offshore waters of the northwestern North Pacific from May to September, with additional concentrations in the Okhotsk Sea in spring and summer. Numerous records in the Sea of Japan in spring, but far fewer in February and March, probably reflect a northward migration towards the Sea of Okhotsk and Kuril Islands; scattered catch positions in February in the Taiwan Strait and the entrance to the Yellow Sea are indicative of a northward migration from unknown wintering grounds (see below). With the exception of a few records at the entrance to the Yellow Sea in October, there is no indication in the Townsend data of a parallel southward migration.

Overall, the north-south migratory movements evident in all three datasets (Maury, Townsend and Brownell *et al.*) provide support for the idea that two largely discrete populations of right whales exist in the eastern and western North Pacific.

Additional details of the apparent patterns of seasonal movement are given below.

Eastern North Pacific

Right whales were rarely caught in the coastal whaling fisheries along the western coast of North America (Scarff, 1986) and so there is less information on the seasonal movements of this species than in the western regions of the North Pacific Ocean. As noted above, the plots (notably those from Maury's data) show a general movement northward in spring and south again in autumn. Major concentrations of summering whales occurred in the Gulf of Alaska and eastern Bering Sea, and numerous animals were also recorded in offshore waters for much of the year. The historical importance of these offshore areas may never be known, and the persistence of the species there today is questionable given the great reduction in the size of the population.

Western North Pacific

Omura (1986) postulated that there were two distinct populations of right whales in the western North Pacific: the 'Sea of Japan' and 'Pacific' stocks (taken here to mean 'sub-populations'). Historical catch data from Japanese coastal whaling villages indicate that both populations moved south in autumn and north again in the spring, with the peak months being September to December and February to April, respectively. The two putative populations were kept apart during migration by the Japanese islands, with the Sea of Japan stock moving along the western coasts, and the Pacific stock travelling off the eastern coasts of the archipelago. Omura (1986) suggested that the Pacific population summered in the Kuril Islands, with some animals moving further northeast and entering the Bering Sea. In contrast, he suggested that the feeding grounds for the Sea of Japan population lay primarily in the Okhotsk Sea, although he recognised that there was little direct evidence to support this belief.

As expected, the data presented here support the idea of a southward movement in the autumn and a return migration north in spring. Many whales moved into the Okhotsk Sea during early summer, an event that presumably coincided with the breakup of ice cover in the area. The Maury data show significant numbers of right whales entering the western Bering Sea by June, and remaining there until September or October, when a general southward movement is again evident in the sightings. However, it is important to reiterate that during the entire period of spring to autumn, right whales were also consistently dispersed in offshore waters across a broad region of the North Pacific. This prominent aspect of the right whale's distribution in the 19th century is largely absent in the modern records, further emphasising the considerable diminution of the species' range.

Whether there were (or are) two distinct sub-populations of right whale in the western North Pacific remains unclear. Catches of right whales by Japanese net whalers at Kawajiri (a whaling village in Yamaguchi Prefecture on the Sea of Japan) dropped sharply after 1859, a situation which Omura (1986) attributes to 'the operation of American whale ships'. This may be a rather simplistic interpretation; presumably the demise of the right whale in the Sea of Japan was due to a combination of Yankee and coastal whaling, with 20th century recovery inhibited to an unknown extent by illegal Soviet catches on the feeding grounds in the Okhotsk Sea.

Wintering/calving grounds

Where North Pacific right whales go in mid-winter, and where they calve (presumably at this time of year), remains unknown. For the putative Pacific and Sea of Japan populations in the western North Pacific, Omura (1986) suggested that the calving grounds probably lay to the south of Japan; specifically, he proposed that right whales calved around or near the Ryukyu Islands (with the implication being that the two populations may have mixed there). There is no reliable evidence with respect to the location of coastal or insular wintering grounds in the eastern North Pacific. Scarff (1986) speculated that right whales from the Gulf of Alaska might migrate to calve 'near the coast of Kamchatka or further south'. We do not believe that any of the Kamchatka coast could have been a calving ground; rather, this area appears to represent a summer feeding habitat, with sightings made in summer or early autumn.

Indeed, there are few data with which to further examine this question. In the western North Pacific, the southernmost sightings from Townsend (1935) are off Taiwan (7 records) and at around 30°N off the Chinese coast (20 records); all are from February or March. A few recent sightings have also been reported from Chichi-jima (Bonin Islands, at 27°N) in March and April (Brownell *et al.*, 2001). In the eastern North Pacific, the situation is even more obscure. There, the data from winter are confined to a handful of 20th century sightings from the western coast of North America, and some sporadic offshore records in the Maury data. As noted by Scarff (1986; 1991), there is no evidence from either historical whaling records or archaeological investigations of aboriginal hunting peoples that the coastal waters of western North America ever contained a calving ground for this species.

Our interpretation of the plots in this regard is that serious consideration should be given to Scarff's (1986; 1991) contention that North Pacific right whales wintered and calved primarily in offshore, not coastal, waters. Indeed, the record is conspicuously marked by a paucity of right whale sightings from *any* nearshore area in winter, despite the considerable likelihood of search effort from 19th century (and earlier) coastal whaling communities in both Japan and North America. In contrast, the apparent historical abundance of right whales in offshore waters (in both Maury and Townsend's data) is too obvious to ignore. In light of this, Scarff (1986, p.57) concludes:

> The recent concentrations of scientific investigations on nearshore populations of right whales off South America, South Africa, eastern North America and Australia may have led to an exaggerated view of the species' coastal tendencies.

We agree that the offshore distribution of right whales has been little studied. We suggest that researchers consider the use of satellite telemetry to locate the many whales that go 'missing' in winter, a question which exists even for the well-studied North Atlantic right whale.

Future work

The analyses presented here provide some direction regarding future work on this species. In light of the small size and highly endangered status of both the eastern and western populations, establishing the present distribution of right whales, and assessment of anthropogenic threats in the habitats where they currently exist, should be a top priority for management. Existing data suggest that the Bering Sea, Okhotsk Sea, the Kuril Islands and the coast of Kamchatka are the areas with the greatest likelihood of finding right whales today. New surveys (including both photo-identification and biopsy components) of these and other regions of the North Pacific should be funded and conducted in the near future.

ACKNOWLEDGMENTS

We are grateful to Richard Merrick and Fred Serchuk for their reviews of this paper.

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Entrapment of harbour porpoises (*Phocoena phocoena*) in herring weirs in the Bay of Fundy, Canada

A.S. Neimanis[*+], H.N. Koopman[*#], A.J. Westgate[*++], L.D. Murison[*] and A.J. Read[++]

Contact e-mail: aleksija.neimanis@usask.ca

ABSTRACT

Harbour porpoises (*Phocoena phocoena*) are small coastal cetaceans vulnerable to mortality in fishing operations. Not all interactions are fatal, however, and each year many porpoises swim into and are subsequently released from herring weirs in the Bay of Fundy, Canada through a targeted release programme. This study examines catch composition, body condition, characteristics associated with mortality and factors affecting entrapment of porpoises in weirs between 1992-2001. A total of 886 porpoises were recorded in weirs during this period. A total of 657 animals were involved in attempted releases: 588 were released alive and 69 were incidentally killed during release. The remainder of the animals swam out on their own or their fates were unknown. Estimated annual mortality represents less than 0.01% of the Bay of Fundy/Gulf of Maine population and only 1.03% of its annual potential biological removal level.

The number of porpoises caught in weirs varied from eight in 1996 to 312 in 2001. Of the 390 animals released with a numbered identification tag, 25 were recaptured in weirs and 4 of those porpoises entered a weir a third time. Males comprised 63.5% of entrapments. Weirs and demersal gillnets captured animals from the same population, but the weir bycatch was biased towards younger, smaller animals. Porpoises that became trapped in weirs exhibited measures of body condition similar to those killed in gillnets and by gunshot wounds in the same waters. None were considered emaciated. Mortality in weirs appeared to be random; porpoises that died during release attempts were of the same age and sex composition and body condition as the individuals that survived. The use of a specialised large-mesh seine significantly increased the probability of successful release. Observations of the stomach contents data of porpoises killed in weirs indicate that porpoises feed while trapped in weirs, but perhaps not at the same rate as animals killed in gillnets. Entrapments peaked in August, concurrent with the highest landings of Atlantic herring, the target species of the weir fishery. Based on a logistical regression model, porpoises are 3.3 times more likely to swim into a weir on a night in which high tide falls during darkness. Weir entrapments do not have a significant effect on this population, largely because of on-going efforts to release porpoises from weirs.

KEYWORDS: HARBOUR PORPOISE; INCIDENTAL CATCHES; FISHERIES; NORTH AMERICA; ATLANTIC OCEAN; CONSERVATION; MORTALITY RATE

INTRODUCTION

As the demand for marine resources increases, particularly in coastal areas, conflicts with marine mammals increase in frequency and severity. Coastal species are particularly vulnerable because of their proximity to human activities. Not all interactions pose a significant threat, however, and each type of interaction needs to be assessed to optimise the use of marine resources while minimising anthropogenic impacts on marine mammal populations.

The harbour porpoise (*Phocoena phocoena*) is a small, coastal cetacean that is vulnerable to fishery interactions throughout its range (Gaskin, 1984; IWC, 1994; Donovan and Bjørge, 1995). These interactions are of particular concern for porpoises from the Bay of Fundy and Gulf of Maine population; large numbers have been killed in sink gillnets set for demersal fish species in these areas in the past (Read and Gaskin, 1988; Read, 1994). Mortality in gillnet fisheries averaged 1,163 porpoises per year from 1994-1998 (Waring *et al.*, 2001). A take reduction plan was implemented in the USA in December 1998, after which gillnet mortality decreased to 270 porpoises in 1999 (Waring *et al.*, 2001). In Canada, total bycatch of porpoises in gillnets was 424 and 101 animals in 1993 and 1994, respectively (Trippel *et al.*, 1996). An estimated 36 porpoises were killed in gillnets in the Bay of Fundy per year from 1995-1999 (Waring *et al.*, 2001). Bycatch in Canadian waters has decreased in recent years because of conservation measures designed to protect overfished groundfish stocks (Trippel *et al.*, 1999). The current abundance estimate for the Bay of Fundy/Gulf of Maine population is 89,700 (53,400-150,900) (Palka, 2000) and the potential biological removal (PBR) for this population is 747 animals per year (Waring *et al.*, 2001). High levels of incidental mortality led the Committee on the Status of Endangered Wildlife in Canada to list porpoises in eastern Canada as threatened (Gaskin, 1992) but in May 2003 this was changed to *Special Concern* (see web at http://www.sararegistry.gc.ca/virtual_sara/files/cosewic/sr%5Fharbour%5Fporpoise%5Fe%2Epdf). This stock is considered strategic by the National Marine Fisheries Service in the United States because prior to 1999, PBR had been exceeded every year (Waring *et al.*, 2001).

Harbour porpoises in the Bay of Fundy are also caught incidentally in herring weirs. Weirs are fixed structures built in shallow water to trap Atlantic herring (*Clupea harengus*). In Canada, the herring weir fishery is restricted to the New Brunswick and Nova Scotian coasts in the Bay of Fundy (Read, 1994) and bycatches in this fishery likely impact only the Bay of Fundy/Gulf of Maine population. Unlike gillnet entanglements, which are almost always fatal, porpoises that enter weirs can be released alive. However, the number of entrapments and subsequent mortality rate have not been well documented. The only published accounts of weir entrapments are found in Smith *et al.* (1983) and briefly in Read (1994). Preliminary data on entrapments up to 1994 were presented to the International Whaling Commission

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[*] *Grand Manan Whale and Seabird Research Station, 24 Rte 776, Grand Manan, NB Canada, E5G 1A1.*

[+] *Department of Veterinary Pathology, Western College of Veterinary Medicine. 52 Campus Dr., University of Saskatchewan, Saskatoon, SK Canada, S7N 5B4.*

[#] *Biological Sciences, University of North Carolina at Wilmington, 601 S. College Road, Wilmington, NC 28403, USA.*

[++] *Nicholas School of the Environment and Earth Sciences, Duke University Marine Laboratory, 135 Duke Marine Lab Rd., Beaufort, NC 28516, USA.*

(IWC) in 1995 (Neimanis *et al.*, 1995). Smith *et al.* (1983) estimated from a questionnaire survey that approximately 70 porpoises are trapped in weirs in the lower Bay of Fundy each year and 39% of these animals die. These authors concluded that porpoises did not feed while in weirs. Read (1994) reported that the number of entrapments per year had increased and noted that more than 100 animals were released alive in 1993. A thorough and updated account of porpoise entrapments in herring weirs is needed to evaluate the impact of the weir fishery on this population of porpoises.

In 1991, researchers at the Grand Manan Whale and Seabird Research Station (GMWSRS) began to assist local weir fishers with the removal of porpoises from their weirs. This cooperative effort, the Harbour Porpoise Release Program (HPRP), was established to minimise porpoise mortality and provide researchers with access to live, free-ranging animals for further study.

This paper provides a comprehensive account of harbour porpoise entrapments in the lower Bay of Fundy since the last detailed report in 1983 (Smith *et al.*, 1983). The aim is to document which segment of the population becomes trapped, its subsequent fate and to determine which factors contribute to entrapment and outcome. This includes a report of total number of entrapments, catch composition, body condition and mortality rates of trapped porpoises. To investigate factors that may contribute to mortality in weirs, basic life history data and body condition of porpoises that survived are compared with those that died. These parameters also are compared for porpoises trapped in weirs with porpoises that died in gillnets in the Bay of Fundy. An analysis of stomach contents collected from animals that died in weirs is provided to re-examine the hypothesis that porpoises are not feeding while trapped (Smith *et al.*, 1983). To better understand factors that may facilitate porpoise entrapment, herring landing data from weirs are presented in relation to the number of porpoises caught; the effects of tidal cycle, moon phase and season on entrapment are also examined. Finally, the impact of weir mortality on the Bay of Fundy/Gulf of Maine population is evaluated.

MATERIALS AND METHODS

Herring weirs

Herring weirs are fixed impoundments built near shore to catch herring (Fig.1). These kidney-shaped structures are comprised of a number of wooden stakes driven into the sea floor from which a 1cm mesh nylon twine is hung. Weirs enclose an average surface area of 1,500m^2 and range from 3-20m in depth at low tide. The opening or mouth of the weir (i.e. the inner bend of the kidney structure) faces towards shore and a twine fence runs from the mouth to shore. When herring follow the shoreline at night, they encounter the fence and are directed into the weir. Once inside, they swim along the perimeter of the weir in a characteristic figure-eight pattern, which leads them away from the weir mouth.

It is believed that porpoises feeding on schools of herring follow the fish into the weir at night. Between one and 14 porpoises have been recorded to have entered a single weir on a given night around Grand Manan. Some leave the weir independently, but most remain trapped in the weir until they are removed. It is unclear why most porpoises remain in the weir, but individuals may be reluctant to swim through the relatively narrow weir entrance, which is located in the shallowest part of the weir. This behaviour is in stark contrast to that of harbour seals (*Phoca vitulina*) which enter and leave weirs at will.

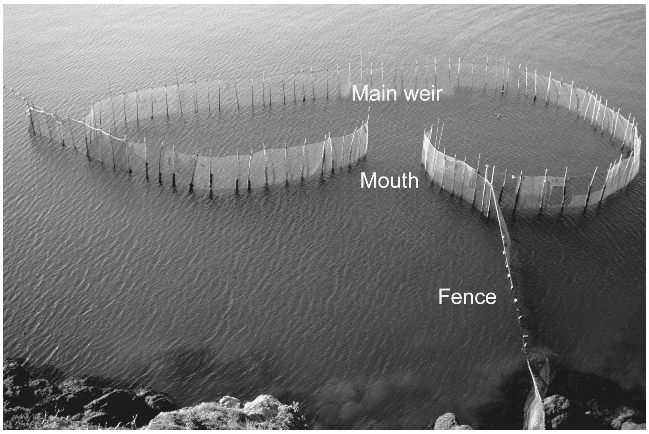

Fig. 1. Overhead view of a herring weir. The fence runs from the shoreline to the mouth and directs herring that are swimming along the shoreline into the main weir.

Removing porpoises from herring weirs

Porpoises are removed from weirs using a seine net and with the direct assistance of the weir operator and seine crew. They can be removed with the herring using a fine mesh purse seine (mesh size 0.75-1.25cm) or released separately with a specialised marine mammal seine. This second net is made from buoyant polypropylene and has a much larger mesh size (7.5cm) which allows herring to pass through while capturing larger animals, such as porpoises, tuna and sharks. Porpoises are typically pursed in the seine, transferred to a boat and transported outside of the weir for release. Occasionally, the seine is deployed as a barrier to sweep animals out of the weir. Porpoises swept out in this manner are released without being handled and no additional data are collected. If an animal dies during seining, an attempt is made to recover the carcass and a necropsy is performed following standard protocols (American Society of Mammalogists, 1961; McLellan *et al.*, 2002).

Processing and releasing porpoises

Basic data are collected from each animal whenever possible, including sex, total length and maximum girth. Tooth eruption is verified to identify calves and mammary glands of females are checked for lactation. A coloured, numbered roto-tag (*Dalton Jumbo Roto-tags*, Dalton E.I.D. Systems, Oxfordshire, UK) is applied to the trailing edge of the dorsal fin for future identification. Finally, porpoises are weighed and then released outside the weir. Whenever feasible, additional samples and data are collected from a subset of animals. Blood samples are obtained from a fluke vessel for a complete blood count, biochemistry profile and other analyses (Koopman *et al.*, 1995; 1999). Since 1998, the heart rate of selected animals is continuously recorded throughout processing and handling procedures are videotaped to record additional information. Some porpoises are fitted with VHF transmitters, combination VHF transmitters/time-depth recorders and/or satellite transmitters (Westgate *et al.*, 1995; Read and Westgate, 1997; Westgate and Read, 1998).

Data analyses

Records of entrapment and the subsequent fate of individual porpoises were collected for 961 animals that entered herring weirs in the Bay of Fundy from 1984-2001 (Table 1). It was not possible to collect a complete suite of data and samples from every animal, as some individuals were released unprocessed, some were observed in weirs but subsequently disappeared and some carcasses were not retrieved from the seine. Thus, sample sizes vary for each analysis performed. Entrapments were recorded between June and October. Prior to 1991, releases were opportunistic and effort was highly variable. By 1992, the release programme had become well-established and began to release porpoises in considerable numbers (Table 1). From 1995, local weirs were monitored daily and date of porpoise entrapments were recorded. Weather permitting, every weir within a 10km radius of North Head, Grand Manan was examined by boat each morning. Not all weirs could be examined every day and the number of local weirs built varied between years. The total number of weirs checked regularly from 1995-2001 varied from 18 to 22. All data analyses were restricted to 1992-2001, when effort levels were comparable. Only eight entrapments were recorded in 1996, so this year was excluded from statistical comparisons among years. All statistical analyses except for logistic regression were carried out using SPSS 10.0 (SPSS Inc, Chicago, IL). Logistic regression was performed using SAS/STAT 6.0 (SAS Institute Inc, Cary, NC). Tests were performed assuming an alpha level of 0.05, unless otherwise stated.

Table 1

Summary of recorded entrapments and subsequent fate of harbour porpoises in herring weirs in the lower Bay of Fundy from 1984-2001.

Year	Released alive*	Died during release*	Swam out	Shot	Fate unknown	Total no. recorded in weirs
1984	4 (4)	0	0	0	0	4
1985	5 (5)	0	0	3	0	8
1986	0	0	0	1	0	1
1987	0	0	0	1	0	1
1988	0	1 (0)	0	4	0	5
1989	1 (0)	1 (0)	0	0	0	2
1990	24 (7)	7 (0)	0	1	9	41
1991	11 (7)	0	1	0	1	13
1992	39 (37)	11 (11)	11	0	11	72
1993	115 (113)	21 (21)	0	8	9	153
1994	41 (40)	11 (11)	5	0	20	77
1995	54 (53)	4 (4)	17	0	7	82
1996	2 (2)	3 (1)	2	0	1	8
1997	23 (20)	2 (2)	3	0	7	35
1998	12 (12)	2 (2)	19	0	1	34
1999	48 (47)	3 (3)	33	0	9	93
2000	12 (12)	0 (0)	3	0	5	20
2001	242 (237)	12 (12)	56	0	2	312
Total	633 (596)	78 (67)	150	18	82	961

*( )=GMWSRS present.

Catch composition

To examine catch composition of porpoises caught in weirs, both animals that were successfully released ($n = 521$) and those that died ($n = 68$) were included. This excluded animals that were released unprocessed, swam out, were swept out or disappeared before seining. Sex ratios were calculated by year and compared using Chi-Square tests ($n = 544$). Since porpoises exhibit sexual dimorphism (Read and Tolley, 1997), body length data were analysed separately for males ($n = 310$) and females ($n = 173$). Mean total lengths were determined by year and compared using ANOVA. As yearly variances were unequal (Levene's test, $p < 0.02$ for both males and females), *post-hoc* comparisons of ANOVA results were made using Tamhane's T2 (Tabachnick and Fidell, 1996). Total lengths were grouped by 5cm increments and length distributions were compared between years for each sex using Kruskal-Wallis tests (Zar, 1974). Age estimates for carcasses collected from weirs were obtained from counts of dentinal growth layers in stained decalcified thin sections of teeth (Bjørge *et al.*, 1995). Mean age and age distribution were compared between 1992 and 1993, when sample sizes were large enough ($n = 9$ and $n = 16$, respectively); ages were not yet available for the 2001 season. An independent samples T-test was used for comparison of mean age and a Kruskal-Wallis test was used for age distribution.

To determine if weir catches were similar in composition to the population of porpoises caught in gillnets, sex ratios, mean lengths, length distributions, mean ages and age distributions of animals caught in weirs were compared to animals killed in gillnets in the lower Bay of Fundy during the same period. All necropsies of gillnet bycatches were performed by the GMWSRS following standard protocols (American Society of Mammalogists, 1961; McLellan *et al.*, 2002). The same statistical analyses were used as described above for weir catch composition. Analyses were restricted to 1992 and 1993, when adequate sample sizes were available from both the weir ($n = 44$ and $n = 108$ for 1992 and 1993, respectively) and gillnet ($n = 19$ and $n = 36$ for 1992 and 1993, respectively) samples. This eliminated the potential influence of interannual variation. Age estimates were only available for dead animals, so sample sizes for age comparisons were 9 and 16 for the weir sample in 1992 and 1993, respectively. Catch composition of porpoises caught in gillnets in 1992 was also compared to that of gillnet mortalities in 1993.

Recaptures

Of the 886 porpoises recorded in weirs from 1992-2001, 390 were fitted with a numbered tag for future identification. Of these, 25 animals were recaptured in a weir at least once. Sex ratio, mean length and length distribution of recaptured animals were determined ($n = 25$). Mean length and length distribution were determined separately for males ($n = 17$) and females ($n = 8$) as above. These results were then compared with those for porpoises that only swam into a weir once ($n = 510$). A Chi-square test, an independent T-test and Kruskal-Wallis analyses were used for sex ratio, mean length and length distribution comparisons, respectively.

Mortality rates

Mortality rates of trapped porpoises were determined for each year and compared among years. Rates were calculated in the following two ways: the number of animals that died in weirs was divided by the total number recorded in weirs, excluding those with an unknown fate (total $n = 814$); the number of animals that died during seining was divided by the total number of porpoises for which a seine was attempted (total $n = 581$). Seines were only included in which personnel from the GMWSRS were present; porpoises that were swept out of weirs were excluded. To examine efficacy of the mammal seine, mortality rate of porpoises seined out with this net ($n = 240$) was compared with that calculated for animals released with the herring seine ($n = 239$) using a Chi-square test. Again, animals swept out were excluded. A further aim was to determine if the number of porpoises present in the seine had any effect on mortality. To avoid variation from type of net used, this analysis was conducted only for porpoises captured in herring seines. Mortality rate during seines for solitary animals ($n = 63$) was compared with multiple animals ($n = 176$) using a Chi-Square test. Animals that were swept out were excluded. In addition, mean number of animals per seine that survived ($n = 195$) was compared with those that died ($n = 44$) using an independent samples T-test.

Body condition of entrapped porpoises

To determine the body condition of porpoises that swim into weirs, total mass (used here as a measure of body condition) was compared among three groups of porpoises: all trapped animals (live releases and mortalities, $n = 99$); porpoises from the Bay of Fundy that died as the result of other human interactions (HI) such as gillnet entanglement or were shot ($n = 15$); and porpoises deemed to be in poor condition ($n = 19$). This latter group represents individuals found stranded on beaches along the mid-Atlantic coast of the United States. They exhibited no evidence of human interaction, presented with a visible depression behind the blowhole and sunken epaxial musculature (Kastelein and van Battum, 1990; Koopman, 2001), and changes in muscle fibre profiles indicate they have starved to death (Stegall *et al.*, 1999). Starvation is associated with a significant decrease in blubber thickness in porpoises from the mid-Atlantic (Koopman *et al.*, 2002). Preliminary comparison of a set of robust animals in good body condition killed in fishing interactions in the mid-Atlantic with robust specimens from the Bay of Fundy indicated that blubber thickness was not significantly different between porpoises < 130cm in length from the two geographical regions (H. Koopman, unpublished data). Thus, emaciated animals from the mid-Atlantic were assumed to be a reasonable comparative group for the weir and gillnet analyses reported here. Total mass increases with body length, and thus values must be corrected for body size prior to comparison. In this dataset, the relationship between body length and body mass was $\text{mass} = (\text{length})^{2.376}$ ($p < 0.001$, adjusted $R^2 = 0.89$). Body mass was compared among the three groups by first linearly regressing $\text{length}^{2.376}$ against mass, and then submitting residual values to ANOVA. Levene's test was used to determine equality of variances between groups tested (Tabachnick and Fidell, 1996). *Post-hoc* comparisons of all ANOVA tests then were made using either Sidak's test (group variances were the same) or Tamhane's T2 test (group variances differed significantly). Most of the stranded porpoises were juveniles and so only porpoises with total body length < 130cm were included in this comparison.

Factors associated with weir mortality

To examine which factors might be correlated with mortality during release, sex ratios, mean lengths and length distributions of porpoises that survived the seine ($n = 483$) were compared with those that did not ($n = 61$). Animals that were swept out were excluded. Comparisons were made using Chi-Square, independent samples T-test and Kruskal-Wallis tests, respectively.

To determine whether poor body condition was associated with an increased chance of mortality, the size-corrected body mass of porpoises that were successfully released ($n = 140$) was compared with those that died during release attempts ($n = 63$). These two groups of porpoises represented the same size classes: mean standard lengths of live (128.4 ± SE 0.75cm) and dead (128.9 ± SE 2.03cm) weir porpoises were not significantly different when compared with an independent samples T-test ($p = 0.816$), thus all animals for which data were available were included. Body mass was compared between live and dead weir porpoises using ANCOVA with $\text{length}^{2.376}$ as the covariate.

Finally, to determine whether porpoises that died in the weir fishery were in the same condition as individuals killed in the local groundfish gillnet fishery, the body condition of porpoises that died during seining was compared with that of porpoises that died in gillnets in the Bay of Fundy. All porpoises in this analysis were dead, so it was possible to include dorsal blubber thickness measured just anterior to the dorsal fin ($n = 59$ and $n = 56$ for weir and gillnet animals, respectively), in addition to girth just anterior to the dorsal fin ($n = 60$ and $n = 56$ for weir and gillnet animals, respectively), and body mass ($n = 63$ and $n = 54$ for weir and gillnet animals, respectively). As above, mass was compared between gillnet and weir mortalities using ANCOVA with $\text{length}^{2.376}$ as the covariate. Blubber thickness is negatively correlated with body size in harbour porpoises (Koopman, 1998) and so both girth and dorsal blubber thickness were compared between gillnet and weir mortalities using ANCOVA with length as the covariate. ANCOVA revealed that blubber thickness showed no relationship with length in this sample ($p = 0.490$), thus the comparison was repeated using ANOVA.

Stomach contents

Stomachs were examined from porpoises killed in herring weirs ($n = 42$) and gillnets ($n = 46$) from 1992-1999 in the Bay of Fundy to determine if porpoises fed while in weirs. Porpoises that were shot were excluded from the analysis because it was not possible to determine if they were shot in weirs or in open water. Stomachs were examined following the methods described in Recchia and Read (1989).

Determination of feeding activity was examined in four different ways. The mass of forestomach contents of animals collected from weirs was compared with those from porpoises killed in gillnets. Content mass was determined by weighing the forestomach full and empty and subtracting the difference. There was a significant correlation ($p < 0.0001$ and $R^2 = 0.68$) between body mass and empty stomach mass. Thus, forestomach content mass was compared between the weir and gillnet samples using ANCOVA with body mass as the covariate. A comparison was also made between the proportion of empty stomachs (operationally defined as a forestomach content mass less than 10% of the empty forestomach mass) of porpoises caught in weirs and gillnets with a Chi-Square test. The proportion of stomachs containing fresh prey was compared between the weir and gillnet sample with a Chi-Square test. Fresh items were defined as ingested prey items in the stomach or oesophagus that still had $> 90\%$ of the flesh intact. The number and composition of otoliths and other prey remnants were determined for each sample. This allowed a prey species count to be assigned to each stomach. Each species of prey found in a stomach was counted as one unit. Species counts were compared between the weir and gillnet samples using an independent samples T-test.

Factors facilitating entrapment

Porpoises probably follow herring, their primary prey, into weirs, so porpoise entrapments should be correlated with the abundance of herring in weirs. Data on herring abundance in weirs were not available, therefore landings were used as a proxy (M.J. Power, Department of Fisheries and Oceans, St. Andrews, NB, Canada, pers. comm.) Porpoise entrapments from 1992 to 2001 are plotted by month with herring landing data from Grand Manan weirs during this same time period (Fig. 4).

Local weir fishermen claim that porpoises are most likely to swim into weirs on nights when high tide falls during darkness and just after a full moon. To determine whether timing of high tide or phase of the moon affected the probability of a porpoise swimming into a weir, each night of the monitoring period was classified according to these two variables. This analysis was restricted to 1995, 1999 and 2001, when there was an adequate number of recorded entrapments ($n = 82$, $n = 93$ and $n = 312$, respectively). Tables published by the Canadian Hydrographic Service (1995; 1999; 2001) provided daily times of high tides in waters around Grand Manan Island. Some light is present before sunrise and after sunset, so darkness was defined as the time span between civil twilights (sun's zenith distance is 96°). Civil twilight times were determined for each night using the Astronomical Almanac (Anon., 1995; 1999; 2001). Greenwich Mean Time (Longitude = 0°) was adjusted to the local time of North Head, Grand Manan Island (66°47'W) by adding four hours and 27 minutes. An hour was then subtracted to account for Daylight Saving Time observed in New Brunswick during the summer months. Nights were categorised according to whether high tide fell between twilights ($n = 158$) or not ($n = 52$).

To examine moon phase effects, nights were divided into four groups corresponding to the quarters of the moon. Dates of each quarter were determined from The Astronomical Almanac (Anon., 1995; 1999; 2001). The moon phase categories were labelled from 1 to 4 in cyclical order with 1 corresponding to the first quarter, i.e. from the new moon to the waxing half moon, and 4 corresponding to the last quarter, i.e. from the waning half moon to the new moon.

The moon affects the marine environment in two ways: illumination and tides. Additionally, the pattern of spring and neap tides is dictated by the position of the moon relative to the earth. Around Grand Manan Island, the greatest tidal amplitudes (i.e. spring tides) occur just after the full and new moons (Canadian Hydrographic Service, 1995; 1999; 2001). During these times, entrances to weirs will be the deepest at high tide. In an effort to separate lunar influences on tidal amplitude from brightness, the four moon phases were grouped to test for each effect (Table 2).

Table 2

Groupings of the four quarters of the moon to test for lunar influence on tidal amplitude (spring versus neap tides) and light level for waters around Grand Manan, NB. The first quarter is defined as the period from the new moon to the waxing half moon and the fourth quarter corresponds to the time period from the waning half moon to the new moon.

Lunar effect	1st quarter (1)	2nd quarter (2)	3rd quarter (3)	4th quarter (4)	Groupings of lunar quarters
Tidal amplitude	Large (spring)	Small (neap)	Large (spring)	Small (neap)	1&3 vs 2&4
Brightness	Dark	Bright	Bright	Dark	1&4 vs 2&3

More than one porpoise may swim into a particular weir on a given night so these entrapments cannot be considered as independent events. The animals may have been foraging in a group or the detection of a porpoise feeding may have attracted other animals to the same herring school. However, local regulations require that weirs must be built at least 305 metres apart, so two animals swimming into two different weirs on the same night were treated as independent events. Porpoise entrapments were viewed as binary response data. On a given night, a particular weir will either catch at least one porpoise ($Y = 1$) or it will not ($Y = 0$).

A logistic regression model describes the effects of a set of explanatory variables on a response variable (Agresti, 1984) and it is therefore appropriate for determining which factors affect porpoise entrapment. The logistic regression model can be generalised as:

$$\text{logit}(p) = \log[p/(1-p)] = \alpha + \beta_1 x_1 + ... + \beta_k x_k$$

where logit is the log odds transformation (i.e. the log of the probability that a given event will occur divided by the probability that it will not) and p is the probability that $Y = 1$ for the values of explanatory variables $X = (x_1, ... x_k)$ (Agresti, 1984). The explanatory variables included in this model were high tide at night, lunar influences on tidal amplitude and nocturnal light level as described in Table 2, Julian date, year and interactions between high tide at night, lunar effects, Julian date and year. As porpoise entrapments in gillnets and weirs peaked in mid-August (Read and Gaskin, 1988; Neimanis *et al.*, 1995; Fig. 4), date of entrapment was thought to be best represented by a quadratic (i.e. parabolic) function. Therefore, Julian date squared and higher order date variables were also considered in the model. Logistic regression models were fitted using the method of maximum likelihood. The null hypothesis (the probabilities of an entrapment occurring versus not occurring were equal when considering the explanatory variables) was tested against the alternative hypothesis (the probabilities were different). The test statistic used was minus twice the difference of the log likelihoods and it represented the joint significance of all explanatory variables ($\alpha = 0.05$). If the −2 Log Likelihood was significant (i.e. the probabilities differed), an analysis of maximum-likelihood estimates was used to assess the significance. Variables that did not contribute significantly to the occurrence of an entrapment ($\alpha > 0.10$) were then excluded from the model. Logistic regression analysis, in the context of a general linear model, was repeated until the most parsimonious model was determined.

Logistic regression analysis revealed that year was a significant variable because of the inclusion of 2001. This was a highly anomalous year (see Table 1). From 17 July to 9 September 2001, at least one porpoise swam into a weir every night. Thus, other factors influencing entrapment could not be evaluated during this year. We therefore excluded 2001 from the logistic regression model in order to determine equations that more accurately modelled porpoise entrapment during average years.

RESULTS

Catch composition

The sex ratio of weir entrapments was biased towards males in 1993 ($p = 0.005$), 1995 ($p = 0.005$), 2001 ($p < 0.001$) and all five porpoises released in 1996 were males (Table 3). The sex ratio did not differ significantly from year to year ($p = 0.212$) and there were no significant differences in sex ratio of the weir and gillnet samples in 1992 and 1993 ($p = 0.518$ and $p = 0.842$, respectively).

Mean length of female porpoises trapped in weirs did not differ significantly among years ($p = 0.513$), nor did it differ significantly from the gillnet sample in 1992 ($p = 0.086$; Table 4). However, mean length of female porpoises caught in weirs was significantly less than that of the gillnet sample in 1993 ($p = 0.002$). Mean length of male porpoises caught in weirs in 1995 was significantly different from males caught in 1992 ($p = 0.001$), 1993 ($p = 0.002$) and 2001 ($p = 0.004$). The mean length of males caught in weirs in 1992 and 1993 was significantly less than that of the gillnet sample from those years ($p = 0.002$ and $p < 0.001$, respectively).

Table 3

Sex ratios of harbour porpoises caught in herring weirs and gillnets from 1992-2001 in the lower Bay of Fundy. Ratios that differed significantly from 1:1 are indicated by *. There was no significant difference in sex ratio of the weir sample between years and sex ratios of the weir and gillnet samples did not differ significantly in 1992 and 1993. Statistical tests were performed at an alpha level of 0.05.

Year	Weir sample Males : Females (*n*)	Gillnet sample Males : Females (*n*)
1992	1.59 : 1 (44)	1.11 : 1 (19)
1993	1.74 : 1 (107)*	1.57 : 1 (35)
1994	0.87 : 1 (43)	1 : 1 (2)
1995	2.7 : 1 (37)*	0 : 1 (2)*
1996	1 : 0 (5)*	-
1997	1.89 : 1 (20)	2 : 1 (6)
1998	1.33 : 1 (14)	-
1999	1.77 : 1 (36)	-
2000	0.50 : 1 (12)	1 : 0 (1)
2001	1.94 : 1 (226)*	1 : 1 (6)
Total	1.72 : 1 (544)	1.22 : 1 (71)

Length distributions of female porpoises captured in weirs did not differ significantly among years ($p = 0.340$). Length distributions of male porpoises in weirs did differ significantly between years ($p < 0.001$), but when 1995 was excluded from the analysis, the distributions were no longer significantly different ($p = 0.426$). Length distributions differed significantly between animals caught in weirs versus gillnets (Fig. 2) in 1992 and 1993 for both females ($p = 0.033$ and $p = 0.001$, respectively) and males ($p = 0.007$ and $p < 0.001$, respectively).

Mean age of animals that died in weirs in 1992 (1.44 ± 0.53 years) was not significantly different ($p = 0.059$) from that in 1993 (2.94 ± 2.86 years). Age distributions also did not differ significantly between years ($p = 0.471$). Mean age of porpoises that died in gillnets was 3.12 ± 1.83 and 3.30 ± 2.19 years for 1992 and 1993, respectively. Mean age and age distribution of porpoises that died in weirs differed significantly from animals that died in gillnets in 1992 ($p = 0.014$ and $p = 0.002$, respectively). However, in 1993, there were no significant differences in mean age and age distribution between the two samples ($p = 0.623$ and $p = 0.209$, respectively). Pooled age distributions from the weir and gillnet samples are presented in Fig. 3.

Recaptures

Of the 390 porpoises that were fitted with a numbered identification tag, 25 of these animals swam into a weir a second time and four animals swam into a weir a third time. Twenty porpoises were recaptured during the same summer and four animals swam back into the same weir. Mean and median number of days between captures were 220.1 and 9, respectively (range = 0 to 3,274 days). Two animals released together from one weir swam into a second weir on the same day. One mature male first tagged in 1992 was later recaptured in 2001. The sex ratio of recaptures was 2.25:1 males to females, although this did not differ significantly from 1:1 ($p = 0.072$) nor was it significantly different from the sex ratio of weir animals caught only once (1.62:1 males to females) ($p = 0.530$). Mean length for recaptured males (123.8 ± 14.5cm) did not differ significantly ($p = 0.134$) from males trapped in weirs only once (128.6 ± 12.7cm). Mean length of females recaptured in weirs (123.1 ± 18.2cm) was not significantly different ($p = 0.283$) from females caught once (129.6 ± 16.7cm). Length distributions did not differ significantly between single captures and recaptures

Table 4

Mean lengths of harbour porpoises caught in herring weirs and gillnets from 1992-2001 in the lower Bay of Fundy. Males and females were analysed separately. For the weir sample, mean lengths for years that share the same upper case letters were not significantly different from each other. There were no significant differences between years in the gillnet sample. Lower case letters denote comparisons between the weir and gillnet samples for a given year. Different lower-case letters indicate significantly different mean lengths between animals caught in weirs versus gillnets. Statistical tests were performed at an alpha level of 0.05.

	Weir sample Mean length (cm) ± st. dev. (*n*)		Gill net sample Mean length (cm) ± st. dev. (*n*)	
Year	Males	Females	Males	Females
1992	125.4 ± 9.2 (26) Aa	127.9 ± 15.3 (17) Aa	138.2 ± 10.2 (8) b	137.9 ± 9.5 (9) a
1993	126.8 ± 10.6 (60) Aa	129.1 ± 9.8 (29) Aa	138.1 ± 10.0 (22) b	141.2 ± 14.4 (15) b
1994	127.0 ± 16.0 (20) AB	124.3 ± 22.9 (20) A	142.5 (1) -	144.0 (1) -
1995	140.3 ± 14.0 (26) B	129.0 ± 24.9 (9) A	-	130.0 ± 46.7 (2) -
1996	113.9 ± 20.4 (5) -	-	-	-
1997	129.5 ± 13.3 (10) AB	119.9 ± 13.3 (7) A	131.6 ± 15.6 (4) -	140.4 ± 27.6 (2) -
1998	121.8 ± 16.3 (8) AB	125.7 ± 28.2 (6) A	-	-
1999	131.4 ± 14.5 (24) AB	136.9 ± 20.5 (10) A	-	-
2000	125.3 ± 7.5 (4) AB	130.5 ± 12.9 (8) A	123.0 (1) -	-
2001	127.4 ± 11.7 (133) A	131.2 ± 15.1 (68) A	143.5 ± 13.4 (3) -	131.3 ± 10.0 (3) -

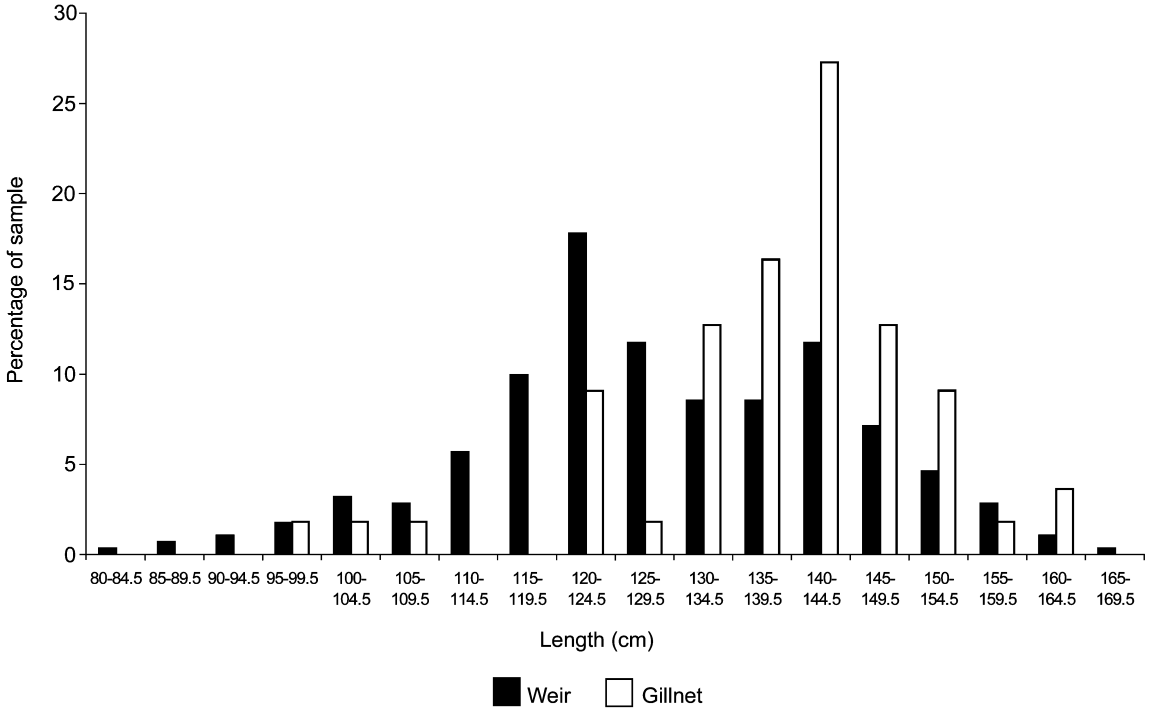

Fig. 2. Standard length distributions (in cm) of harbour porpoises trapped in herring weirs or killed incidentally in groundfish gillnets in the Bay of Fundy from 1992-2001. Animals in each length class are represented as a percentage of the total number of animals in each sample.

($p = 0.159$ and $p = 0.514$ for male and female comparisons, respectively). Both immature and mature animals were represented in the recaptured sample.

Mortality rates

Overall mortality for porpoises recorded in weirs was 9.5% and mortality rate of porpoises for which a seine was attempted was 12.0% (Table 5). Mortality rate for animals seined out using the mammal seine (2.5%) was significantly lower ($p < 0.001$) than that for animals removed with the herring seine (18.4%). Mortality rate of solitary porpoises seined out with the herring seine (19.0%) did not differ significantly ($p = 0.879$) from that of seines involving more than one porpoise (18.2%). Mean number of porpoises present in successful seines (2.13 ± 2.37) did not differ significantly ($p = 0.113$) from the mean number of animals in herring seines in which at least one animal died (2.45 ± 3.27).

Table 5

Annual mortality rates for harbour porpoises trapped in herring weirs in the lower Bay of Fundy from 1992-2001.

Year	Percentage mortality in herring weirs[1]	Percentage mortality during seining[2]
1992	18.0% (11/61)	22.9% (11/48)
1993	20.1% (29/144)	16.5% (22/133)
1994	19.3% (11/57)	27.7% (13/47)
1995	5.3% (4/75)	10.5% (4/38)
1996	42.9% (3/7)	33.3% (1/3)
1997	7.1% (2/28)	9.1% (2/22)
1998	6.1% (2/33)	14.3% (2/14)
1999	3.6% (3/84)	8.3% (3/36)
2000	0% (0/12)	0% (0/12)
2001	3.9% (12/310)	5.3% (12/228)
Total	9.5% (77/814)	12.0% (70/581)

[1] All porpoises that died in herring weirs divided by the total recorded in weirs excluding those with an unknown fate. [2] Porpoises that died during seining divided by the total number for which a seine was attempted. Only seines where GMWSRS personnel were present were included.

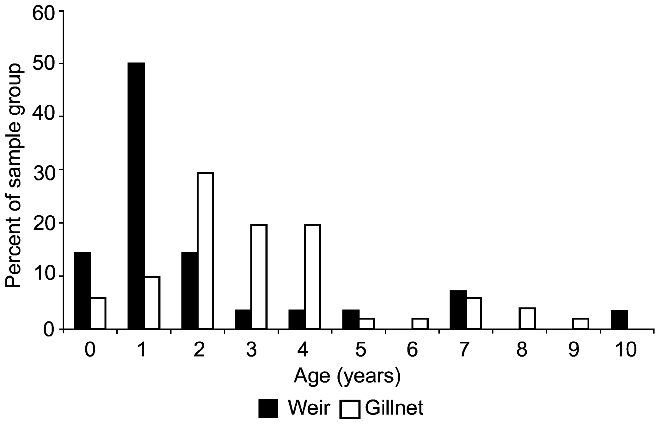

Fig. 3. Age distribution of harbour porpoises killed in herring weirs and gillnets in the lower Bay of Fundy from 1992-1999. Animals in each age class are represented as a percentage of the total number of animals in each sample.

Body condition

Porpoises that became trapped in weirs were in the same body condition as other porpoises in the Bay of Fundy; all juvenile (< 130cm) porpoises were in significantly ($p < 0.001$) better condition than stranded, emaciated animals. Body size was a significant factor in the analysis ($p < 0.001$). Once corrected for body size, juvenile weir and HI porpoises had a similar ($p = 0.85$) body mass (adjusted means: weir 31.99 ± SE 0.32kg; HI 31.52 ± SE 0.82kg) that were both significantly ($p < 0.001$) heavier than those of the juvenile stranded sample (adjusted mean 21.29 ± SE 0.74kg).

Porpoise characteristics associated with mortality

Sex ratio did not differ significantly ($p = 0.872$) between porpoises that survived the seine (1.73:1 males: females) and animals that died during seining (1.65:1 males: females). Mean length of female porpoises that survived the seine

(129.0 ± 16.2cm) did not differ significantly ($p = 0.636$) from mean length of females that died (130.8 ± 20.8cm). Mean length of males that survived seining (128.3 ± 12.8cm) did not differ significantly ($p = 0.645$) from males that died (129.3 ± 11.8cm). Length distributions did not differ significantly between the two samples either ($p = 0.764$ and $p = 0.697$ for females and males, respectively).

Body condition did not differ significantly between animals that survived the seine versus those that died. Once the significant effect ($p < 0.001$) of body size was removed, body mass of released porpoises (adjusted mean 41.29 ± SE 0.35kg) and weir mortalities (adjusted mean 41.24 ± SE 0.52kg) were not significantly different ($p = 0.945$).

There were no significant differences ($p = 0.585$) in girth between porpoises killed in the weir fishery (adjusted mean 86.48 ± SE 0.54cm) and those from groundfish gillnets (adjusted mean 86.92 ± SE 0.56cm) once the effect of body size ($p < 0.001$) was removed. Mean blubber thickness of weir (mean 21.1 ± SE 0.8mm) and gillnet mortalities (mean 19.9 ± SE 0.8mm) was not significantly different ($p = 0.241$). However, once body size was accounted for ($p < 0.001$), porpoises that died in the weir fishery (adjusted mean 43.0 ± SE 0.4kg) possessed slightly higher ($p = 0.004$) mean body masses than animals from gillnets (adjusted mean 41.2 ± SE 0.4kg).

Stomach contents

Using body mass as a covariate, forestomach content mass of animals that died in weirs was significantly less than that of animals killed in gillnets ($p < 0.001$). However, there was no significant difference between the proportion of animals with empty stomachs that died in weirs (21.4%) versus those that died in gillnets (14.0%) ($p = 0.349$). Only one stomach of the weir-caught sample (2.4%) contained fresh prey compared to 18 stomachs (36%) from the gillnet sample ($p < 0.001$). Herring comprised 97.9% of the prey in the weir sample, but only 78.8% of the prey in the gillnet sample (Table 6). Excluding empty stomachs, the prey-species count for weir-caught animals (1.21) did not differ significantly ($p = 0.078$) from that of animals that died in gillnets (1.43).

Table 6

Proportion of prey items ingested by harbour porpoises killed in herring weirs and gillnets in the western Bay of Fundy from 1992-1999 as indicated by prey otoliths and hard parts.

Prey species	Weir sample	Gillnet sample
Clupea harengus	0.979	0.788
Merluccius bilinearis	0.005	0.122
Gadid sp.	0.009	0.009
Urophysis sp.	0	0.004
Myxine glutinosa	0	0.065
Cephalopod sp.	0.005	0.013
S. scombrus	0.002	0

Factors associated with entrapment

Porpoise entrapments mirrored herring landings, with both showing peaks in August (Fig. 4). The final logistic regression model included the intercept ($p = 0.010$), tide phase at night ($p = 0.006$), moon light ($p = 0 .050$), Julian date ($p = 0.015$), Julian date squared ($p = 0.011$) and moonlight by Julian date ($p = 0.032$). It is represented by the following equation:

$$\text{Logit}(p) = -47.799 + 1.1926(\text{tide}) - 6.7703\beta + 0.40955(\text{date}) - 0.000963(\text{date})^2 + 0.03314\beta(\text{date})$$

where tide = 1 if high tide falls during the night or tide = 0 if it does not and date is the numerical day of the year, with 1 January as 1. Solving the equation for β (moonlight) with a given tide and day of the year will give the logit (L) for that day. The anti-logit represents the probability that a porpoise will swim into a weir. Using the following equation: $p = e^L/1{+}e^L$, one can determine the probability of a porpoise swimming into the weir on that given day.

Year, tidal amplitude and other interactions were not significant ($p > 0.10$) and were therefore excluded from the model. The odds ratio of an entrapment occurring on a night when high tide falls during darkness was 3.3. Odds ratios for moonlight, Julian date, Julian date squared and moonlight*date are 0.001, 60.1, 0.91 and 1.4, respectively.

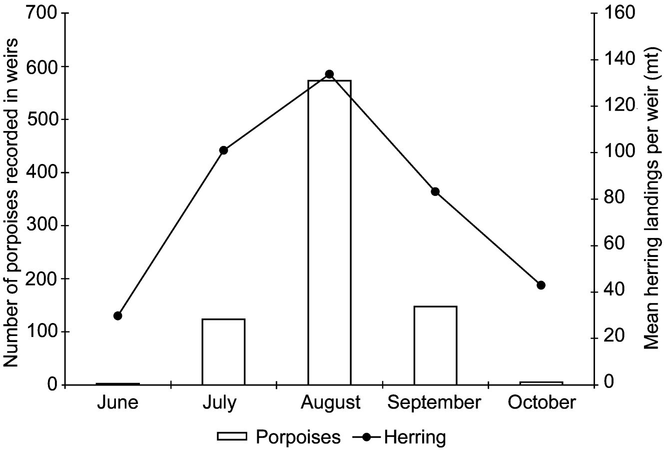

Fig. 4. Monthly distribution of harbour porpoises recorded in herring weirs around Grand Manan versus mean herring landings (mt) per Grand Manan weir from 1992-2001.

DISCUSSION

The weir fishery in the lower Bay of Fundy has an insignificant impact on the Bay of Fundy/Gulf of Maine population of harbour porpoises. With the assistance of the porpoise release programme, weirs kill less than 0.01% of the population each year. Since the inception of the HPRP, weir mortality accounts for 1.03% of PBR each year (0% in 2000 to 3.88% in 1993).

With the exception of 1995, the population of animals that entered herring weirs was similar from year to year. Mean age and age distributions for both sexes did not differ between years, nor did mean length and length distributions for trapped females. If 1995 is excluded, the same was true for male porpoises. However, the weir sample in 1995 was over-represented by large males and these males were significantly longer than males caught in 1992, 1993 and 2001. In general, males are over-represented in the weir sample, comprising 63.2% of the total porpoises examined, although sex ratio was only significantly male-biased in 1993, 1995, 1996 and 2001. The biological significance of the shift in composition of weir entrapments in 1995 is unknown.

Porpoises captured in herring weirs represent a slightly different subset of the general population than animals entangled in gillnets. Sex ratio did not differ significantly between the weir and gillnet samples and the range of lengths and ages were similar between weir (84.5-167cm and 0-10 years) and gillnet (97-163cm and 0-9 years) animals. However, there were significant differences in mean length and age and their respective distributions. With the exception of mean length of female porpoises in 1992, mean length and length distributions differed significantly between the weir and gillnet samples for both sexes. Weir-caught porpoises had a greater proportion of smaller animals than the gillnet sample and mean length was shorter for the weir group. Likewise, weir animals from 1992 had a younger mean age and a greater proportion of younger animals than porpoises caught in gillnets in 1992. There was no significant difference in age parameters between the two groups in 1993. We conclude that weirs and gillnets capture animals from the same general population, but weirs are biased towards younger, smaller animals.

Only a small proportion (6.4%) of porpoises fitted with an identification tag swam into weirs a second time and of those, 16% were recaptured a third time. It is not possible to predict which animals are more likely to swim into a weir again. The catch composition of recaptures was not significantly different from animals that were only trapped in weirs once. However, a recaptured porpoise has a higher probability of swimming into a weir a third time than a released animal has of becoming trapped a second time.

The conclusion that weirs capture animals from the same general population as animals taken in other human interactions is further strengthened when considering body condition data. There were no significant differences in body condition between animals caught in weirs and those killed in gillnets or shot, but both were in significantly better condition than the stranded and emaciated sample. These results demonstrate that the porpoises observed entrapped in weirs exhibit the same body condition as other porpoises in the Bay of Fundy, and that they are also much fatter than porpoises known to be in poor body condition. If porpoises trapped in weirs were debilitated from illness or starvation, they would be expected to be in poor condition. Andersen (1974) concluded that 90% of porpoises caught in Danish trap nets were sick, many with lung parasites and pneumonia. Chronic pneumonia is typically accompanied by a loss in body condition (e.g. Ettinger and Feldman, 2000), which was not evident here. Blood analyses (haemogram and chemistry) did not reveal overt indications of illness (Koopman *et al.*, 1995; 1999), although cortisol levels were higher than in other cetaceans. Increased levels of cortisol can accompany fever or illness (e.g. Lorenz and Cornelius, 1993), but based on the absence of other haematological or gross evidence of disease, we believe these increases are caused by the stress of entrapment and release. Andersen (1974) also found that many porpoises in Danish trap nets died within one hour to two days after capture. Our recapture and electronic tagging data do not support a similar situation for porpoises caught in herring weirs. Mean satellite tag duration to date has been approximately 70 days and the longest track duration was 212 days (Westgate and Read, 1998). The longest period between recaptures was nine years. We therefore conclude that Andersen's hypothesis (1974) does not apply to porpoises caught in herring weirs in Canada. Further investigations using blood parameters and serology, histopathology, microbial cultures and samples for parasitic analyses can be carried out to examine the health of this population in more detail.

Mortality rates of porpoises in weirs were considerably less than the 39% estimate published by Smith *et al.* (1983). The overall mortality rate of porpoises in weirs was 9.5% of all recorded entrapments. This includes 18 porpoises that were shot. Seining porpoises from weirs does pose a risk to the animals, as mortality of porpoises for which a seine was attempted was 12%. This value is higher than overall reported mortality because it excludes animals that swam out on their own or were swept out. However, without a dedicated effort to remove porpoises during seining, mortality would probably be higher. The probability of mortality during seining can be further minimised by using the marine mammal seine, which has proven to be highly effective in the safe release of porpoises. Only 2.5% of porpoises that were seined out with the mammal net died, versus 18.4% that died in the herring seine net. The larger mesh size of the mammal seine allows herring to escape, leaving only porpoises behind in the seine. During seining, billows and pockets constantly form in the seine net due to the effects of tidal currents. If a porpoise becomes trapped in a pocket below the surface, it will die unless it can surface. The mammal seine is lighter than the herring seine and it floats to surface once the weights on the bottom of the seine are pulled up. This allows porpoises caught in billows beneath the water to swim in the net to the surface to breathe. The risk of mortality has been further reduced in recent years by placing swimmers in the weir to assist porpoises that become trapped in the folds and billows of the seine.

Mortality appears to be a random event among porpoises captured in weirs. There were no significant differences in sex ratio, length parameters or body condition of porpoises that survived the seining versus those that died. The mass of porpoises that died in weirs was slightly greater, but condition did not differ in blubber thickness or body girth, compared to porpoises that died in other human interactions in the Bay of Fundy. There was no relationship between body condition and the likelihood of mortality in porpoises trapped in weirs. We therefore conclude that the animals that perish during release attempts represent a random subsample of the population caught in weirs.

Some porpoises eat while trapped in weirs, but perhaps not at the same rate as animals captured in gillnets. The proportion of empty stomachs did not differ significantly between animals that died in weirs versus those that perished in gillnets. However, forestomach content mass of the weir sample was significantly less than that of the gillnet sample, indicating that weir-caught animals were either eating less or had not eaten for a longer period of time before death. Trapped porpoises have been observed to vomit when they become entangled in the herring seine. This behaviour will negatively bias measurements of stomach content mass. However, it is not known if a similar proportion vomit when they become entangled in groundfish gillnets. Porpoises in weirs were eating herring almost exclusively before they died, but it is not possible to infer if they were eating the herring before or after they entered the weir, as many animals are seined within 24 hours of entering a weir. The proportion of fresh prey items was significantly less in the weir sample than the gillnet sample, suggesting that animals in weirs had not fed as recently as animals in gillnets. However, the one weir-caught porpoise with fresh prey items in her stomach had been in the weir for at least 30 hours. She was a lactating female and fed while trapped. We conclude that porpoises trapped in weirs feed less than porpoises in open water, but animals with increased energy demands may have little choice and must eat while trapped.

Porpoise entrapment is most certainly correlated with herring abundance in weirs, as illustrated in Fig. 4. Entrapment numbers peak in August, concurrent with the highest herring landings from Grand Manan weirs. Since the implementation of the HPRP in 1991, herring landings per Grand Manan weir peaked in 1993, closely followed by 2001 (M.J. Power, Department of Fisheries and Oceans, St. Andrews, NB, Canada, pers. comm.). These years also corresponded to the highest numbers of porpoise entrapments (Table 1). 2001 was unprecedented in terms of porpoise entrapments and it was also anomalous in terms of amount of herring landed in June. Herring landings per Grand Manan weir in June 2001 were at least 2.5 times higher than any other recorded landings for June since 1978. Most harbour porpoises of the Bay of Fundy/Gulf of Maine population are believed to enter the Bay in the early summer, i.e. June and July (Gaskin, 1992; Palka *et al.*, 1996). Perhaps the unprecedented numbers of herring in weirs early in the season attracted more porpoises to the inshore waters around Grand Manan, where they remained for the 2001 season.

Certain environmental factors were also found to affect the probability of a porpoise swimming into a weir on a given night. Entrapment was related to tidal cycle (i.e. water depth at night), moon phase (i.e. moonlight level associated with moon phase) and time of year. Water depth varies with tidal cycle. High tides that fell during darkness as opposed to daylight significantly increased the probability (3.3:1) that a porpoise swam into a weir. High tides that occurred during the day had no effect on porpoise entrapment because porpoises have only been known to enter weirs during the night. This is most likely related to prey movement. During the day, herring are found in deeper water and it is only at night that they move inshore and become vulnerable to the weir fishery (Anthony, 1972). The Bay of Fundy experiences some of the world's greatest tidal amplitudes (> 10m) and weirs are built very close to shore with their entrances facing shoreward. Although none of the sea floor at the entrances of weirs monitored was ever exposed at low tide, the water level may have been shallower than some critical level for porpoises.

Lunar influence on light levels at night was also found to contribute significantly to the probability of weir entrapments. However, this variable was involved in a higher order interaction in the regression model, and must be interpreted by solving the regression equation for a given set of circumstances. The significance of lunar influence is not surprising, as herring behaviour is affected by light levels. Herring move furthest inshore on dark nights (Anthony, 1972).

Time of year also affects the probability of porpoise entrapments in weirs. An increase in local porpoise abundance is inferred from gillnet bycatches (Read and Gaskin, 1988; Trippel *et al.*, 1996). These authors found that the majority of bycatches in gillnets occurred in August. The HPRP is usually busiest during the last two weeks of August and mean date of entrapment events peak in August (Fig. 4). As previously mentioned, this also coincides with the highest herring landings from local weirs. It is therefore expected that porpoise entrapment will most likely occur when local porpoise densities and inshore landings of their primary prey are highest.

At present, we are only beginning to understand the nature of porpoise entrapments in herring weirs. It is not clear why 1995 was an anomalous year regarding the composition of porpoises or why 2001 was unprecedented in terms of numbers of entrapments. Nor is it clear why porpoises appear reluctant to leave weirs during the daytime. Further research on the ecology and behaviour of this species may help answer these questions.

The HPRP has become a successful, immediate mitigation solution for porpoise entrapment in weirs. However, like whale disentanglement, we view this as a triage procedure to minimise porpoise mortality in the short-term. Ultimately, we hope to develop strategies to reduce or eliminate porpoise entrapment altogether. The success of such long-term mitigation of bycatch requires the input and cooperation of the weir fishery.

Given the positive relationship established with weir fishermen, cooperation towards long-term mitigation of porpoise entrapment should be possible. As a first step, formal interviews with weir fishermen were conducted in the summer of 2003 to identify and discuss possible solutions. As with any such project, it will likely be some years before these mitigation measures can be implemented. In the mean time, continuation of the HPRP will provide opportunities to further study, monitor and help reduce incidental mortality in this porpoise population.

ACKNOWLEDGEMENTS

This manuscript is dedicated to the memory of the late Dr David Gaskin, whose vision and pioneering research introduced us all to the study of harbour porpoises. The Harbour Porpoise Release Program has been developed and refined over the past decade with the assistance of all members of the Grand Manan Whale and Seabird Research Station. The authors would particularly like to thank Patrik Börjesson, Tara Cox, Dave Johnston, Rob Ronconi, Rick Smith, Michelle Theberge, John Wang and Sarah Wong for participation in and dedication to the Release Program. This study, and the Release Program, would not have been possible without the participation and cooperation of the weir fishermen of the Bay of Fundy. The Harbour Porpoise Release Program has been supported by many sources including: the Body Shop, Connors Brothers Limited, Chicago Zoological Society, the Duke University Marine Laboratory, Fundy Community Foundations, Humane Society of the United States, International Fund for Animal Welfare, New England Aquarium, United States Marine Mammal Commission, Whale and Dolphin Conservation Society, World Wildlife Fund Canada and donations from individuals. Funding for the scientific components of the Release Program has been provided by Greenpeace International, the IWC, National Geographic, WhaleNet and the United States National Marine Fisheries Service. Porpoises were handled according to approved Duke University Animal Care and Use Committee Protocols, and all work was conducted under approved and authorised permits issued by the Department of Fisheries and Oceans, Canada.

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J. CETACEAN RES. MANAGE. 6(1):19–24, 2004

Updated estimates of harbour porpoise (*Phocoena phocoena*) bycatch in the Danish North Sea bottom-set gillnet fishery

MORTEN VINTHER AND FINN LARSEN

Department of Marine Fisheries, Danish Institute for Fisheries Research, Charlottenlund Castle, DK-2920 Charlottenlund, Denmark
Contact e-mail: fl@dfu.min.dk

ABSTRACT

The bycatch of harbour porpoise in the Danish North Sea bottom-set gillnet fisheries between 1987-2001 is estimated using two methods involving extrapolation of observer data. When observed entanglements are extrapolated to fleet level based on target species landings, the annual bycatch was estimated to be in the range of 2,867-7,566 harbour porpoise with a mean of 5,817. When observations are extrapolated based on fishing effort, estimates are in the range of 3,887-7,366 porpoises with a mean of 5,591. Both methods estimate a significant reduction in bycatch in the most recent years due to a decrease in both effort and landings. However, the reduction is less pronounced with the effort based method.

KEYWORDS: HARBOUR PORPOISE; FISHERIES; GILLNETS; BYCATCH; NORTH SEA

INTRODUCTION

Bycatches in Danish fisheries have been monitored using observer programmes since 1992. High bycatches of harbour porpoises (*Phocoena phocoena*) in the North Sea bottom-set gillnet fisheries for turbot, cod, hake and plaice were documented by Vinther (1995; 1997; 1999), who estimated the average total annual bycatch in the period 1994-1998 at 6,785 porpoises (Vinther, 1999). The total bycatch was well above the level agreed by ASCOBANS (Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas) in 1997 as acceptable (ASCOBANS, 1997), and led to the formation of a Danish action plan to reduce bycatch of porpoises in the North Sea (Ministry of Environment and Energy, 1998). The action plan recommends the use of acoustic alarms (pingers) as a primary means of mitigation, and from 2000, use of pingers became mandatory in the Danish wreck fishery in the North Sea (ICES sub-area IV) in the months August-October.

Vinther (1999) extrapolated the observed bycatch number per landed weight of target species to the total fleets' landings of target species. This method assumes implicitly that the fish Catch Per Unit Effort (CPUE) is constant during the period. However, the stock sizes and fish quotas of the target species have changed considerably during the last ten years (ICES, 2003). Such changes may violate the assumption of constant CPUE and the bycatch estimates may be seriously biased. This paper explores the possibilities of taking changes in fish CPUE into account in extrapolating observed bycatch to the total fleet.

The total Danish North Sea porpoise bycatch for the period 1987-2001 is estimated using the 1992-1998 bycatch observations presented in Vinther (1999), and additional bycatch observations from the period since 1998. Two methods for extrapolation are used; the landings-based method used by Vinther (1999); and an effort-based method. The underlying assumptions for these methods as well as sampling strategies for the relevant fisheries are also discussed.

MATERIALS AND METHODS

Materials and methods used in this paper are in most cases similar to those presented by Vinther (1999) and therefore details are only provided for additional data and methods. The time series of observer data has been extended with data from additional fishing trips since 1998, including data from 10 trips in the cod fishery, 8 trips in the plaice fishery, 8 trips in the sole fishery and 1 trip in the turbot fishery. Statistics for the whole sampling period (1992-2001) are presented in Table 1. Although fish landings and harbour porpoise bycatch were monitored from more than 5,500km of net, the observer coverage has been rather low. The average sampling activity was highest in the turbot fishery where, on average, 1.1% of the annual landings were monitored. Lowest sampling coverage was in the plaice fishery with just 0.3% coverage. For safety and cost/benefit reasons, larger vessels were preferred for monitoring such that smaller vessels were under-sampled as presented for the cod fishery in Fig. 1. Mean bycatch numbers per landed target species weight, using the same stratification and method as in Vinther (1999), are presented in Table 2. Landings and effort statistics for the total Danish North Sea set-net fleet were separated into fisheries using cluster analysis on the species composition for each individual trip (see Vinther, 1999). For the hake fishery, it should be noted that data are available from only two trips, both in 1997; thus the estimated bycatch rate should be treated with caution. For the plaice fishery, it should be noted that all porpoise bycatch was in the first quarter of the year and that 17 of the 21 porpoises bycaught were taken on two trips in January-February 1998; thus the bycatch rate for this fishery should also be treated with some caution.

Observed bycatch in this study is recorded as numbers per km-hours of net fished. Such an effort measure is however not available for the total fleet and an indirect way of extrapolation of observed bycatch to total fleet level must be applied. In addition, data for several years have to be merged, as sampling within a year has been too limited for an

Table 1

Mean annual fleet landings (in tonnes) for the Danish North Sea set-net fleet given by fishery and period, and corresponding observer sampling activity and porpoise bycatch numbers.

	Total fleet (mean annual)			Observed fleet (summed for all years)					
Fishery and period	Quarter of the year	Target species landings (tonnes)	Total landings (tonnes)	No. of trips	No. of sampling units	Target species landings (tonnes)	Length of nets (km)	Mean observer coverage (%)	No. of harbour porpoise
Cod (1993-2000)	1	3,241	3,699	18	280	68.5	356	0.3	36
	2	1,965	2,263	14	215	49.4	243	0.3	5
	3	2,296	2,661	58	681	188.3	1,175	1.0	86
	4	2,266	2,571	28	393	144.1	887	0.8	22
Hake (1997)	2-3	90	212	2	32	3.1	122	3.4	4
Plaice (1994-2001)	1	1,607	1,943	9	61	61.6	498	0.5	21
	2	1,217	1,718	12	33	8.5	157	0.1	0
	3-4	479	607	3	3	0.2	7	0.0	0
Sole (1992-2000)	1-4	768	926	22	68	8.2	875	0.1	0
Turbot (1993-2000)	2	280	489	13	110	24.4	945	1.1	78
	3	67	124	5	41	6.2	301	1.2	77
Other (1993-2000)	1-4	-	94	-	-	-	-	-	-
Small vessels	1-4	-	2,500	-	-	-	-	-	-
Total	1-4	14,276	19,807	184	1,917	562.6	5,565		329

Table 2

Bootstrap estimates of porpoise bycatch (in numbers per 1,000 tonnes landed target species weight) by fishery and period, and corresponding mean target species CPUE (in kg per day at sea).

Quarter and fishing method	Bycatch (numbers per 1,000 tonnes)			CPUE
	Mean	95% CI	CV %	Mean
Cod (1993-2000)				
1, 2, 4, wreck	33	8-81	53	438
3, wreck	502	186-1,188	46	418
1, 3, other	577	364-1,112	29	449
2, 4, other	218	135-357	24	413
Stratified	281		20	
Turbot (1993-1999)				
2	3,211	2,233-4,590	18	
3	12,417	6,786-19,453	27	
Stratified	5,067		16	250
Hake (1997)				
3	1,332	310-4,139	59	234
Plaice (1994-2001)				
All	295	176-484	26	681
Sole (1992-2000)				
All	0			186

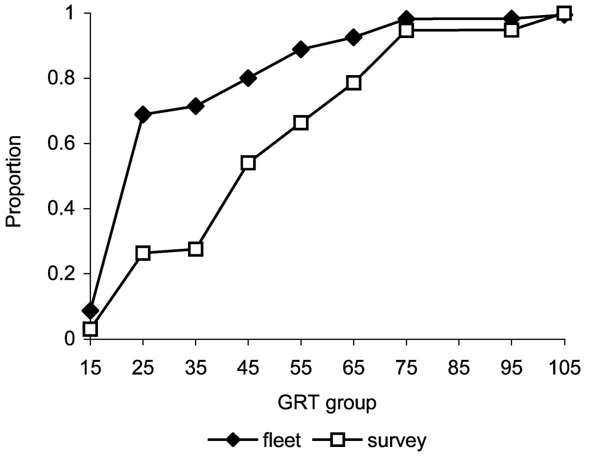

Fig. 1. Proportion of cod landings by vessel GRT group in the fleet and survey, 1993-2001.

estimate of total annual bycatch. Two methods for extrapolation are applied in this study: the ‘landings’ method; and the ‘effort’ method. The ‘landings’ method is the same applied in Vinther (1999) and assumes constant CPUE for the target species during the whole period, as effort is derived from the fleet’s landings and the observed CPUE from the sampling period.

‘Landings’ method

$$total\ \ bycatch_{year} = \frac{\sum_{y=Y1}^{Y2} obs\ \ bycatch_y}{\sum_{y=Y1}^{Y2} obs\ \ effort_y} * \frac{\sum_{y=Y1}^{Y2} obs\ \ effort_y}{\sum_{y=Y1}^{Y2} obs\ \ landings_y} * fleet\ \ landings_{year} \Leftrightarrow$$

$$total\ \ bycatch_{year} = \frac{\sum_{y=Y1}^{Y2} obs\ \ bycatch_y}{\sum_{y=Y1}^{Y2} obs\ \ landings_y} * fleet\ \ landings_{year}$$

where *total bycatch*$_{year}$ is the estimated bycatch within a fishery for a given year; *obs bycatch* and *obs effort* are the observed bycatch and effort from surveys; and *Y1* and *Y2* are the first and last year in a survey for a specific fishery.

Total annual bycatch was estimated by fishery and season (see Table 1), however, the fishery and season indices have been left out of the notation for clarity.

The ‘landings’ method was used for an estimate of the 1994-1998 average bycatch and later for an estimate of annual bycatch in the period 1990-2000[1].

However, the fisheries’ annual landings (Fig. 2) and effort varied considerably in the period 1987-2001, and an assumption of constant CPUE of the target species is clearly

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[1] In litt. Danish Minister of Food, Agriculture and Fisheries to EU Commissioner Fischler (ASCOBANS AC8/Doc. 18).

not met, as CPUE of cod, hake and turbot have varied by a factor of around two (Fig. 3). Plaice CPUE seems more stable, even though the annual landings have varied by a factor of five.

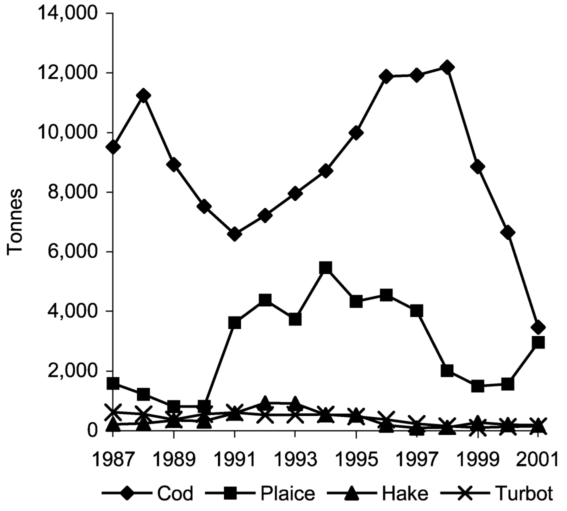

Fig. 2. Annual landing weights of the target species in four Danish North Sea set-net fisheries for the period 1987-2001.

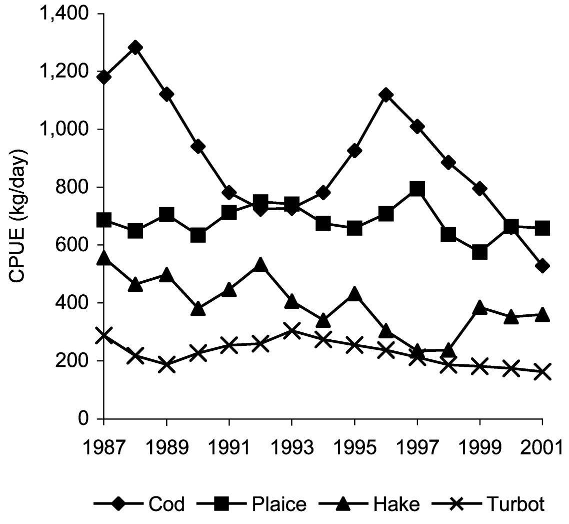

Fig. 3. Target species CPUE (in kg landings per day at sea) for the Danish North Sea set-net fleet for the period 1987-2001.

The 'effort' method estimates bycatches on the basis of annual fleet effort and observer data on bycatches and fish landings. This method also assumes constant CPUE, but just in the observer period.

'Effort' method

$$\text{total bycatch}_{year} = \frac{\sum_{y=Y1}^{Y2} \text{obs bycatch}_y}{\sum_{y=Y1}^{Y2} \text{obs landings}_y} * \frac{\sum_{y=Y1}^{Y2} \text{fleet landings}_y}{\sum_{y=Y1}^{Y2} \text{fleet effort}_y} * \text{fleet effort}_{year}$$

Definitions are as for the 'landings' method.

The fleets show a highly variable fish CPUE (Fig. 3), which also should be reflected in the observer data. A generalised linear model (GLM), using a Gamma error distribution and a log link function, was used to model the observed, or survey CPUE (kg/length of net) in the cod fishery. This fishery has had the most extensive observer coverage (1,701 hauls) and has a highly variable historical CPUE. The cod fishery takes place over shipwrecks or similar objects with a relatively high density of cod, using relatively short lengths of nets; or on other bottom types with a relative low density of cod, using relatively longer chains of nets. The highest catches in the non-wreck fishery are obtained when cod are moving actively around, e.g. during spawning migration, such that a possible seasonal effect on CPUE might be different for the two types of cod fishery. The model includes an overall cod density term (year), the two types of fishery (bottom) and an interaction effect between season (quarter) and type of fishery (quarter*bottom):

Model: CPUE = year bottom quarter*bottom.

All model variables are categorical.

RESULTS

All parameters in the GLM model for the survey cod CPUE were highly significant (Table 3). The estimated year factor for survey CPUE (Fig. 4) follows the trend in the fleet CPUE reasonably well, taking the relatively low sampling intensity for the years after 1997 into account. The estimated Spawning Stock Biomass (SSB; ICES, 2003) can be seen as a proxy for the 'fishable' biomass of cod, as the gillnet fishery mainly targets the larger cod. There is a high correlation between the ICES SSB estimate and total fleet CPUE (Fig. 4).

Table 3

Likelihood ratio statistics for testing the significance of each effect specified (type III analysis) in the survey cod CPUE model.

Effect	Degrees of freedom	χ^2	Probability>χ^2
Bottom	1	380.6	<0.0001
Year	7	256.5	<0.0001
Bottom*quarter	6	23.3	0.0007

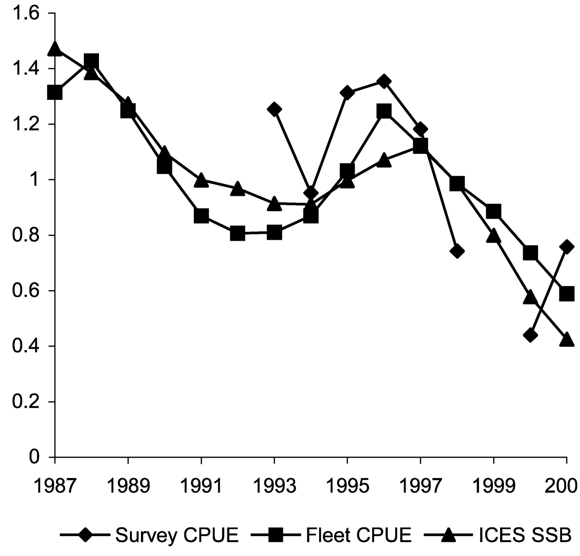

Fig. 4. Survey CPUE (year index from GLM model), total fleet CPUE index (target species landings per day at sea) and ICES cod Spawning Stock Biomass for the period 1987-2001. All values are scaled to the series mean.

The estimated total porpoise bycatch in the period 1987-2001 using the two methods is presented in Table 4. The overall mean bycatch for the two methods is quite

Table 4

Estimated annual bycatch (in numbers) of harbour porpoise in the Danish North Sea set-net fisheries by fishery and quarter for the years 1987–2001 for both the 'landings' and the 'effort' methods of extrapolation.

Fishery	Quarter	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	Mean
'Landings' method																	
Cod, wreck	1, 2 and 4	136	146	116	115	96	98	96	109	122	153	154	152	117	84	40	116
	3	320	595	468	133	188	314	529	526	654	660	650	752	443	397*	259*	459
Cod, other	1 and 3	2,034	2,028	1,732	1,154	1,037	1,125	1,416	1,394	1,680	1,883	1,813	1,983	1,582	1,029	535	1,495
	2 and 4	268	459	318	384	326	361	332	423	454	583	614	580	368	336	175	399
Hake	All	283	319	451	437	776	1,233	1,212	718	704	246	119	144	357	272	242	501
Turbot	2 and 3	3,122	2,818	1,905	2,792	3,084	2,676	2,731	2,782	2,413	1,896	1,193	770	533	685	742	2,009
Plaice	All	468	362	239	242	1,066	1,290	1,105	1,614	1,281	1,342	1,188	594	441	463	872	838
Sole	All	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
All	All	6,630	6,727	5,230	5,257	6,573	7,099	7,421	7,566	7,308	6,762	5,731	4,974	3,840	3,266	2,867	5,817
'Effort' method																	
Cod, wreck	1, 2 and 4	97	99	89	104	102	117	116	123	117	121	130	148	126	106	67	111
	3	276	405	383	173	291	386	606	555	568	475	587	738	511	570*	405*	462
Cod, other	1 and 3	1,410	1,342	1,217	919	1,076	1,307	1,603	1,578	1,546	1,472	1,514	1,943	1705	1,420	950	1,400
	2 and 4	236	323	294	401	386	443	428	456	435	445	538	565	411	413	261	402
Hake	All	119	160	212	268	405	541	697	493	381	189	119	142	217	181	158	285
Turbot	2 and 3	2,719	3,229	2,547	3,067	3,033	2,577	2,245	2,534	2,366	1,999	1,402	1,034	737	985	1,144	2,108
Plaice	All	465	380	231	260	1,018	1,172	1,014	1,627	1,325	1,292	1,018	636	521	475	903	822
Sole	All	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
All	All	5,322	5,938	4,973	5,191	6,312	6,543	6,709	7,366	6,737	5,991	5,308	5,206	4227	4,149	3,887	5,591

* Bycatch is overestimated, as the effect of the use of pingers has not been taken into account.

similar for each fishery except for the hake fishery where the 'effort' method gives a 43% reduction. The hake fishery illustrates clearly the difference between the two methods as the observer programme includes just one year's (1997) data for this fishery. Therefore, the two methods estimate the same bycatch for 1997, but the 'effort' method estimates a relatively lower bycatch for years with a higher CPUE than in 1997. The time series had the lowest recorded CPUE in 1997 (Fig. 3), such that the 'effort' method estimates a considerably lower bycatch for the whole period.

The decrease in CPUE for cod and turbot in the most recent years (see Fig. 3) results in a higher estimated bycatch in these years for the 'effort' method, such that the decline in bycatch due to lower landings of target species is less than with the 'landing' method.

DISCUSSION

The estimates of porpoise bycatch presented here rest on a number of assumptions, some common to both methods employed in this study and some special to one or the other. This discussion section focuses on the most important of these assumptions and, where possible, ways of resolving the problems identified are recommended.

Two assumptions are common to both methods employed. The first is that porpoise densities in the areas covered by the fisheries are constant in the extrapolation period. Changes in porpoise densities in the areas covered by the fisheries could be caused by changes in porpoise population size, changes in porpoise distribution, changes in the distribution of the fisheries, or a combination of these. There are currently no data to explore changes in porpoise distribution or abundance, but the results from SCANS-II, an abundance survey of small cetaceans planned for 2005-2006, can potentially provide information on this. There are no indications of systematic changes in fisheries distribution that could give rise to the downward trend in total bycatch seen since the mid-1990s, and it is hard to imagine how this, although theoretically possible, could happen in practice. A more parsimonious explanation for the reduced bycatch is the reduction in effort which has taken place in most fisheries.

The second assumption common to both methods is that the observer data are representative for the whole fleet, not just regarding bycatch rate but also regarding target species CPUE and fishing area. This assumption is probably to some degree violated, as primarily larger vessels are chosen for the observer programme for cost/benefit and safety reasons. The larger vessels tend to fish further offshore and bycatch rates in these areas may be different from bycatch rates in more coastal areas. At present there are insufficient data to evaluate the effect of this bias in observer coverage, and an effort to include the smaller vessels in future sampling programmes is recommended.

A known bias common to both methods employed is created because the effects on bycatch rates of using acoustic alarms (pingers) in the wreck fishery are ignored. Use of pingers since August 2000 has been mandatory in the Danish cod wreck-fishery in August-October, and the effect of using pingers is reported to be close to a 100% reduction in bycatch in the observed part of the wreck fishery (Larsen, 1999; Larsen *et al.*, 2002). However, there are no ways to assess the efficacy of pinger use in the unobserved part of the wreck fishery. Thus the bycatch numbers for the wreck fisheries in Table 4 are overestimated by an unknown amount for both methods employed.

Another potential source of bias common to both methods stems from the extremely uneven distribution of bycaught porpoises on trips in the plaice fishery. As mentioned earlier, 17 of the 21 animals recorded in this fishery were taken by the same vessel on two trips in January-February 1998, although sampling has covered 24 trips in 8 years. Two types of nets (called 'snehvidegarn' and 'bastardgarn') are used in this fishery, and all porpoise bycatches have been taken in the 'bastardgarn'. The relative occurrence of the two net types in the fleet is, however, not known, so it is not possible to extrapolate observed bycatch rates to the 'bastardgarn' effort only. For this reason, the total porpoise bycatch in the plaice fishery is over-estimated by an unknown amount. We recommend that an effort is made to establish the relative occurrence of these two net types in the fishery, so that extrapolations can be carried out on a basis that better reflects how the fishery is performed.

A negative bias common to both methods results from missed bycatches or drop-outs, i.e. entangled animals which are shaken out or spontaneously fall out of the nets before being observed and recorded. Such drop-outs will most often go undetected, unless the observer is watching the nets as they are hauled out of the water. However, the observers in the sampling programme from which these data originate, are usually busy working up samples of the catch and have little time to watch the nets being hauled. In some bycatch observer programmes, where an effort has been made to scan for drop-outs, the proportion of porpoises found floating at the surface out of the total number recorded is relatively high (e.g. Bravington and Bisack, 1996; Tregenza *et al.*, 1997). How large a fraction of the entanglements go undetected probably depends on a number of factors such as observer routines, the strength of the netting, type of net hauler employed, sea state during hauling, soak time as well as others, which means that drop-out rates cannot be used from one fishery on another fishery. Thus we recommend that an effort is made to establish drop-out rates for the relevant Danish fisheries.

A specific assumption for the ‘landings’ method is constant target species CPUEs during the whole period for which porpoise bycatch is estimated. Fig. 3 shows that this assumption is clearly violated, which was the main reason for developing the ‘effort’ method of extrapolation.

The ‘effort’ method also assumes constant target species CPUEs, but only during the years covered by the observer programme. However, CPUE may vary considerably even within a relatively short period, as shown in Fig. 3 and confirmed for cod by the GLM results. For the cod fisheries, the change in CPUE during the observer period 1993-2000 has varied between 661 and 1,119kg per day with the highest values in the middle of the period. Sampling activity has not been equally distributed between years, but was highest in 1993 and 1997, which represent a low and a high CPUE year. Therefore the effect of a varying CPUE is somewhat levelled out. The assumption of constant CPUE has, however, been violated and the bycatch estimates might be biased. As an alternative, the observer time series could have been divided into more, but shorter series. However, the gain of using a long observer time series, in contrast to two shorter series, is reduced sampling variance. This is illustrated by the plaice fishery, where sampling has been modest throughout the whole sampling period 1994-2001, but as CPUE seems relatively stable in this fishery (Fig. 3) even a sparse sampling programme can give rise to an estimate of total bycatch with a low sampling variance.

If the observed bycatch rate were measured as bycatch per days at sea, it would be possible to extrapolate directly to the effort for the whole fleet without the assumption of constant CPUE during the observer programme. There are, however, a number of problems associated with that approach. For cost/benefit and safety reasons, larger vessels are chosen for the observer programme and the observed bycatch rate would have to be adjusted with an unknown vessel-size factor if effort was recorded as days at sea. Moreover, on some of the observer trips, a mix of fishing gears have been used (with and without pingers) of which only data from nets without pingers were used. So the observed days at sea must be adjusted for the period using pingers and divided by various types of gear when different species were targeted on the trip. The major problem is, however, the way the effort measure, ‘days at sea’, is defined. ‘Days at sea’ is a derived value from logbook and sales slip data and includes the number of days between the date of the first fishing operation (actually the date for the first catch record in the logbook) and the date when the fish is landed (date on the sales slip), plus one. For fisheries where the nets are set on one trip and hauled during the next trip (e.g. in the turbot and plaice fisheries) the ‘days at sea’ is misleading for the total effort. To avoid problems caused by the ‘days at sea’ definition, it is assumed that the logbooks and sales slips have been completed in the same way throughout the period, such that the catch statistics can be used to calculate fleet CPUE. We believe this is a reasonable assumption, but have no way of verifying whether it is actually true.

To use ‘days at sea’ as an unbiased estimator of effort for extrapolating the observed bycatch rates to fleet level, the net length used per day is assumed to be constant throughout the period of extrapolation. It could be expected that fishermen seeing a decrease in CPUE would try to compensate for this by increasing the number of nets used per day. However, this seems not to be the case, judging from information from the observed part of the fleet and from the correlation between the ICES SSB index and the CPUE for the Danish fleet. The reason is that most of the time at sea is used for steaming between the fishing positions (e.g. shipwrecks) and to set, haul and clean the nets. Less time is actually used for handling the landings, especially for the larger and ‘rare’ fish like cod, hake and turbot. Therefore a decrease in landings per net does not necessarily free much time that could be used to deploy more nets on a trip. Consequently, we feel that this assumption is justified.

The ICES assessment of the North Sea cod has been criticised, but it is nevertheless comforting that there is a high correlation between the ICES SSB index and the CPUE for the Danish fleet (Fig. 4). Assuming that a higher stock size gives a higher CPUE, the correlation indicates that the ‘days at sea’ effort used throughout the period is fairly unbiased. It also indicates that the fishermen are not increasing the number of nets operated as a reaction to decreasing CPUE, since this would have resulted in relatively stable fleet landings per day, more or less independent of the stock size of cod.

There has been a downward trend in trip duration from approximately six days in 1987 to three days in 2001 in the cod fishery. Taking the definition of ‘days at sea’ into account, this trend might have biased the estimate of total bycatch, as a relatively larger part of the trip duration will be used on steaming from the fishing ground to the harbour on shorter trips, assuming that the same fishing grounds are chosen. The real CPUE has therefore been underestimated in the most recent years, with the shorter trips leading to an over-estimated bycatch using the ‘effort’ method.

The two methods extrapolate observed bycatch to total fleet level for a rather long period and the time series of estimated bycatch should be treated with caution for years without an observer programme. Both methods give a similar average bycatch, but the ‘effort’ method is less optimistic about the reduction in bycatch in the most recent years. We believe that the ‘effort’ method is a more appropriate way to extrapolate the observer data to the whole fleet, but raise caution about continuing the extrapolation without obtaining new data from the relevant fisheries. The severe reductions in cod quotas since 1998 have changed the fishing practices of the Danish gillnet fleet in the North Sea such that the data analysed here may no longer represent the current situation in the fisheries with respect to bycatch of harbour porpoises.

ACKNOWLEDGEMENTS

We are grateful to Dr Anna Rindorf, Dr Steen Munch Petersen and two anonymous reviewers for helpful comments to this manuscript. The work was partly funded by the EC through contract numbers FAIR-CT05-0523 and DG-XIV 97/0006.

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Humpback whale (*Megaptera novaeangliae*) occurrence in the Mediterranean Sea

A. FRANTZIS[*], O. NIKOLAOU[*], J.-M. BOMPAR[+] AND A. CAMMEDDA[#]

Contact e-mail: afrantzis@otenet.gr

ABSTRACT

Humpback whales were considered extremely rare in the Mediterranean Sea until recently. Only two confirmed records were known from a period of more than 100 years and both were from the western basin. However, nine new observations spread across both Mediterranean basins have been recorded since 1990. This increase in numbers and range during a relatively short period of time seems to be a new, growing trend, suggesting that the occurrence of humpback whales in the Mediterranean Sea is no longer accidental, but occasional. It coincides with the recovery of some stocks of the expanding North Atlantic population after their depletion during a long period of whaling. The true reason behind increased humpback whale entries in the Mediterranean Sea and their exact origin cannot be known until new occurrences are properly photo-identified and sampled genetically.

KEYWORDS: HUMPBACK WHALE; MEDITERRANEAN; EUROPE; DISTRIBUTION; RANGE

INTRODUCTION

The humpback whale (*Megaptera novaeangliae*) has always been considered an extremely rare species in the Mediterranean Sea (Aguilar, 1989; Notarbartolo di Sciara and Demma, 1997). Aguilar (1989) concluded that only two documented and confirmed records existed for the Mediterranean Sea until 1989: a historical 'catch' near Toulon (southeastern France) in 1885 and a sighting reported in 1986. Since 1990, the number of humpback whale observations in the Mediterranean Sea has increased and the range of sighting locations has expanded to cover both basins of the Mediterranean Sea. This paper reports six recent unpublished records (five sightings and one stranding) including the first occurrence of humpback whales in the eastern Mediterranean basin, and discusses the possible reasons of increase in this species' occurrence in the Mediterranean Sea.

METHODS

In order to gather data on humpback whale occurrences in the Mediterranean Sea, a search of unpublished opportunistic sightings or strandings was made by contacting a large number of Mediterranean cetologists working in the field or involved in national stranding networks. Photographic documents were collected in the same way and analysis was performed in order to determine whether any of the humpback whales observed and photographed since 1998 were resighted in the observations made in the following years. Details of the trailing edges of the flukes, the right and left dorsal fin area, and the left pectoral fin pigmentation were used for comparisons with the available material from each sighting, since the ventral part of the flukes had been photographed in only one case. In that particular case, photo-identification comparisons were made with 5,341 and 2,998 individual whales from the North Atlantic Humpback Whale Catalogue (NAHWC), and the Years of the North Atlantic Humpback (Yonah) Catalogue, respectively.

RESULTS

From 1990 to 2004, six sightings and three entanglements of humpback whales have been recorded in the Mediterranean Sea (Table 1; Fig. 1). On 6 March 1990, a single humpback whale entered the small and shallow Bay of Aiguablava, Catalonia, Spain. It seemed to be a large individual and was recorded a few hundred meters from the shore (Aguilar, pers. comm.). On 2 October 1992, a young humpback whale was found dead, entangled in fishing nets in the Gulf of Gabès, Tunisia (Chakroun, 1994). One more humpback whale was found entangled in fishing nets in the following year, on 21 May 1993 off Cavalaire, France. This was a young female that measured 7m and weighed 2,600kg. Later in August of the same year, two humpback whales of similar size were filmed off Toulon (Sears pers. comm.). On 24 January 1998 a single humpback whale was observed inside the shallow Gulf of Oristano (west Sardinia, Italy). The whale was followed, while travelling at a distance of about 30-400m from the coast over depths of 5-22m. Its total length was estimated to be 7-8m.

An exceptional sighting of a single humpback whale was reported on 17 April 2001 in the Bay of Tolo, which is encompassed by the Argolikos Gulf, Myrtoon Sea, Greece. This was the first time that a humpback whale has been recorded in the eastern Mediterranean basin. The whale was sighted almost daily on a regular basis until 15 May 2001. Throughout its stay the whale remained 0.5-5km offshore, over depths that ranged from 30-140m. Feeding behaviour was observed repeatedly. One more humpback whale was sighted in Greece on 19 July 2002 in the strait between Lefkada and the Meganisi Islands, Ionian Sea. It was observed at 200 to less than 50m from the coast, over depths of less than 50m. No feeding behaviour was observed and underwater photos showed that the whale was emaciated. No resightings were reported from the area during the next few days. About two weeks later, on 4 August 2002, a humpback whale was sighted in the west Adriatic Sea, 3.5km off Senigallia, Italy (Affronte *et al.*, 2003). That whale was also

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[*] *Pelagos Cetacean Research Institute, Terpsichoris 21, 16671 Vouliagmeni, Greece.*
[+] *Groupe d'Étude des Cétacés en Mediterranée, 13 bld du Redon, Chloris, bat A, F-13009 Marseille, France.*
[#] *International Marine Centre, Località Sa Mardini 09072, Torregrande (Oristano) and Dipartimento di Geoingegneria e Tecnologie Ambientali, Facoltà di ingegneria, Piazza d'armi, Cagliari, Italy.*

Table 1

Documented records of humpback whales in the Mediterranean Sea. F: Female; M: Male.

Date	Location	No of animals	Sex	Size	Type of record	Source of information
Nov. 1885	Toulon, SE France	1	?	6.8m	Catch in nets (found dead?)	Aguilar (1989)
14 Mar. 1986	Off Majorca, Catalonia, Spain	2	F? + calf?	?	Sighting	Aguilar (1989)
Mar. 1990	Bay of Aiguablava, Catalonia, Spain	1	?	Adult?	Sighting	Aguilar, pers. comm.
2 Oct. 1992	Gulf of Gabès, Tunisia	1	?	8m	Catch in nets (found dead)	Chakroun (1994)
21 May 1993	Off Cavalaire, SE France	1	F	7m	Catch in nets (found alive)	Bompar (2000)
Aug. 1993	Off Toulon, SE France	2	?	?	Sighting	Sears, pers. comm.
24 Jan. 1998	Gulf of Oristano, W Sardinia, Italy	1	?	7-8m	Sighting	This paper
17 Apr. 2001	Bay of Tolo, Myrtoon Sea, Greece	1	?	8-11m	Sighting	This paper
19 Jul. 2002	SW Lekfada Island, Ionian Sea, Greece	1	?	?	Sighting	This paper
4 Aug. 2002	Off Senigallia, Adriatic Sea, Italy	1	?	~9m	Sighting*	Affronte *et al.* (2003)
5 Apr. 2003**	North of Tartous, Syria	1	M	7.85m	Floating dead	Saad (2004)
17 Feb. 2004	SE Corfu Island, Ionian Sea, Greece	1	F	7.2m	Catch in nets (found dead)	This paper

*The sighting of 4 August 2002 could be a resighting of the whale observed on 19 July 2002 (see text for details). **This record became known at page proof stage and is not considered elsewhere in the paper. Saad, A. 2004. First record of a humpback whale stranding on the coast of Syria (Eastern Mediterranean). FINS Newsletter of ACCOBAMS 1(1):10. [Available from: *www.accobams.org*].

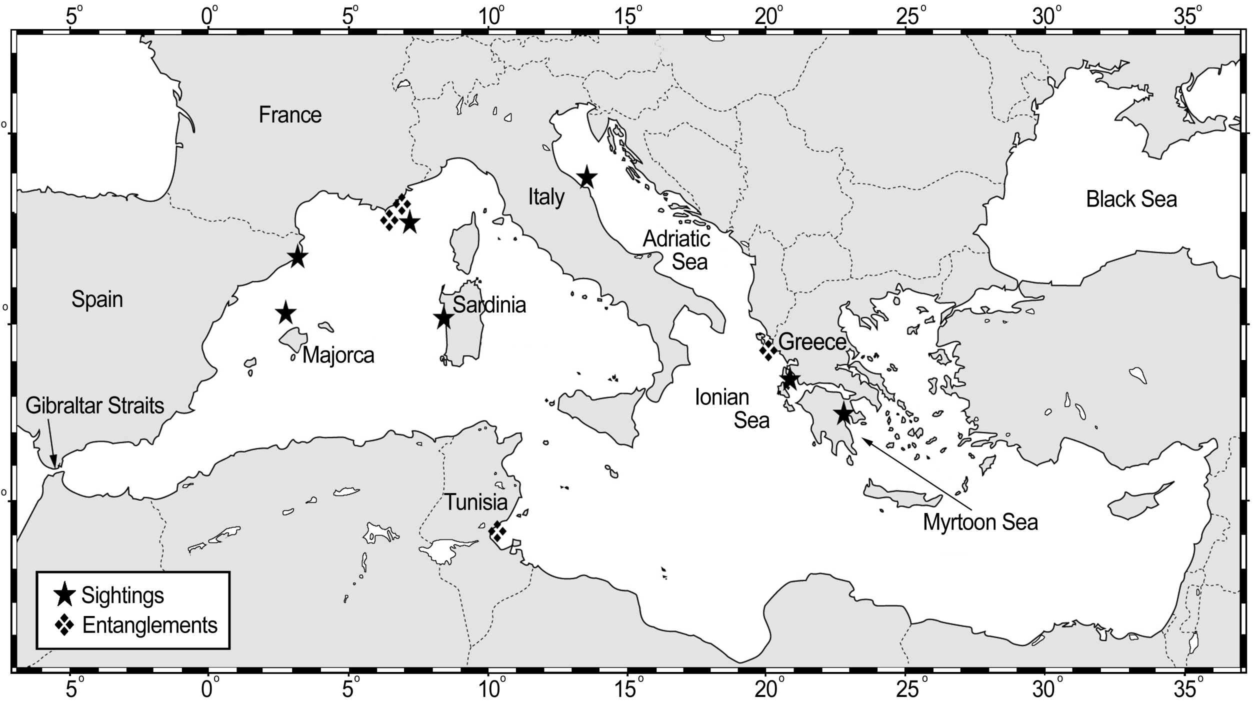

Fig. 1. Location of humpback whales recorded in the Mediterranean Sea.

emaciated to the point that its survival was doubtful. Its size was estimated to be about 9m. Finally, the carcass of a young female humpback whale was reported floating on 17 February 2004 in the Bay of Lefkimmi (SE Corfu Island, Ionian Sea), and eventually washed ashore two days later. The total length of this whale was 7.2m. Nets (probably pelagic driftnets) around its mouth and a rope that was found attached to its tailstock indicated entanglement and interaction with fisheries. Barnacles on its genital area suggested that this whale was slowly moving for many days before its death (Clapham, pers. comm.). Skin samples were collected for future DNA analysis and comparison.

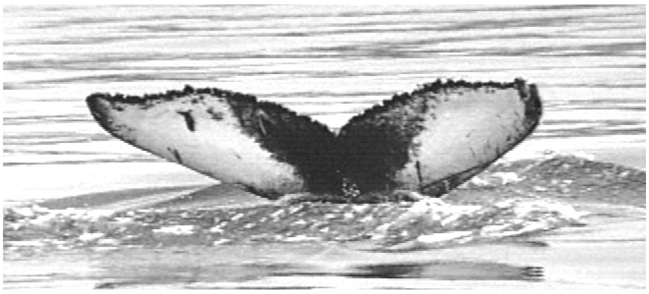

Fig. 2. Pigmentation pattern and trailing edge of the flukes of the humpback whale recorded in the Bay of Tolo (Argolikos Gulf, Myrtoon Sea, Greece) in 2000. This whale has been catalogued as a new entry (#4923) in the North Atlantic Humpback Whale Catalogue (NAHWC).

The whale observed in the Bay of Tolo is the only one for which good photos of the ventral part of the flukes are available (Fig. 2). No matches were found with any humpback whale photo-identified in the North Atlantic, therefore the whale was added to the NAHWC catalogue as new entry #4923 (Seton, pers. comm.). These comparisons included the photo-identified whales (up to 2002) in the most neighbouring population unit to the Mediterranean Sea, which winters around the Cape Verde Islands (Rice, 1998; Hazevoet and Wenzel, 2000). Attempts to match the most recent Mediterranean records (1998, 2001, 2002 and 2004) showed that all sightings made in different years concerned different individuals. Due to the poor quality of the available photographs no definitive conclusions could be drawn regarding the comparison between the two whales observed in 2002. Nevertheless, taking into consideration the very similar black pigment patterns on the right pectoral fin, some similar scarring and pigmentation patterns on the right dorsal and head areas, the emaciated state of both whales, and the time and geographical distance between the two sightings, it seems likely that the whale observed in the east Ionian Sea on 19 July 2002 was the individual observed 16 days later in the north-west Adriatic Sea. The shorter trajectory (which is almost a straight line) linking these two sighting points is about 510 nautical miles and corresponds to an average speed of 1.33 knots for a whale travelling continuously with a north-west heading. This is an underestimate of the real speed, which seems reasonable for a whale exploring new areas or looking for its way back to the Atlantic. Migrating humpback whales swim at average speeds of 2.2 to 8.2 knots (Clapham and Mead, 1999).

DISCUSSION

Considering that only two records were known until 1989 for a period of more than a century, it was correct to believe that the humpback was an extremely rare species in the Mediterranean Sea (Aguilar, 1989). The data in this paper show an average of roughly one record every one and half years since 1990. These numbers clearly suggest that the occurrence of humpback whales in the Mediterranean Sea is no longer accidental, but occasional. It is comparable to that of the four other cetacean species (minke whale, *Balaenoptera acutorostrata*, killer whale, *Orcinus orca*, false killer whale, *Pseudorca crassidens* and rough-toothed dolphin *Steno bredanensis*) that are traditionally considered occasional visitors to the Mediterranean Sea (Notarbartolo di Sciara and Demma, 1997; Bompar, 2000; Notarbartolo di Sciara, 2002). There are relatively few records (from less than 10 to about 20) for all the above-mentioned species, which apparently have no resident population in the Mediterranean Sea (Notarbartolo di Sciara and Demma, 1997; Notarbartolo di Sciara, 2002). It is reasonable to assume that the latter is also the case for Mediterranean humpback whales, which should be still regarded as wanderers of the North Atlantic population (Aguilar, 1989). Various studies indicate a recovery of some stocks of this population after their severe depletion by commercial whaling (Clapham and Mead, 1999; Waring *et al.*, 1999; IWC, 2002; 2003). The historical presence of humpback whales off the European Atlantic coasts is well documented, although few records are available from the 20th century (see Aguilar, 1989), and the actual status and structure of the eastern North Atlantic stocks are unclear (Clapham and Mead, 1999). Animals from Iceland and Norway visit the West Indies for breeding (Stevick *et al.*, 1999; IWC, 2002), but genetic results have shown that some animals (at least from Iceland) do not breed in the West Indies (Valsecchi *et al.*, 1997). Some whales use the waters around the Cape Verde Islands as breeding and calving habitats, and a recent match provided direct evidence of a link with the feeding grounds off Iceland during the boreal summer (Hazevoet and Wenzel, 2000; Jann *et al.*, 2003). The available data and the lack of any photo-identification matching prevent any links being made between the humpback whales observed in the Mediterranean and any particular Atlantic stock. Surprisingly, the 11 Mediterranean records are spread quite homogeneously throughout the year in all seasons (Fig. 3), therefore no link with particular migration movements towards feeding or breeding grounds can be established. Most of them concern young animals (Table 1) that may have entered the Mediterranean Sea either after an early separation from their mother, or during their first independent migration cycle (Clapham and Mead, 1999).

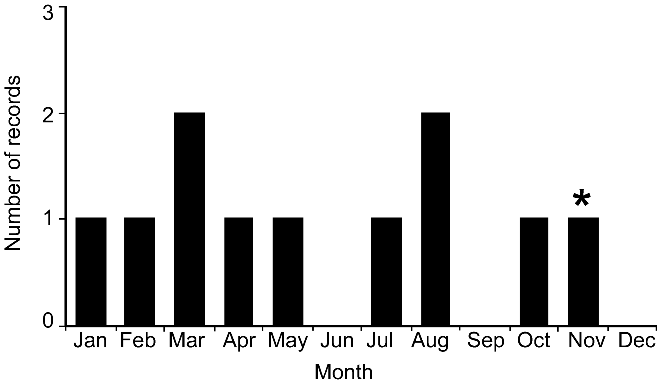

Fig. 3. Distribution of humpback whales recorded in the Mediterranean Sea by month. The asterisk indicates the historical record in 1885.

The sudden increase of observations since 1990 can be attributed to: (1) a real increase in humpback whale entries in the Mediterranean Sea; (2) an increased ‘effort’ in their recording; and (3) both (1) and (2) occurring simultaneously. There is no doubt that during the past decade, the number of scientists and laypersons who study, observe and record cetaceans has increased significantly in the region, as well as the circulation of information among them. Therefore, the rate of humpback whale entries in the Mediterranean might have always been the same, with their occurrence passing unnoticed previously. However, the humpback whale is an unmistakable species and the majority of Mediterranean records occurred very close to the coasts. Consequently, similarly conspicuous and exceptional events would have been known in the past, since the zoologists and naturalists of the 19th and early 20th centuries (like Gervais, Risso, Companyo and others) were very keen on reporting whale sightings or strandings in the northwestern Mediterranean Sea (Bompar, 2000). They had very motivated or even paid personnel *in situ*, because of their high interest in enriching museum collections. Despite the many records from other occasional Mediterranean species (Bompar, 2000), there is only one humpback whale record before 1986 (Aguilar, 1989). Similarly, the increased effort over the last few years did not result in an equivalent increase of the other occasional whale species.

The explanation for humpback whale entries in the Mediterranean can only be the subject of speculation. Aguilar (1989) proposed that, driven by the pursuit of food or by specific water temperature gradients, some individuals might follow the migration pattern of Atlantic fin whales (*Balaenoptera physalus*) towards the Ligurian Sea (northwestern Mediterranean). New data show that seasonal fin whale migration through the Gibraltar Straits is unlikely to occur (Notarbartolo di Sciara *et al.*, 2003). Nevertheless, temporal or permanent changes in the oceanographic conditions in the Gibraltar Strait and the entire Mediterranean Sea combined with the merely explorative nature of humpback whales could be the cause of the recent entries. Drastic changes in the physical characteristics of the Mediterranean waters may result from relatively small evolutionary changes in heat and freshwater budgets across the sea surface (Béthoux and Gentili, 1999). Increase in water temperature and slight climate changes that are already known to occur (Béthoux *et al.*, 1998; Béthoux and Gentili, 1999) could progressively or even suddenly remove oceanographic barriers that prevented humpback whales from entering the Mediterranean Sea in the past. Another plausible explanation for recent humpback whale occurrence could be a spillover from an expanding North Atlantic population. Increased numbers of births may have resulted in an increase in the attempts of young individuals to explore new grounds. The distribution of humpback whale populations seems to be very dynamic, and there are reasons to believe that humpback whales have only 'recently' colonised or shifted to two of their largest actual breeding grounds in the world, in Hawaii (Herman, 1979) and the northern West Indies (Mattila *et al.*, 1994; Reeves *et al.*, 2001). Although the possibility of a new colonisation of the Mediterranean Sea by humpback whales cannot be rejected, there are not enough data to support such a scenario. However, if this were true, the apparently high number of entanglements in fishing nets might be a serious impediment for the colonisation.

Thousands of kilometres away from their usual feeding and breeding grounds, humpback whales have reached the western and eastern Mediterranean repeatedly during the last years. This seems to be a new, growing trend, rather than an old phenomenon that passed unnoticed during the previous decades. It is not known if these whales are able to find their way back to the Atlantic Ocean, or if they stay in the Mediterranean Sea and die sooner or later. It is important to increase the scientific effort, so that new humpback whale sightings are properly identified photographically and genetically. Only then will comparisons with the Atlantic stocks provide answers to the questions that arise each time a humpback whale is observed in the Mediterranean Sea.

ACKNOWLEDGEMENTS

We acknowledge a grant to Pelagos Institute from the Vodafone Group Foundation and Vodafone-Panafon Hellenic Telecommunications Company S.A. for the preparation of this work and the financial support of fieldwork from Oracle Corporation and Oracle Hellas through CAFAmerica grant. We wish to thank A. Aguilar, R. Sears and P. Clapham for their personal communications; R. Seton, Project Manager of NAHWC, for her enormous matching work on the whale from Tolo Bay; B. Jann and N. Øien for having looked for the same whale in their fluke catalogues; J. Robbins for her comments regarding the Greek records; the biology student Ch. Kassara, the reporter Z. Mousios, G. Michaelidis and Ch. Chalikias from Lefkas Diving Centre, the director of Greenpeace Greece N. Charalambides and the photographer K. Koulouris for sending information and photos of the whales in Lefkada Island, in the Bay of Tolo and in the Bay of Lefkimmi; the Mayor of Lefkimmi, A. Gardikiotis, for supporting sampling activities in the field; P. Stevick and P. Domenici for useful discussions and comments, and B. Piergallini from the International Marine Centre (Oristano) for useful advice on the Sardinian field operations. Finally, we are particularly grateful to P. Clapham and R. Reeves for their help and comments on an earlier draft of this manuscript and to two anonymous reviewers who helped improve the submitted manuscript with their comments.

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J. CETACEAN RES. MANAGE. 6(1):29–32, 2004

A note on genetic isolation of Mediterranean sperm whales (*Physeter macrocephalus*) suggested by mitochondrial DNA

Violaine Drouot[*#++], Martine Bérubé[+], Alexandre Gannier[*#], John C. Goold[++], Robert J. Reid[**]
and Per J. Palsbøll[+]

Contact e-mail: violainedr@hotmail.com

ABSTRACT

Thirteen sperm whales were sampled, using sloughed skin, in the Mediterranean Sea during six distinct encounters. Individuals were discriminated using the results of molecular sexing, mitochondrial control region sequencing and microsatellite genotyping (3 loci). Samples from 57 specimens were available from sperm whale strandings on northern European coasts. The first ~200bp of the mitochondrial DNA (mtDNA) control region of each sample were sequenced and three different haplotypes were identified. The frequency of each haplotype was significantly different between the Mediterranean Sea and the eastern North Atlantic, suggesting that sperm whales in the two areas comprise different maternal entities.

KEYWORDS: SPERM WHALE; GENETICS; EUROPE; MEDITERRANEAN SEA; NORTH ATLANTIC; STOCK IDENTITY

INTRODUCTION

The sperm whale (*Physeter macrocephalus*) has a cosmopolitan distribution. Genetic analyses to date have detected comparatively low levels of variation in mitochondrial (mt) DNA on a global scale. While the levels of variation at nuclear loci are similar to those reported in other large whales, the degree of divergence within ocean basins among putative sperm whale populations is low in both genomes. For example, genetic inter-oceanic differentiation was detected in mtDNA between North Atlantic and Pacific sperm whales by Lyrholm and Gyllensten (1998) but no genetic heterogeneity was detected within North Pacific sampling areas. In contrast, Richard *et al.* (1996) detected significant levels of genetic heterogeneity among sperm whale pods (mature females accompanied by immature male and female individuals) at the Galapagos Islands, possibly due to matrilineal pod structure of sperm whales at low latitudes.

Sperm whales are the second most common large whales observed in the Mediterranean Sea after fin whales (*Balaenoptera physalus*). A central question to the management and conservation of the species in the Mediterranean Sea, where abundance and movements through the Strait of Gibraltar are poorly known, is whether sperm whales in the Mediterranean are isolated from the eastern North Atlantic populations. This study represents a first attempt to test the hypothesis of a homogeneous distribution of genetic variation among sperm whales in the Mediterranean and eastern North Atlantic. To test for any deviation from the null-hypothesis, nucleotide sequences from the first part of the maternally inherited mtDNA control region were collected and analysed.

MATERIAL AND METHODS

Sample collection

Skin samples ($n = 36$) were collected in the Mediterranean Sea during 1998, 1999 and 2001 summer surveys. All samples were collected as sloughed skin from free-ranging sperm whales observed in six distinct groups encountered in four different areas of the Mediterranean Sea: the Tyrrhenian Sea; the Ionian Sea; the North western Basin; and the Balearic Sea (Table 1 and Fig. 1). Group composition was extrapolated from the estimated size of the animals (Rice, 1989). All sightings were assumed to be of different groups unless at least one individual of the group was re-sighted (based on photo-identification).

Samples from the eastern North Atlantic (Table 2) were available from animals stranded in Scotland ($n = 26$), Ireland ($n = 4$), Belgium ($n = 5$) (Holsbeek *et al.*, 1999; Joiris *et al.*, 1991), Netherlands ($n = 3$) (Holsbeek *et al.*, 1999), Norway ($n = 2$) and Denmark ($n = 17$) (Kinze *et al.*, 1998). As was the case in the Mediterranean Sea, samples from the North Atlantic were from a wide-ranging area (Fig. 1), thereby ensuring that sampling was not biased to only a single group of sperm whales in either area (see Richard *et al.*, 1996). The Atlantic samples were mostly from male animals, some of them mass stranded (Table 2). After collection, samples were preserved in a saturated sodium chloride solution with 20% dimethyl sulfoxide.

Laboratory analysis

Two different methods of DNA extraction were employed: a standard phenol/chloroform extraction protocol (Sambrook and Russell, 2001) and an extraction kit (DNeasy Tissue Kit™, Qiagen Inc.). The first ~200bp of the 5’ end of the mtDNA control region was amplified using a forward primer (MT4F) designed by Arnason *et al.* (1993) and Bp16071R (5’-CCTCAGTTATGTTATGATCATGGGC-3’). This approach was necessitated by the degraded nature of DNA extracted from the sloughed skin samples. The initial symmetric PCR amplifications were carried out in a total volume of 20 μL consisting of: 0.2 μM of each dNTP; 67mM Tris-Cl (pH 8.8); 2mM $MgCl_2$; 17mM NH_3SO_4; 10mM β-mercaptoethanol; 0.1μM of each primer; 0.4 units of *Taq* DNA polymerase. Negative and positive controls were included to detect possible contamination as well as loading

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[*] *Centre de Recherche sur les Cétacés, Villa Aurelia, Ch. Moyennes Breguières, BP715, 06600 Antibes Cedex, France.*

[+] *Conservation Genetic Laboratory, Ecosystem Science Division – ESPM, University of California at Berkeley, 151 Hilgard Hall, #3110, Berkeley CA 94720-3110, USA.*

[#] *Centre de Recherche sur les Cétacés, Marineland, 306 Avenue Mozart, 06600 Antibes, France.*

[++] *Institute of Environmental Science, University of Wales Bangor, Wales, LL57 2UW, UK.*

[**] *SAC Veterinary Services, Drummondhill, Stratherrick Road, Inverness, IV2 4JZ, UK.*

Table 1

Details of the sperm whale tissue (sloughed skin) samples obtained from the Mediterranean Sea (Group composition: Ad=Adult; Juv=juveniles; N-b=new-born; N_{SS}=Number of sloughed skin samples collected; N_I=Number of individuals genetically identified) and results of molecular sexing (F=female; M=male; U=undetermined) and allele size (bp) for the 3 microsatellite loci analysed (*where electrophoresis was performed but no sufficient result was obtained).

Gp	Date of encounter	Location	Gp size	Group composition	N_{SS}	N_I	Sex	EV001	GATA 053	GT011
A	21/06/98	Tyrrhenian Sea	5	3 Ad,	4	3	F	121/128	200/216	*
				1 Juv,			F	120/124	200/216	*
				1 N-b			U	121/134	202/223	*
B	14/07/98	Ionian Sea	7	4 Ad,	3	1	F	121/139	*	*
				2 Juv,						
				1 N-b						
C	05/08/99	Northwestern basin	1	1 Ad	1	1	M	128/141	207/223	*
D	29/06/00	Ionian Sea	5	3 Ad,	5	4	F	128/130	208/223	108/114
				1 Juv,			F	120/130	199/212	101/114
				1 N-b			F	120/124	212/224	*
							M	124	212/223	100/115
E	08/07/01	Baleares	6	2 Ad,	22	3	F	124	203/211	116/120
	12/07/01			2 Juv,			F	124	203/211	114/118
	13/07/01			1 N-b			M	124	203/207	119/121
F	19/08/01	Northwestern basin	1	1 Ad	1	1	1 M	124	*	116/120
Total			**24**		**36**	**13**				

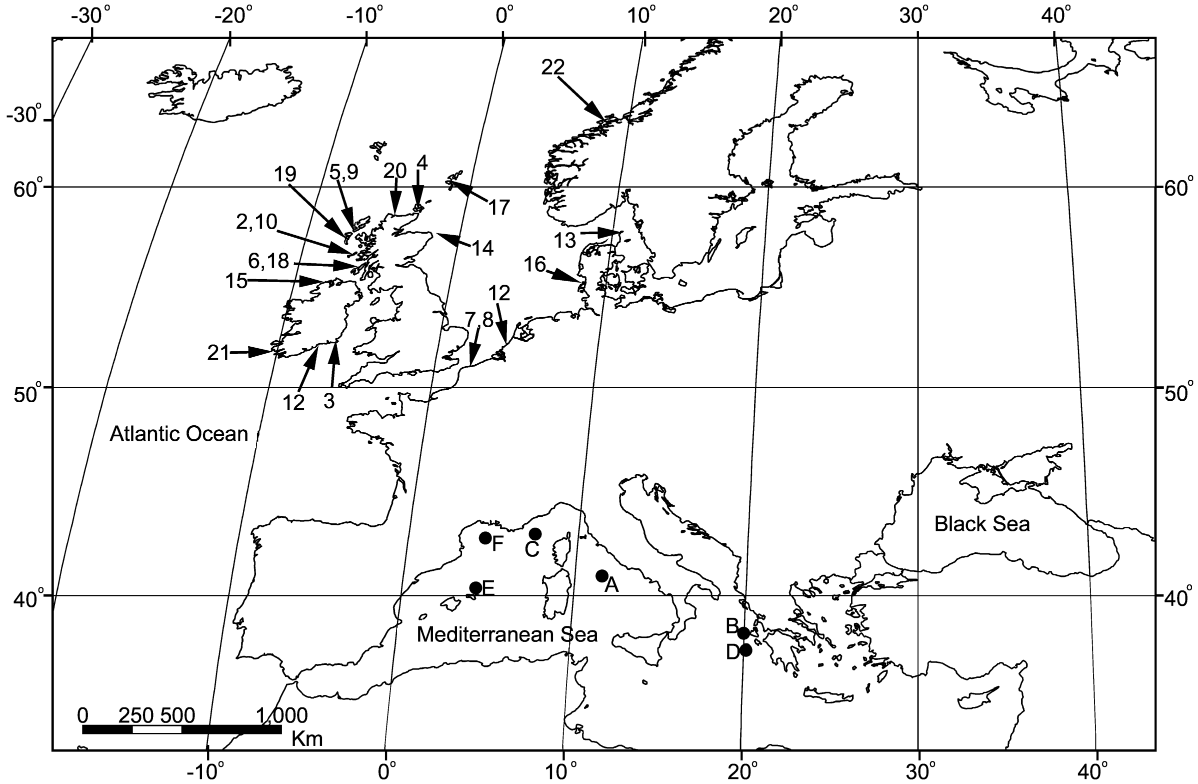

Fig. 1. Map showing the sampling sites in the Mediterranean Sea (letters refer to sightings described in Table 1) and along the North eastern Atlantic coast (numbers refer to stranding described in Table 2).

errors. Reactions were performed on thermal cyclers (MJ Research Inc.) and consisted of 2 minutes of denaturing at 94° Celsius, followed by 28-33 cycles of denaturing at 94° Celsius, for 1 minute; annealing at 54° Celsius for 1 minute and extension at 72° Celsius for 4 minutes. Cycling sequencing was conducted according to the manufacturer's instructions (Dye Terminator Cycle Sequencing Ready Reaction Kit™, Applied Biosystems, Inc.). The order of sequencing products was resolved using an Applied Biosystems ABI Prism™ 377 automated sequencer. Sex was determined using the multiplex approached presented by Bérubé and Palsbøll (1996). To ensure only sloughed skin samples collected from different individuals were included in the test we determined the genotype at three microsatellite loci; EV001 (Valsecchi and Amos, 1996), GATA 053 (Palsboll *et al.*, 1997) and GT011 (Bérubé *et al.*, 1998). Amplifications were conducted as described in the original primer notes with fluorescent end labelling. The

Table 2

Details of tissue samples obtained from sperm whales stranded on the Northeastern Atlantic coast. N=Number of animals sampled (MS=Mass stranding). Type of tissue used (necropsy): S=Skin; M=Muscles; Sex (from field observation) M=male; F=Female; H=haplotype (as defined in Table 3), *n*=number of samples, empty cells correspond to failed DNA extraction or sequencing.

ID No.	Date of stranding	Country	Location	N	Tissue	Sex	Size range (m)	H (*n*)
1	12/02/89	Belgium	Unknown	1	M	M	Adult	2 (1)
2	21/11/93	Scotland	Lochailort, Highland	1	S	M	15	2 (1)
3	26/09/93	Ireland	Kilmore Quay, Wexford	1	S	F	11.07	-
4	07/12/94	Scotland	Backaskail Bay, Sanday Orkney	11 (MS)	S	All M	12-13.4	1(8), 2(3)
5	13/03/94	Scotland	Hougharry, Western Isles	1	M	M	13	2 (1)
6	27/03/94	Scotland	Campa Islay, Argyll	1	M	M	16	1 (1)
7	18/11/94	Belgium	Koksijde	3	M	M	14.4-15.4	1(2), 2(2)
8	18/11/94	Belgium	Nieuwpoort	1	M	M	18.2	-
9	19/01/95	Scotland	Benbecula Western Isles	1	M	M	14	-
10	23/03/95	Scotland	Carse of Ardersier, Highland	1	S	M	13.7	1 (1)
11	15/06/95	Ireland	Youghal, Co. Donegal	1	S	F	7.22	1 (1)
12	12/01/95	Netherlands	Scheveningen	3	M	M	15.2-15.4	1 (3)
13	25/01/96	Denmark	Hulsig	1		M	13.1	2 (1)
14	28/01/96	Scotland	Cruden Bay, Grampian	6 (MS)	S	All M	12.1-12.8	1(1), 2(5)
15	20/03/96	Ireland	Tory Island, Co. Donegal	1	S	M	14.8	-
16	27/03/96	Denmark	Rømø	16 (MS)	S	All M	11.9-13.2	1(9), 2(7)
17	08/09/96	Scotland	Mousa, Shetland	1	M		'Large'	3 (1)
18	27/12/96	Scotland	Traigh Angus, Islay Strathclyde	1	S	M	14.4	2 (1)
19	20/03/98	Scotland	West Gerinish, Western Isles	1	S	M	~10	-
20	06/08/98	Scotland	Bettyhill, Highland	1	S	M	12.2	1 (1)
21	20/05/99	Ireland	Brandon, Co. Kerry	1	S	M	~15	-
22	28/04/99	Norway	Storjford	1	S	M	15.4	1 (1)
23	2001	Norway	Unknown	1	S			2 (1)

amplification products were separated and sized using an Applied Biosystems ABI Prism™ 377 automated sequencer.

Data analysis

The assessment of the degree of genetic differentiation between Mediterranean and Atlantic samples was based on the comparison between the observed and expected mtDNA haplotype frequencies and was tested using a G-test (or likelihood ratio test) for goodness of fit (Sokal and Rohlf, 1995).

RESULTS

DNA was extracted successfully from all sloughed skin samples from the Mediterranean Sea and for 52 of the necropsy samples from the eastern North Atlantic sperm whales. Approximately 25mg of tissue was sufficient to extract DNA from necropsy samples, however 40mg was needed from sloughed skin samples. Whale sex was successfully identified in 31 sloughed skin samples. Samples for which microsatellite loci analyses failed or where the genotype at a locus was ambiguous were discarded from the statistical analysis. From the combined results of the sexing, mtDNA control region sequences and microsatellite loci genotypes 13 samples were included from the Mediterranean Sea (Table 1), all of which differed at a minimum of one of the three loci.

Approximately 200 nucleotides of the 5’ end of the mtDNA control region were successfully sequenced. Only two polymorphic sites were identified, defining three distinct different haplotypes (Table 3). Haplotype 1 was most frequent and observed in 64% of all samples, followed by haplotype 2, which was observed in 35% of all samples. Haplotype 3 was rare and observed only in a single individual (Table 3). The polymorphic nucleotide position defining the last haplotype has not previously been reported. The comparison between the Atlantic and Mediterranean populations was based on the proportion of each different haplotype among the samples collected in each area. All individuals sampled in the Mediterranean shared the same haplotype: haplotype 1 (Table 3). In contrast haplotype 1 was observed in 54% of the Atlantic individuals and haplotype 2 in 44%. Thus, while no nucleotide diversity was observed in the Mediterranean samples (one unique lineage), three haplotypes were observed among the eastern North Atlantic samples (nucleotide diversity of 1.5). The frequencies of haplotypes were significantly different between the Mediterranean Sea and eastern North Atlantic ($G_{[2df]} = 14.0$, $p < 0.01$).

Although females and immature whales in other areas have been seen to form long-term stable groups within which there is substantial genetic similarity (Dillon, 1996; Richard *et al.*, 1996; Dufault *et al.*, 1999), it is unlikely that the results have been affected by intra-group homogeneity in mtDNA control region as the Mediterranean sequences appeared to be monomorphic (a single haplotype). However, the difference in the haplotype frequency was also tested by including only one sample from each group. Although this reduced by half the sample size from the Mediterranean Sea, the difference was still significant ($G_{[2df]} = 12.3$, $p < 0.01$).

DISCUSSION

Although this analysis found a low level of intra-specific variation in the mtDNA control region as reported by Lyrholm *et al.* (1996), the spatial distribution of this variation was not homogenous. The significant level of divergence between the Mediterranean Sea and the eastern North Atlantic is consistent with the notion of restricted movement of groups between the two areas and suggests a resident sperm whale population in the Mediterranean Sea. A similar discreteness in the distribution of variation at the mtDNA control region has previously been observed between the Mediterranean and the eastern North Atlantic in striped dolphin, *Stenella coeruleoalba*, (Archer, 1996) and fin whale (Bérubé *et al.*, 1998).

Table 3

The mitochondrial control region haplotypes detected in the study and the respective frequencies in the two sampling areas.

Nucleotide position*	9	58	62	105	121	184	Sampling area
Published**	C	T/C	C/T	C/T	C/T	T/C	
Haplotype 1	C	T	T	C	C	T	Mediterranean Sea (n=13) Eastern North Atlantic (n=28)
Haplotype 2	C	T	C	C	C	T	Eastern North Atlantic (n=23)
Haplotype 3	A	T	T	C	C	T	Eastern North Atlantic (n=1)

*Position with reference to the sequences reported by Lyrholm and Gyllensten (1998); **Lyrholm and Gyllensten (1998).

The exclusive maternal inheritance of the mitochondrial genome means that the results reflect different maternal structures between the two areas, but nothing with respect to inter-breeding between the two areas. Differentiation in mtDNA haplotypes is consistent with the behaviour of female sperm whales, which have been observed to show fidelity to areas. The observation of newborn calves in different areas of the Mediterranean basin also suggests that females remain in the Mediterranean Sea to breed. However, these data cannot answer the question of whether Atlantic male sperm whales enter the Mediterranean to breed with females, or whether 'resident' males and females co-exist in the Mediterranean without interbreeding with Atlantic animals. Visual surveys in the area of the Strait of Gibraltar, the only possible passage between the two areas, suggested that sperm whales are present in the area for foraging rather than migratory purposes and do not support the hypothesis of a consistent migration pattern through the Strait (De Stephanis *et al.*, pers. comm.). Widespread inter-breeding with Atlantic males seems unlikely but additional analyses of nuclear mendelian inherited loci are needed to establish conclusively whether Mediterranean Sea sperm whales form a distinct population from the eastern North Atlantic or two different maternal entities as concluded from this preliminary study.

ACKNOWLEDGMENTS

We are particularly grateful to Drs Carl Kinze (Zoological Museum, University of Copenhagen, Denmark), Emer Rogan (University College of Cork, Ireland), Junita Karlsen and Fernando Ugarte (University of Tromsø, Norway), Claude Joiris, Ludo Holsbeek, Sophie Laran (University VLB of Brussels, Belgium) and Thierry Jauniaux (University of Liège, Belgium) for providing eastern North Atlantic samples. Thanks are also due to the members and volunteers of Groupe de Recherche sur les Cétacés (GREC) for their help with collecting the samples in the Mediterranean Sea, and two anonymous reviewers.

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Abundance of Atlantic white-sided dolphin (*Lagenorhynchus acutus*) during summer off northwest Scotland

K. Macleod

Sea Mammal Research Unit, Gatty Marine Laboratory, St Andrews, Scotland, Fife, KY16 8LB

Contact e-mail: km53@st-andrews.ac.uk

ABSTRACT

A shipboard cetacean survey was conducted in July/August 1998 within an area to the west of Scotland, UK, commonly known as the Atlantic Frontier. The aim of the survey was to document the distribution and abundance of cetaceans to provide baseline population data for an area that is being increasingly explored and developed by oil companies. A double platform 'independent observer' (IO) method was used to estimate the abundance of the Atlantic white-sided dolphin (*Lagenorhynchus acutus*) using standard line-transect and distance sampling methodology. Previously, uncorrected Atlantic white-sided dolphin abundance was estimated as 27,194 (CV = 0.29) from this survey. This paper presents abundance estimates corrected for $g(0) < 1$ using a direct duplicate method. The value of $g(0)$ was estimated to be 0.61 (CV = 0.09). The abundance in two strata was estimated as 21,371 (CV = 0.54) to the west of the Outer Hebrides and 74,626 (CV = 0.72) in the Faroe Shetland Channel. The high CVs are the result of small sample sizes, particularly of the duplicate data set. However, the abundance estimates represent the first for this species to the northwest of Scotland and adds to existing baseline abundance estimates for small cetaceans in UK waters. The results could be useful for planning future surveys that aim to calculate more precise abundance estimates. These results, together with opportunistic sightings data collected during other surveys, suggest that the waters to the west of Scotland are an important habitat for the Atlantic white-sided dolphin. Presently, threats to this species in the area are relatively unknown but a baseline population estimate will be an integral part of any management regime should there become a need in future.

KEYWORDS: ATLANTIC OCEAN; ABUNDANCE ESTIMATE; G(0); SURVEY-VESSEL; WHITE-SIDED DOLPHIN

INTRODUCTION

The Atlantic white-sided dolphin (*Lagenorhynchus acutus*) is widely distributed in temperate and sub-polar waters of the North Atlantic. Its range extends from eastern Labrador (Rice, 1998), Greenland (Kapel, 1975) and Spitzbergen, *ca* 77°N (Øien, 1996) in the north, to North Carolina, 35°N (Palka *et al.*, 1997), the Azores (McBrearty *et al.*, 1986) and the Straits of Gibraltar (Hashmi and Adloff, 1995) in the south. The abundance of the Atlantic white-sided dolphin throughout its range is presently unknown. The estimates available within parts of its range are almost exclusively from the western North Atlantic (Table 1). Only some of these estimates take account of school size bias (Baylock *et al.*, 1995) and the probability of detecting a group on the trackline (Baylock *et al.*, 1995; Waring *et al.*, 1998) whilst estimates in the Gulf of St Lawrence (Kingsley and Reeves, 1998) were uncorrected for either. In the northern North Atlantic, an approximate abundance of 37,622 Atlantic white-sided dolphins (no variance calculated) was estimated from the North Atlantic Sightings Surveys (NASS) in Icelandic and adjacent waters in 1987 (Sigurjónsson and Víkingsson, 1997). A preliminary abundance of 20,444 (95% CI = 12,714-32,874) *Lagenorhynchus* sp. was estimated from aerial surveys conducted in Icelandic waters in 2001 (NAMMCO, 2002).

There is very little information about the abundance of the Atlantic white-sided dolphin in the northeast Atlantic, with only two unpublished estimates (Hughes *et al.*, 1998; O'Cadhla *et al.*, 2001). Around Britain, its distribution is centred in offshore waters beyond the continental shelf-edge off the western coast, although it does occur in the Celtic Sea and central North Sea in small numbers (Hammond *et al.*, 2002). It is probably one of the most abundant offshore odontocetes off northwest Scotland (Macleod *et al.*, 2003) but without baseline abundance estimates the status of this species cannot be assessed. Fisheries are a known source of mortality of Atlantic white-sided dolphins although direct and indirect takes are largely unquantified. Bycatch has been recorded in pelagic and mid-water trawls to the south and west of Ireland (Couperus, 1998; Morizur *et al.*, 1999) and is likely to occur in other areas of this species' range. Atlantic white-sided dolphins have been observed feeding around trawls during towing − a behaviour that may increase their susceptibility to being caught (Morizur *et al.*, 1999). There are occasional direct takes in drive fisheries on the Faroe Islands (e.g. Bloch and Hoydal, 1990; Gallien *et al.*, 2001). Additionally, waters to the west of Scotland and Ireland, commonly referred to as the Atlantic Frontier, are the frequent focus of oil and gas development and seismic surveys during the exploration phase. There is growing evidence that many marine mammals respond to acoustic

Table 1

Estimates of abundance of Atlantic white-sided dolphins in the western North Atlantic.

Region	Relevant period	Estimate (CV)	Reference
Cape Hatteras to Nova Scotia: shelf and shelf-edge	1978-1982	28,600 (0.21)	CeTAP (1982)
Northern Gulf of Maine and lower Bay of Fundy	July-Sept 1991 and 1992	20,400 (0.63)	Baylock *et al.* (1995)
Southern Georges Bank to southeast Scotian Shelf	June-July 1993	730 (0.47)	Baylock *et al.* (1995)
Virginia to the Gulf of St Lawrence	July-Sept 1995	27,200 (0.43)	Waring *et al.* (1998)
Gulf of St Lawrence	August-Sept 1995	11,740 (0.47)	Kingsley and Reeves (1998)
Georges Bank to the Gulf of St Lawrence	July-August 1999	51,640 (0.38)	D. Palka, pers. comm. in Waring *et al.* (1998)

and physical disturbance associated with industrial development (Harwood and Wilson, 2001). Sighting rates of Atlantic white-sided dolphins were significantly reduced when airguns were firing compared to when they were not during seismic surveys in UK waters between 1998 and 2000 (Stone, 2003).

This paper presents an abundance estimate for Atlantic white-sided dolphins in waters off northwest Scotland. Data were collected during an Independent Observer (IO) line-transect survey, conducted in an area of the Atlantic Frontier to the west of the Outer Hebrides and in the Faroe-Shetland Channel during July-August 1998. The aim was to record the distribution of cetaceans in the region (Macleod *et al.*, 2003) and to estimate the abundance of cetacean species where possible. The precision of the abundance estimate is low and the problems with it are discussed. However, the estimate is considered an important baseline since it represents the first published for this species in these waters and the wider northeast Atlantic.

METHODS

Data collection

The survey was conducted from 13 July −14 August 1998 on board MV *Neptun* searching at an average speed of 10 knots. The area surveyed included the outer continental shelf to offshore waters, extending from the Outer Hebrides in the south to the Shetland and Faroe Islands in the north (Fig. 1). Indices of abundance estimated from the Sea Birds at Sea Team off the west of Scotland (Weir *et al.*, 2001) suggested that the relative density of cetaceans throughout the region differed and so two strata were defined (Fig. 1), one to the west of the Outer Hebrides (A) and one in the Faroe-Shetland Channel (B). The survey design was based on predetermined saw-tooth tracklines with a random start point.

The ship provided two separate observation platforms, for which mean eye height above sea level was measured as 7.3m (lower team) and 9.6m (upper team). An IO method, involving two independent teams of observers, was used to record distance sampling data. The two teams of four observers remained the same throughout the survey and were isolated from one another, both visually and acoustically, during surveying periods. The teams were chosen so that the experience of the observers on each was comparable. The lower team of observers communicated between themselves using two-way radios since the bridge house separated the port and starboard bridge wings. On both platforms, the observers were rotated through the observation positions and a ‘rest position’ on their platform every 30 minutes. Searching was primarily carried out using the naked eye but binoculars were used by both platforms intermittently to scan greater distances and aid species identification and group size estimation. The survey was carried out in ‘passing mode’ in which the vessel did not approach sighted cetaceans. A sighting was defined as a single individual or a group of individuals, which appeared in close proximity spatially and were engaged in the same general behaviour. Sub-groups were defined as sightings using this definition in the event of encounters with large aggregations of animals. Radial distances and sighting angles were measured using *Fujinon* 7 × 50 reticle binoculars and angleboards, respectively, mounted on the railings of each observation platform. Visual estimates of distance were only recorded when the sighting was close to the ship, rendering both the horizon and animals outside the field-of-view of the binoculars. A two-day training period for observers took place before the survey to practise angle and distance estimation using the equipment and by eye, each taken in turn to check estimates to surrounding vessels or headlands against the radar.

Survey effort continued throughout daylight hours (generally 06:00-21:00) but was suspended when sighting conditions were unsuitable (Beaufort sea state >4 or poor visibility). All sightings, effort and environmental data were recorded onto data sheets. Sightings data included time of initial cue, position (latitude and longitude), species identification and certainty, group size (min, max and best) and other associated notes. Effort and environmental data, including sea state and swell height, were logged at the beginning, end and at 30-minute intervals or whenever conditions changed throughout the day. The data collected by the lower team were also entered directly into a laptop running the LOGGER (IFAW, 1992-1994) program and linked via an NMEA interface to a Garmin II Plus Global Positioning System (GPS).

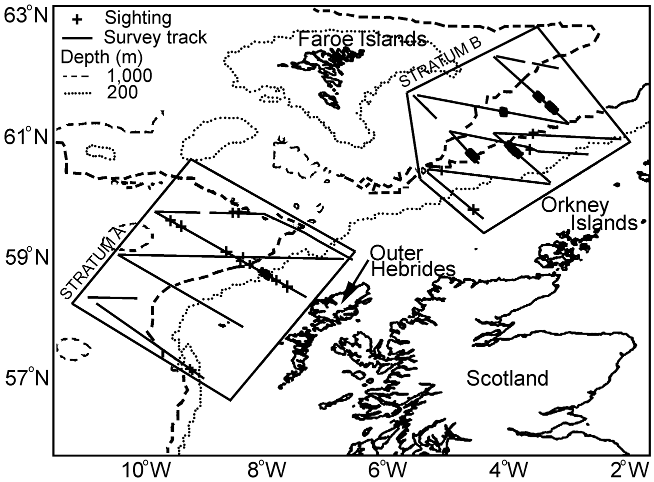

Fig. 1. Survey area showing transects surveyed on effort and distribution of sightings of the Atlantic white-sided dolphin.

Data analysis

The abundance estimate was corrected for sightings missed on the trackline to compensate for violation of one of the fundamental assumptions of distance sampling theory −that all objects are detected on the trackline with certainty (Buckland *et al.*, 2001). The value of the detection function at zero distance, $g(0)$, and absolute abundance were calculated using the direct duplicate method (Palka, 1995).

Duplicate identification

Duplicate sightings were identified from the Atlantic white-sided dolphin sightings recorded by the upper and lower platforms. They were identified by comparing the times of sightings, estimates of distance and sightings angle, best estimates of group size and group headings. A duplicate required there to be an exact match in time, or within a minute, of sighting times of the initial cue; sighting angles to be within 5° of each other; and sighting distances to be within one reticle eye division. Best estimates of group size could vary but the range had to be within two animals. This is particularly important when there were a large number of groups aggregated in a relatively small area and short space of time. The species identification had to correspond for both teams of observers. Definite duplicates were determined after the survey was completed.

Abundance estimation

Applying the direct duplicate method, corrected animal abundance from a stratified survey is estimated by:

$$N = \sum_{i=1}^{2} D_i \cdot A_i = \sum_{i=1}^{2} \frac{D_{iup} \cdot D_{ilo}}{D_{idup}} \cdot A_i \tag{1}$$

where

N = estimated abundance of animals, corrected for $g(0)$, across strata;
D_i = estimated density of animals, corrected for $g(0)$, within stratum i (n/km^2);
A_i = area of stratum i (km^2);
D_{iup} = density of animals as seen by the upper team, not corrected for $g(0)$, within stratum i;
D_{ilo} = density of animals as seen by the lower team, not corrected for $g(0)$, within stratum i;
D_{idup} = density of animals as seen by both teams (duplicates), not corrected for $g(0)$, within stratum i.

where

$$D_{iup} = \frac{n_{iup} \cdot f_{iup}(0) \cdot E(s_{iup})}{2L_{iup}} \tag{2}$$

and

n_{iup} = number of sightings detected by the upper team, within stratum i;
$f_{iup}(0)$ = estimated probability density function of perpendicular distances for the upper team, within stratum i, evaluated at zero ($f(0) = 1/\mu$ where μ is the effective strip half-width);
$E(s_{iup})$ = estimated group size, within stratum i; for the upper team; and
L_{iup} = total survey effort for the upper team, within stratum i.

Similarly, equation 2 was used to estimate D_{ilo} and D_{idup}.

Only sightings and effort collected in Beaufort sea state 2 or below were used in the analysis. For each dataset, $f(y)$ was modelled from perpendicular distance data pooled over strata because of the limitations imposed by small sample sizes in stratum A and of the duplicate dataset. The reticle distances and angles were converted to radial distances (km) using the equation of Lerczak and Hobbs (1998). The radial distances were converted to perpendicular distances ($d = r.\sin\theta$); histograms of their distributions for both platforms and the duplicate data are shown in Fig. 2. Each dataset was modelled in Distance 3.5 (Thomas *et al.*, 1998) using the combinations of key functions and series expansions identified as model robust (Buckland *et al.*, 2001). The data were grouped into distance intervals for analysis (Buckland *et al.*, 2001). The need for, and number of, adjustment terms in the series expansion were assessed by the Likelihood Ratio test (Buckland, 1987; Buckland *et al.*, 2001) and Akaike's Information Criteria (AIC). The best model was judged from visual inspection of the model fit and on the basis of the lowest value of AIC. Expected group size was estimated using size-bias regression of the log of cluster size against $g(y)$ for data pooled across strata. Encounter rates and density were estimated for each stratum and observation platform. Additionally, estimates of density were combined over strata, weighted by stratum areas.

Variance of $f(y)$, $E(s)$ and uncorrected density for each platform was estimated empirically in Distance 3.5. Confidence intervals (CI) and coefficients of variation (CV) of the corrected abundance estimates were calculated using a nonparametric bootstrap. Within each bootstrap replicate, transects were resampled with replacement independently within each stratum, and estimates of D_{iup}, D_{ilo} and D_{idup} and therefore D were calculated. 999 replicates were performed and confidence limits were calculated as the 2.5 and 97.5th percentile of the bootstrap distribution. The bootstrap procedure was complicated using stratum A transects by the small sample sizes. When there were no duplicate sightings on a transect, D_{Adup} was zero and, following from equation 1, D and consequently N were infinity. These resamples were excluded for the calculation of the CV and CI for density in stratum A. The delta method (Buckland *et al.*, 2001) was also used to estimate the CV of density estimates as a comparison to the bootstrap estimates, particularly given the problems associated with the small sample sizes for stratum A.

Estimating g(0)

The value of $g(0)$ can be estimated by:

$$g_i(0) = g_{iup}(0) + g_{ilo}(0) - \left(g_{iup}(0) \cdot g_{ilo}(0)\right) \tag{3}$$

where

$$g_{iup}(0) = \frac{n_{idup} \cdot f_{idup}(0)}{n_{ilo} \cdot f_{ilo}(0)} \tag{4}$$

and $g_{ilo}(0)$ can be calculated similarly. Given that $f(0) = 1/\int_{y=0}^{w} g(y)dy$:

$$g_{iup}(0) = \frac{n_{idup}}{n_{ilo}} \cdot \frac{\int_{y=0}^{w} g_{ilo}(y)dy}{\int_{y=0}^{w} g_{idup}(y)dy} \tag{5}$$

and

$$g_{ilo}(0) = \frac{n_{idup}}{n_{iup}} \cdot \frac{\int_{y=0}^{w} g_{iup}(y)dy}{\int_{y=0}^{w} g_{idup}(y)dy} \tag{6}$$

The shape of the detection curve $\int_{y=0}^{w} g(y)dy$ was modelled by estimating the probability density function, $f(y)$, using perpendicular distances y in Distance 3.5. The CV of each $g(0)$ estimate was obtained from 100 bootstrap resamples of

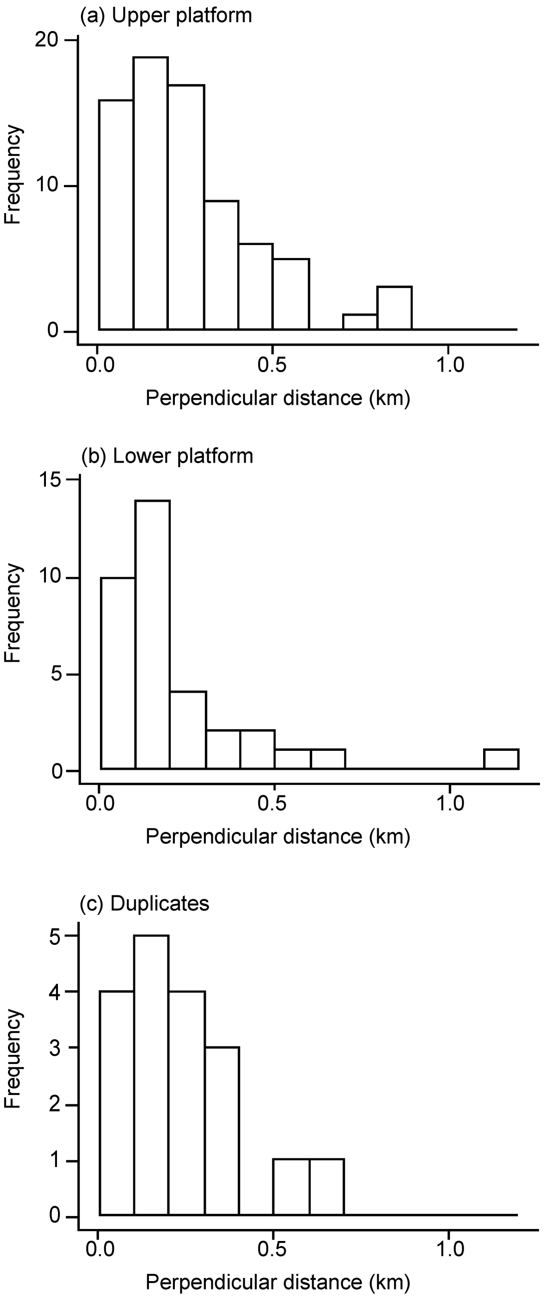

Fig. 2. Frequency histograms of the perpendicular distances (km) for the (a) upper, (b) lower and (c) duplicate sightings.

the lower, upper and duplicate datasets. A sample within each stratum was defined as all effort completed on a single day (one transect), so as to increase independence between samples. Resampling was conducted across strata.

RESULTS

Survey effort in strata A and B amounted to 1,082km and 1,074.6km, respectively. The total survey effort in Beaufort sea state 2 or less was 627.5km in stratum A and 800km in stratum B. The upper team recorded more sightings than the lower platform and the sample sizes for each were $n_{lo} = 35$ and $n_{up} = 79$. From these datasets, only 18 definite duplicates were identified. Examination of the histograms of sighting perpendicular distances (Fig. 2) led to the right truncation of both the upper and lower teams data at 0.8km and 0.64km, respectively, to remove outliers. The resulting sample sizes were $n_{up} = 73$ and $n_{lo} = 34$. The histogram of duplicate data showed no obvious outliers and was not truncated ($n_{dup} = 18$).

The data were analysed as grouped. Histograms of perpendicular distances to sightings (Fig. 2) suggested some responsive movement of white-sided dolphins to the approach of the research vessel, with fewer detections in the first 100m than the second. Consequently, when modelling $f(y)$, the first interval was chosen to be wide enough that animals moving away from the trackline would still be in the first interval. The half-normal key function, without a series expansion, was the best model of $f(y)$ for each of the three datasets. The point estimates of $f(0)$ for the upper, lower and duplicate data were 2.77 (CV = 0.09), 3.16 (CV = 0.14) and 2.95 (CV = 0.20), respectively (Table 2). The plots of the probability density function for the lower, upper and duplicate data are shown in Fig. 3. The effective strip half-width (μ) for the upper and lower platforms was 360m (SE = 30) and 320m (SE = 40), respectively.

Encounter rates were higher in stratum B compared to stratum A and on the upper platform compared to the lower platform (Table 2). Cluster size bias was significant for the upper and lower platforms, which meant that larger schools were more likely to be detected at distance than smaller schools. The regression estimates of group size of Atlantic white-sided dolphins for the upper platform was 5.26 (SE = 0.53) and 6.78 (SE = 0.68) for the lower platform (Table 2). The mean group sizes, pooled over strata, for the lower and upper platforms were 6.06 (SE = 0.57) and 4.97 (SE = 0.49), respectively. Observed group sizes estimated by the upper platform ranged from 1-22 individuals compared with 2-16 individuals for the lower platform.

Estimates of density and abundance, not corrected for $g(0)$, are shown for each stratum and observation platform in Table 3. The uncorrected abundance estimate for the whole survey area using the upper platform data was 32,947 (CV = 0.30). Uncorrected abundance estimates using the lower and duplicate data were 22,213 (CV = 0.35) and 7,609 (CV = 0.37), respectively. The values of $g(0)$ were estimated from the pooled data across strata, and were $g_{up}(0) = 0.48$ (CV = 0.10), $g_{lo}(0) = 0.26$ (CV = 0.17) and $g(0) = 0.61$ (CV = 0.09). During the bootstrap resampling of stratum A

Table 2

Summary of parameter estimates for the upper, lower and duplicate data. The encounter rate (n/L) was estimated by stratum; remaining parameters estimated from data pooled over strata.

Stratum	Platform	n	L (km)	No. transects	A (km^2)	n/L (SE)	f(0) (SE)	E(s) (SE)	μ (SE)
A	Upper	13	627.5	7	54,720	0.02 (0.01)	2.77 (0.25)	5.26 (0.53)	0.36 (0.03)
	Lower	5				0.008 (0.003)	3.16 (0.44)	6.78 (0.68)	0.32 (0.04)
	Duplicate	3				0.005 (0.003)	1.95 (0.39)	4.65 (0.60)	0.34 (0.07)
B	Upper	60	800.2	9	45,213	0.07 (0.02)	2.77 (0.25)	5.26 (0.53)	0.36 (0.03)
	Lower	29				0.04 (0.01)	3.16 (0.44)	6.78 (0.68)	0.32 (0.04)
	Duplicate	15				0.02 (0.006)	1.95 (0.39)	4.65 (0.60)	0.34 (0.07)

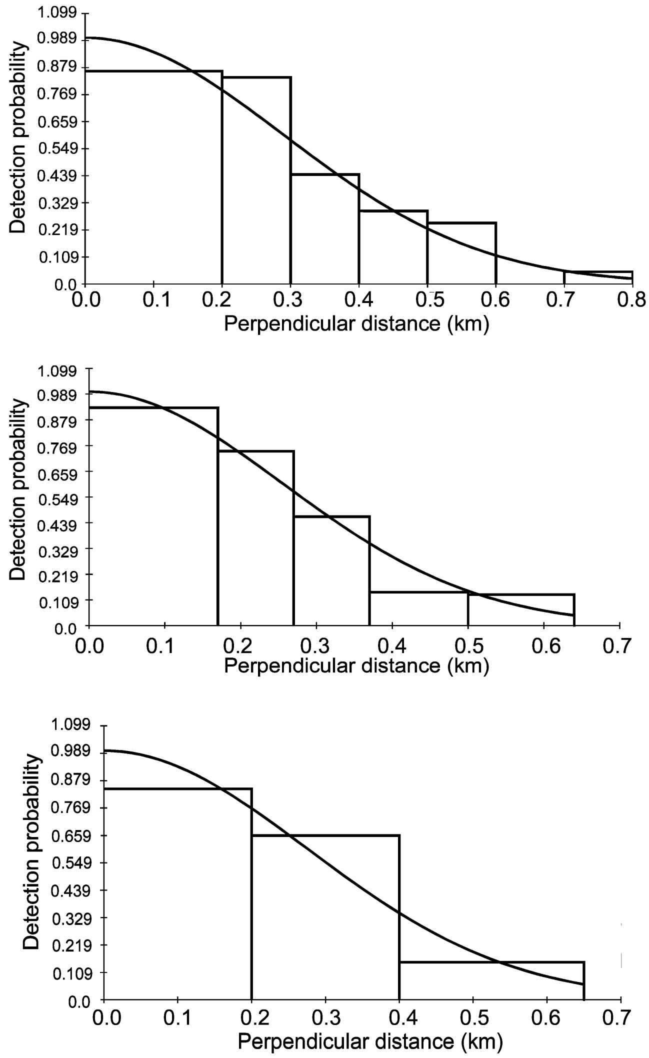

Fig. 3. Sightings distribution of the Atlantic white-sided dolphin against perpendicular distances for the upper (top), lower (middle) and duplicate (lower) data. The data are fitted with a half-normal key function and the fitted curve indicates the estimated probability density function.

transects, 20.3% of the resamples were infinity due to zero duplicate sightings on some transects. These resamples were removed and the calculation of CV and 95% CI of density were based on 796 resamples. The bootstrap estimate of CV will be an underestimate and the delta estimate is considered a more realistic value for stratum A. In stratum A, the corrected animal density and abundance was estimated to be 0.39 (Bootstrap CV = 0.54 and 95% CI = 0-0.74; Delta CV = 0.96) and 21,371 (Bootstrap CV = 0.54, 95% CI = 0-40,659), respectively. In stratum B, density was estimated as 1.65 (CV = 0.72, 95% CI = 0.19-5.42; Delta CV = 0.66) white-sided dolphins and an animal abundance of 74,626 (CV = 0.72) corrected for $g(0) < 1$ (Table 3).

Table 3

Summary of uncorrected and corrected density and abundance estimates. CVs were estimated using bootstrap resampling.

Stratum	Platform	Uncorrected density (CV)	Uncorrected abundance (CV)	Corrected density (CV)	Corrected abundance (CV)
A	Upper	0.15 (0.55)	8,217 (0.55)	0.39 (0.54)	21,371 (0.54)
	Lower	0.08 (0.47)	4,668 (0.47)		
	Duplicate	0.03 (0.64)	1,795 (0.64)		
B	Upper	0.55 (0.34)	24,730 (0.34)	1.65 (0.72)	74,626 (0.72)
	Lower	0.39 (0.40)	17,545 (0.40)		
	Duplicate	0.13 (0.39)	5,814 (0.39)		

DISCUSSION

The corrected abundance estimate presented here represents one of two for this species in the eastern North Atlantic. O'Cadhla *et al.* (2001) estimated the abundance of Atlantic white-sided dolphins to be 5,490 (CV = 0.43) in an area of approximately 100,000km^2 to the west of Ireland and over the Rockall Trough during August 2000. The value of $g(0)$ used to correct this estimate was obtained from a pooled dataset of Atlantic white-sided and common dolphin (*Delphinus delphis*) sightings. The corrected estimates presented here are 21,371 (CV = 0.54) Atlantic white-sided dolphins to the west of the Outer Hebrides and 74,626 (CV = 0.72) in the Faroe-Shetland Channel. A combined abundance of 11,760 (CV = 0.26) Atlantic white-sided and white-beaked dolphins (*L. albirostris*) for the North Sea and adjacent waters was estimated from shipboard double platform surveys in July 1994 (Hammond *et al.*, 2002). This estimate is, however, mainly representative of the North Sea, which is an area of relatively low density of Atlantic white-sided dolphins compared to white-beaked dolphins.

The abundance estimates available suggest that northwest Scotland has the greatest summer abundance of Atlantic white-sided dolphins in surveyed British waters. Estimates of density support this and suggest that, at least during summer, animal density appears to increase towards the northern end of the range of this species. Existing density estimates indicate a gradient from western Ireland of 0.046 animals/km^2 (O'Cadhla *et al.*, 2001), to the Outer Hebrides of 0.39 animals/km^2, peaking in the Faroe-Shetland Channel at 1.65 animals/km^2. The importance of the west coast is further reflected by the difference in encounter rates (density estimates for the North Sea are not available) of Atlantic white-sided dolphins between the North Sea and western Britain. Group encounter rates recorded by the upper team were 0.07/km (SE = 0.01) in the Faroe-Shetland Channel and 0.02/km (SE = 0.02) to the west of the Outer Hebrides. Within the North Sea, an encounter rate of only 0.005/km was estimated for the Atlantic white-sided dolphin from data collected during July 1994 (Borchers *et al.*, 1995). Atlantic white-sided dolphins occur year-round in these waters but spatial trends in densities may vary seasonally. Observations recorded during seabird surveys off northwest Scotland, show a large increase in numbers of white-sided dolphins during August, with twice as many dolphins recorded as in any other month (Weir *et al.*, 2001).

The assumption that all animals are detected on the trackline with certainty is usually violated during cetacean surveys and the value of $g(0)$ has been found to be less than one for a range of species (Hammond *et al.*, 2002). These data were originally analysed assuming that $g(0) = 1$ (Hughes *et al.*, 1998) and by using a single dataset of unique sightings formed by combining both the upper and lower platforms and eliminating duplicates ($n = 102$, and assuming $f(y)$ was the same for both platforms) across all sea states (0-4). The combined estimate of abundance for the Faroe-Shetland Channel and the west of the Outer Hebrides was 27,194 (CV = 0.29), which is considerably lower, but more precise, than the corrected estimates. In addition to missing animals on the trackline, the histograms of perpendicular distances to white-sided dolphin sightings (Fig. 2) suggested some responsive movement of the dolphins away from the research vessel, which further violates the basic distance sampling assumptions. Evasive behaviour by Atlantic white-sided dolphins has been documented in the North Atlantic (Palka and Hammond, 2001) and it leads to underestimation of density. Future surveys for Atlantic white-sided dolphins should use a survey methodology that enables an estimate of *g*(0) robust to responsive movement. Such a method was used during the Small Cetacean Abundance in the North Sea and adjacent waters (SCANS) (Hammond *et al.*, 2002) survey and involved two teams of observers on independent platforms, but one team (the 'tracker') searched farther ahead of the vessel than the other. The assumption was that sightings recorded by the 'tracker' platform had not yet reacted to the approach of the survey vessel (Buckland and Turnock, 1992). However, the method cannot fully account for responsive movement if the assumption routinely fails. Palka and Hammond (2001) have recently developed a different approach to account more fully for responsive movements based on animal orientation data and this approach should be considered for future surveys for Atlantic white-sided dolphins.

Another source of bias in the corrected abundance estimate is the identification of duplicate sightings. Delays in recording the first time of sightings (since this was not automated) and potential changes in the swimming direction of groups between the initial sighting by one team and detection by the other makes identification of definite duplicates difficult. The reticle binocular readings were also more prone to error than the angle board measurements since swell sometimes made it difficult to hold them steady. These problems were kept in mind when trying to identify duplicates. The duplicate dataset was also small, with just 18 sightings used for modelling the detection function. Buckland *et al.* (2001) suggested sample size minima of forty sightings, but the half-normal model of the duplicate data appeared to fit the data well and the data showed no spurious outliers or 'spikes'. However, bias in the estimation of $g_{dup}(y)$ from such a small sample size can arise when using the direct duplicate method (Palka, 1995). A third observer on the IO survey, who could have determined duplicates at sea, as used on SCANS (Hammond *et al.*, 2002) may have been more successful at duplicate identification than post-survey.

The value of *g*(0) varied between the upper (0.48) and lower platforms (0.26). The probability of detection would be expected to be greater for the upper platform because of the advantage which height above sea level confers on the ability of observers to sight cetaceans. Observers on the higher platform can probably see animals further away from the vessel than the lower platform and this is reflected in the slight differences in the estimates of effective strip-half width ($\mu_{up} = 360$m (SE = 30) and $\mu_{lo} = 320$m (SE = 40)). The probability of detection for observers on the higher platform may also be better than the lower platform when sea state increases. Although the observer teams were chosen so that the level of experience on each was similar, there will be individual observer variability that will influence overall team efficiency and the detection function for that team. Similarly, differences in environmental conditions, such as the amount of glare, may differ between platforms thus affecting the detection function. An assumption of using distance sampling data to model the detection function is that detection of an object depends solely on its distance from the trackline. In reality, many variables are likely to affect the detection probability (Marques and Buckland, 2003). The double platform survey method and modified mark-recapture model used for analysis in this study, also assume that the detections by the platforms are independent but this is often not the case (for example, the behaviour of a group of dolphins may increase the probability of detection by one team but also the other). If heterogeneity in the detection probability is not modelled, then abundance estimates will be negatively biased (Borchers *et al.*, 2002). There are two methods which can be used to minimise heterogeneity and these are stratification of the data by covariates and incorporating covariates into *f*(0) estimation (Marques and Buckland, 2003). Stratification of small datasets is not possible. Future surveys should ensure that covariates are recorded so that heterogeneity can be modelled and unbiased abundance estimated.

The combined estimate of *g*(0) for the white-sided dolphin was 0.61 (CV = 0.09) which is similar to the value of *g*(0) generated from SCANS for *Lagenorhynchus* sp. of 0.65 (Hammond *et al.*, 1995). The SCANS estimate of *g*(0) for the white-beaked dolphin (*L. albirostris*) was 0.71 (Hammond *et al.*, 1995). It would not be appropriate to use a combined *Lagenorhynchus* sp. *g*(0) as a general multiplier of abundance estimates for either the white-beaked or Atlantic white-sided dolphin. In the northeast Atlantic, the observed behaviour of these species in the field suggests that their detection probabilities would be different. Both *L. albirostris* and *L. acutus* are active swimmers but, from personal observations, the white-sided dolphin tends to be seen in large aggregations making them easier to detect. On the other hand, *L. albirostris* is slightly larger, has a more prominent dorsal fin, and the ability to detect this species is influenced less by Beaufort sea state (Macleod, 2001). The Atlantic white-sided dolphin appears to exhibit evasive behaviour in the presence of vessels whereas white-beaked dolphins commonly approach vessels (Palka and Hammond, 2001). An estimate of *g*(0) for the Atlantic white-sided dolphin in the western Atlantic was 0.62 and was calculated from shipboard line transect surveys in the Gulf of Maine/lower Bay of Fundy (Palka *et al.*, 1997).

The abundance of the Atlantic white-sided dolphin was highest in the Faroe-Shetland Channel, although a number of biases in the estimates have been highlighted. The corrected estimates also have substantial uncertainty. The non-parametric bootstrap assumes that the sampling units from which resamples are drawn are independent. A sampling unit was defined as the length of transect surveyed in one day so that this assumption was not seriously violated. However, this resulted in a number of transects (Table 2) below the recommended minima of 15-20 (L. Thomas, pers. comm.) for the bootstrap procedure which may give rise to unreliable estimates of variance.

Data from dedicated cetacean surveys in these waters was previously limited to the NASS conducted in the late 1980s and 1990s (Gunnlaugsson and Sigurjónsson, 1990; Buckland *et al.*, 1992). However, the focal species of these surveys were whales and coverage in Scottish waters was extremely low. Therefore, the NASS data were of little value to the planning stages of this survey and estimating required coverage. The abundance estimates presented are the first for this region and can aid the planning of future surveys with aim of estimating more precise abundance (e.g. Hammond and Macleod, 2003). Offshore waters west of the Outer Hebrides and Northern Isles probably have the highest densities of this species in British waters, which suggest that the area provides an important habitat for them. Abundance estimates are important in areas undergoing rapid industrialisation that may have adverse effects on local populations. They are also vital to assessing the sustainability of removals caused by other sources, particularly fisheries bycatch. Baseline abundance estimates should be considered against subsequent estimates as a way of long-term monitoring of cetacean populations and as an integral part of an assessment of the status of a species.

ACKNOWLEDGEMENTS

The survey was funded by the Whale and Dolphin Conservation Society (WDCS) and Greenpeace, UK and made possible by the great efforts of Mark Simmonds (WDCS) and Elizabeth Murray (Greenpeace). Thanks to the observers onboard: Holly Arnold, Marijke de Boer, Rachel Irish, Chris Mackins, Elizabeth Murray, Kirsti Norris, Justine Pert, Mark Simmonds and Martin Stanley; and the captain and crew of MV *Neptun*. Special thanks to Tiago Marques (University of St Andrews) for his programming skills that enabled part of the analysis. Thanks to Phil Hammond (Sea Mammal Research Unit) for reviewing an early draft of the manuscript and to Len Thomas (University of St Andrews) and an anonymous reviewer for their helpful comments. Part of these data were collected using the software LOGGER developed by the International Fund for Animal Welfare (IFAW) to promote benign and non-invasive research.

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Status, ecology and conservation of Irrawaddy dolphins (*Orcaella brevirostris*) in Malampaya Sound, Palawan, Philippines

BRIAN D. SMITH[*], ISABEL BEASLEY[#], MARIEL BUCCAT[+], VICTOR CALDERON[+], RODERICK EVINA[+], JOSEPH LEMMUEL DE VALLE[+], ANGELA CADIGAL[+], EMMALYN TURA[+] AND ZHULJAKIM VISITACION[+]

Contact e-mail: bsmith@wcs.org

ABSTRACT

A geographically isolated population of Irrawaddy dolphins was recently discovered in Malampaya Sound, Palawan, Philippines. Line-transect surveys conducted in April-November 2001 covered 884km of trackline in the entire Sound and resulted in a total population estimate of 77 individuals (CV = 27.4%), confined to the inner portion (133.7km^2). For all Irrawaddy dolphin sightings, where ecological data were collected ($n = 48$), the mean temperature was 30.2°C, depth 6.5m, salinity 28.3ppt and turbidity 2.2NTUs. Significantly higher turbidity, lower salinity and shallower depth were recorded in the inner Sound compared to adjacent waters. Bottlenose dolphins *Tursiops* sp. (probably *truncatus*) were observed in waters just outside of where Irrawaddy dolphins were recorded. During the study, at least two Irrawaddy dolphins were accidentally killed in bottom-set nylon gillnets used to catch crabs, locally called *matang quatro*. Reports from local fishermen also indicated that as many as three additional animals may have been killed in these nets during the same period. These findings strongly suggest that the Irrawaddy dolphin population in Malampaya Sound is in immediate danger of extirpation due to low numbers, limited range and high mortality. This is the only known population of the species in the Philippines and the nearest known other population is in northern Borneo, some 550km to the south. Recommendations for conserving the population include that: (1) socioeconomic alternatives be developed to promote the conservation goal of reducing the incidence of dolphin entanglement in *matang quatro* gillnets; (2) gillnet free zones be established in core areas of dolphin distribution; (3) Irrawaddy dolphins be promoted as a flagship species of environmental health in the Sound; (4) a long-term programme be established to monitor the dolphin population; and (5) additional investigations be conducted to determine if Irrawaddy dolphins occur in other areas of the Philippines.

KEYWORDS: IRRAWADDY DOLPHIN; SURVEY-VESSEL; ABUNDANCE ESTIMATE; INCIDENTAL CATCHES; GILLNETS; ASIA; CONSERVATION; PHOTO-ID; HABITAT

INTRODUCTION

Irrawaddy dolphins (*Orcaella brevirostris*) are among the cetaceans at greatest risk of population extirpation and perhaps species extinction. They are patchily distributed in shallow, nearshore tropical and subtropical marine waters of the Indo-Pacific, from northeastern Australia in the south, north to the Philippines (Dolar *et al.*, 2002) and west to northeastern India (Stacey and Leatherwood, 1997; Stacey and Arnold, 1999). Their marine distribution is concentrated in estuaries and semi-enclosed water bodies (i.e. bays and sounds), generally adjacent to mangrove forests. Freshwater populations occur in three river systems: the Mahakam of Indonesia; the Ayeyarwady (formerly Irrawaddy) of Myanmar (formerly Burma); and the Mekong of Laos, Cambodia and Vietnam. Irrawaddy dolphins also occur in partially isolated brackish or fresh-water bodies, including Chilka Lake in India and Songkhla Lake in Thailand.

Little information is available on the range-wide status of Irrawaddy dolphins. Recent surveys indicate declines in the range and abundance of the Mekong and Mahakam freshwater populations (Smith and Jefferson, 2002). The latter was classified as Critically Endangered in the 2000 IUCN Red List (Hilton-Taylor, 2000) after surveys recorded only a few tens of dolphins, confined to a 152km segment in the middle reaches of the river (Kreb, 2000).

Interest in Irrawaddy dolphins of Malampaya Sound can be attributed to the fact that it is the only known population of the species in the Philippines and because threats from accidental killing in fishing gear, habitat degradation (both in the estuary and surrounding watershed), and possible prey depletion from over-fishing and the destruction of fish spawning grounds are prevalent and expected to increase (Dolar *et al.*, 2002). The species was first documented as occurring in Malampaya Sound during an investigation of dugongs (*Dugong dugon*) in 1986 (Kataoka *et al.*, 1995). The first dedicated cetacean survey of the area, conducted in June-July 1999, recorded 17 sightings during 230 linear km of search effort and calculated a mean encounter rate of 7.4 dolphins/100km (SE = 2.9) and mean group size of 5.3 dolphins (SE = 1.1; Dolar *et al.*, 2002). All sightings were made in shallow waters (76% less than 6m deep) of the inner Sound.

Malampaya Sound was proclaimed a protected area in June 2000 (National Integrated Protected Areas Programme (NIPAP), 2000). The Sound encompasses approximately 230km^2, divided into inner and outer portions by 13 rocky islands (Fig. 1). Maximum depth is about 16m in the inner Sound and 46m in the outer Sound. The surrounding landscape is characterised by high hills, with altitudes ranging from 100-500m, and dominated by the 1,013m tall Mt Capoas on the western side. Steep topography and a highly indented shoreline contour, with many small and large bays, coves and inlets create complex wind patterns that vary greatly according to area, season and time of day (due to convection forces). Seasonal climate is largely determined by the southwest monsoon rains, with the wettest months in July-September. Freshwater inflows come from

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[*] *Wildlife Conservation Society, Asia Freshwater and Coastal Cetacean Program, 27/16 Soi Naya, Moo 1 Muang, Rawai, Phuket 83130, Thailand.*

[#] *School of Tropical Environment Studies and Geography, James Cook University, Townsville, Queensland 4811, Australia.*

[+] *Kabang Kalikasan Ng Pilipinas (World Wildlife Fund – Philippines), 23 Maalindog Street, U.P. Village, Diliman, Quezon City 1101, Philippines.*

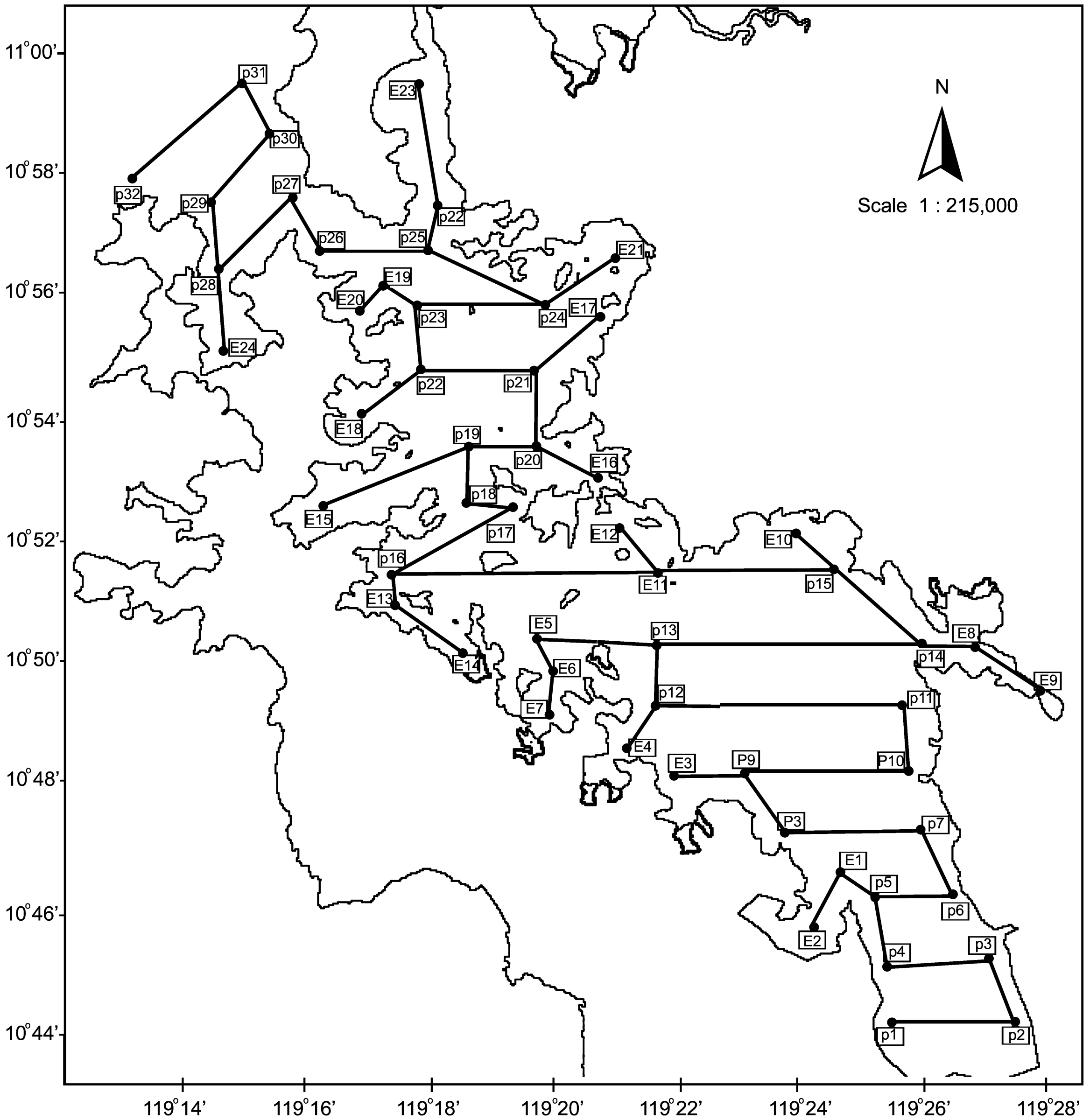

Fig. 1. Map of Malampaya Sound showing tracklines for dolphin surveys conducted during April-November 2001.

numerous rivers draining into the inner Sound and extensive mangroves dominate the shore in this area. Fishing is the principal source of income and employment, with at least 5,000 fishermen dependent on at least 60 commercially valuable species. However, fish production within the Sound is believed to have declined dramatically in recent years (NIPAP, 2000).

MATERIALS AND METHODS

Line-transect surveys

Line-transect surveys were conducted from April to November 2001 to assess the abundance and distribution of Irrawaddy dolphins. During the surveys, three observers stood watch at all times, one stationed on each of the port and starboard sides, searching with handheld binoculars (Fujinon 7×50 with an internal compass) and naked eye from the beam to about 10° past the bow, and one in the centre searching by naked eye, except to focus on visual cues (e.g. splashes), in about a 20° cone in front of the bow. The centre observer also served as the data recorder. Five observers rotated through these positions approximately every 30 minutes or at the end of transect line endpoints, giving each observer about an hour of rest for every 90-minute period spent actively searching for dolphins (i.e. on-effort). The vessel crew and resting observers were instructed to keep dolphin sightings a secret until the on-effort observers saw them, or the entire dolphin group passed well behind the vessel beam. When this happened, the sighting was classified as off-effort and not included in the line-transect analysis. The survey vessel was a 10.3m double outrigger, with a beam of 2.3m on main hull, and equipped with a 40hp four-cylinder diesel motor. The eye-height of the centre observer was about 3.8m above the waterline, while the eye heights of the port and starboard observers were about 3.2m above the waterline. A Global Positioning System (GPS) was used to determine position, speed, course and the distance covered along the trackline. These data, along with Beaufort sea state and the presence or absence of fog and/or rain, were recorded on a standardised effort log. Data entries were made after all observer rotations and any substantial change in vessel course or sighting conditions.

The transect lines were designed to systematically search the entire Sound according to the most unbiased route (i.e. one that avoids following environmental contours; Fig. 1). The transect line was divided into legs, which generally followed a ladder-like grid, with the primary or longer legs normally running east to west across the Sound, ending about 1km from shore, and spaced 2km apart. Connecting doglegs were oriented as close as possible to perpendicular in orientation to the primary legs. Extension legs were added to some endpoints (and occasionally in the middle of longer legs) so that waters within bays and behind islands could also be visually searched. Extension legs were followed to the end while on-effort but line-transect data were not collected during the return trip. The 2km spacing between the primary legs was based on prior experience with the sighting characteristics of Irrawaddy dolphins (see Smith *et al.*, 1997; Smith and Hobbs, 2002), published accounts of their generally cryptic surfacing behaviour (see Mörzer Bruyns, 1966; Stacey and Arnold, 1999) and distance estimates made during practice surveys, which indicated that the dolphins would probably not be observed at a distance greater than 1,000m during Beaufort 3 sea state conditions.

There was some unavoidable compromise in the trackline design since the doglegs did indeed follow shoreline contours, but this potential bias was balanced by the need to search areas hidden within the numerous small bays and inlets that could not be searched from the primary or extension legs. Nevertheless, potential sighting biases along the dogleg lines were evaluated by examining possible clumping in dolphin distribution close to the shoreline and by comparing encounter rates recorded along the dogleg lines versus those along the primary and extension lines.

Vessel speed was maintained at 8-10km h^{-1}. Survey effort was generally stopped when sea state conditions became greater than Beaufort 3, or when rain or fog affected ability to detect dolphin surfacings at a distance of less than 1km. The decision to suspend survey effort when sea state conditions were greater than Beaufort 3 was made after considering the generally cryptic surfacing behaviour of Irrawaddy dolphins, the strong spatial heterogeneity of wind patterns in the Sound and the improbability of making a sufficient number of sightings in Beaufort 4 conditions or greater to allow stratification according to sea state. Simply deleting the survey effort conducted while sighting conditions were poor from the line-transect analysis, as done by Jefferson *et al.* (2002) for estimating the abundance of finless porpoises (a cetacean that exhibits somewhat similar inconspicuous surfacing behaviour), could have resulted in a severely biased abundance estimate. Spatial coverage would have been uneven (especially near Mt Capoas where convection winds often created poor survey conditions), while dolphin distribution within the Sound may have been clumped. When the sea state exceeded Beaufort 3 along one transect line, acceptable conditions were sometimes found along another line located in the lee of the wind. The vessel returned to the earlier line when the wind abated.

The protocol for suspending survey effort when sea state conditions were greater than Beaufort 3 was unavoidably violated due to poor weather during the three August surveys, but a single composite survey was put together *post hoc* from transect lines covered in good conditions. Effort from the first August survey was mostly used, but transect lines surveyed in Beaufort 4 conditions during this survey were substituted with those that were surveyed in Beaufort 3 or less conditions during the second or third surveys, whichever one was completed first.

The survey programme was designed to obtain coverage during the pre-monsoon, monsoon (southwest) and post-monsoon seasons. Considering that previous studies had only observed Irrawaddy dolphins in the inner portion of the Sound (south of the P15-P16 transect line; Fig. 1) and the importance of maximising the number sightings to ensure a reasonably precise abundance estimate, the survey protocol was established to cover the entire Sound (*ca* 154 linear km) during the first survey of each season. Then during subsequent surveys of that season, if no sightings were made north of the P15-P16 transect line, searching would only occur along transect lines of the inner portion and a 4km wide buffer strip extending to the north (*ca* 101 linear km, inclusive of transect lines P16-P17-P18-P19 and E15-P19-P20-E16; Fig. 1).

When a dolphin group was sighted, information was recorded on a standardised sighting form that included entries for geographic position, time of sighting, Beaufort sea state and estimated distance and relative angle from the bow to the dolphin group. Distances were estimated by eye (see below for details on training exercises to reduce distance estimation biases) and relative angles were determined from the difference between the vessel course (as measured by the GPS in magnetic degrees) and the bearing to the dolphin group (as measured by the internal compass in the binoculars). The recorder also occasionally checked the vessel bearing according to the internal compass in handheld binoculars to ensure that there were no major discrepancies between the two readings. Survey effort was then suspended and the vessel turned towards the dolphins to obtain a more accurate estimate of group size and to take photographs of individuals for identification purposes (see below). After finishing these tasks, the vessel returned off-effort (i.e. while not actively searching for new dolphins) to the position where it left the trackline; movements of the sighted dolphin group were tracked during this time to avoid double counting the animals when search effort was resumed.

For oceanic line-transect surveys of cetaceans, sighting distances are generally estimated using the number of binocular reticles from the animal cluster to the horizon (e.g. see Kinzey and Gerrodette, 2001). Because the horizon was not visible in the Sound, this technique could not be used. Following the example of Jefferson and Leatherwood (1997) and Jefferson *et al.* (2002), a laser range finder (*Bushnell Yardage Pro* 1000) was used in training observers to more accurately estimate sighting distances by eye and to investigate potential distance estimation biases. Although a laser range finder cannot obtain a reading from a surfacing dolphin, training exercises were conducted using other objects on the water, such as fishing gear, boats and buoys. One observer estimated the distance to an object while another simultaneously recorded the actual distance with the laser range finder. The results were kept secret until after 20 trials, when the information was shared so that observers could improve the accuracy of their future estimates. These exercises were periodically conducted for all observers throughout the survey programme.

Dolphin density (D) and its associated coefficient of variation (CV) were estimated using the program DISTANCE 3.5 and according to the line-transect formula in Buckland *et al.* (1993):

$$\hat{D} = \frac{n\,\hat{f}(0)\,\hat{E}(s)}{2L} \text{ and } \hat{CV} = \sqrt{\frac{\hat{\mathrm{var}}\,(n)}{n^2} + \frac{\hat{\mathrm{var}}\,[\hat{f}(0)]}{[\hat{f}(0)]^2} + \frac{\hat{\mathrm{var}}\,[\hat{E}(s)]}{[\hat{E}(s)]^2}} \tag{1}$$

where n = number of on-effort sightings; $f(0)$ = probability density value at zero perpendicular distance; $E(s)$ = unbiased estimate of group size; L = length of transect lines surveyed; and var = variance. Abundance (N) was then calculated from the density estimate (D) according to: $\hat{N} = \hat{D} * A$, where A = size of the survey area. Data from all surveys were pooled to estimate $f(0)$. This value was then used to calculate abundance estimates stratified according to pre-monsoon, monsoon and post-monsoon seasons and a composite estimate for all three seasons combined. All other parameters and their associated variances were estimated empirically.

The Distance program plotted histograms of sighting distances and comparisons were made of the Akaike's Information Criterion (AIC) for uniform, hazard-rate and half-normal models to determine which one most closely fit the empirical data. Various levels of truncation for dolphin groups sighted at distances far from the trackline were experimented with and the level that resulted in the best fit with the theoretical model was chosen. One of the primary assumptions of line-transect theory is that all objects have the same probability of detection at the same distance. Since this assumption is violated when surveying species that occur in clusters (because large clusters have a higher probability of being detected far from the trackline in comparison to smaller ones), a size bias correction was used for $E(s)$, calculated from the log of estimated group sizes regressed against the detection probability estimated from the fit of the selected model to the pooled data (see Buckland *et al.*, 1993).

Dive and surface time study

A major assumption of line transect theory is that all animals are observed on the trackline (i.e. $g(0) = 1$; see Buckland *et al.*, 1993). This assumption is often violated when surveying cetaceans because the animals may be submerged or behave cryptically when they are within the observers' field of view, or the observers' attention may be directed elsewhere. This potential bias was investigated using radial sighting distances recorded during the surveys and group dive and surface times collected from land-based observation sites.

Four land-based observation sites were chosen (Malampaya Sound Protected Area Office (PAO), Agpay, Logpond, and Pancol; see Fig. 2), based on the criteria that they overlooked a variety of habitat types where dolphins were frequently found. Two observers searched for dolphins from these sites (not concurrent with the vessel surveys), one with binoculars and the other by naked eye, alternating every 10 minutes to reduce fatigue. Once a dolphin group was sighted, using a stopwatch, observers recorded group dive and surface times. Groups were defined as any cluster of dolphins observed in apparent association, moving in the same direction and often, but not always, engaged in the same activity. Dive times were defined as the interval when no animals were visible at the surface for longer than one second, while surface times were defined as the interval in between.

Photo-identification

In addition to conducting line-transect surveys, the feasibility of using photo-identification for assessing the Malampaya Irrawaddy dolphin population was investigated. When lighting conditions were adequate and the animals were within a reasonable range of the vessel (e.g. < 25m) photographs were taken of the dorsal fin and flukes with a *Canon* EOS-5 QD camera, equipped with a *Canon* EF 100-400mm F4.5-5.6 lens and image stabiliser. Ektachrome Elite 100 ASA colour slide film was used.

In photo-identification studies, dolphins are typically identified from nicks, scars, scratches, deformities and pigmentation features located on or in the region of the dorsal fin (e.g. Würsig and Jefferson, 1990). However, during preliminary surveys some of these features were also observed on the flukes, which sometimes became visible when the dolphins made steep inclined dives while foraging. Since the dorsal fins often appeared to have few marks that would allow individuals to be identified, attempts were made to photograph and use distinguishing features on the flukes for identification purposes.

Photographs were classified as poor (dolphin either not in view or the image not of sufficient quality for identifiable features to be discerned if they were present), moderate (dorsal fin or flukes could be seen clearly and, if present, obvious diagnostic features would probably be discerned) and good (dorsal fin or flukes could be seen and, if present, subtle diagnostic features would probably be discerned). Moderate and good slides were then classified according to the presence of recognisable distinguishing marks: absent (no features available for identification), fair (sufficient marks discerned for probable identification) and excellent (marks could be easily distinguished for reliable identification).

Ecology

During line transect surveys, information was collected on water surface temperature (with a standard laboratory thermometer), salinity (with an *Atago* Hand Refractometer), turbidity (with a *Hanna* HI 93703 Microprocessor Turbidity Meter) and depth (with a *Speedtech* Hand-held 400 KHs depth sounder) at all leg endpoints and at the locations where dolphins occurred. Samples were taken at leg endpoints due to convenience (as these generally corresponded with our observer rotations). It is recognised that this may have caused a small bias toward nearshore conditions. Factorial ANOVAs were used to investigate the effect of area (inner and outer Sound) and season (pre-monsoon, monsoon and post-monsoon) on temperature, salinity, depth and turbidity. A similar factorial design was also used to test for differences among the ecological samples collected at transect line waypoints in the inner Sound and those collected at the locations of dolphin sightings.

RESULTS

Distribution and abundance

During three surveys of the entire Malampaya Sound (total area 230.7km^2), one each in the pre-monsoon, monsoon and post-monsoon seasons, Irrawaddy dolphins were observed only in the inner portion (total area 133.7km^2; Fig. 2). Sightings of the species were confined to the same area during the other four surveys (two each during the pre-monsoon and post-monsoon seasons), which included search effort in the inner Sound and buffer zone (24.0km^2) only. Altogether 50 Irrawaddy dolphin sightings were recorded during survey effort used for the line-transect analysis (mean group size = 5.3, SD = 2.9, range = 1-15). Six sightings were made of bottlenose dolphins *Tursiops* sp. (probably *truncatus*; mean group size = 5.1, SD = 3.2, range = 1-9), 2 and 4 in the monsoon and post-monsoon seasons respectively. All were in the outer Sound and buffer zone, with the exception of one that was in the far northern portion of the inner Sound.

Based on seven complete surveys of the inner Sound conducted during the pre-monsoon, monsoon and post-monsoon seasons, covering a total of 578.1km of trackline, the overall abundance of Irrawaddy dolphins was estimated, using a Fourier series uniform + cosine model (see Burnham *et al.*, 1980), to be 77 individuals

(CV = 27.4%). This figure was similar to the seasonally stratified estimates (67, CV = 38.6%; 78, CV = 78.1% and 81, CV = 31.7% for pre-monsoon, monsoon and post-monsoon seasons, respectively, and well within their 95% confidence intervals; Table 1). There were no significant differences among the stratified estimates (Chi-square Prob = 0.5149).

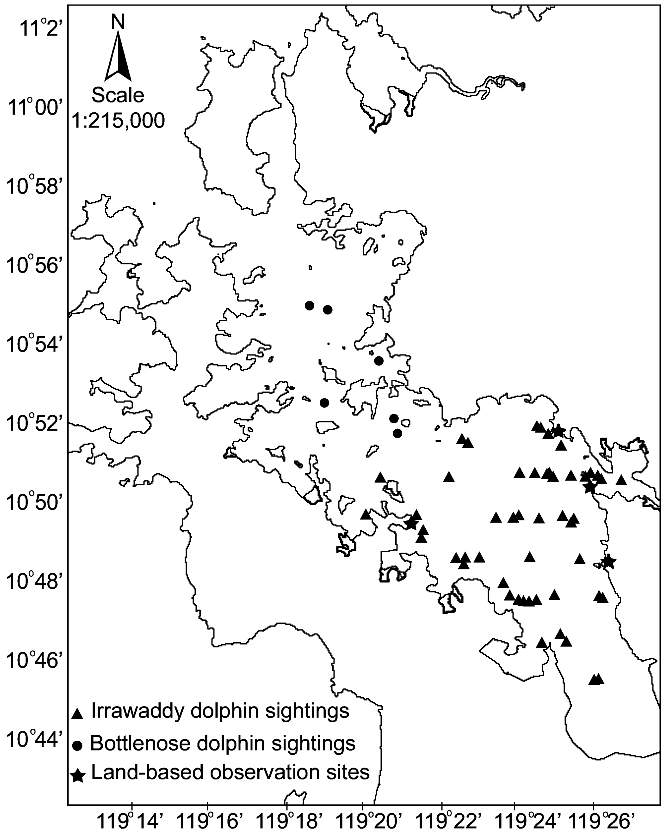

Fig. 2. Map of Malampaya Sound showing the locations of Irrawaddy and bottlenose dolphin sightings and land-based observation sites.

Photo-identification

A total of 524 dorsal fin photographs of Irrawaddy dolphins were taken during 29 encounters. Overall photographic quality was low, with 91.0% classified as poor, 5.3% as moderate and 3.7% as good. The large number of poor photographs resulted from the dolphins being too far away, or the image being out of focus. Of the 44 good and moderate quality photographs, 38.6% had no distinguishing marks, 11.4% were classified as fair and 50.0% as excellent. From the fair and excellent photographs, 17 Irrawaddy dolphins were identified. Two of these (OBRE03 and OBRE05) were re-identified once each during the study. In addition, 27 fluke photographs were taken. Photographic quality was similarly low, with 81.5% classified as poor, 11.1% as moderate and 7.4% as good. Two of these had fair distinguishing marks and three excellent, resulting in identifications of four individuals. No re-identifications were made from fluke photographs.

Ecology

For all Irrawaddy dolphin sightings in which ecological data were collected ($n = 48$) the mean surface temperature was 30.2°C (SD = 1.3, range = 27.0-32.5), depth 6.5m (SD = 3.1, range = 1.5-15.1), salinity 28.3ppt (SD = 4.7, range = 14.0-34.0) and turbidity 2.2 nephelometric turbidity units, NTUs (SD = 2.2, range = 0-9.6). These values were not significantly different from those collected at transect line waypoints in the inner Sound (DF = 247), although temperature was almost significant at Prob. = 0.0848, with slightly lower temperatures recorded at the survey waypoints. There were significant differences, however, in the ecological data collected in the inner Sound during different seasons for temperature (F = 28.83, Prob. = 0.0000; Tukey-Kramer Multiple-Comparison (TKMC) Prob. < 0.05 for Apr = Aug, Apr = Oct/Nov, Apr = Aug and Oct/Nov) and salinity (F = 21.70, Prob. = 0.000; TKMC Prob. < 0.05 for Apr and Aug = Oct/Nov, Apr = Aug and Oct/Nov) (Table 2, Fig. 3).

For the five bottlenose dolphin sightings where ecological data were collected, the mean temperature was 29.7°C (SD = 0.3; range = 29–30.0), depth 23.9m (SD = 7.0; range = 14.4–31.7), salinity 30.6ppt (SD = 1.5; range = 29.0-32.0) and turbidity 0.2NTUs (SD = 0.2; range = 0-0.5).

For surveys of the entire Sound (one each during April, August and October), there were significant differences (DF = 158) between ecological data collected in the inner and outer Sound (the latter inclusive of the buffer zone) for depth (F = 164.54, Prob. = 0.0000), salinity (F = 19.27, Prob. = 0.0000) and turbidity (F = 21.53, Prob. = 0.0000), with depth and salinity greater in the outer Sound and turbidity greater in the inner Sound. There were also (Table 2, Fig. 3) significant differences among seasons (DF = 2, 153) for temperature (F = 4.29, P = 0.0154, TKMC Prob < 0.05 for Apr = Aug, Apr = Oct, Apr = Aug and Oct) and salinity (F = 15.03, P = 0.0000; TKMC Prob. < 0.05 for Apr = Oct, Apr = Aug and April and Aug = Oct).

DISCUSSION

Distribution

The absence of Irrawaddy dolphin sightings in the outer Sound and buffer zone, during this study and others (see Dolar *et al.*, 2002, and unpublished reports of WWF-Philippines), and the close agreement among

Table 1

Composite and seasonally stratified abundance estimates of Irrawaddy dolphins in Malampaya Sound from line transect surveys conducted during April-November 2001 (*ESW* = effective strip width, S = mean groups size, *N* LCI and *N* UCI = Estimated population size at lower and upper 95% confidence intervals, respectively).

Survey	*n*	*L* (km)	*ESW*	*S*	*E(s)*	*D*	%CV	*N* LCI	*N* UCI	*N*
Pre-monsoon (Apr)	11	243.7	274.9	5.27	6.07	0.4983	38.57	32	139	67
Monsoon (Aug)	4	78.7	274.9	5.27	6.07	0.5612	78.14	18	311	74
Post-monsoon (Oct/Nov)	14	255.7	274.9	5.27	6.07	0.6045	31.67	44	139	81
Composite for all lines	30	578.1	274.9	5.27	6.07	0.5730	27.43	45	130	77
Composite for primary and extension lines	26	480.2	301.1	5.27	5.78	0.5209	29.93	39	124	70

Table 2

Summary of ecological parameters recorded during surveys in the Inner Sound, buffer zone, Outer Sound and for Irrawaddy dolphin sightings (PPT=parts per thousand, NTU=nephelometric turbidity units). Parameters are only reported for the first August survey due to poor weather conditions that compromised survey effort during subsequent surveys of the monsoon season.

		Temp (°C)				Depth (M)				Salinity (PPT)				Turbidity (NTU)			
Survey/sample	N	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
14-17 Apr																	
Inner Sound	24	30.0	1.4	28.0	33.0	6.4	4.8	0.7	15.7	31.8	1.3	29.0	34.0	3.7	4.6	0.0	20.7
Buffer zone	5	30.2	1.1	29.0	31.0	12.7	6.0	8.5	22.9	31.4	0.5	31.0	32.0	0.3	0.3	0.0	0.7
Outer Sound	20	30.0	1.1	28.0	31.5	28.0	11.8	9.8	49.1	32.1	1.3	29.0	34.0	0.3	0.3	0.0	0.8
Dolphins	3	31.5	1.8	30.0	33.5	6.0	4.7	1.0	10.4	31.0	3.6	27.0	34.0	3.9	3.8	1.2	8.2
19-20 Apr																	
Inner Sound	27	31.4	0.9	30.0	33.0	6.6	4.6	0.9	15.4	31.3	1.5	27.0	36.0	3.6	5.0	0.4	25.2
Buffer zone	6	30.3	1.3	29.0	32.0	18.1	7.2	9.5	26.5	32.7	1.2	31.0	34.0	0.5	0.8	0.1	2.0
Dolphins	5	31.6	0.5	31.0	32.0	6.4	2.7	4.1	10.8	31.0	0.7	30.0	32.0	1.5	0.7	0.8	2.5
22-24 Apr																	
Inner Sound	29	31.2	0.9	28.5	33.0	6.2	4.5	0.8	15.3	31.7	1.8	25.0	34.0	3.6	3.6	0.0	12.5
Buffer zone	5	31.1	0.5	30.5	32.0	17.6	8.1	8.6	25.6	32.4	0.5	32.0	33.0	1.1	0.9	0.5	2.6
Dolphins	9	31.2	0.7	30.0	32.5	5.0	1.9	2.3	7.7	32.0	2.1	27.0	34.0	2.9	1.9	1.1	6.1
8-11 Aug																	
Inner Sound	30	29.3	1.0	27.0	31.0	6.2	4.4	1.3	11.7	29.1	5.9	5.0	33.0	4.5	8.5	0.3	46.8
Buffer zone	6	29.2	0.6	28.0	29.5	19.9	6.5	11.7	27.1	29.8	7.3	15.0	34.0	4.4	8.9	0.6	22.7
Outer Sound	20	29.8	0.9	28.0	31.5	30.1	12.3	10.1	49.7	32.9	1.3	30.0	34.0	0.9	0.9	0.1	3.0
Dolphins	8	29.6	1.4	27.0	31.0	7.1	3.1	2.1	11.8	30.0	4.0	21.0	32.0	2.3	3.1	0.4	9.2
8-11 Oct																	
Inner Sound	29	29.2	1.0	28.0	31.0	5.8	4.5	0.8	14.5	27.4	2.7	21.0	31.0	2.9	5.1	0.0	22.6
Buffer zone	6	29.6	1.4	28.0	31.5	17.4	7.2	9.1	26.5	29.2	0.8	28.0	30.0	0.1	0.2	0.0	0.4
Outer Sound	21	29.7	1.1	27.5	31.0	26.9	15.1	6.1	51.2	31.0	2.1	25.0	34.0	0.0	0.0	0.0	0.0
Dolphins	7	29.6	0.9	28.0	30.5	6.8	2.9	2.4	10.9	28.3	1.7	26.0	30.0	0.7	0.9	0.0	1.9
13-16 Oct																	
Inner Sound	30.0	28.6	0.9	27.0	31.0	6.3	4.6	0.9	16.1	24.8	6.0	3.0	30.0	5.1	12.7	0.0	68.0
Buffer zone	6.0	29.6	0.5	29.0	30.0	17.0	7.3	9.2	26.5	29.7	0.8	29.0	31.0	0.0	0.0	0.0	0.0
Dolphins	8.0	29.1	0.8	28.0	30.0	6.0	3.2	3.2	11.0	25.1	3.5	18.0	29.0	2.2	2.5	0.0	6.3
6-12 Nov																	
Inner Sound	21.0	29.9	1.5	26.8	33.0	7.2	4.5	1.6	14.8	25.2	6.9	8.0	33.0	2.4	4.5	0.0	17.9
Buffer zone	6.0	29.4	0.8	28.5	30.5	16.6	7.1	8.5	25.6	31.8	1.3	30.0	33.0	0.3	0.7	0.0	1.7
Dolphins	8.0	29.5	1.1	27.5	31.0	8.6	3.9	3.9	15.1	21.4	3.6	14.0	25.0	2.6	3.0	0.1	9.6

abundance estimates from surveys conducted during pre-monsoon, monsoon and post-monsoon seasons, strongly suggest that the population is resident within the inner Sound. A plot of all on-effort sightings appears to show a slight affinity of the animals for shoreline areas (Fig. 2), but dolphins were also frequently observed in mid-water, often in close proximity to *bukatots* (fixed lift nets operating from a semi-permanent structure made of wood pilings). These structures probably aggregate fish and therefore may actually enhance mid-water habitat for the dolphins.

Evaluation of precision and biases of abundance estimates

An evaluation of the assumption that $g(0) = 1$ (i.e. all dolphins are detected on the trackline) was conducted using dive and surface time information from land-based observations and the radial distance estimates to dolphin clusters from the survey vessel. A mean dive time of 11.9sec ($n = 5{,}510$, SD = 18.6, range = 1-259) and mean surface time of 1.3sec ($n = 5510$, SD = 0.7, range = 1-14) were recorded from 90 dolphin groups at four observation sites. The mean group size was 4.7 dolphins (SD = 1.9; range = 2-10), which is close to the mean group size recorded during the line-transect surveys (5.3; interestingly the same figure was reported by Dolar *et al.*, 2002) and there was no significant difference between the two samples (Mann-Whitney Test Prob. = 0.2876).

According to distance estimation data from the surveys, a decline in sighting frequencies occurs past 466.7m (Fig. 4). The mean vessel speed for all seven surveys was 8.8 km/hr or 2.44 m/sec. This means that, on average, it took 190.6sec, to cover the distance where it can be assumed that dolphins available on the surface would have a high probability of being detected (otherwise there would have been a decline in the sighting rate before this distance). A cumulative frequency distribution of dive times indicates that, while surveying along 466.7m of trackline, 99.9% of dolphin groups would be available for detection at least once (only seven dives were recorded longer than 190.6sec.) and, on average (according to the mean dive time plus the mean surface time (13.2sec) for a complete dive cycle), during 17 occasions for a total of 22.1sec on the surface. The dolphins would also be available for detection during the same number of surfacings and for the same amount of time while inside the second 466.7m increment (i.e. between 466.7m-933.4m distance from the vessel) where the proportion of animals detected was still relatively high (85.7%). Although the behaviour of Irrawaddy dolphins was relatively inconspicuous, their relatively short surfacing intervals ensured a very high probability of detection on the trackline (at least during Beaufort 3 sea state conditions or less; see below). The short surfacing intervals recorded during this study should not be extrapolated to Irrawaddy dolphins in others areas. The results may have been related to the shallow depth of the inner Sound. Stacey and

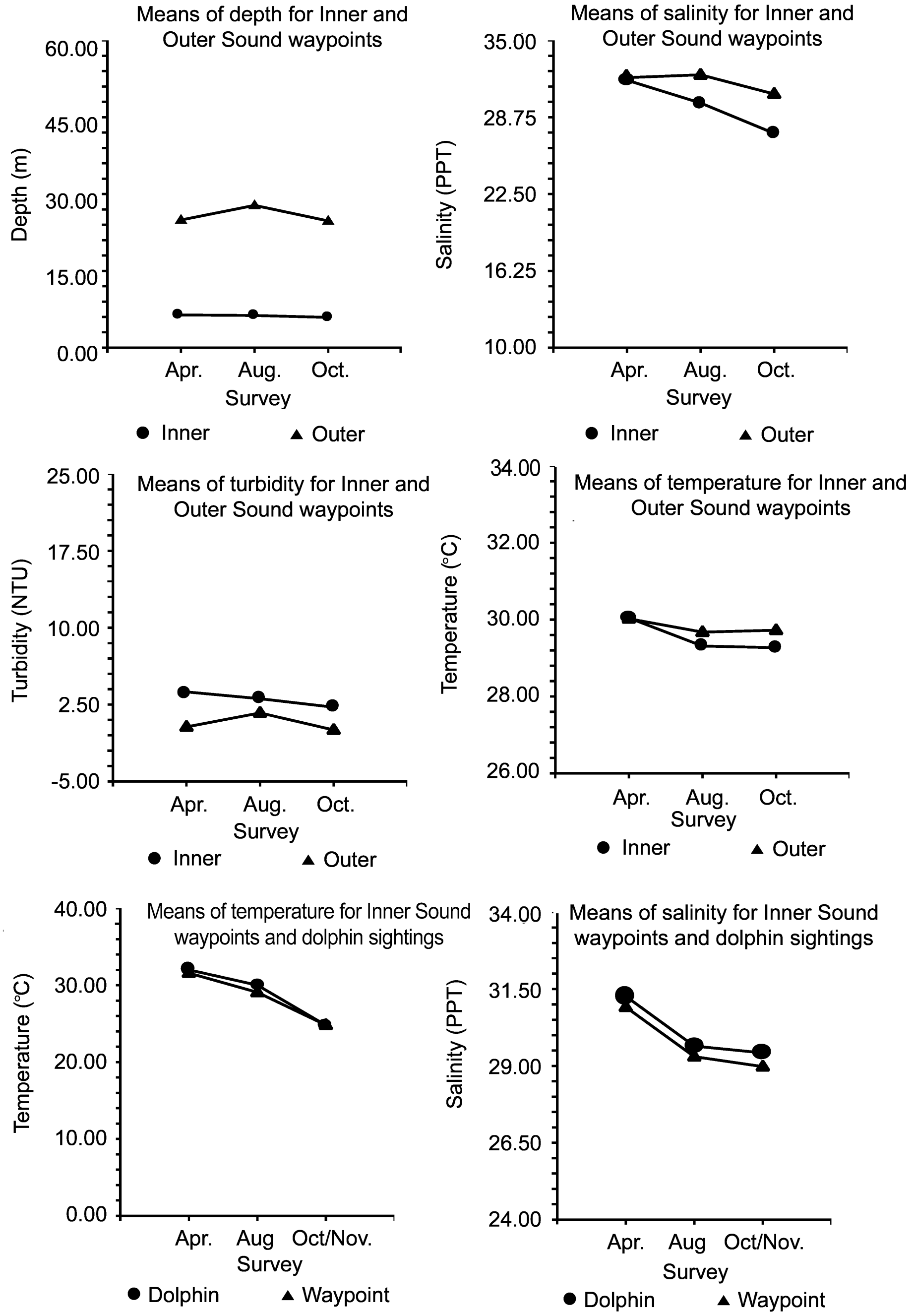

Fig. 3. Plots of means for ecological data collected in the inner and outer sounds (top four plots) and at waypoints and dolphin sightings in the inner sound only (bottom two plots).

Hvenegaard (2002) recorded much longer dive times for the species in the Mekong River, Laos (mean = 115.3sec, SD = 59.1, $n = 277$), where the mean depth of dolphin habitat was 18.4m.

Due to the large number of sightings made at perpendicular distances relatively close (0-200) and far (401-800m) from the trackline, but lack of sightings at middle distances (201-400; Fig. 5), a high truncation level (40%) was necessary to obtain a good fit of the theoretical model (Fig. 6). Three possibilities can explain these field results. The first one relates to differences in the ability to detect the animals according to their behaviour. The dolphins often behaved in an extremely cryptic manner, barely breaking the surface with the top of their head. While exhibiting this behaviour they could generally only be detected at close distances. During other times, while feeding and socialising, the animals were much more visible, showing their dorsal fins and flukes and sometimes splashing. They could then easily be observed from far distances (especially considering their short group dive intervals; see above). Another explanation was that observers alternated between searching with binoculars and naked eye. The tendency while searching with binoculars was to emphasise detecting dolphins at far distances, while observers searching with naked eye were probably unable to detect dolphins at distances greater than 200m. This may have resulted in a lack of searching coverage at middle distances (i.e. > 200m and < 400m), although this problem would be expected to have a more profound effect on radial sighting distances than on perpendicular sighting distances. A third explanation could have been avoidance behaviour at far distances and attraction behaviour at close distances. However, this is considered unlikely since the dolphins were only observed reacting evasively to the survey vessel when approached quite close (e.g. < 25m) to take photographs (and never when a depth reading was taken with the

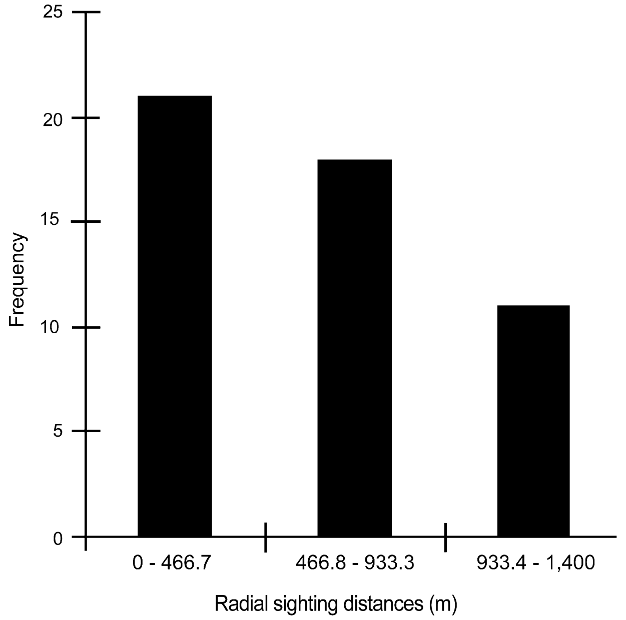

Fig. 4. Histogram of sighting frequencies divided into three equal radial distance increments.

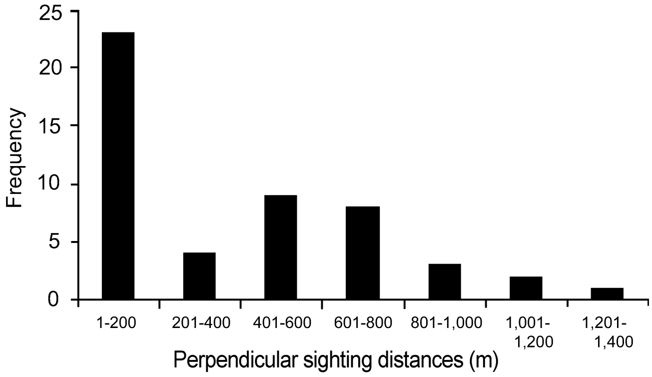

Fig. 5. Frequency of perpendicular sighting distances recorded during dolphin surveys ($n = 50$).

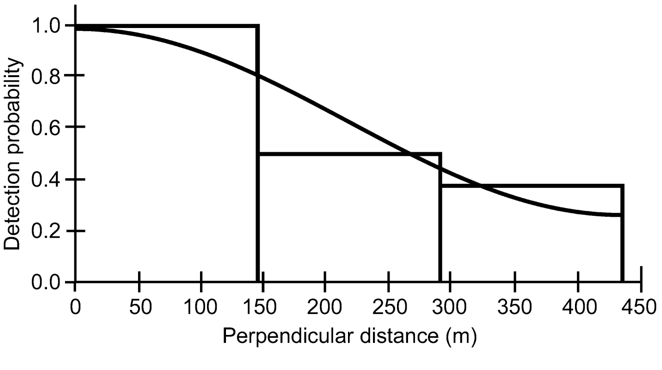

Fig. 6. Detection probability plot for the composite abundance estimate from all transect lines in the inner Sound.

hand-held 400kHz depth sounder) and they were never observed to be attracted to the boat, except occasionally when the motor stopped and the vessel drifted while not conducting survey effort. A similar lack of evasive or attraction behaviour was observed while collecting dive and surface time data from the land-based observation sites. Regardless of the factors that contributed to the poor fit of the theoretical model without substantial truncation, the level of precision for the composite abundance estimate was sufficient to determine that the population size is dangerously small.

The accurate estimation of sighting distances is one of the primary assumptions of line-transect theory (Buckland *et al.*, 1993). A regression of estimated versus measured distances to objects on the water showed a significant relationship between the two ($F = 745.7$, $\text{Prob.} = 0.0000$, $R^2 = 0.7104$) and a small negative bias among distances measured > 100m (Fig. 7). If it is assumed that observers estimate distances to dolphins with the same degree of accuracy as the objects that were used for the distance estimation experiment, then the abundance estimate of 77 dolphins may have a slight positive bias. The distance estimates could have been calibrated to dolphin clusters recorded during the surveys, but the high residual variance among the estimated values in the experiment, and the fact that the measured values were only on average 4.8% greater than the estimated ones, implied that the resultant correction factor would probably be unreliable and the difference negligible. A large portion of the residual variance and negative bias could be accounted for by differences in the distance estimation abilities of individual observers. Perhaps not surprisingly, those observers who estimated distances most accurately in the experiment also had the highest sighting rates during the dolphin surveys. This may partially mitigate the apparent negative bias indicated in the experiment (which used pooled data from all observers). It also suggests that calibrations should be considered for individuals, rather than for the entire observer group. This was not possible due to small sample sizes relative to the number of individuals who participated as observers.

Due to the apparent slight clumping of dolphin sightings in nearshore areas and the possibility that including dolphins observed along the dogleg transect lines could positively bias the resulting abundance estimate, a separate estimate using only data from the primary and extension transect lines was calculated. While this estimate was slightly lower (65 dolphins, CV = 30.1%), the general overlap in its range (95% CI = 36-116) versus the estimate that includes all transect lines (95% CI = 45-130) and the broad agreement between overall encounter rates for the dogleg lines (0.0725 sightings km^{-1}; $n = 7$) versus the primary and extension lines (0.0895 sightings km^{-1}; $n = 43$) indicates that including sightings from the dogleg lines would not cause a significant bias. The similarity in sighting rates, combined with the extreme shoreline complexity of the Sound, also implied that deleting sightings made on dogleg lines would unnecessarily reduce sample size and possibly introduce a slight negative bias, because substantial areas of dolphin distribution in waters not visible from the primary or extension lines would be excluded from the analysis.

To a certain extent, distance sampling can compensate for animals missed due to poor sighting conditions (Jefferson and Leatherwood, 1997), however, Beaufort sea state can severely affect line-transect abundance estimates, especially with cryptic species; see for example Palka (1996). Measures were taken to minimise this bias by generally suspending survey effort when sea state conditions were greater than Beaufort 3 or, during the monsoon season, patching together data from three surveys to achieve a single complete survey conducted in Beaufort 3 conditions or less. In the latter case, due to the strong spatial heterogeneity of wind patterns (especially affected by convection forces near Mt Capoas) and the possibility that a clumped distribution of dolphins within the Sound could lead to a biased abundance estimate, for the line-transect analysis it was considered important to use only data from complete surveys conducted in Beaufort 3 conditions or less. The wisdom of both of these measures was reinforced by the fact that no Irrawaddy dolphin sightings were made during the survey effort conducted in Beaufort 4 conditions (93.8km in the inner Sound – only 16.2km of these data were used in the line-transect analysis). In comparison, the overall encounter rate for survey effort conducted in Beaufort 3 conditions or less was 0.0865 sightings km^{-1}. If the detection rate for survey effort conducted in Beaufort 4 conditions was comparable, six or seven sightings should have occurred during the 93.8km of survey effort conducted during these conditions.

The distribution of survey effort in the inner Sound used for the abundance estimate was 11.6%, 19.1%, 31.4%, 15.9% and 2.8% for Beaufort sea states of 0-4, respectively. Although there were insufficient data to stratify abundance estimates according to sighting conditions, a chi-squared test showed no significant difference between the actual and expected number of sightings according to sea state (Prob. = 0.4048, DF = 4).

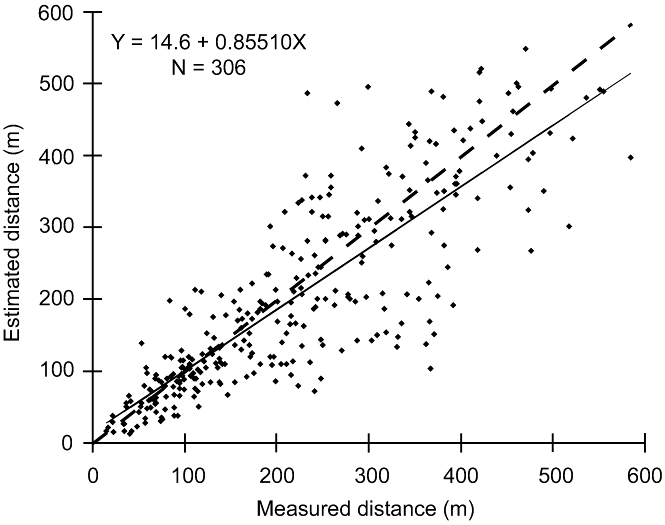

Fig. 7. Estimated sighting distances to fixed objects versus measured distances from laser range-finder readings. The solid line fits the empirical data while the dashed line is theoretical assuming no bias.

Feasibility of photo-identification

Preliminary indications are that photo-identification is a feasible research technique for studying Irrawaddy dolphins in Malampaya Sound (also see Parra and Corkeron, 2001) but that its application will be labour intensive and fairly expensive. Problems with this technique included the lack of distinguishing marks, cryptic surfacing behaviour and avoidance of the research vessel by the dolphins upon close approach. Stacey and Hvenegaard (2002) reported similarly poor results from photoidentification efforts on Irrawaddy dolphins in the Mekong River of Laos; out of 629 photographs taken, only 7.5% were of sufficient quality to distinguish identifiable marks and less than one-quarter of these exhibited marks that could be used to identify individuals.

While line-transect surveys proved to be a much more useful technique for assessing Irrawaddy dolphin abundance, photo-identification can provide valuable data on other aspects of the dolphin population that are difficult (or impossible) to obtain using other methods. Home range, habitat use and social affiliations can be investigated according to re-identifications and the frequencies at which individuals occur in particular locations and in the same groups (see Würsig and Jefferson, 1990 and other papers in Hammond *et al.*, 1990). Knowledge of these parameters is important for developing effective strategies to reduce human-dolphin conflicts and for evaluating the effects of dolphin kills. For example, the death of a single individual can have severe negative consequences on the survivability of other individuals in complex cetacean societies (especially females with dependent calves), but these issues are difficult to assess without information on individuals within the population.

Environmental preferences

Results of the ecological investigation indicate the restricted environmental preferences of Irrawaddy dolphins in the Sound, which probably explains their confinement to the inner portion. Interspecific competition with bottlenose dolphins occurring in the buffer zone and outer Sound may also play a role. Irrawaddy dolphins appear to be particularly adapted to shallow inshore waters, characterised by relatively low salinity and high turbidity, in comparison to areas located closer to open water (i.e. the buffer zone and outer Sound).

These results reinforce the notion of the population's vulnerability to local disturbances. Unlike some other dolphin species, whose environmental preferences are more flexible and thereby allow them to occupy a greater range of habitat (e.g. bottlenose dolphins; see Shane *et al.*, 1986; Wells and Scott, 2002), Irrawaddy dolphins appear to be obligatorily adapted to relatively rare and circumscribed ecological conditions – deep pools of large rivers and protected nearshore marine environments (including appended lakes) with substantial freshwater inputs (see reviews in Stacey and Leatherwood, 1997; Stacey and Arnold, 1999; Smith and Jefferson, 2002). High salinity, however, does not appear to have direct adverse effects, as there was no difference between the mean salinity values recorded for the outer and inner portions of the Sound during April (when freshwater inputs were particularly low), while the dolphins still remained confined to the latter area. This implies that the affinity of the dolphins for low salinity waters is likely due to ecological preferences (probably related to prey), rather than to physiological intolerance to high salinity conditions.

Mortality and population viability

Estimating human-caused mortality of dolphins in the Malampaya population is an extremely difficult task due to the sporadic reporting of accidental kills. The situation is confounded by the value of dolphin carcasses for human consumption, which ensures that most deaths probably go unreported, or that reports are received well after the remains of the animal have been disposed. Between February and August 2001, two dolphins were confirmed accidentally killed in bottom-set nylon gillnets used for catching crabs – locally known as *matang quatro* nets. Unconfirmed reports from fishermen indicate that three additional dolphins may have been killed in gillnets during this seven-month period. For small cetaceans, it is generally recommended that yearly removals should not exceed 1-2% of the overall population size (Wade, 1998). Using a minimum estimate of two dolphins killed per year, considered extremely conservative, this works out to be 2.6% of the population, according to the best estimate of abundance made during line transect surveys

(77 dolphins). It has been argued that, when evaluating the potential effects of mortality on dolphin populations, the minimum abundance estimate, rather than the best, should be used for calculating mortality rates (e.g. see Taylor and Gerrodette, 1993). Using the minimum abundance estimate (45 dolphins), the yearly mortality rate would then be 4.4% of the population size. This figure should probably still be considered low because it considers only the two confirmed kills made during seven months.

Considering that the small size of the Malampaya population already means it is vulnerable to extirpation, due to demographic stochasticity, inbreeding depression and catastrophic environmental and epizootic events (see Soulé and Wilcox, 1980; Gilpin and Soulé, 1986; Lynch, 1996), the present rate of accidental killing will almost certainly lead to its extirpation unless immediate action is taken to reduce or eliminate human-caused deaths.

Conservation

The Irrawaddy dolphin population in Malampaya is the only one known of the species in the Philippines and its extirpation would represent a significant loss of cetacean diversity in the region. Of paramount importance is to eliminate, or drastically reduce, dolphin mortality from entanglement in gillnets. Similar to the situation of the vaquita, a critically endangered porpoise isolated in the upper Gulf of California, Mexico (see Rojas-Bracho and Taylor, 1999; Rosel and Rojas-Bracho, 1999; Taylor and Rojas-Bracho, 1999), this will require action at a socio-economic level, as well as assistance from cetacean and fishery scientists (Reeves *et al.*, 2003). A number of high priority recommendations relevant to conservation are discussed below.

(1) Developing socio-economic alternatives to dramatically reduce or eliminate the incidence of dolphin entanglement in matang quatro *gillnets*

The economic status of fishermen in Malampaya Sound is generally poor (see National Integrated Protected Areas Programme (NIPAP), 2000), and the *matang quatro* fishery for crab provides substantial local employment. This fishery requires little monetary investment and is therefore an attractive option for the most economically impoverished of local fishermen. It would be unacceptable (and probably counter-productive) to prohibit this fishing technique without providing alternatives that ensure an equal or greater income. While more information is needed about the feasability of alternative employment options and the details of the *matang quatro* crab fishery (e.g. number of fishermen and income generated by the fishery, differences between the efficiency of traps versus *matang quatro* nets, market trends for crabs in the Philippines, etc.), immediate action must be taken to provide alternative employment if Irrawaddy dolphins are to be conserved in Malampaya Sound (Reeves *et al.*, 2003). Alternatives could include developing the green mussel fishery, improving the efficiency of crab pots, promoting grow-out pens for groupers and other economically valuable fishes and developing community-based ecotourism.

(2) Establishing gillnet free zones in core areas of dolphin distribution

While providing employment alternatives for gillnet fishermen is clearly the most important first measure that should be taken, without the ultimate closure of this fishery, conservation prospects for the dolphins are poor. A likely scenario is that, as current gillnet fishermen take up other occupations, immigrants or a new generation of local villagers, will begin using *matang quatro* nets again. Concurrent with promoting employment alternatives should be regulations prohibiting gillnet use. This should proceed in a step-wise fashion, starting in areas that are easily monitored and where the dolphins occur most often. As more gillnet fishermen choose to pursue other forms of employment, more areas would then become closed. The success of this approach will depend on the close cooperation of regulating authorities, conservation organisations and local fishing communities, and enforcement to ensure that everyone abides by the same rules. A major challenge will be to convince local people that gillnet free zones offer benefits to them and thus deserve their support. Such benefits might include revenues from nature tourism, permission to use non-destructive fishing techniques and the fact that protection of fish breeding or nursery areas can enhance fisheries outside the zone (see Reeves *et al.*, 2003).

(3) Promoting Irrawaddy dolphins as a flagship species for environmental stewardship of Malampaya Sound

The presence of Irrawaddy dolphins is a strong reminder that judicious stewardship is critical to preserve an environment that supports abundant and diverse fish and crustacean communities. Promoting Irrawaddy dolphins as a flagship species will require strengthening community awareness programmes, with an emphasis on educating local people on regulations regarding resource use and on promoting the linkages between dolphin conservation and sustainable fisheries.

Research and monitoring

In addition to direct conservations measures, it is important that research is continued, particularly in terms of monitoring whether conservation measures are working. Some high priority recommendations is this regard are discussed below.

(1) Strengthening the capacity of local scientists

Education and infrastructure development are required so that local scientists and resource managers can provide the stimulus and expertise for dolphin conservation. It is essential that local workers develop the ability to independently devise, conduct, analyse and effectively communicate the results of research and monitoring activities.

(2) Establishing a long-term programme to monitor the dolphin population

Monitoring abundance, distribution and mortality is critical for measuring the efficacy of conservation measures. Line-transect surveys have been shown to be an appropriate technique for assessing dolphin distribution and abundance in the Sound. It is important that the standardised survey protocol developed during the present study be consistently applied and that effort and observations be painstakingly documented. The low precision, typical of wildlife studies investigating small populations, can dramatically affect the ability to achieve statistical significance with inter-survey comparisons (Taylor and Gerrodette, 1993). This means that several surveys will need to be conducted each year and that an appropriate α probability level should be based on the consequences of failing to reject the null hypothesis of no trend when it is indeed false (see Gerrodette, 1987; 1993) — probably set at not less than 0.10, considering the small size of the Malampaya population. Monitoring mortality will also be essential. Researchers face formidable challenges in this area because fishermen may be reluctant to report accidental kills, due to fear of prosecution or future restrictions on fishing activities. They also have a strong motivation to keep dolphin carcasses, due to their value as a source of meat. Scientists should establish a community-based reporting network to encourage fishermen to report incidental catches and recover carcasses for examination and necropsy. The fishermen should be assured that they will not be prosecuted for reporting accidental kills and a campaign should be initiated for convincing local villagers to utilise alternative food sources.

(3) Continuing photo-identification efforts for both Irrawaddy and bottlenose dolphin in selected areas
As new identifications are made and previously identified animals are re-identified, the photo-identification catalogue compiled during this study will become a more valuable tool for guiding management considerations. Due to the difficulties of applying this technique to Irrawaddy dolphins in the Sound, a relatively small area accessible by small paddleboat should be targeted for emphasis (e.g. nearshore waters between Old Guinlo and Agpay). This will provide initial information on habitat use and site fidelity, which can be followed up by more extensive photo-identification efforts as the photo-id catalogue is enlarged and as per the availability of funds and trained personnel.

(4) Conducting additional investigations to determine if Irrawaddy dolphins occur in other areas of the Philippines
The small size of the Irrawaddy dolphin population in Malampaya Sound means that it is extremely vulnerable to extirpation. The loss of genetic variation in small populations can result in decreased fecundity and reproductive success, smaller offspring size, slower growth rates and reduced survivorship (Ralls *et al.*, 1986). The prospects for survival of the population would be greatly enhanced by the mixing of individuals from one or more other populations, even if this occurred only very occasionally (assuming that the new immigrants were adapted to similar environmental conditions; see Lynch, 1996). Both for evaluating the long-term viability of the Malampaya population and considering the need for protecting other populations, should the species be found to occur elsewhere in the Philippines, a concerted effort should be made to identify other areas in Palawan and adjacent islands (e.g. in the Calamian Group to the north and Balabac Island and the Pangutaran Group to the south) where Irrawaddy dolphins might occur. This investigation should initially be conducted using interview surveys (see Aragones *et al.*, 1997) and by selecting potential sites, based on knowledge of the oceanography, bathymetry and ecological features where Irrawaddy dolphins are already known to be present (see above and reviews in Stacey and Leatherwood, 1997; Stacey and Arnold, 1999), with follow-up at-sea surveys conducted using standardised techniques.

(5) Investigating the population identity of Irrawaddy dolphins in Malampaya Sound
Wildlife conservation should aim to preserve the full range of genetic variation within species. The nearest area where another population of Irrawaddy dolphins is known to occur is northern Borneo, some 550km to the south. Evidence from skull morphology suggests that there are probably two sub-species or species of *Orcaella*, one occurring in South and Southeast Asia and another in Australia and Papua New Guinea (Beasley *et al.*, 2002). Throughout their range, there may also be numerous genetically distinct populations. Information on the population identity of Irrawaddy dolphins in Malampaya Sound would be useful for evaluating the viability and evolutionary significance of the population (see Dizon *et al.*, 1992). Population identity should be investigated using both morphologic and genetic techniques.

ACKNOWLEDGEMENTS

This project was a component of the Malampaya Sound Ecological Studies Project implemented by Kabang Kalikasan Ng Pilipinas (World Wildlife Fund – Philippines). We acknowledge the assistance with logistics given by the project manager, Terry Aquino, and staff of the Malampaya Sound Protected Area Office and KKP offices in Puerto Princessa and Quezon City. We appreciate the help of the captain and crew of our research vessel, Cipriano Galang, Greg Gacoba and Jessie Macahipay and local villagers of Malampaya Sound. We also thank Dr Thomas Jefferson (Southwest Science Fisheries Center, National Marine Fisheries Service, NOAA, P.O. Box 271, La Jolla, CA 92038 USA) and two anonymous reviewers for providing valuable comments on an earlier draft of the manuscript. The Wildlife Conservation Society provided funding to the first author during the later stages of preparing this paper.

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A review of animal and human health concerns during capture-release, handling and tagging of odontocetes

S.A. NORMAN*, R.C. HOBBS*, J. FOSTER#, J.P. SCHROEDER+ AND F.I. TOWNSEND++

Contact e-mail: Stephanie.Norman@noaa.gov

ABSTRACT

The capture-release of odontocetes allows for tag deployment which provides an opportunity to study behaviour and habitat use by free-ranging animals, as well as clinical assessment of the animal and tissue collection. This review recognises those elements that are common to most capture and tagging projects, identifies collective knowledge of animal and human health concerns during handling of odontocetes and provides guidelines for safer handling techniques. Handling during tagging projects can involve chase, capture, restraint, manipulation, tag application, often removal from the water and release at the capture site. The risk of injury during capture will be reduced by using experienced personnel, adequate technical support and proper equipment. For the duration of the handling process, the animal's stimulus response should be monitored as well as its cardiovascular and respiratory function. Stress response of the odontocete is monitored by behavioural assessments, physiological monitoring and/or blood sampling. Possible complications from tag placement may include infection at the implant site leading to tag failure, behavioural alterations in response to tag placement and tag rejection. During handling of an odontocete, there is the potential for disease transmission between humans and the animal. Exposure to diseases is minimised by wearing protective clothing and gear and exercising caution when working around the animal's blowhole.

KEYWORDS: DISEASE; LIVE-CAPTURE; RADIO-TAGGING; SATELLITE TAGGING; STRESS

INTRODUCTION

Tagging and tracking odontocetes allows biologists and wildlife managers to study behaviour and assess habitat and resource use by free-ranging animals. Current techniques for long-term (e.g. longer than a week) tag attachments to smaller odontocetes require that animals are captured and held for a brief period of time while the tag is attached (Irvine *et al.*, 1982; Würsig, 1982; Tanaka, 1987; Tanaka *et al.*, 1987; Scott *et al.*, 1990; Martin *et al.*, 1993; 2001; Hanson *et al.*, 1998; Hanson and DeLong, 1999; Martin and Smith, 1999; da Silva and Martin, 2000; Ferrero *et al.*, 2000; Richard *et al.*, 2001). Capture of animals for tag deployment also provides an opportunity for clinical health and body condition assessment of the animal (Wells *et al.*, In press), determination of gender, collection of teeth for age determination (Hohn *et al.*, 1989), evaluation of contaminant burdens (Schwacke *et al.*, 2002) and assessment of reproductive condition (Wells, 2003). This review attempts to identify those elements that are common to most capture and radio- and/or satellite-tagging projects, to codify collective knowledge regarding animal and human health concerns and to provide guidelines for safer handling practices. This document is intended as a reference for experienced researchers and a resource for inexperienced researchers contemplating new projects, but is by no means a comprehensive review of specific projects or all odontocete species to be tagged. It should be kept in mind that the capture and handling of odontocetes can vary greatly amongst and within species and locations and that different species may have some different handling and monitoring requirements. Although this review describes procedures requiring varying skill levels and makes recommendations regarding their application, reference to this document should not be considered a substitute for including experienced personnel in the field party. A general rule of thumb is that the degree to which an animal is compromised increases with the amount of handling and the length of time that the animal is handled. An individual animal may not display any obvious outward signs of being compromised beyond a threshold from which it cannot recover, as evidenced by the occasional sudden death of an animal that otherwise outwardly appears to be tolerating handling. Thus, handling should be kept to the minimum necessary to complete the research objectives.

Clearly, capture-release is not the only option and in cases where capture and handling is not feasible or a short-term, less invasive attachment is desired, remote deployment methods such as suction cups and barbed or toggled attachments may suffice especially for larger cetaceans (e.g. blue whales (*Balaenoptera musculus*), northern bottlenose whales (*Hyperoodon ampullatus*), bowhead whales (*Balaena mysticetus*), Dall's porpoise (*Phocoenoides dalli*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), killer whales (*Orcinus orca*), narwhals (*Monodon monoceros*), northern right whales (*Eubalaena glacialis*) and sperm whales (*Physeter macrocephalus*) – Watkins *et al.*, 1984; 2002; Watkins and Tyack, 1991; Baird, 1994; Baird and Hanson, 1997; Mate *et al.*, 1997; 1998; 1999; 2000; Hooker *et al.*, 2000; Laidre *et al.*, 2002). Although remote attachment methods are not reviewed here, this review will give researchers faced with a choice of methods more information on the tradeoffs involved with a capture and handling technique.

Handling during tagging projects can involve chase, capture, restraint, manipulation, tag application and often removal of the animal from the water, followed by release at the capture site. Several health concerns should be kept in mind during these phases:

(1) physical injury of both humans and animals during the handling process;

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* *National Marine Fisheries Service, National Marine Mammal Laboratory, 7600 Sand Point Way NE, Seattle, WA 98115, USA.*
# *International Marine Research Foundation, 4226 South 375th Place, Auburn, WA 98001, USA.*
+ *Marine Mammal Research Associates, 620 W. Anderson Rd., Sequim, WA 98382, USA.*
++ *Bayside Hospital for Animals, 251 NE Racetrack Rd., Fort Walton Beach, FL 32547, USA.*

(2) the physical and physiological effects of capture on the animal;
(3) possible complications from tag placement;
(4) the potential for disease transmission between humans and animals.

These concerns are addressed and suggestions are provided to minimise disease transmission and injury to all individuals involved. Methods are outlined for assessing and monitoring the animal during handling and tagging in the field.

Evolution in the marine environment, as a herd animal subject to predation, provides marine mammals with unique adaptations to cope with acute, short-term stressors and maintain homeostasis (St Aubin and Dierauf, 2001). However, some aspects of tagging operations, such as restraint in a sling and close proximity to humans, fall outside of the normal range of adaptation. The immediate and long-term effects of handling on the health and behaviour of these animals is unclear, but given the considerable cost for each tag deployment and subsequent data collection and the implicit assumption that the tagged animals are representative of the population of interest, close monitoring of the animal's biological parameters and risk reduction during handling become important.

RISK OF INJURY DURING HANDLING

Capture

The capture itself can be dangerous to both humans and the animals. Several techniques for capturing and handling white whales (*Delphinapterus leucas*) and other small odontocetes have been described. Seine nets have been used to capture species such as bottlenose dolphins (*Tursiops truncatus* —Asper, 1975), killer whales (*Orcinus orca*) and botos, or Amazon River dolphins (*Inia geoffrensis* —da Silva and Martin, 2000). Breakaway hoop nets have been successfully used in capturing bottlenose dolphins and porpoises (Ridgway, 1966; Asper, 1975; Hanson, 1998). In certain areas, pilot whales (*Globicephala melas* —Heide-Jørgensen *et al.*, 2002) and white whales (Sergeant and Brodie, 1969; Martin and Smith, 1992) can be driven to shore at low tide or into shallow water where they can be grounded or captured with a hoop net over the head (Orr *et al.*, 2001). Entanglement in a drift or set-net has been used to capture white whales (Orr *et al.*, 2001) and narwhals (Dietz and Heide-J¢rgensen, 1995; Dietz *et al.*, 2001), and an encirclement technique along with high speed net deployment has also been used to capture white whales in the Cook Inlet, Alaska region (Ferrero *et al.*, 2000) and pelagic dolphin schools in the eastern tropical Pacific Ocean (Perrin *et al.*, 1979; Jennings *et al.*, 1981). A human handling a cetacean may be struck by a thrashing tail or rostrum, or injured by netting or restraining devices. Removing it from a net or other capture device can also be stressful or traumatic to the animal and operators. Handlers can eliminate much of their own risk by minimising carelessness and planning handling practices with forethought. Once restrained, white whales, as well as narwhals, seldom continue to struggle for more than a few minutes (Orr *et al.*, 2001; Heide-Jørgensen, pers. comm.). Smaller species such as harbour (*Phocoena phocoena*) and Dall's porpoise may struggle (Norris and Prescott, 1961) for a longer period of time than larger species such as white whales and killer whales (Walker, pers. comm.). In contrast to most odontocetes, botos are much more comfortable laying on their side than on their bellies and consequently struggle less (da Silva and Martin, 2000). Tucuxis (*Sotalia fluviatilis*), however, are more nervous than botos when being handled (Martin, pers. comm.).

During capture and handling procedures, a cetacean sometimes incurs physical injuries such as abrasions, lacerations, contusions, or injuries resulting from malpositioning or unruly behaviour (Spraker, 1982). The risk of some of these injuries may be reduced by using skilled, experienced personnel and adequate technical support and proper equipment to handle the animal, removing or covering hazardous objects in the work area, and placing padding around its body (sand, foam pads, or other appropriate material) if it is fully or partially removed from the water.

Monitoring of animals caught in a net

In net captures, there are significant risks of death if an animal is trapped below the water surface for too long, or of aspiration of water into the lungs if the animal is trapped near the water's surface and allowed to struggle for a prolonged period (Walker, pers. comm.). If a significant volume of water has been aspirated at the water's surface while struggling in a net, the animal may have difficulty getting sufficient oxygen into its bloodstream and may die during handling or after release. Further, aspiration of seawater into the lungs may introduce infection that can result in pneumonia. On the other hand, cetaceans trapped below the surface may not struggle at all or give little indication of entanglement. A capture team should remain aware of the possibility of multiple entanglements, and that an apparently single animal may have been accompanied by one or more animals unseen below the surface. The following procedures should be undertaken to reduce these risks:

(1) nets in the water should be watched continuously for movement or dips in the floatline and should be patrolled regularly in cases where the full length is not visible from a single vantage point;
(2) in murky or turbid water, where the full depth of the net is not visible from the surface, the net may be raised periodically until the lead line is visible or, alternatively, the cork line may be closely observed for movement or sinking;
(3) nets should not be left in the water unattended and should be removed from the water when not in use;
(4) mesh size should be selected to limit the risk of capturing non-target animals;
(5) net dimensions should be limited to the minimum necessary for the operation to reduce the chance of multiple captures (e.g. during captures of white whales, limited water visibility has prevented detection of non-target animals trapped under the surface with the target animal);
(6) entangled animals should be quickly supported and removed from the net;
(7) team size should be sufficient to provide the appropriate number of people for support of each animal in the water, assuming the maximum number of animals likely to be caught in each set, with at least enough additional people held in reserve on a mobile vessel to get to a place where another animal strikes the net;
(8) the field team should make contingency plans for handling multiple captures, releasing excess animals, captures of mother-calf pairs, and net handling, while the team is occupied with tagging of captured animals;

(9) capture operations should not be conducted in marginal weather conditions;
(10) the person responsible for setting and retrieving the net should have a high level of familiarity with the area, including water depth, current patterns, tide, seafloor type, presence of entangling objects and non target individuals, etc;
(11) every effort should be made not to set a net around very young calves and their mothers, or other potentially compromised individuals, unless they are the targets of the study as nursing calves are especially sensitive to capture and handling;
(12) a cetacean found entangled should be brought quickly to the surface and supported with its blowhole well exposed and protected from waves washing over −if the animal does not clear its blowhole and begin breathing normally, or it begins breathing in a weak, laboured or unusual manner, respiration can sometimes be stimulated by light splashing of the forehead region with water.

The design of the net (e.g. length, depth, mesh size and twine, type and size of lead and float lines, single panel or multipanel, how it hangs) is dependent upon the species to be captured and the field situation. There is no substitute for experience with nets and capture techniques. Rather than incurring unnecessary risk, prolonged trial and error should be avoided by consulting knowledgeable researchers involved with a similar species and field setting. Even with the best advice, however, it can take several field seasons to develop a reliably efficient and safe technique for a novel situation (Ferrero *et al.*, 2000; Orr *et al.*, 2001). A simulated capture, using the boats and equipment, may help avoid errors during the actual event.

Moving and restraining a captured odontocete

An animal should be lifted using an appropriately designed sling. A compartment syndrome can develop if an animal is restrained without appropriate support. This syndrome is a condition in which increased pressure within a limited space compromises the circulation and function of the tissues within that space (e.g. muscle compressed by an ill-fitting sling), as a result from trauma, prolonged recumbency or physical activity (Matsen, 1975). In order of preference:

(1) the animal should be restrained and moved while fully supported in water;
(2) the animal should be restrained by a sling, stretcher or straps, or intentionally grounded and held or moved while partially supported in water;
(3) in water that is over the handlers' heads, the animal may be placed on floating mats for disentanglement and monitoring;
(4) once on a mat, the animal may be towed to the processing vessel or shoreline;
(5) depending on its weight, the animal should be either carried in a sling or stretcher or restrained and moved while properly supported on a sponge rubber mattress (e.g. placed on the deck of a ship, a pallet, gurney, or sledge − a hoist can lift the animal onto the processing vessel);
(6) if on a beach, the animal can be carefully dragged on a tarp or blanket if the substrate is reasonably smooth.

The animal's comfort can be maintained during tagging and blood drawing by: protecting the eyes and blowhole from direct sunlight, dirt and debris; allowing the flippers to lie in a natural position either tucked or extended; keeping the skin moist; and distributing and minimising pressure to the abdominal and thoracic cavities. Care should be taken to ensure that water does not enter the blowhole, as cetaceans do not have a mechanism to expel water from the lungs. Further, water may carry infectious microorganisms into the lungs. Pressure can be minimised by padding as discussed above, or in the case of intentional grounding, by grounding the animal at the deepest depth that still allows necessary control of the animal. If sand or debris adheres to the corneal surface, the eyes should be rinsed with salt water. If the substrate allows, holes can be dug under the flippers, or if the animal is to be held in a sling or stretcher, holes are cut to allow the flippers to protrude. A pressure sprayer provides an excellent way to keep a cetacean's skin moist throughout all procedures, but buckets and sponges work as well. Moistened towels may also be used. When the air temperature and humidity are high, and if the water temperature is significantly higher than typical for the species, the water being used for moistening should be cooled if possible.

Physical and chemical restraint or sedation

Physical restraints should be made of pliable material and be broad enough to avoid pressure points. Cargo lifting straps (5-10cm width), broad nylon straps (5-10cm width), canvas or nylon slings that are fleece or foam covered, head nets with foam covered rims and tail ropes with garden hose sleeves are some examples of restraints. Tail ropes may also be made of heavy cotton or 3.5cm soft braided nylon. However, if the animal is struggling, nylon rope might create burns unless covered by a protective material such as a hose. Restraints should be simple and convenient to use, and, most importantly, easy and quick to remove when the procedure is completed, or if the animal should be immediately released. Restraints should be checked regularly to ensure they are not too tight. If the restraint squeezes, lifts, bears the weight of the animal or puts even moderate pressure along the length of the animal, then the animal should be checked at regular intervals (e.g. every 5mins) and shifted if possible to redistribute its weight avoiding subsequent pressure sores. Finally, some thought should be given as to how the restraint will fall off the animal if it should swim away with a restraint device still in place.

In general, once a cetacean is properly restrained, it tends to calm down (time frame is variable by species). In some instances sedation may be used to assist in restraint. In very rare instances, chemical immobilisation may be needed, which has been accomplished by careful use of various chemical agents (Joseph and Cornell, 1988; Reidarson *et al.*, 1998). While chemical immobilisation should not be administered to animals that are to be immediately released, they may be useful in cases where the animal is fractious or longer duration procedures are anticipated. Sedatives and other chemical agents are ideally only administered by veterinarians, as several of these agents are controlled substances and response of an animal to them may, at times, require resuscitation or other medical intervention such as administration of reversal agents. The risk of using a sedative, tranquiliser or other chemicals on a potentially stressed odontocete must be weighed against the benefits gained from using such agents to achieve a research objective. Temperature and respiration should be monitored at regular intervals. Adequate positioning of the animal is needed to prevent ischemia or compartment syndrome due to inappropriate weight bearing. Where it is not feasible to have a veterinarian in the field, at least one member of the tagging team should be trained by a veterinarian to estimate proper doses, identify symptoms of overdose, and be prepared to abort tagging efforts if the animal is in unacceptable or life-threatening distress. Precapture training and preparation can reduce the risk of loss of a valuable research animal, decrease the time and effort involved in capture, ensure the release of a healthy animal and improve the quality of the resulting data.

PHYSICAL AND PHYSIOLOGICAL EFFECTS ON THE ODONTOCETE

One of several methods may be used to monitor a cetacean's condition throughout the handling process. A collective group of signs must be monitored since no one sign will necessarily indicate trouble. Once caught, the animal should be observed for a period of time (5-10mins), during which time minimal data gathering may begin, to see if it becomes immediately distressed. When a period of observation is not feasible or significantly prolongs the holding time, one member of the team should be given the responsibility to monitor the animal closely. The individual should record the specific times of respirations and note the strength of breaths and body posture, particularly as the animal is being brought aboard the boat or beached and preliminary measurements are being taken. Depending on the primary objectives of the project, the most critical procedures should be done first (e.g. length measurements, photographs, blood drawing) in the event the animal has to be released back to the water prematurely. An animal in distress may exhibit an arching of the body (e.g. flukes and head bent upward while breath holding) often followed by thrashing, or may have very shallow, erratic respirations (Walker, pers. comm.). If so, it should be splashed with a bucketful of seawater over the melon to stimulate breathing. If there is no response, it should be returned to the water (with the potential for drug administration) and monitored until it returns to normal. It can then be released or evaluated for tagging. Behavioural criteria such as response to stimuli can be assessed as follows.

Assessing stimulus response

An animal should appear to be aware of its surroundings and respond fairly readily to stimuli. Gentle tapping near the eye should elicit a blink (Geraci and Lounsbury, 1993). A blank, unresponsive 'stare' warrants real concern and should prompt immediate action to be taken by the individual monitoring the animal.

Monitoring cardiovascular function

Cardiovascular and respiratory function can be roughly evaluated in the field by monitoring heart and respiratory rates, respectively (Table 1). A stethoscope (for smaller species e.g. harbour or Dall's porpoise) or hand (larger species e.g. bottlenose dolphin and larger) may be placed firmly in the axillary region (where the pectoral flipper joins the body wall; Fig. 1) to detect a heartbeat and determine heart rate (Geraci and Lounsbury, 1993). In small species, one can sometimes see a heart rate 'flutter' externally in the axillary region just behind the pectoral flipper. Heart rate may also be monitored with a heart rate sensor that allows data to be recorded and stored continuously during the handling process. Heart rate can and should vary considerably, even under normal conditions. For example, the heart rate of a bottlenose dolphin increases to a rate of 70-100 beats per minute (bpm) just after inspiration. As the animal continues to hold its breath, heart rate falls to between 30-40 bpm until the next breath (Ridgway, 1972). The rate will remain low regardless of the length of the apneustic plateau. Thus a normal respiratory rate of 2-3 times a minute will be accompanied by an increase, then decrease in heart rate as just described (Ridgway, 1972). If this normal sinus arrhythmia is absent, a pulse that is rapid or weak signals the onset of cardiovascular deterioration (i.e. shock, hyperthermia).

Monitoring of respiratory function

Monitoring of respiratory function should begin as soon as the animal is captured in the net. Respiratory rate for smaller species is usually 2-3 respirations per minute (rpm), but may increase to 6-8 rpm in excited individuals, and 1-2 rpm for larger species (Geraci and Lounsbury, 1993). The researcher should come into the field with the best available knowledge of average and maximum breath intervals for the species. Irregular or increased respiratory rate (i.e. >10 and 6, respectively) can signal respiratory fatigue and distress. In some species, however, respiratory intervals may become prolonged. In bottlenose dolphins, for example, an interval between respirations that extends to >1-1.5min with little respiratory chest movement occurring, is cause for concern. These 'ineffective respirations' may require immediate action on the part of the individual monitoring the animal. Changing the animal's position (e.g. from lateral to sternal recumbency) and splashing water on the melon can improve the respiratory rate and quality. One should also watch for a cetacean keeping its blowhole open and breathing in a rapid, shallow manner that often indicates stress. If this behaviour occurs or the animal's respiration 'shuts down', sometimes a

Table 1

Physical parameters for monitoring condition of cetacean during handling process (adapted from Ridgway, 1972).

Parameter	Normal range	Interpretation of abnormal values
Heart rate (bpm[1])	70-100 (after inspiration) 30-40 (during breath hold)	<30 = hypothermia >100 = hyperthermia or shock
Respiratory rate (rpm[2])	2-3 (resting smaller species)	>8 = distress (respiratory fatigue or hyperthermia)
	6-8 (excited smaller species)	
	1-2 (resting larger species) 3-6 (excited larger species)	>6 = distress (respiratory fatigue or hyperthermia)
Body temperature (°C)	36.5-37 (small to medium species)	<35.6 = cardiovascular collapse[3] >40 = impending hyperthermia

[1] bpm = beats per minute (depends on size of species, as smaller animals may have faster rates);
[2] rpm = respirations per minute (depends on size of species, as smaller animals may have faster rates);
[3] white whales and some larger whales may have lower core body temperatures.

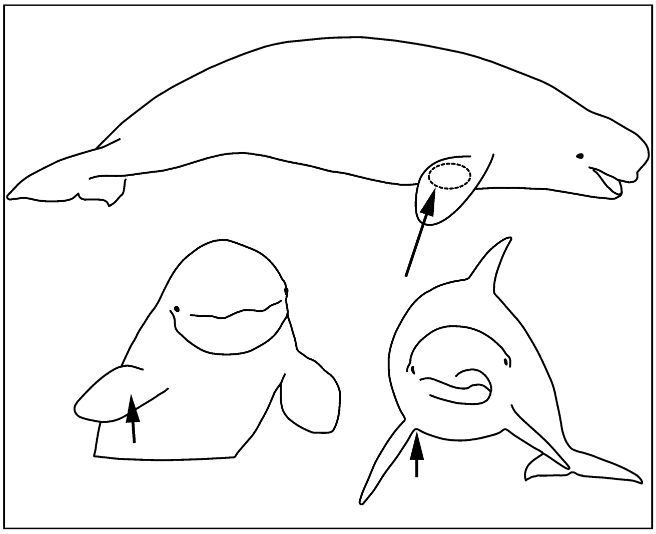

Fig. 1. Head-on and lateral views of odontocetes demonstrating placement of stethoscope or hand on the thoracic wall in the axillary region for heart rate determination (arrows).

light spraying of the head region with water will stimulate resumption of a normal respiratory pattern. Splashing a sponge full of water directly over the blowhole area will often stimulate a respiration when the dolphin is 'breath holding' (i.e. holding its breath longer than a minute). If these methods are not successful, the animal can be placed back in the water, and if it settles down, an attempt to finish the needed procedures can be made in the water. Alternatively, it may be prudent to release the animal without further procedures once the animal resumes normal respiratory patterns.

Body temperature monitoring

Body temperature can be monitored when practical to do so by insertion of a flexible temperature probe into the anal opening (at least 15-25cm or more depending on the animal's size). With the animal properly restrained, another person can reach under the animal and insert the probe. In males, one must avoid placing the probe near the gonads to prevent a false reading of hypothermia due to the presence of a vascular plexus that cools the testes (Rommel *et al.*, 1994). If insertion of a probe is not practical, the pectoral flippers, flukes and dorsal fin may be felt frequently to help assess changes in body temperature. This is the most commonly used method to monitor body temperature in a field setting. In small to medium-sized species, the normal body temperature range is 36.5°-37°C. Temperatures below 35.6°C signal the onset of hypothermia or cardiovascular shock in some species (Geraci and Lounsbury, 1993).

Hyperthermia is less likely to occur during capture and handling of cetaceans if proper procedures are followed and the team is vigilant in monitoring the status of the animal. Although hyperthermia may damage all organ systems, the nervous and reproductive systems are the most sensitive (Fowler, 1996). This is less likely to happen in the polar latitudes. Clinical signs of hyperthermia observable in the field include a shallow, rapid respiratory rate, increased heart rate, increased body temperature (>40°C) and decreased blood pressure. The core body temperature is one of the most sensitive and accurate measures of an animal's change above its thermal neutral-zone into hyperthermia. If the animal's temperature rises more than 1.5-2°C, cold water or crushed ice should be applied to the flukes, flippers and dorsal fin. The ability to monitor blood pressure may be impractical or limited in a field setting. Preventative measures are usually the best approach, such as providing shade and keeping the animal cool, particularly if longer procedures are anticipated. It is also advisable to keep handling procedures as brief as possible and, if practical, try to schedule captures in the early morning when the ambient temperature is cooler.

A member of the technical support team should be trained to record respiration times – the time will allow the recorder to inform appropriate staff when an inordinately long time has passed since the last breath – and heart rate, although the latter may be difficult to monitor depending on the species and blubber thickness. Ideally, vital signs for an animal should be recorded continuously over the entire span of the event (e.g. every 5-10mins during each hour, or preferably, continuously) with directions to notify the veterinarian of any changes in that animal's baseline rates. Subtle changes may be difficult to recognise, so it may be necessary to rely more on trends over time or other indicators (e.g. behaviour, posture, alertness, reflexes, analysis of blood sample in the field if practical) to ensure the animal's status is not deteriorating.

STRESS RESPONSE MONITORING

A stress response brought about by chase and capture has been shown to trigger changes in the hematological and plasma chemical constituents of some cetaceans (St. Aubin and Geraci, 1988; 1989). These physiological imbalances may impair an animal's immune system, rendering them more susceptible to pathogenic organisms that otherwise might not pose a threat. Changes in blood constituents were noted in two white whales caught for tagging and then recaptured several days (19 and 24) later for removal of data loggers (St Aubin *et al.*, 2001). Both animals showed evidence of white blood cell responses consistent with inflammation and stress. Since the interval between chase and collection was the same for the two whales, the observed hematological changes were assumed to be due to tissue damage and repair from satellite tag application. The response of blood constituents to handling and tagging operations suggests that these procedures represented an immune system challenge under these conditions.

Behavioural assessments may be used in the field to recognise acute stress. Anxiety is one of the most common manifestations presented by animals under stress, although passivity may also be a sign as well as increased respirations (Hanson *et al.*, 1998). It is prudent to try to minimise the chase duration and expedite handling procedures to prevent acute intense or prolonged stress. Although a suspected case of capture myopathy was reported by Colgrove (1978), it is rarely encountered in cetaceans (Schroeder *et al.*, 1985a).

Individuals resighted or recaptured after several months or years appear to be completely healed (Orr *et al.*, 1998) suggesting that for at least some individuals, handling activities do not severely compromise survivability. However, no direct comprehensive, controlled studies on cetaceans have been conducted to determine if survivability has been compromised to some extent. Long-term survivability and reproduction of bottlenose dolphins studied in Sarasota Bay, Florida, for more than 30 years does not appear to have been compromised due to capture-release techniques utilised in that project, given that more than 40% of the dolphins first tagged in 1970-1971 were still observed more than 30 years later, and the population size of the resident dolphin community has increased significantly during this period (Wells, 1991; 2003). Furthermore, health assessment and monitoring of four generations of resident bottlenose dolphins have not found any complications from capture-release. Blood sampling of individuals before (e.g. as soon after capture as practical) and several days after tagging would allow for evaluation of the physiological impact of these activities on stress indicators in the blood. This would require more sample handling but is of great research value. Some other cetacean species, however, may not be as resilient or as easily handled as bottlenose dolphins.

Treating shock and allergic reactions

The use of medications to treat shock, potential infection or allergic reactions is rarely needed. They may be used under conditions where quick release is not possible and there will be an extended period of time before the animal can be returned to the environment. Corticosteroids may be used to treat shock during which time an animal's condition may be declining rapidly (i.e. body temperature < 35°C, respiratory rate > 8bpm). Epinephrine can be used for adverse or allergic reactions to other medications administered such as antibiotics. Signs of an allergic reaction may include: agitation, increased heart rate, difficulty in obtaining blood samples due to circulatory collapse, possible swelling at an injection site or of soft tissues of the head. Once treated, the tagging team should support the animal until it is stable (e.g. respiratory rate and body temperature have normalised), after which tagging procedures should be aborted and the animal released. If it is not feasible to have a veterinarian available during tagging procedures, it may be logistically difficult or impossible to obtain timely veterinary support in an emergency situation at a remote tagging site.

POSSIBLE COMPLICATIONS FROM TAG PLACEMENT

Tag attachment systems range in impact from suction cups, to surgical implantation of attachment pins or the entire tag, to attachment of the tag by barbed spear. Except for the suction cup, each of these involves piercing of the skin and blubber and possibly other structures as well. Health considerations include maintenance of aseptic conditions at the tag placement site during deployment, preventing introduction of microorganisms into the pin tract, preventing pressure necrosis by the tag and its pins, minimising tissue reaction to the tag materials with subsequent tag rejection and promoting wound healing (see Irvine *et al.*, 1982; Scott *et al.*, 1990; Wells, 2002).

Dermatological effects of tag attachment

Several studies have characterised the rate of cutaneous wound healing in bottlenose dolphins (Brown *et al.*, 1983; Bruce-Allen and Geraci, 1985; Geraci and Smith, 1990) and white whales (Geraci and Bruce-Allen, 1987; Geraci and Smith, 1990). Cutaneous wound healing was found to take place at a quicker rate in bottlenose dolphins than in white whales. Geraci and Smith (1990) stated that two principal factors decrease the life of an implant: infection and movement in the tissue. Infection may be minimised by including a slow-release, broad-spectrum antibiotic in the implant and preventing contamination while embedding the implant. Movement of a tag or attachment pin or spear within a tissue can be decreased by careful engineering of the attachment to distribute pressure and minimise jerking due to variation in hydrodynamic drag such as during exit and re-entry of the tag during a breathing cycle. With an implanted tag, this can be accomplished by embedding the head of the tag deeply and with a sufficient number of anchors to stabilise the implant within the tissue. Much is still unknown about the effects of epidermal thickness, ambient temperature, salinity and stress on cutaneous wound healing. These factors could have implications with regard to healing of wounds caused by placement of transmitter devices.

Steps to help minimise implant rejection during attachments using surgically implanted pins

A suggested procedure for skin preparation, bolt placement and tissue boring is described below. This may not be feasible in all cases, but before a researcher chooses to simplify the suggested procedures, careful thought should be given to weighing the importance of speed and convenience of the tagging process against the increased risk to the animal. As noted earlier, an animal compromised during handling may not behave in a manner representative of the population, thus providing data that are possibly misleading and wasting a research opportunity.

Before placing a tag, a site must be chosen that is reasonably devoid of blood vessels so there is minimal impact to heat-exchanger vessels and the general blood supply of the area (Lander *et al.*, 2001). To accomplish this, the dorsal fin (or ridge, if working with a finless species such as white whales) of a dead specimen should be examined. It is recommended to use a sterile hypodermic needle to probe anticipated pin sites to determine if major blood vessels, particularly arteries, are present before actual pin placement on a live animal (Chilvers *et al.*, 2001; Lander *et al.*, 2001).

Prior to any puncture or pin placement, the skin surface should be cleaned properly with an antiseptic such as isopropyl alcohol or 10% Povidone-Iodine solution (Poviderm Solution, Vetus c/o Burns Veterinary Supply, Inc., Dallas, Texas)[1] (Westgate *et al.*, 1998), then the area should be surgically scrubbed twice. This technique involves starting at the point of pin placement and, using small circular scrubbing movements, working circumferentially to the periphery, being careful not to touch the centre of the scrubbed area with the same gauze sponge once it has reached the outer limits of the sterile field. This will prevent contamination of the sterile field's centre with organisms from the periphery. Upon completion of the first scrub, a second should be completed in the same manner. In a field situation, a researcher may find it challenging to maintain sterility of the pin placement site while working with a live animal in a less than ideal environmental setting (e.g. seawater washing over the sides of the boat or deck).

The area of pin placement is then locally anesthetised with lidocaine HCl 2% with epinephrine 1:100,000 using a 21 or 22-gauge needle. Some individuals have found this to be more effectively accomplished using an injector needle gun (Miltex Inc., Bethpage, New York[1]). In larger species such as the killer whale, difficulty may be encountered trying to inject directly into the tough dermis. Alternatively, the anesthetic agent may be injected into the base of the dorsal fin using a 2-inch 19-gauge needle so that the anesthetic may be drawn up the fin from the injection site. Following placement of the local anesthetic, another surgical scrub of the area is performed following the same technique as for the first two. All equipment (tags, pins, bolts) is cleaned with

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[1] Use of trade names does not imply endorsement by the National Marine Fisheries Service.

isopropyl alcohol prior to utilisation. Holes for pins are typically established using specialised hole cutters, similar to a laboratory cork borer, that have been cleaned and disinfected prior to each use. The borehole should be made slightly smaller in diameter than the pin to be used to reduce bleeding and loosening at the tag site and to prevent disruption of the healing process (Hanson, 2001). If nuts are used on the ends of the pins to secure the transmitter package, one must be cautious not to overtighten the nuts as compression and ischemic necrosis may occur at the attachment site. A small amount of triple antibiotic ointment (i.e. polymixin B-bacitracin-neomycin) may be applied to the tag entry point. Injections of antibiotics are not routinely given after tag placement unless the animal has sustained numerous abrasions/lacerations during capture.

Behavioural effects of tag placement

There are reports of telemetry devices leading to changes in behaviour of an animal subject through increased drag or discomfort of the attached instrument package (Irvine *et al.*, 1982; Tanaka, 1987; Tanaka *et al.*, 1987; Scott *et al.*, 1990). Increases in drag from 12-15% to as much as 27% are reported for a dorsal fin mount tag on a harbour porpoise model (Hanson, 1998). Würsig (1982) found that radio-tagged dusky dolphins (*Lagenorhynchus obscurus*) were not bothered by the transmitters, but seemed to swim slower than normal after two or more days post-tagging. Foraging and attendance behaviour of female Antarctic fur seals (*Arctocephalus gazella*) was altered with instrumented animals, but the biological significance and long-term effects of telemetry devices is unclear (Walker and Boveng, 1995; Boyd *et al.*, 1997). Dorsally attached transmitters did not seem to affect behaviour in white whales (Richard *et al.*, 1997).

POTENTIAL FOR DISEASE TRANSMISSION

Although there are few reports of disease transmission between marine mammals and humans (Geraci and Ridgway, 1991), one must always be aware of the possibility of being exposed to new diseases (e.g. Buck and Schroeder, 1990; Tryland, 2000). Many bacteria are shared by humans and cetaceans, and some cause disease in both, however, there is not an excessive health risk to humans from association with cetaceans. Early studies (Johnston and Fung, 1969) suggested humans could be a source of infection to cetaceans in a captive environment, but it is not certain if the same risk exists during handling of wild animals. Several species of bacteria have been recovered from stranded cetaceans that have been associated with a variety of infections in humans (Buck, 1984). Similarly, marine mammals harbour microflora that are commensal and usually pose no health threat under normal circumstances. Under conditions of stress the animal may be debilitated or immunosuppressed and be predisposed to infection by these organisms or others encountered in the environment. Exposed mucous membranes and cut skin surfaces are especially prone to potential pathogens. The same may apply to humans handling the animals.

Most bacteria associated with marine mammals are not a public health concern. More thorough discussions of potentially zoonotic diseases between humans and marine mammals are available in Geraci and Ridgway (1991), Higgins (2000) and Cowan *et al.* (2001), however, a few warrant special mention here since they are more commonly recognised pathogens of humans (Schroeder *et al.*, 1985b; Suer *et al.*, 1988; Palmer *et al.*, 1991). These organisms could be infectious for persons with compromised immune function or could be inoculated into cuts, bites or abrasions. Infections with *Brucella* sp. have been reported in cetaceans (Ross *et al.*, 1996; Miller *et al.*, 1999). Although there have only been two published reports of a marine *Brucella* isolate infecting a human (Brew *et al.*, 1999; Sohn *et al.*, 2003), one should recognise its zoonotic potential. *Mycobacterium marinum* has been transmitted to a human via a dolphin bite on a finger and thus should be considered zoonotic (Flowers, 1970). Blastomycosis infection of a veterinarian's hand followed examination of an infected bottlenose dolphin (Cates *et al.*, 1986). *Erysipelothrix rhusiopathiae* is a pathogen that causes cutaneous infarcts or generalised septicemia in many species of cetaceans and contributes to pain, swelling and/or more generalised illness in humans (Medway, 1980). This pathogen was once thought to be the cause of seal finger, but subsequent studies have implicated a mycoplasma (Stadtlander and Madoff, 1994). Marine morbilliviruses have not shown to be infectious to humans (Cowan *et al.*, 2001).

Steps to minimise exposure

Personnel handling cetaceans must exercise caution to diminish exposure to potentially hazardous microorganisms. Care should be exercised when working around the head region of cetaceans. One should avoid being in the path of the blowhole exhalation due to the risk of exposure to microorganisms, particularly those that are known to be human pathogens. Cases of such transmission have not been demonstrated, nevertheless, the potential still warrants mention. Surgical masks may be worn if handlers will potentially be exposed directly in the face to a cetacean's breath or bodily fluids. However, this is often not a realistic option in the field setting especially during activities such as capture and release. Exposed mucous membranes and cut or abraded skin surfaces are especially prone to potential pathogens. Gloves and other gear such as strong, heavy footwear should be worn to protect against abrasions, cuts and bites. Personnel should avoid touching their eyes and face during handling to minimise transmission of an organism to mucous membranes. Handlers should thoroughly wash their hands in disinfectant between and after handling animals. Immunocompromised or pregnant field staff should avoid direct exposure to cetaceans.

In the very rare instances of infection acquired from a cetacean, the infection may begin subtly following an encounter with the animal. If an infectious condition is suspected, the handler should present the history of contact with a marine mammal to the physician, as the clinician may not think to ask about such an exposure.

CONCLUSION

Anecdotal evidence suggests that most capturing and tagging of odontocetes occurs without incident to either the animal or the handler. Handlers must be aware, however, of the potential health risks to both cetaceans and humans. These risks will be reduced by including knowledgeable, experienced personnel in all aspects of the project, keeping handling times to the minimum necessary to safely complete the objective, careful planning which includes contingencies for potential problems, having adequate personnel and equipment on hand to meet the contingencies, as well as maintaining vigilance when monitoring the animal's condition throughout the handling process. It is important to use caution when working around the animal in order to avoid cuts, abrasions and other wounds that might facilitate the transfer of a potential pathogen between cetaceans and humans. Care in handling procedures will result in fewer lost animals and more reliable data.

ACKNOWLEDGEMENTS

The authors would like to thank Brad Hanson, Tony Martin, Michael Scott, Bill Walker and Randy Wells for their insights and helpful comments on the manuscript. Karna McKinney of the Alaska Fisheries Science Center Graphics Unit kindly prepared the figures. Reviews of this document were made by Brian Fadely and Kim Shelden, both biologists at the National Marine Mammal Laboratory (NMML) of the Alaska Fisheries Science Center, NMFS, NOAA. Doug DeMaster, Jim Lee and Gary Duker provided valuable reviews as did Stephen Raverty, Ministry of Agriculture, Food and Fisheries, Abbotsford, British Columbia, Canada. Finally, the authors would like to thank Sam Ridgway, an anonymous reviewer and Greg Donovan for very constructive comments and suggestions. This report was funded by the National Marine Fisheries Service, Contract # 40-AB-NF-112103.

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Behavioural responses of humpback whales (*Megaptera novaeangliae*) to whalewatching boats near Isla de la Plata, Machalilla National Park, Ecuador

MEIKE SCHEIDAT[*], CRISTINA CASTRO[+], JANIRA GONZALEZ[+] AND ROB WILLIAMS[#]

Contact e-mail: scheidat@ftz-west.uni-kiel.de

ABSTRACT

Machalilla National Park, on the coast of mainland Ecuador, supports a growing whalewatching industry that focuses on Southern Hemisphere humpback whales, which spend the austral winter (June-September) in this area. This study was designed to measure short-term reactions of whales to the whalewatching vessel activity typically seen in this area for two reasons: (1) to identify the nature of whales' avoidance response, if any, in order to draft whalewatching guidelines that help local mariners identify when they may be disturbing whales; and (2) to quantify the magnitude of any avoidance response, to examine how this relatively understudied population behaves around boats compared with whales in other whalewatching areas. A shore-based theodolite tracking team created a 'natural' experiment to observe relationships between whalewatching traffic and whale behaviour in 1998 and 1999. Swim speed and path directness of humpback whales were measured in the absence of boats, and how those parameters changed when boats arrived was recorded. When whales entered the study area accompanied by boats, a record was made of how their behaviour changed after the boats left. Humpback whales reacted to the approach of whalewatching boats by increasing swim speed significantly, and adopted a much more direct path after boats left. Future research is needed to determine whether responses vary with number, proximity or type of vessel. Similarly, future studies are recommended to determine whether different age-sex classes vary in vulnerability to disturbance. Meanwhile, this study enables provision of much-needed, practical advice to local operators who are concerned that they may be disturbing whales: one way that mariners can tell if they are causing disturbance is if they need to increase their vessel's speed to keep pace. The average behavioural responses measured were strong enough to recommend that Machalilla National Park adopt precautionary management procedures to limit number and proximity of vessels.

KEYWORDS: HUMPBACK WHALE, SOUTH AMERICA, WHALEWATCHING, BEHAVIOUR, SHORT-TERM CHANGE

INTRODUCTION

Researchers first observed humpback whales (*Megaptera novaeangliae*) in Machalilla National Park, Ecuador (Fig. 1) in the late 1980s (Félix and Haase, 2001). These animals are thought to be contiguous with a larger Southern Hemisphere humpback whale stock, and typically spend June to September in the area to calve and mate (Scheidat *et al.*, 2000). A preliminary abundance estimate for this breeding population, based on capture-recapture statistics from an ongoing photo-identification study, is 400 animals (Scheidat *et al.*, 2000). For a number of logistical and other reasons, little information is available on humpback whales from Central and South America.

The tourism industry in Ecuador recognised the economic value of these whales only five years ago. Whalewatching is a particularly lucrative industry in many parts of the world, and is often cited as an economic alternative to whaling (e.g. Hoyt, 1995). However, it has been recognised for many years that harassment by vessels can have both short- and long-term effects on humpback whales (e.g. Norris and Reeves, 1978). Short-term effects have the advantage of being easily demonstrated in terms of avoidance and aggressive behaviours, although whether long-term effects occur is more significant at the population level. Absence of proper controls makes it more difficult to create causal linkages between long-term human activity and changes in abundance and distribution of animals.

Repeated disturbance of critical behaviours such as feeding, resting and mating can reduce the biological fitness of the population. While on the mating and calving grounds, humpback whales rely on blubber reserves obtained during the feeding season, and therefore may be exceedingly vulnerable to energetic costs of repeated disturbance. Young calves are especially dependent on sufficient time with their mothers to suckle and rest. For them, any disruption carries energetic costs. Studies that quantify the nature and extent of short-term behavioural responses to human disturbance can be useful for alerting researchers to potential population-level effects while they are still reversible. Monitoring the extent of disturbance in breeding areas is especially important.

Despite the relatively recent commercialisation of whalewatching off Ecuador, interest in humpback whales as a tourist attraction has increased dramatically. The waters around Isla de la Plata are becoming known as a good destination for seeing humpback whales, although tourist activity in the waters around Isla de la Plata is not restricted to whalewatching alone. Vessels generally leave Puerto Lopez (Fig. 1) between 0800 and 1000 and arrive in the waters around the island one to two hours later. Tourists may spend several hours on the island and most vessels leave again between 1400 and 1600 to return to the mainland. Vessels typically stop to observe opportunistically whenever sightings of humpback whales are made, rather than searching for whales.

Park managers and conservationists are now concerned that any harassment of whales by whalewatchers could disrupt their reproductive and social activities and, ultimately, displace the animals from the area. However, all realise that whalewatching has become an important part of the local economy. The need to protect the humpback whales

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[*] *FTZ Westküste Büsum, Hafentörn, 25761 Büsum, Germany.*
[+] *Casilla 17-21-872, Quito, Ecuador.*
[#] *Sea Mammal Research Unit, Gatty Marine Laboratory, St Andrews, Fife KY16 8LB, Scotland.*

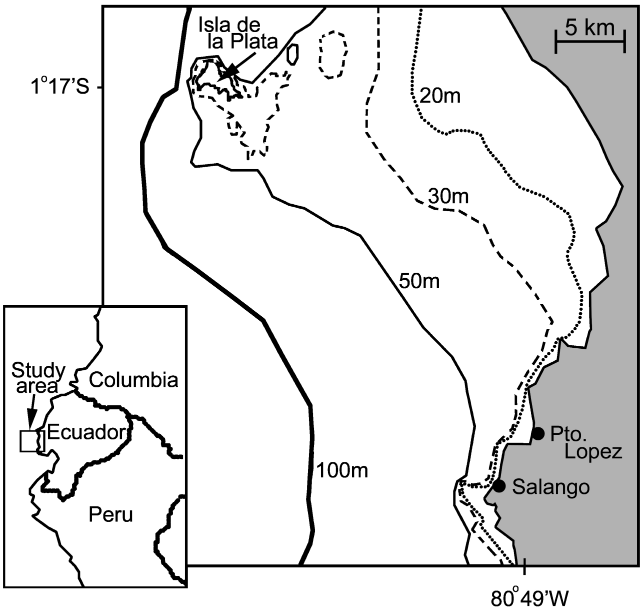

Fig. 1. Map of study area

has become a necessity, not only from the whales' perspective, but also to provide an alternative and potentially sustainable income to a coastal community in a developing country.

It is difficult to determine reactions of cetaceans towards vessels. For example, if using a vessel then the observation platform may also be influencing the effect to be measured. In a few locations, land-based studies are possible. This approach provides an independent platform that has no influence on the whales' behaviour. It also allows the use of theodolites to measure speed or features of the whale's path in the absence of boat traffic (Würsig *et al.*, 1991).

Isla de la Plata was used as such a platform to begin to examine whether the activity of local whalewatching vessels is altering certain aspects of the behaviour of humpback whales. A brief pilot study in 1997 located a cliff on the eastern point of Isla de la Plata as a reliable place from which to observe whales and whalewatching vessels. From here, groups of whales were observed before and/or after encounters with whalewatching boats. The study was designed to measure certain short-term reactions of humpback whales to typical vessel activity in this area. The two aims were: (1) to identify the nature of any avoidance response in order to draft whalewatching guidelines to help local mariners identify when they may be disturbing whales; and (2) to quantify the magnitude of any avoidance response, to see how this relatively understudied population behaves around boats compared with whales in other whalewatching areas.

METHODS

The behaviour of humpback whales and the activity of whalewatching boats were observed from Isla de la Plata between 7 July and 28 August 1998, and between 27 July and 27 September 1999. The 90m height of the observation point, Escalera, allowed long-range observation of several groups, and enabled the team to monitor groups long enough to obtain pre-, during- and post-exposure observations. The height of the observation point was obtained using a detailed contour map of the island provided by Fundación Natura, Ecuador. The focal plane of the theodolite was established as 91.5m above mean sea level (including the height of the theodolite and tripod).

As whales or groups of whales entered the study area, the centroid of the group was tracked using a WILD theodolite (with automatic vertical index) mounted on a tripod, using the methods described in Würsig *et al.* (1991). An example of a typical tracking is shown in Fig. 2. Whales were tracked from the moment they entered to the moment they exited the field of view. A group of whales was defined as animals that were swimming within three body lengths of one another. Vertical and horizontal angles at each theodolite reading were measured to the nearest 20 seconds of arc. Time was recorded to the nearest second. Group activity, group size and group composition (number of calves present) were noted at the beginning and end of each observation session and if these parameters changed. The horizontal distance from the observation point at sea level to the whales was calculated using the trigonometric relationships between the vertical and horizontal angles of sightings and the known height of the theodolite (Davis *et al.*, 1981). Approximate fixes (for example, those made on the 'footprint' left by the whale) were omitted from all calculations. Changes in height of the water level were ignored, but are negligible (< 1%) due to the height of the cliff and the small tidal movements in the study area. In many studies, this is key, since percent errors in cliff height and swim speed tend to be approximately equal (Würsig *et al.*, 1991).

Two candidate response variables were calculated. The mean swim speed of the group was averaged across a tracking session using the distance between two points and the time taken to cover this distance. The directness index of a group of whales was calculated dividing the 'crow's flight' distance (between the first and the last position of a tracking session) by the cumulative surface distance covered between all recorded positions (Fig. 3). This index is equivalent to the milling index of Tyack (1982). Its value ranges from 1 (when animals move in a straight line) to 0 (when animals swim in a circle, that is, end up in the starting position).

Tracking sessions were targeted when humpbacks entered the study area unaccompanied by whalewatching boats. When a boat or boats approached focal animals, the position of each boat was recorded at least twice during a theodolite tracking session. The maximum number of boats ever observed near whales was nine (Scheidat, unpublished data). However, during the natural experiments, the number of boats accompanying the whales was either one or two. The whales continued to be tracked when associated boats left the focal animals, in order to obtain a sample of whale behaviour under pre-, during, and post-exposure conditions. Subsequent analyses were restricted to interactions when observation time with and without vessels were each 20 minutes long, and when at least five whale positions were recorded under each traffic condition.

RESULTS

Tracking humpbacks and whalewatching vessels

In 1998 and 1999, a total of 73 opportunistic observations were made under a variety of traffic conditions. On 27 of those occasions, natural experiments occurred that enabled comparison of pre- to during-exposure ($n = 12$) or during- to post-exposure ($n = 15$) behaviour. These pairing categories are mutually exclusive, such that a during-vessel behavioural observation is compared with either pre-vessel or post-vessel behaviour, but not both. Consequently, the pre- to during-treatment samples are statistically independent from

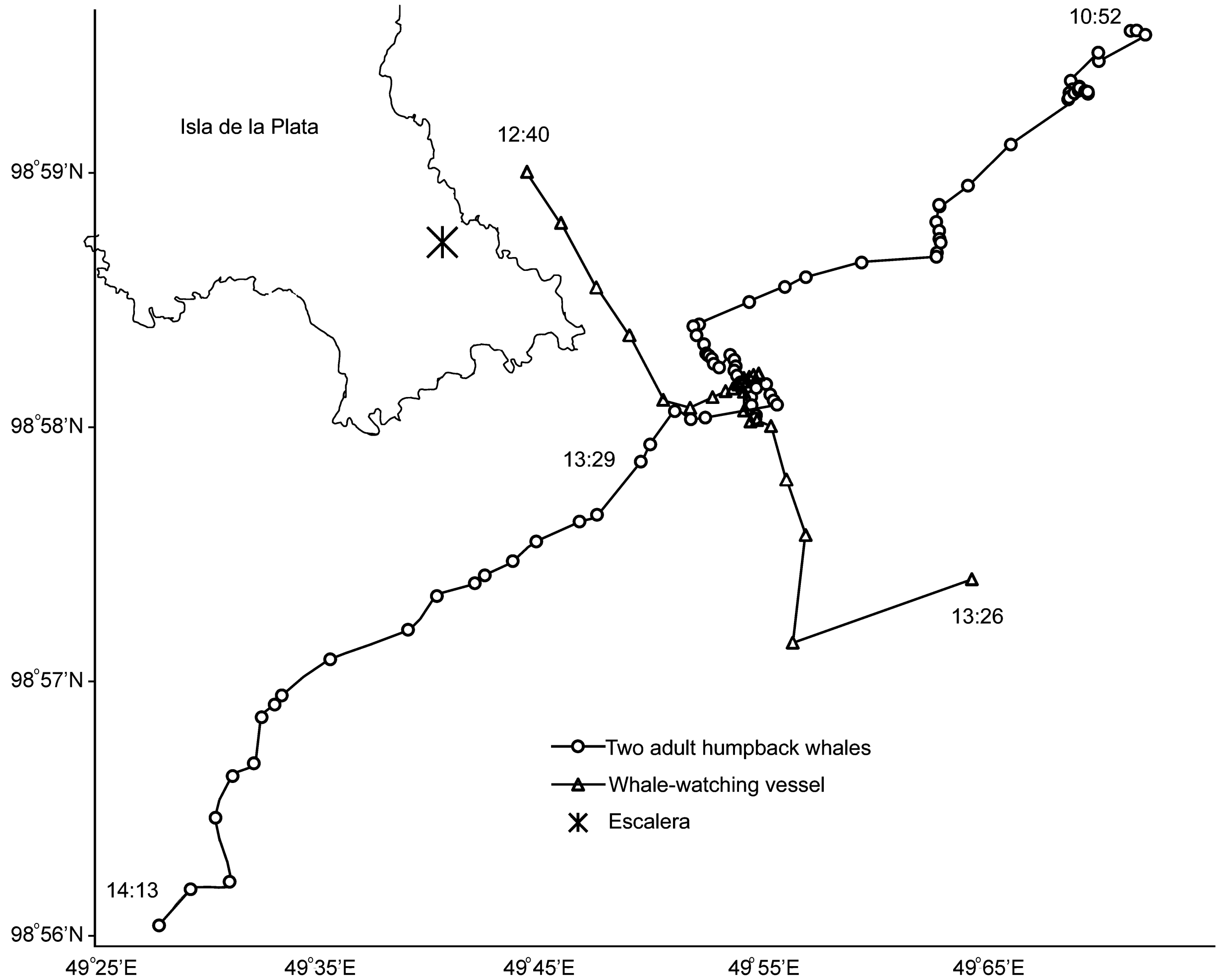

Fig. 2. Example of a typical theodolite tracking from the Isla de la Plata. The observations were made from the observation point Escalera at a focal height of 91.5m. The tracking took place on the 25 August 1999. A group of two adult whales was first noted at 10:52 moving slowly in a southwesterly direction. They were joined by a whalewatching vessel at 12:54 that stayed with the group until 13:18. The humpbacks showed milling and resting behaviour until about 13:28 when they started to travel. The map is presented using a UTM (Universal Transverse Mercator) projection for zone 17 (78°W to 84°W). The x-Axis represents the distance in metres from the central meridian of zone 17 (81°W) and the y-Axis represents the distance in metres from the South Pole.

the during- to post-treatment samples: each group of whales serves as its own control, and a day's tracking session of a group yields only one pair of observations.

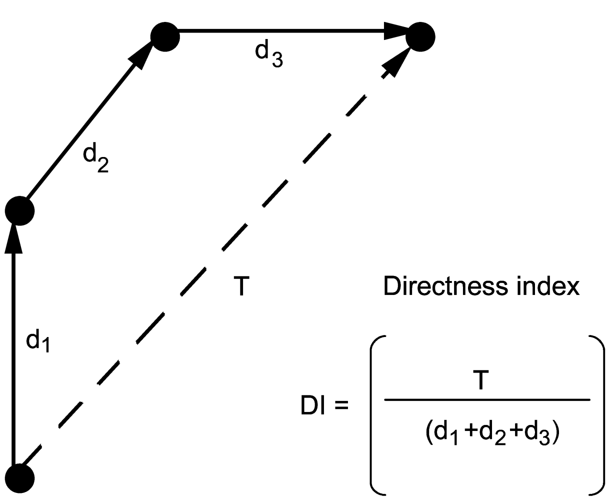

Fig. 3. A sample swimming path with four surfacings (●) and three dives (d_i), showing a measure of path predictability called the directness index. The directness index is the ratio of the track diameter (T) to its perimeter.

Pre-, during and post-treatment observations were each 20 minutes long with at least five theodolite positions recorded. The remaining, opportunistic observations varied widely in terms of track length and number of positions recorded. Consequently, the analyses presented here are restricted to those observations of natural experiments, where local whalewatching traffic conditions form the treatment.

Changes in swim speed and path directness

Histograms of speed and directness index revealed some evidence of positive skew. Rather than performing Kolmogorov-Smirnov tests, which often fail to detect true deviations from the normal distribution in small samples (Zar, 1998), non-parametric tests were performed. The Wilcoxon matched-pairs test (the non-parametric equivalent of the paired t-test) was chosen as the most conservative way to analyse these data while retaining sufficient power to detect a true effect (Stewart-Oaten, 1995).

On 12 occasions, whale behaviour was recorded in the absence of vessel traffic and during subsequent exposure to either one ($n = 6$) or two ($n = 6$) whalewatching boats. In these cases, mean speed of humpback whale groups increased significantly from 2.97km h^{-1} to 4.52km h^{-1} during the vessel interaction (Wilcoxon test for paired data; $Z = 2.04$, $p = 0.041$; Fig. 4). No significant change in directness index was observed ($Z = 0.94$, $p = 0.346$; Fig. 4).

On 15 additional occasions, humpback whales entered the study area already accompanied by one ($n = 11$) or two ($n = 4$) whalewatching vessels. In these cases, whale behaviour was recorded in the presence of the whalewatching traffic as well as after the vessel(s) left. Whale behaviour was compared during the 20 minutes immediately before the boat left, to the first 20 minutes of behaviour recorded post-treatment. Speed did not decline significantly after the vessel left (Wilcoxon-test for paired data; Z = 1.70, $p = 0.088$). Whales' paths, however, became significantly more direct (from a mean directness index of 0.59 during interactions to a mean of 0.76 after the whalewatching boat left, Z = 2.22, $p = 0.027$; Fig. 4).

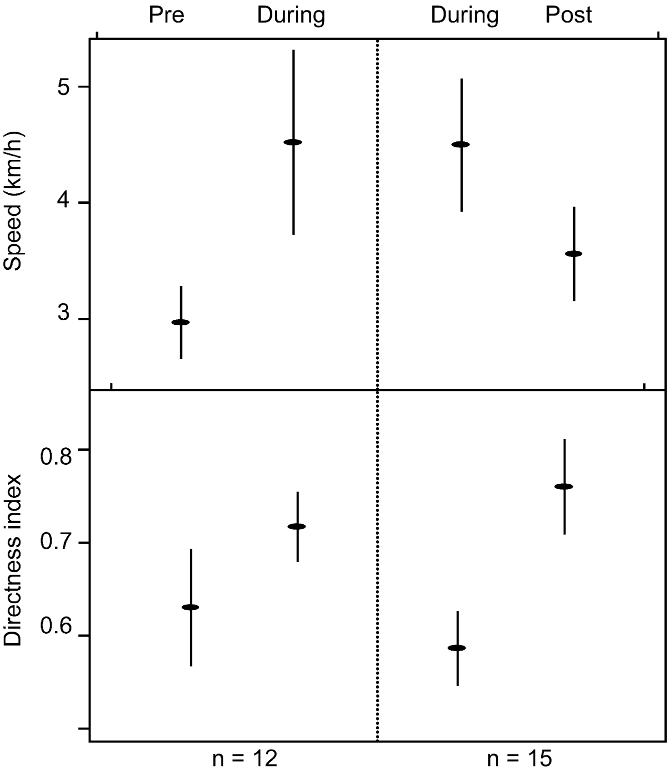

Fig. 4. Behavioural responses (mean +/- SE) of humpback whales after the approach (left) or departure (right) of whale-watching traffic.

DISCUSSION

The study successfully addressed the goal of identifying the nature and magnitude of certain short-term behavioural responses of humpback whales to whalewatching boats near Isla de la Plata, Machalilla National Park, Ecuador. The data show that these humpback whales increased swim speed when approached by local whalewatching boats. Au and Green (2001) and Bauer *et al.* (1986) have reported that Hawaiian humpbacks also responded to the presence of boats with a stereotyped response of increasing swim speed. Weak evidence ($p = 0.088$) was found that swim speed decreased in the 20mins of observation after boats left. Whilst this is not significant at the $p = 0.05$ level, it is suggestive of the prediction that the effect is short lived. However, additional experiments are required to confirm or deny this.

Similarly, while our results show that the whales adopted a more predictable path after the boats left, this may not be evidence of immediate recovery after short-term disturbance but may reflect a statistical artefact of our [namely, visual] versus the whale's [ostensibly acoustic] perception of the node dividing no-boat from boat traffic conditions. Without a causal linkage between decreasing path directness and boats approaching, it is difficult to account for the adoption of a straighter path after the boat leaves. Further experiments are required that *inter alia* record the position of a greater number of surfacings, to demonstrate a convincing causal relationship between the presence of vessels and the directness of whales' paths.

A further limitation of the present study is that it was limited to cases with only one or two vessels approaching the focal group. A similar study on killer whales in Canada, suggested that whales may perceive one boat differently than many (Williams *et al.*, 2002). In reality many boats may arrive at the same time in our study area. Further experiments are required to test the effects of a large number of boats 'crowding' whales.

Clearly, cetaceans display a wide range of reactions to human activity. For example, they may approach a vessel, move away from a vessel or apparently not react at all. Cetacean reactions will not all be visible to a human observer, as changes in behaviour or swimming speed are; nor are they necessarily problematic in their own right. However, long before whales show responses that are obvious at the surface, they are likely to react at a physiological level. Despite this, for practical reasons, when investigating reactions of whales to humans, as in the present study, we usually rely on behaviour that is noticeable and measurable, treating this an indicator of potentially important physiological changes.

Reactions of humpback whales to various types of vessels vary considerably among populations, locations and time of year. Watkins *et al.* (1981) reported that passage of a tanker within 800m did not disrupt feeding animals; humpback whales generally seem less likely to react when actively feeding compared to resting or when engaged in other activities (Krieger and Wing, 1984; Krieger and Wing, 1986). In contrast, in a study of the effects of vessel noise on humpback whales summering in Alaska, Baker and Herman (1989) demonstrated a number of significant responses including increases in dive durations and orientation away from the path of moving boats, often at ranges of up to 3-4km.

Bauer (1986) and Bauer and Herman (1986) reported short-term reactions of breeding humpback whales to vessels in Hawaiian waters. Results differed among age-sex class, depending whether the sighting consisted of singers, other lone animals, mothers or calves. In general, Hawaiian humpback whales attempted to avoid vessels by making longer dives, swimming away from the path of the vessel, and sometimes by demonstrating agonistic behaviours. Some agonistic behaviours were observed in the current study, such as charging or tail slapping, between whales as well as towards vessels. These results are not presented however because it is difficult to ascertain whether rates of surface-active behaviour were collected in an unbiased way (for example that independent events were assigned to a given individual, that bouts of surface activity were recorded as a single event, and/or that events recorded under high-traffic conditions were as likely as when whales were unaccompanied by boats).

Some data on long-term changes in behaviour or habitat-use by humpback whales appear contradictory. On the summer feeding grounds off Cape Cod, humpback whales remain for extended periods and return annually despite exposure to many ships, fishing vessels and whalewatching boats (e.g. Beach and Weinrich, 1989;

Clapham *et al.*, 1993). However, there is some indication that humpback whales do change habitat use in response to human disturbance. For example, Herman (1979) suggested that humpback whale density may be inversely related to the daily amount of boat traffic and to the local amount of human activity in Hawaii.

Despite this, there is evidence that the Hawaiian population is increasing, suggesting that any long-term negative effects are not apparent at the level of the population (Bauer *et al.*, 1993).

The annual return of humpback whales to feeding and breeding grounds *per se* is unconvincing evidence that whalewatching traffic is not disruptive, since strong residency patterns can be found with both weak and strong levels of disturbance; even highly localised whaling activities often failed to disrupt conservative migratory traditions (e.g. Chittleborough, 1965). While strong site fidelity may be interpreted as evidence that animals are fairly tolerant of human disturbance and will probably not change their habitat due to vessel presence, it may equally indicate the extreme importance of some areas to the biology of the whales.

One of the challenges in studying behaviour is to take into account individual variation when arriving at general conclusions. The present study attempted to address this by targeting observations of a wide range of subjects: lone animals, mother-calf pairs and groups of up to six adults. To the best of our knowledge, no group is represented more than once in the analyses. Ideally, photo-identification studies should be undertaken to allow focal animals to be identified. Unfortunately, limited resources prevented this. However, we recognise the need for this to occur in future studies, although care must be taken to ensure that any disturbance associated with taking photographs does not confound the results. Knowledge of individuals also allows more targeted experiments to be carried out. Combined studies can have both practical and cost benefits, the latter being particularly important in a developing country. Finally, experimental approaches to a variety of individuals and groups are required to confirm whether the 85 whales observed in this study behaved in a way that typifies the population of approximately 400 animals.

The fact that Machalilla humpback whales respond to the arrival of whalewatching vessels by increasing their swim speed is cause for concern. As expected for whales on their breeding grounds, no feeding has been observed. Consequently they must rely on fat reserves to meet their high energetic demands − the females to calve and lactate, and the males to engage in active reproductive displays. Some long-term Hawaiian studies suggest that mother-calf pairs become proportionally less frequent close to shore when recreational boating increases (Glockner-Ferrari and Ferrari, 1985; 1990; Salden, 1988). Mother-calf pairs may be especially vulnerable to disturbance, since some potential avoidance responses (of increased swim speeds and longer dive times, for example) may be beyond the physiological limits of the calf, and because calves may have less opportunity to suckle if the mother is forced to increase her speed or to change her behaviour from resting to travelling.

Williams *et al.* (2002) measured behavioural responses of northern resident killer whales to an experimental whalewatching boat, and found that animals generally evaded the boat by adopting a more circuitous path. This evasive response, when compared with a wider range of opportunistic observations, tended to increase in magnitude as boats got closer. The results from the current study, however, suggest that humpback whales respond to whalewatching boats with a stereotyped tendency to increase swim speed (c.f. Bauer, 1986; Au and Green, 2001).

Of course, fasting puts breeding humpback whales in a qualitatively different context than foraging killer whales. It is unwise to equate swimming faster with a costlier behavioural response than swimming further to get where one wants to go. Neither is it appropriate to speculate whether fasting animals (which vary widely in the thickness of their blubber layers) are less able to cope with repeated short-term disturbances than foraging animals (which may vary widely in their foraging efficiency and prey availability). However, the nature and apparent strength of humpbacks' response to disturbance is striking. It is noteworthy that a variety of studies have detected increased swim speeds as a stereotypical response of baleen whales to vessel traffic (Bauer, 1986; Corkeron, 1995; Au and Green, 2001), given the unlikelihood that that this response can be successful in mitigating disturbance from motorboats. Similarly, it is interesting to note that humpback whales increased speed by over 50% (Fig. 2) in this study, and perhaps as much as 300% in Hawaii (Au and Green, 2001), while the mean response of male northern resident killer whales to a single whalewatching boat was to adopt a path that was 13% less direct (Williams *et al.*, 2002).

Ultimately, studies of whale behaviour around boats are limited by their ability to estimate the extent to which short-term behavioural changes affect the fitness of individuals. Continuing monitoring on the level of the individual is critical to detect any long-term effects of human disturbance. Photo-identification data from this study show that some individual humpback whales are sighted repeatedly throughout a season, as well as between years (Scheidat *et al.*, 2000). On the one hand, this means that individuals are potentially exposed to repeated disturbance, not only on a single day but for up to several months during one year. The area around Isla de la Plata seems to form critical habitat for humpback whales. This makes it both especially important to whales as well as an area where whales are likely to be exposed to disturbance. Between-year site fidelity of some animals may allow for repeated disturbance, and potentially habituation. On the other hand, the wide range of group composition observed during this two-year study suggests that it is unlikely that the observed sample is pseudo-replicated. Further photo-identification studies along the Ecuadorian coast, as well as comparisons with unpublished datasets from other nearby breeding grounds are needed. Should those efforts succeed, it will then be possible to look at vessel impacts at the level of individuals. It is hoped that the findings from this study provide a useful starting point to estimate the cost of this relatively new vessel traffic to some whales in poorly studied waters.

Mobley *et al.* (1999) found that whalewatching is not having an effect on the apparently slow recovery of the Hawaiian humpback whale population. Before reaching similar conclusions for humpback whales in Ecuador, longer-term monitoring and a concerted, collaborative effort to test a wider variety of traffic conditions is require. In the meantime, it seems sensible to manage whalewatching activity in Ecuador as though short-term behavioural responses signify underlying disturbance that may have currently undemonstrated long-term impacts. Experimental studies to determine which whalewatching boats and activities elicit the weakest behavioural responses are strongly recommended. Although local whalewatching guidelines exist, they are not legally binding, and reflect perceived rather than demonstrated impacts of vessel traffic. In order to produce biologically relevant guidelines, experimental testing of relationships between whale behaviour and vessel type, number and proximity are encouraged. In the meantime, it has been recommended that boats limit their closest approach to 100m, and that no more than two boats be allowed within 1,000m at a time until future experimental studies identify more appropriate guidelines. Similarly, local whalewatch operators have been made aware that if they have to speed up their boats to keep up with whales, then this may be a sign that the whales are disturbed. Cooperation between environmental agencies and local stakeholders is especially critical for managing whalewatching in this developing country, where sustainability of the whalewatching industry may be a bigger concern among decision-makers than the well being of the whales.

ACKNOWLEDGEMENTS

This study was financed with grants from the German Academic Exchange Program (DAAD), the Jane Marcher Foundation, as well as yaqu pacha e.V. (Organisation for the Conservation of Aquatic Mammals in South America). We would like to thank the volunteers who worked under often difficult conditions on the island. We would like to thank also the Machalilla National Park for permission to work from the island, and local whalewatching operators for logistical support throughout this study.

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Errata

Behavioural responses of humpback whales (*Megaptera novaeangliae*) to whalewatching boats near Isla de la Plato, Machalilla National Park, Ecuador

Mieke Scheidat, Cristina Castro, Janira Gonzalez and Rob Williams

Journal of Cetacean Research and Management 6(1): 63-68
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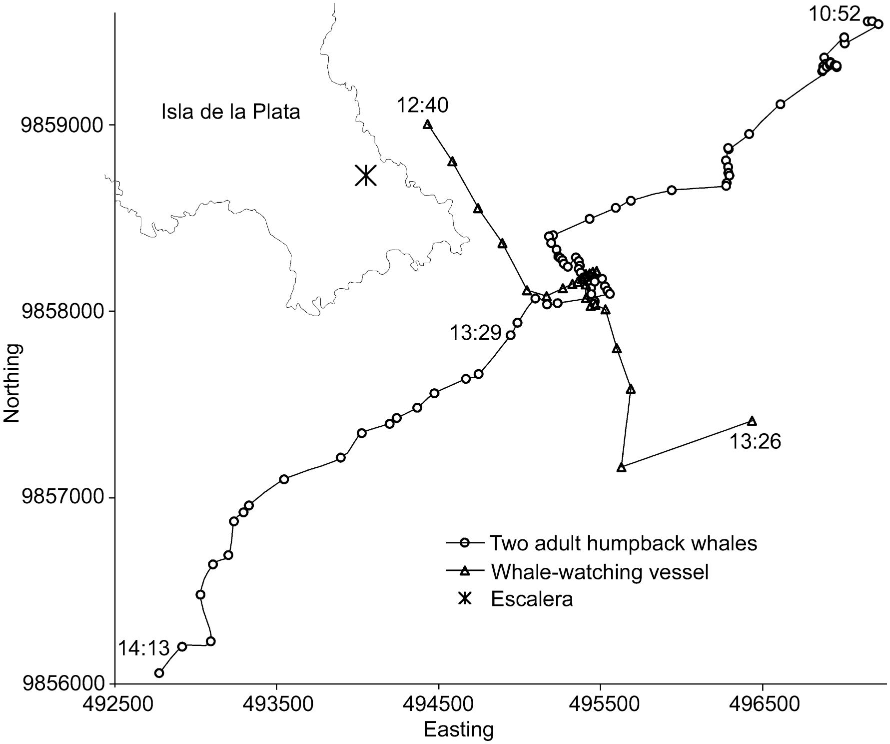

Fig. 2. Example of a typical theodolite tracking from the Isla de la Plata. The observations were made from the observation point Escalera at a focal height of 91.5m. The tracking took place on the 25 August 1999. A group of two adult whales was first noted at 10:52 moving slowly in a southwesterly direction. They were joined by a whale-watching vessel at 12:54 which stayed with the group until 13:18. The humpbacks showed milling and resting behaviour until about 13:28 when they started to travel. The map is presented using a UTM (Universal Transverse Mercator) projection for zone 17 (78°W to 84°W). The x-axis represents the distance in metres from the central meridian of zone 17 (81°W) and the y-axis represents the distance in meters from the South Pole.

J. CETACEAN RES. MANAGE. 6(1):69–77, 2004

Chronic interaction between humans and free-ranging bottlenose dolphins near Panama City Beach, Florida, USA

AMY SAMUELS[*+] AND LARS BEJDER[+#]

Contact e-mail: asamuels@whoi.edu

ABSTRACT

‘Swim-with’ activities, in which humans enter the water to interact with free-ranging cetaceans, are a popular form of nature tourism; however, there is considerable disagreement as to whether these encounters constitute a threat to the animals. At the request of the US Marine Mammal Commission, a systematic study was designed to quantify effects of swim-with activities on the behaviour of bottlenose dolphins in waters near Panama City Beach, Florida. Certain dolphin behaviours were identified as indicative of chronic interaction with humans, and based on presence of these behaviours, at least seven dolphins were identified that permitted people to swim nearby. Because these dolphins accepted food handouts from people, they were considered to be conditioned to human interaction through food reinforecement. Specific human-dolphin interactions that posed a risk for dolphins or humans were identified, and it was calculated that human interaction put a specific juvenile dolphin at risk once every 12 min, including being fed by humans once every 39-59 min. Humans interacting with that dolphin were estimated to be at risk once every 29 min. Although the study was of limited duration, the observations were so clear-cut and the nature of interactions so potentially hazardous it was concluded that food provisioning was the probable basis for swimming with free-ranging dolphins near Panama City Beach, Florida, and therefore, human interaction at this location was likely to be harmful to the dolphins and in clear violation of the US Marine Mammal Protection Act.

Of equal importance to the findings of this study is the methodology. A systematic behavioural methodology was designed that can be adapted to study potential impacts of nature tourism on coastal communities of cetaceans in which individuals are readily distinguished. The focus was on the behaviour of individual animals in order to describe and quantify in-water interactions between dolphins and humans, to make behavioural comparisons for the same individual dolphins in the presence and absence of swimmers, and to make behavioural comparisons for individual dolphins in the same region that do and do not interact with swimmers. Coupled with standard photo-identification techniques, these methods can be used to identify the class of animals, or proportion of a local community, that is more likely to interact with, be detrimentally affected by, and/or avoid human interaction. Sequential observations of the same individuals taken over time can be used to document habituation or sensitisation to human interaction.

KEYWORDS: BEHAVIOUR; BOTTLENOSE DOLPHIN; CONSERVATION; WHALEWATCHING; HUMAN INTERACTION

INTRODUCTION

Over the past several decades, there has been a significant shift in public attitudes towards cetaceans in many countries. For centuries, these animals were considered a resource to be exploited by humans, but in the 1970-80s, strong anti-whaling and pro-conservation sentiments became prevalent particularly in parts of North America, Europe and Australasia (e.g. Duffus and Dearden, 1993; Samuels and Tyack, 2000). In the USA, this viewpoint was manifested in the Marine Mammal Protection Act of 1972 (MMPA, 16 U.S.C. §1361 *et seq.*), which made the federal government responsible for conserving and protecting marine mammal species. With this shift in public sensibility came increasing emphasis on ‘non-consumptive’ uses of marine mammals. For example, cetaceans have become popular tourist attractions, and commercial operators now provide many ways for members of the public to view and interact with whales and dolphins at sea. Hoyt (2001) reported that cetacean-focused tourism is a $US1 billion industry attracting more than nine million people per year in 87 countries and territories.

Tourism focusing on free-ranging cetaceans is a type of ‘nature tourism’, which encompasses a variety of ways people can enjoy wild animals in natural areas (e.g. Newsome *et al.*, 2002). By implication, tourism focusing on nature is often presumed to be ‘ecotourism’; however, in the strictest sense, ecotourism is a specialised subset of nature tourism, and the label is reserved only for those activities that are ecologically sustainable, environmentally educative and contributing to the conservation of biodiversity (e.g. Goodwin, 1996; Newsome *et al.*, 2002). There is a trend towards another form of nature tourism, ‘adventure tourism’, that is, ‘instead of being satisfied with looking at nature, people want to interact with nature’ (Simmonds, 1991, p.664). The proliferation of hands-on adventures targeting cetaceans in the wild has prompted expressions of concern from such organisations as the International Whaling Commission (IWC, 1995). With respect to nature tourism focusing on cetaceans in US waters, the National Marine Fisheries Service (NMFS) and the Marine Mammal Commission (MMC) find themselves having to apply general statutory provisions under the MMPA to address these new potential threats to marine mammals in the wild (NMFS, 2002).

The public is offered opportunities around the world for close encounters with cetaceans at sea. At some locations, tourism is based on feeding: for example, tourists at several sites in Australia offer fish to free-ranging dolphins (e.g. Connor and Smolker, 1985; Orams, 1994; Corkeron, 1998). This practice continues despite extensive documentation that feeding by humans is often harmful to the animals (reviewed in Orams, 2002). For example, detrimental effects on behavioural patterns are associated with food provisioning in several primate species (e.g. Wrangham, 1974; Southwick *et al.*, 1976; Brennan *et al.*, 1985; Altmann and Muruthi, 1988;

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[*] *Department of Conservation Biology, Daniel F. and Ada L. Rice Conservation Biology and Research Center, Chicago Zoological Society, Brookfield IL 60513 USA.*

[+] *Biology Department, Woods Hole Oceanographic Institution, Woods Hole MA 02543 USA.*

[#] Present address: *Biology Department, Dalhousie University, Halifax, Nova Scotia B3H4J1 Canada.*

Fa, 1988). Aggression and injury to humans and destruction of human property are associated with food provisioning in such terrestrial species as black and grizzly bears (*Ursus americanus, U. arctos*: Craighead and Craighead, 1971; Gunther, 1994); African elephants (*Loxodonta africanus*: Moss, 1988); coyotes (*Canis latrans*: Bounds and Shaw, 1994); vervet and Barbary monkeys (*Cercopithecus aethiops, Macaca sylvanus*: Brennan *et al.*, 1985; Fa, 1992); and cassowaries (*Casuarius casuarius*: Kofron, 1999).

Harmful effects of uncontrolled feeding by humans were recently demonstrated for free-ranging cetaceans as well. In Western Australia, low survivorship of calves was associated with tourist feeding of female bottlenose dolphins, *Tursiops* sp. (Wilson, 1994; Mann *et al.*, 2000). Such findings led wildlife managers to impose stringent regulations in order to minimise the impact of feeding on dolphins at existing sites (Wilson, 1994; 1996), and to prohibit all new feeding programmes in the state (Western Australian Wildlife Conservation Act No.1950, Closed Season Notice for Marine Mammals, 1998). In the USA, NMFS amended regulations under the MMPA in 1991 to specify that feeding, or attempting to feed, marine mammals in the wild constitutes a form of 'take' (50 C.F.R. §216.3). This decision was upheld in a 1993 court ruling, based on substantial evidence that feeding free-ranging cetaceans can alter their natural behaviour and increase their risk of injury or death (Bryant, 1994).

'Swim-with' activities, in which humans enter the water for the purpose of interacting with free-ranging whales and dolphins, are another popular form of cetacean-focused tourism. Swim-with activities target at least 20 cetacean species worldwide, and new programmes are initiated on a regular basis (e.g. Samuels *et al.*, 2000; 2003; Hoyt, 2001). The list includes such well-known situations and species as dusky (*Lagenorhynchus obscurus*), bottlenose (*T. truncatus*) and common (*Delphinus delphis*) dolphins in New Zealand (e.g. Barr and Slooten, 1998; Yin, 1999; Constantine, 2001), and Atlantic spotted (*Stenella frontalis*) and bottlenose dolphins in the Bahamas (e.g. Ransom, 1998). There are also a number of less familiar sites and species, including Hector's dolphins (*Cephalorhynchus hectori*) in New Zealand (e.g. Bejder *et al.*, 1999), dense beaked whales (*Mesoplodon densirostris*) near the Canary Islands (e.g. Ritter and Brederlau, 1999), dwarf minke whales *(Balaenoptera acutorostrata*) in Australia's Great Barrier Reef (e.g. Arnold and Birtles, 1999) and humpback whales (*Megaptera novaeangliae)* in the South Pacific (e.g. Constantine, 1998; Orams, 1999). In the USA, members of the public swim with wild bottlenose dolphins (*T. truncatus)* in Florida and wild spinner dolphins (*S. longirostris)* in Hawaii (e.g. Frohoff and Packard, 1995; Flanagan, 1996; Wursig, 1996; Ford, 1997; Samuels and Bejder, 1998; Colborn, 1999; Driscoll-Lind and Ostman-Lind, 1999; Forest, 2001).

There is considerable disagreement among wildlife managers, tour operators and scientists as to whether impacts of swim-with activities on targeted animals are harmful, beneficial or neutral. Opponents of swim-with activities suggest that the increased tolerance of humans and vessels that sometimes results from interaction may compromise free-ranging cetaceans by disrupting natural behaviour and/or increasing the animals' vulnerability to vessel strikes, entanglement and vandalism (e.g. Spradlin *et al.*, 1998). Moreover, interacting with free-ranging cetaceans is contrary to the 'respect the wild in wildlife' principles proposed by the National Watchable Wildlife Program (Duda, 1995, p.23) whose 'look but don't touch' guidelines have been adopted by many managers of terrestrial wildlife. There are also documented dangers for humans who enter the water to interact with cetaceans in the wild (e.g. Shane *et al.*, 1993; Santos, 1997). In contrast, some advocates of swim-with activities maintain that the animals have a choice as to whether or not they will interact with humans (e.g. Dudzinski, 1998), and others suggest that close encounters with free-ranging animals may enhance respect for wildlife, leading to environmental activism and benefits for nature areas (e.g. Orams, 1997). These points are countered by observations that the careful plans needed to safeguard the animals and to realise these potentials are not always in place (e.g. Duffus and Dearden, 1993; Amante-Helweg, 1996; Kinnaird and O'Brien, 1996), and that no tourism is ecologically neutral (e.g. Isaacs, 2000). The controversy is further confounded by the fact that not all swim-with situations are the same. Each of the four basic types of in-water encounters – involving free-ranging cetaceans that are 'lone sociable', food provisioned, habituated and unhabituated – is likely to result in different interactions, responses and impacts (e.g. Samuels *et al.*, 2000).

In the USA, swim-with activities remain a legislative 'grey area'. 'Harassment' was defined in the 1994 amendments to the MMPA to mean 'any act of pursuit, torment, or annoyance which …has the potential to injure… or …disturb a marine mammal …in the wild by causing disruption of behavioural patterns…'. NMFS interpreted this definition to include swim-with activities and issued guidelines accordingly. However, that interpretation has been challenged because swimming with free-ranging cetaceans was not specifically named (e.g. Baur *et al.*, 1999). NMFS (2002) published an Advance Notice of Proposed Rulemaking to address concerns about human interactions, requesting comments from the public regarding a need for specific regulations to protect free-ranging marine mammals. Comments received are currently under review.

As a step towards resolving the controversy over commercial swim-with activities in US waters, the MMC requested a study designed to evaluate systematically how chronic in-water interactions with humans affect the behaviour of free-ranging bottlenose dolphins. A pilot study was conducted near Panama City Beach, Florida, where free-ranging bottlenose dolphins have frequent encounters with clients of commercial tour operators and members of the public (Samuels and Bejder, 1998). The boat-based study was conducted from the perspective of individual dolphins in the region. A concurrent and complementary shore-based study was conducted from the perspective of tourists seeking interactions with dolphins in Panama City Beach waters (Colborn, 1999).

METHODS

Sighting records

A total of 29 hours was spent searching for and observing dolphins in waters near Panama City Beach, Florida, during five days from 4-9 August 1998. The study was focused on dolphins in the vicinity of the southwest shore of Shell Island in the St Andrews State Recreation Area (30°07'N, 85°43'W). This site, hereafter referred to as 'Interaction Beach', was where nearly all human-dolphin interactions reportedly occurred. Interaction Beach and environs comprised an area of less than 1 n.mile^2. The entire study area was approximately 24 n.mile^2 with Interaction Beach near the centre, and including St Andrews Bay, Panama City Harbor Channel and Gulf of Mexico waters up to 1.5 n.miles offshore of Shell Island and Biltmore Beach.

On 4-5 August, the research effort emphasised photo-identification surveys; thereafter (6-9 August) effort emphasised extended focal follows of selected dolphins. Time was divided between identifying and observing dolphins that had interactions with humans on a regular basis and dolphins that did not interact with humans.

Standard photo-identification techniques were used (Würsig and Jefferson, 1990) to photograph dolphin dorsal fins during brief, close approaches in a 21ft boat with an outboard engine. Each image was automatically stamped with the date and time so that the photographed dolphin could be associated with behavioural records. For each dolphin, or dolphin 'school' (defined as a short-term aggregation in Connor *et al.*, 1998) encountered, an estimate of school size and presence or absence of certain behaviours was recorded to gauge the animals' tolerance of close encounters with humans. Certain behaviours were considered to be indicative of chronic interaction with humans (Table 1).

A 200mm lens was used and a limited number of close approaches was made; therefore it was not possible to obtain adequate identification photos for each dolphin encountered. Only high-quality identification photos were analysed, which included nearly all photographs of dolphins that had interactions with humans on a regular basis but only a subset of photographs of dolphins that did not. As a result, the estimate of the number of dolphins that did not interact with humans was based on school-size records.

Table 1

Dolphin behaviours indicative of chronic human interaction

Behaviour	Description
Remain close	Remain within touching distance of one of more humans that are in the water or in a vessel.
Head up	Approach with head out of water to within 2m of vessel or human.
Beg	Approach with head out of water and open mouth to within 2m of vessel or human.
Lunge at vessel	Vertical lunge with open mouth and head and flippers out of water within 2m of a vessel.
Follow vessel	Rapid travel within 2m of the side or stern of a vessel that is moving at speed (but not riding the bow wave).
Accept food	Accept fish handouts or other food items from humans.

Focal follows

During 4-9 August, a protocol was developed for focal-animal sampling of dolphin behaviour (Altmann, 1974). Since the goal of this fieldwork was to design a systematic protocol for describing and quantifying in-water interactions between dolphins and humans, the protocol was revised and behavioural measures were added over the course of the week. As a result of changes in the protocol, specific findings reported below are sometimes based on different sample sizes.

Standard behavioural sampling techniques (defined in Altmann, 1974) were used that have been adapted for studying dolphin behaviour and human-dolphin interactions (Samuels and Spradlin, 1995; Samuels and Gifford, 1997). Focal follows of individual dolphins, or temporarily cohesive dolphin schools, were conducted for periods ranging from 30 mins to 2 hrs 11 mins. Note that a group-level focus is typically not appropriate for behavioural sampling because an observer cannot continuously monitor all the behaviour of all individuals in an aggregation of animals (Altmann, 1974). However, this method was adequate for the present study because it was possible to continuously monitor whether any focal dolphins had interactions with humans. In this study, group-level behavioural sampling was used only for dolphins that did not exhibit any of the behaviours listed in Table 1. All focal follows were conducted in the vicinity of Interaction Beach where dolphins were so often surrounded by tourist vessels, even to the exclusion of the research vessel, that it is unlikely the presence of a research vessel had a significant effect on the behaviour of focal dolphins.

At regular intervals throughout each follow, the following specific information about focal dolphin(s) was recorded.

(1) Number of dolphins in the school: a 10m chain rule was used to define a temporarily cohesive group of dolphins (Smolker *et al.*, 1992).
(2) Activity: standard activity categories were used: rest, travel, forage, mill and socialise with other dolphins. A 'human interaction' activity state was added to encompass the behaviours indicative of chronic interaction with humans (Table 1). During follows of focal schools, 'predominant group activity' was recorded (Mann, 1999) and it was noted whether any individuals were engaged in a different activity.
(3) Location: the general location of the focal dolphin(s) was recorded using a 1 by 1 n.mile grid system superimposed on a chart of Panama City Beach waters.

The sampling interval at which these data were recorded varied according to dolphin activity. During rest, travel, forage and mill, information for the focal dolphin(s) was recorded at each surfacing (every 2-3 mins). However, discrete surfacing bouts were difficult to identify when dolphins were interacting with humans. Preliminary observations of human-dolphin encounters indicated that an interaction with a human occurred on average once per minute. Therefore, in close proximity to human activity, the above information was recorded for the focal dolphin as point samples at 1-min intervals.

In addition to these data taken at regular intervals, other behavioural information was recorded for the focal dolphin(s). Due to vessel crowding near Interaction Beach, it was not possible to record consistently all details of all interactions involving focal dolphin(s); therefore the following behavioural sampling rules were adopted (defined in Altmann, 1974):

(1) a record was made of whether focal dolphin(s) exhibited any of the human interaction behaviours (Table 1) at least once during each sampling interval (one-zero sampling);
(2) details of social interactions with humans or with other dolphins that involved focal dolphin(s) were recorded on an *ad libitum* basis, i.e. whenever possible;
(3) the number of fish (or other food items) offered to focal dolphin(s) was recorded;
(4) the numbers of vessels and human swimmers within 10m of focal dolphin(s) were recorded as scan samples at 5-min intervals;
(5) identification photos were taken periodically to confirm presence of focal dolphin(s). Identification photos were taken opportunistically when a close approach could be made without disturbing dolphin behaviour.

Analysis of risky behaviour

During focal follows, certain human-dolphin interactions were observed that may cause injury, illness or death to the dolphin or the human (Table 2). These risky behaviours were identified following the *Watching Wildlife* guide (Duda,

Table 2

Human-dolphin interactions that may result in injury, illness or death.

Code	Type of interaction	Sources of risk include:
Risk to dolphin		
D1	Human and dolphin make physical contact (or are within touching distance).	Human may inadvertently touch vulnerable body parts of dolphin; human may be aggressive and injure dolphin; human may cause dolphin to behave submissively; potential for disease transmission.
D2	Dolphin is in close proximity to vessels.	Dolphin may be injured by propeller, hit by moving vessel, or injured by an object that falls or is dropped from a vessel.
D3	Dolphin is in close proximity to deployed fishing gear.	Dolphin may be entangled or hooked; dolphin may learn to steal fish from fishers.
D4	Human feeds dolphin.	Dolphin may ingest tainted fish or inappropriate food; young dolphins may not learn appropriate foraging skills.
D5	Human offers object to dolphin.	Dolphin may ingest object and sustain internal injuries.
Risk to human		
H1	Human and dolphin make physical contact (or are within touching distance).	Dolphin may inadvertently touch vulnerable body parts of human; dolphin may be aggressive and injure human; potential for disease transmission.

1995), *Report to Congress on Results of Feeding Wild Dolphins: 1989-1994* (Bryant, 1994), a study of human-dolphin interactions in captive swim-with-dolphin programmes (Samuels and Spradlin, 1995), reports on the potential for interspecific disease transmission (e.g. Buck and Schroeder, 1990; Geraci and Ridgway, 1991) and common sense.

RESULTS

Number of dolphins that has chronic interaction with humans

Assessing each dolphin's tolerance of human interaction was unambiguous. Dolphins categorised as having chronic interactions with humans were repeatedly observed to make close approaches to vessels and to display the behaviours indicative of human interaction listed in Table 1. In contrast, dolphins that did not interact with humans showed no interest in swimmers or vessels (except to bow ride), and performed none of the human-interaction behaviours. Because of the prevalence of 'accept food' (Table 1) among dolphins that has interactions with humans on a regular basis, these dolphins were considered to be conditioned to human interaction by food reinforcement. [In Samuels and Bejder (1998), conditioned dolphins were mislabelled as 'habituated', following colloquial but inaccurate usage in the wildlife literature (Nisbet, 2000). Documenting the occurrence of 'habituation' required sequetial measures over time showing a waning in response as individuals learn that there are neither adverse nor beneficial consequences to occurrence of the stimulus (Thorpe, 1963). It was, therefore, incorrect to use the term in reference to dolphins attracted to human interaction by food reinforcement.]

Based on sighting records, a minimum of 89 dolphins was encountered during the five days. Behavioural assessment indicated that seven dolphins identified photographically, or 8% of all dolphins encountered, had interactions with humans on a regular basis. Six of the seven conditioned dolphins were identified during the first two days when the research effort emphasised photo-identification work. Since research emphasis was subsequently shifted to focal follows (a method that limits the number of dolphins encountered), there were likely to be a greater number of conditioned dolphins than the seven identified. Some conditioned dolphins were frequent visitors to Interaction Beach: six of the seven dolphins were sighted there on multiple days, and one dolphin was identified on four of the five days. During the study period, none of the conditioned dolphins was identified in schools with dolphins that did not interact with humans.

Differences in the behaviour of conditioned dolphins and dolphins that did not interact with humans

Nine focal follows were conducted for a total of 12 hours (Table 3). Conditioned dolphins were observed in four focal-individual follows (6h 32min); a single juvenile dolphin was the focus of three of those follows (5h 53min). Also conducted were one focal-individual and four focal-group follows (5h 28min) to monitor the behaviour of at least 27 dolphins that did not interact with humans.

Despite the brief observation period, the data indicated that the lives of conditioned dolphins were strikingly different from those of dolphins that did not interact with humans. It is estimated that conditioned dolphins were engaged in interactions with humans during approximately 77% of the time they were observed (i.e. one or more human-interaction behaviours occurred in 188 of 245 1-min point samples on 6-9 August). In contrast, dolphins that did not interact with humans never exhibited any of those behaviours (i.e. human-interaction behaviours occurred in none of a total of 85 surfacing intervals on 6-9 August).

Dramatic differences in ranging patterns were documented even though all focal follows were conducted within the same region. Conditioned dolphins remained at the same location, i.e. within the < 1 n.mile^2 area consisting of Interaction Beach and the adjacent Panama City Harbor Channel. All recorded 'travel' by conditioned dolphins was from vessel to vessel. Because conditioned dolphins typically approached any new vessel that arrived in the area, it was possible to keep the focal dolphin in view by looking for that dolphin alongside the most recently arrived vessel. In contrast, focal dolphins that did not interact with humans travelled distances of several nautical miles along the Gulf coast or into the bay; they moved through Interaction Beach, without stopping or showing any interest in human activities.

Profile of a juvenile dolphin conditioned to human interaction

The study focused on one conditioned dolphin, 'HiMidLo', so named for three distinctive nicks in the dorsal fin. HiMidLo was selected for intensive study because this juvenile was ever-present in the vicinity of Interaction

Table 3

Summary of focal follows of conditioned dolphins and dolphins that did not interact with humans.

Date	Name/ description	Duration (hr:min)	Summary
Conditioned dolphins			
5 Aug	HiMidLo	1:40	Remained at Interaction Beach; accepted fish handouts; suspected underwater feeding by tour operator.
6 Aug	HiMidLo	2:02	Remained in the channel adjacent to Interaction Beach; accepted fish handouts; rejected bologna.
7 Aug	HiMidLo	2:11	Remained at Interaction Beach from early morning until tour boats arrived; accepted fish handouts from vessel travelling at speed; suspected underwater feeding by tour operator.
9 Aug	HiNick	0:39	Remained at Interaction Beach; accepted fish handouts.
Total duration		6:32	
Dolphins that did not interact with humans			
5 Aug	Group of 5	1:45	Cohesive group travelling southeast in Gulf along Biltmore Beach and Interaction Beach; no interest in humans (except to bow ride).
6 Aug	Trio #1	0:55	Cohesive group travelling northeast into St Andrews Bay; no interest in humans.
7 Aug	Single dolphin	1:42	Distinctive dolphin with damaged dorsal fin, moved through several dolphin schools while travelling southeast in Gulf along Biltmore Beach and Interaction Beach; no interest in humans.
9 Aug	Group of 15	0:30	Cohesive group travelling southeast in Gulf along Shell Island; apparently disturbed by our approach (repeated tail slaps) but calmed within 15 mins; no interest in humans.
9 Aug	Trio #2	0:36	Cohesive group first sighted at Interaction Beach, travelling southeast in Gulf close to Shell Island; no interest in humans.
Total duration		5:28	

Beach, and it was suspected that the daily life of this dolphin was typical of that of many conditioned dolphins in the region. In addition, there were concerns regarding the effects of chronic human interaction on the behaviour and well being of an immature dolphin. Based on body size, HiMidLo was estimated to be a 4-5 yr old, an independent juvenile but not fully mature. The sex of this dolphin was not determined.

HiMidLo was encountered in the vicinity of Interaction Beach on four of five field days, including prior to the arrival of any tour boats on the morning of 7 August. Focal follows of this dolphin were conducted on three days (Table 3). HiMidLo was engaged in interactions with humans or vessels during approximately 75% of observation time (161 of 214 point samples on 6-7 August), and the dolphin was within 10m of humans in the water during 55% of observation time (125 of 228 intervals on 6-7 August). On average, there were 4 swimmers, 2.6 boats, and 1.3 jet skis within 10m of HiMidLo (73 5-min scans on 5-7 August). Maximums of 34 swimmers (5 August) and 14 vessels (6 August) were recorded within 10m of HiMidLo (Fig. 1). HiMidLo moved from boat to boat approximately once per 3 min (45 of 130 intervals on 7 August).

In contrast to the high proportion of time spent interacting with humans, HiMidLo was observed socialising with other dolphins only twice: an affiliative interaction involving ‘gentle rubbing’, and a presumably agonistic interaction involving a tail slap when another dolphin approached the boat where HiMidLo was begging. HiMidLo was observed to forage naturally only once (‘pinwheel feeding’); whereas, this juvenile was fed by humans on average once per 39-59 min (6-9 times during the 5h 53mins of focal observations; Table 4; Fig. 2). The range in values was due to three instances of unconfirmed feeding in which people appeared to hide their interactions with dolphins in response to the presence of observers and/or the recent citation of a local commercial operator for feeding wild dolphins (NOAA, 1999). It was surmised from the dolphin’s behaviour that these additional feeding events occurred.

HiMidLo had numerous risky encounters with humans in only three days of focal observations (Table 4). It was estimated that interactions with humans put HiMidLo at risk once per 11.8 min (30 interactions in 5h 53mins of focal observations), and that humans in the water with HiMidLo were at risk of injury by the dolphin once per 29.4 min (12 interactions in 5h 53min). Some interactions posed multiple risks to the dolphin, as when humans fed the dolphin from a vessel that was moving at speed with fishing gear deployed.

Focal observations of a second dolphin indicated that the behaviour of HiMidLo was likely to be typical of conditioned dolphins at Interaction Beach. The second conditioned dolphin, ‘HiNick’, was judged by size to be an adult. HiNick was identified at Interaction Beach on three days, and this dolphin’s behaviour was observed for a total of 39 mins on 9 August. HiNick interacted with humans during 87% of focal observation time (27 of 31 point samples in which activity was known), was in close proximity to swimmers during 81% of the time (25 of 31 intervals), and was being fed by humans during 61% of the time (19 of 31 point samples) or once every 13 mins (3 separate feeding events in 39 min). *Ad libitum* observations of other conditioned dolphins were consistent with observations of HiMidLo and HiNick.

DISCUSSION

Food provisioning of free-ranging dolphins near Panama City Beach, Florida

Observations suggested that human interaction was likely to be harmful to dolphins in waters near Panama City Beach. Despite the brief observation period, a high rate of uncontrolled food provisioning by humans was documented, and numerous encounters with humans were recorded that put conditioned dolphins at risk of injury, illness or death. Dolphins like the juvenile, HiMidLo, may have been in additional danger if provisioning and human interaction

Fig. 1. A typical scene at Interaction Beach in which jet skis, boats and swimmers surround a dolphin. The woman on the stern of the boat to the left had been feeding fish to the dolphin. (Photo credit: L. Bejder).

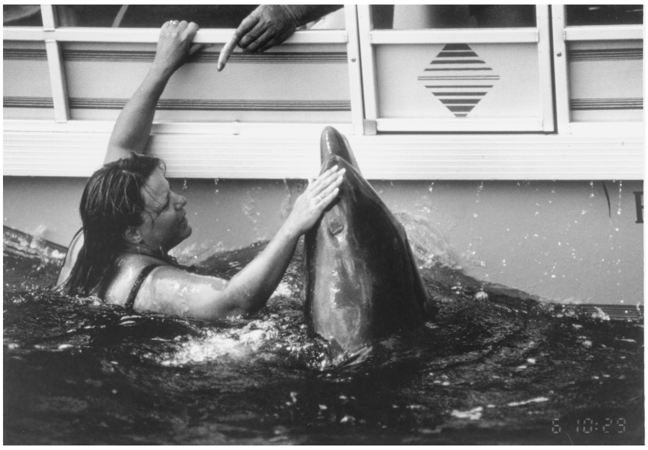

Fig. 2. A woman in the water alternately petted, embraced and fed fish to HiMidLo during the focal follow on 6 August in the Panama City Harbor Channel. Her actions put both the woman and the dolphin at risk of injury (Table 2). (Photo credit: L. Bejder).

have interfered with development of the foraging and social skills necessary for survival in the absence of provisioning.

Given that the methodology limited the number of dolphins encountered, there was likely to be a greater number of conditioned dolphins than those identified in this study. A longer-term study would be needed to ascertain the actual proportion of the local community that is affected by human activity.

Numerous encounters at Interaction Beach were observed in which humans in the water were at risk of injury by the dolphins. The death of a Brazilian swimmer from injuries inflicted by a bottlenose dolphin (Santos, 1997) showed that this is a risk to be taken seriously. Such encounters can also have detrimental consequences for the animals involved. In Brazil, dolphin aggression was provoked by human misbehaviour; nevertheless, an intensive effort was needed to protect the dolphin from further harassment or retribution (Santos, 1997). Other instances of human misbehaviour have had less fortunate consequences for the animals. For example, animals lured by food to approach human activities have sometimes been killed when they became aggressive or destructive (e.g. bears: Gildart, 1981; coyotes: Bounds and Shaw, 1994).

The frequent and uncontrolled feeding of dolphins, both by commercial operators and members of the public, indicated that food provisioning was the basis for in-water encounters between humans and dolphins in Panama City Beach waters. Since dolphins that receive food handouts will indiscriminately approach any vessel or swimmer, even people who did not feed dolphins were able gain close access. It should be noted that food provisioning does not occur at all locations where people swim with free-ranging dolphins. For example, feeding is clearly not a component of swimming with bottlenose, common, dusky and Hector’s dolphins in New Zealand (e.g. Barr and Slooten, 1998; Bejder *et al.*, 1999; Constantine, 2001), spinner dolphins in Hawaii (e.g. Driscoll-Lind and Ostman-Lind, 1999; Forest, 2001; Wursig, 1996), spinner and bottlenose dolphins in Japan (e.g. Dudzinski, 1998), or spotted and bottlenose dolphins in the Bahamas (e.g. Ransom, 1998). However, the situation at Panama City Beach suggests that feeding by humans may be the basis for swimming with free-ranging dolphins at some other sites. At locations like Panama City Beach where food provisioning is used to sustain swim-with activities, human interaction is likely to be harmful to the dolphins.

Studying the effects of swim-with activities on the behaviour of coastal cetaceans

The stated goal of this fieldwork was to design a systematic study to investigate effects on free-ranging dolphins of chronic in-water encounters with humans. It was found that Panama City Beach, Florida, was not a suitable site for such a study because feeding was so prevalent. The effects of food provisioning are so pervasive that it would be difficult to tease apart which effects on dolphin behaviour were due to in-water interactions with humans and which were due to food provisioning. Nevertheless, on the basis of observations, it is concluded that the risks to conditioned dolphins in Panama City Beach waters were so clear that immediate enforcement action would be justified without further study. Should enforcement take effect, further study would be warranted to monitor the behaviour and welfare of potentially dependent dolphins after food handouts were stopped.

A systematic behavioural methodology focusing on individual dolphins was designed that can be adapted to study potential impacts of nature tourism on coastal communities of cetaceans in which individuals are readily distinguished. Methods like these have long been standard for studying effects of human activities on terrestrial animals (e.g. Altmann and Muruthi, 1988), but have only recently been applied to impact assessment studies for cetaceans (e.g. Allen and Read, 2000; Mann and Kemps, 2003; Nowacek *et al.*, 2001). In contrast, most studies of swim-with activities have focused on responses by schools of cetaceans and/or responses to vessel approaches (reviewed in Samuels *et al.*, 2000). These emphases are necessary first steps, in part dictated by methodologies used (e.g. distant, shore-based observations; in-water or tour vessel-based observations). However, as several researchers have noted, findings from such studies are often limited, and more refined, longitudinal investigations are needed (e.g. Constantine, 2001; Ransom, 1998; Samuels and Bejder, 1998; Yin, 1999). Noteworthy attributes of well-designed studies for evaluating impacts of tourism on free-ranging cetaceans are reviewed in Bejder and Samuels (2003).

In this study, behavioural sampling methods were selected that provided a focus on individual animals. These methods were designed to: (1) describe and quantify in-water

Table 4

Risky encounters between humans and HiMidLo during focal follows.

Date	Time	Human-dolphin interaction (durations based on # point samples elapsed)	Risk to dolphin or human? (Codes from Table 2)
5 Aug	10:39	Suspected underwater feeding by diver from tour boat (6 min)	D4
	10:58	Face-to-face with swimmer	D1, H1
	11:15	Surrounded by 8 boats, 5 jet skis	D2
	11:27	Fed from boat (5 min)	D2, D4
	11:35	Surrounded by 8 boats, 2 jet skis	D2
	11:35	Fed from boat (2 min)	D2, D4
6 Aug	8:47	Petted by swimmer	D1, H1
	8:51	Suspected feeding from tour boat; dolphin tossed fish near boat (1 min)	D2, D4
	9:01	Surrounded by 5 boats, 5 jet skis	D2
	9:16	Chin slap near swimmer	H1
	9:19	Anchor dropped near (on?) head	D2
	9:25	Close follow and lunge at boat while underway (3 min)	D2
	9:26	Surrounded by 3 boats, 8 jet skis	D2
	9:41	Surrounded by 3 boats, 11 jet skis	D2
	9:51	Hit by man in face while begging at boat	D1, D2
	9:53	Petted by swimmer	D1, H1
	9:55	Petted by swimmer	D1, H1
	10:24	Submissive flinch in response to 'swimmer on top of' dolphin	D1, H1
	10:27	Swimmer alternated feeding and petting (4 min) (Fig. 2)	D1, D4, H1
	10:35	Offered bologna from boat; dolphin rejected bologna and left (1 min)	D2, D4
7 Aug	7:44	Close follow to boat underway with fishing lines out	D2, D3
	8:04	Close follow to boat underway with fishing lines out	D2, D3
	8:11	Close approach to propeller of boat underway	D2
	8:18	Close follow to boat underway with fishing lines out	D2, D3
	8:30	Close follow to boat underway	D2
	8:46	Close follow to boat underway	D2
	8:53	Begging while boaters dangled hat and sunglasses	D2, D5
	8:56	Fed from boat while underway; dolphin followed and lunged at boat (6 min, 5 fish seen)	D2, D4
	9:14	Suspected underwater feeding by tour operator (intermittent for >51 min)	D4
	9:16	Tail slap near swimmer	H1
	9:29	Leap near swimmer	H1
	9:43	Abrupt dive near swimmer	H1
	9:45	Face to face with swimmer	D1, H1
	10:01	Fed near swimmer (1 min)	D1, D4, H1

interactions between cetaceans and humans, including types and frequencies of interactions; (2) make comparisons of behaviour, ranging and association patterns for the same individuals in the presence and absence of swimmers; and (3) make similar comparisons for individuals in the same region that do and do not interact with swimmers. These methods were coupled with standard photo-identification techniques to (4) estimate the proportion of the local community, and identify particular classes of animals, that are tolerant of human interaction.

The focus on individual animals allowed rapid assessment of the detrimental effects of human interaction on local dolphins. In addition, this focus provided a tentative profile of the vulnerability of the juvenile age class to the detrimental effects of tourism. In a longer-term study, data obtained using methods like these can be used to determine which animals are more likely to interact with, be detrimentally affected by, or avoid swimmers. Such a study conducted over time would provide valuable information on the short-term, seasonal and long-term impacts of swim-with encounters on the daily lives of individual cetaceans, on animals of different gender, age class, activity state, or reproductive condition, and on cetacean communities. Sequential observations of the same individuals taken over time can be used to document habituation of sensitisation of cetaceans to human interaction.

ACKNOWLEDGEMENTS

We thank T. Spradlin and K. Colborn for assistance in the field. We are also grateful to J. Drevenak, R. Hardy, F. Townsend, and the staff at Gulf World for providing equipment, logistical support, and/or local information during fieldwork. P. Best, R. Constantine, P. Corkeron, S. Dawson, M. Gosliner, R. Hofman, D. Laist, R. Mattlin, T. Spradlin, J. Twiss and P. Tyack provided helpful comments on previous versions of this manuscript. This research was funded by MMC Contract T53700904, and conducted under the NMFS Office of Protected Resources General Authorisation #913-1468. This is contribution #11068 of the Woods Hole Oceanographic Institution.

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J. CETACEAN RES. MANAGE. 6(1):79–86, 2004

Geographic and temporal variations in strandings of beaked whales (Ziphiidae) on the coasts of the UK and the Republic of Ireland from 1800-2002

COLIN D. MACLEOD, GRAHAM J. PIERCE AND M. BEGOÑA SANTOS

Department of Zoology, University of Aberdeen, Tillydrone Avenue, Aberdeen, AB24 2TZ, UK.

Contact e-mail: c.d.macleod@abdn.ac.uk

ABSTRACT

This study analyses published records of beaked whale strandings from the coasts of the UK and the Republic of Ireland between 1800 and 2002. Strandings of northern bottlenose whales (*Hyperoodon ampullatus*) were lowest in April and highest in September. The number of strandings between months differed significantly from an even spread over all months of the year, with more strandings between July and October. Most strandings in late summer and autumn occurred on North Sea coasts and their stomach contents included the squid *Gonatus fabricii*, which is found only in more northern waters. This suggests that these whales may be migrating southward at this time of year. Most strandings of Sowerby's beaked whales (*Mesoplodon bidens*) also occurred in late summer and autumn, although this was not significant. Strandings of Cuvier's beaked whales (*Ziphius cavirostris*) occurred almost exclusively on the Atlantic coasts of the UK and in Ireland. There were significantly more Cuvier's beaked whale strandings than expected in January and February and in June and July. A Cuvier's beaked whale which stranded in northern Scotland in February contained similar prey to two whales stranded in north-western Spain at the same time of year, suggesting this animal could have been feeding in more southern waters prior to stranding. Seasonal patterns of strandings of northern bottlenose and Cuvier's beaked whales were significantly different with more of the former stranding in August to October and more of the latter from November to July. This is consistent with a hypothesis of temporal segregation between the two species to reduce potential competition for prey.

KEYWORDS: DISTRIBUTION; MIGRATION; NORTHERN HEMISPHERE; MOVEMENTS; NORTHERN BOTTLENOSE WHALE; SOWERBY'S BEAKED WHALE; CUVIER'S BEAKED WHALE; STRANDINGS; NORTH ATLANTIC; NORTH SEA

INTRODUCTION

The beaked whales (Ziphiidae) are one of the most enigmatic families of marine mammals and much of their biology remains unknown. This is due, in part, to their oceanic distribution and preference for deep waters beyond the shelf edge. As a result, much of what is currently known about beaked whales comes from the study of stranded animals. Oceanic cetaceans may strand for two basic reasons, they may die or become incapacitated and be carried by currents to where they strand, or they may make 'navigational errors' and travel into shallow waters, where they become disoriented or suffer from a lack of suitable food, and subsequently strand (e.g. Klinowska, 1985). Around the coasts of Britain and Ireland, the main surface currents are dominated by the North Atlantic Drift (North Atlantic current) and travel north along the western coasts with branches of currents heading into coastal regions, such as the English Channel, Bristol Channel, Irish Sea and the Sea of the Hebrides. Strandings in these areas may represent passively transported individuals. In terms of potential 'navigational errors', there are a number of locations which could act as 'traps' for oceanic cetaceans around the coasts of the UK and Ireland, such as the North Sea (considered by Smeenk (1997) to be a sperm whale trap).

Records of stranded cetaceans can be used to infer information on a variety of aspects of their biology, such as distribution, diet, timing and routes of migrations, contaminant burdens, reproduction and genetic composition of populations (e.g. Katona *et al.*, 1988; Law *et al.*, 1996; Smeenk, 1997; MacLeod, 2000; Santos *et al.*, 2001a; b). However, data obtained from strandings have a number of inherent biases and weaknesses which must be taken into account when interpreting such data (Klinowska, 1985). Using information from a large number of stranding events collected over a long time period in any analysis will counteract some of the limitations, in particular reducing the importance of a small number of 'atypical' strandings events, and may allow the separation of regular patterns from random variation. However, with oceanic species, such as beaked whales, strandings are relatively rare events and long time series of data are required to obtain sufficient numbers of strandings to investigate any aspect of their biology.

One of the longest continuous time series of strandings records comes from the coasts of the UK and the Republic of Ireland (Klinowska, 1985). Systematic records have been kept continuously since 1913 and have been published on a regular basis (Harmer, 1914; 1915; 1916; 1917; 1918; 1919; 1921; 1923; 1924; 1927; Fraser, 1934; 1946; 1974; Sheldrick, 1989; Sheldrick *et al.*, 1994). In addition, non-systematic records of beaked whale strandings are available at least as far back as 1800 (e.g. Sowerby, 1804). To date, six species of beaked whales have been recorded stranded on the coasts of the UK and Ireland (e.g. Harmer, 1927; Herman *et al.*, 1994; Berrow and Rogan, 1997).

Beaked whale strandings around the UK and Ireland have previously been investigated on a number of occasions (e.g. Fraser, 1974; Evans, 1980; Sheldrick, 1989; Sheldrick *et al.*, 1994; Berrow and Rogan, 1997). Some of these studies have provided summaries of strandings over relatively short time periods (Fraser, 1974; Sheldrick, 1989; Sheldrick *et al.*, 1994). Berrow and Rogan (1997) discussed the seasonal pattern of strandings of beaked whales, amongst other cetacean species, in Ireland while Evans (1980) mentions beaked whales as part of general discussions of cetacean distribution in the northeast Atlantic. None of these previous investigations have covered the whole area (UK and Ireland) and/or time-frame (1800-2002) encompassed by this study, nor included any statistical analysis of beaked whale strandings patterns. For three of these species (the northern bottlenose whale *Hyperoodon ampullatus*, Sowerby's beaked whale *Mesoplodon bidens*, and Cuvier's beaked whale *Ziphius cavirostris*), the number of individuals recorded is now sufficient to allow a detailed investigation of geographic and temporal variations in the number of strandings.

In UK and Irish waters, northern bottlenose whales have been sighted in the deep water beyond the shelf edge both to the west of Ireland and Scotland and to the north in the Faroe-Shetland channel (Pollock *et al.*, 2000; O Cadhla *et al.*, 2001). In addition, this species has been sighted as far north as the Norwegian Sea, and in the past was hunted in this area and around Iceland (Sigurjónsson and Vikingsson, 1998; Bjorke, 2001). Similarly, Sowerby's beaked whale has been sighted in deep water to the west of Ireland, to the north and west of Scotland and as far north as 71°30'N in the Norwegian Sea (Carlström *et al.*, 1997; Pollock *et al.*, 2000; O Cadhla *et al.*, 2001). In addition, both species have been recorded in winter months (November to February) in the Bay of Biscay to the south of the UK and Ireland (Brereton and Williams, 2001). The nearest location where Cuvier's beaked whales have commonly been sighted is in the southern Bay of Biscay, around 600-700km to the south of the UK coast (Williams *et al.*, 1999). However, Cuvier's beaked whales are also known to occur in the deep waters to the west of Ireland (Pollock *et al.*, 2000; O Cadhla *et al.*, 2001) and there is one sighting as far north as northern Scotland (Evans, 1992). Nevertheless, there are no confirmed records of Cuvier's beaked whales further north than this in the northeastern Atlantic.

In other parts of the world, beaked whale occurrence is known to vary throughout the year. For example, along the Pacific coast of Japan, Baird's beaked whales (*Berardius bairdii*) appear in May, densities increase during the summer and this species disappears from these waters by December (Kasuya and Miyashita, 1997). Similarly, Cuvier's beaked whales were caught throughout the year in southern Japan, but were caught in more northern areas only from July to December (Nishiwaki and Oguro, 1972). In the northeast Atlantic, an annual southward movement of northern bottlenose whales has been inferred from variations in catches in a number of fisheries which took this species. In the Norwegian Sea, a southward migration apparently begins in July, with most northern bottlenose whales having left this area by August, although some remain behind until November (Mitchell and Kozicki, 1975). Around Iceland, catches of northern bottlenose whales occurred from the end of March until the start of November, with the greatest numbers being caught at the start of July (Sigurjónsson and Vikingsson, 1998). Similarly in the Faroes, although whales may be caught all year round in the local drive fishery, the majority of northern bottlenose whales have been caught from August to October, and this is presumed to coincide with an autumn migration past the east of the Faroes (Bloch *et al.*, 1996). Previous studies have suggested that there are also seasonal movements of beaked whales around the UK and Ireland. For example, Mitchell and Kozicki (1975) suggested that strandings of northern bottlenose whales reflect a north-south seasonal movement in UK and Irish waters. However, other authors, such as Hooker (1999), have suggested this evidence is not clear-cut and that other explanations for the observed strandings patterns are possible, such as offshore-nearshore movements.

In the present study, records from the UK and Ireland were analysed to identify geographical and seasonal patterns in strandings of three species of beaked whales, which may relate to variations in patterns of distribution of living animals in the area. Published information on stomach contents of stranded beaked whales from the northeast Atlantic were used to infer the direction and extent of movements of beaked whales prior to stranding.

In addition, MacLeod *et al.* (2003) hypothesised that *Ziphius* and *Hyperoodon* species occupy a similar dietary niche with a high potential for competition between species from these two genera. If this is true, there must be some mechanism to limit competition. They proposed spatio-temporal segregation, with each species either occurring in different areas or occupying the same areas at different times. Within the northeast Atlantic, northern bottlenose whales and Cuvier's beaked whales occupy a similar dietary niche, feeding on similar-sized individuals of the same species (e.g. Santos *et al.*, 2001a; b), so conforming with the hypothesis of dietary niche overlap with high potential competition between these species. Therefore, the strandings data were investigated to see if there was any support for the proposed spatio-temporal segregation.

METHODS

Data on beaked whale strandings between 1800 and 2003 were collated from published records (Harmer, 1914; 1915; 1916; 1917; 1918; 1919; 1921; 1923; 1924; 1927; Fraser, 1934; 1946; 1974; Sheldrick, 1989; Herman *et al.*, 1994; Sheldrick *et al.*, 1994; Berrow and Rogan, 1997; Smiddy, 1997; MacKay *et al.*, 1998; 2002; Berrow and Storer, 2001; Berrow and Dalebout, 2002; Berrow *et al.*, 2002; Quigley and Flannery, 2002), on-line databases (Irish Whale And Dolphin Group Strandings Database: *www.iwdg.ie/strandings*) and recent unpublished data (SAC Veterinary Science Division, unpublished data; Natural History Museum, unpublished data). For each record, information on the location, date and species was entered into a database. For almost all records, published locations were listed only by place names or map references which were converted into latitude and longitude.

Strandings were not sub-divided by cause of stranding. Accurately identifying the true cause of a stranding is often not possible. For example, animals which make navigational errors may then suffer from a lack of available food which can affect their health and subsequently lead to death from starvation or infection. In such a case, the original navigational error cannot be picked up from an examination of the carcass except possibly through the location of the animal. Conversely, illness or poor health may lead to navigational errors and subsequent stranding.

Similarly, there was no separation of records into live or dead strandings. Within the area covered by this study, there are regions with relatively remote coastlines, such as parts of Ireland or northwest Scotland, where strandings may not be detected for several days, or more, after they occur. Therefore, although the condition of an animal when it was found is known, it is not always certain whether it was alive or dead when it came ashore. This is particularly true of older records when communications were not as fast as today.

Finally, unless individuals were specifically known to have stranded together at the same place and time, they were considered separate data points. This avoided any possible problems in trying to ascertain whether individuals which stranded in close proximity in space and/or time were in reality linked in any way and if so how. Although this may lead to some stranding events being counted twice, the number of such possible double counts was low (under 2% of all cases).

The area under consideration was separated into seven separate regions (Fig. 1): northern Atlantic coasts (Cape Wrath to John O'Groats, including the Northern Isles); mid-Atlantic Coasts (from a line approximating the Islay Front between Ireland and southern Scotland to Cape Wrath, including the Western Isles); southern Atlantic coasts (from Cork in Ireland along south-west and west Irish coasts to the Islay Front area); Irish Sea Coasts (from Cornwall on the west coast of the UK and Cork in Ireland north through the Irish Sea to the Islay Front); the northern North Sea coasts (John O'Groats to just south of Sunderland and consisting of the northern half of the UK North Sea coastline); southern North Sea (the southern half of the UK North Sea coasts) and English Channel coasts (from Dover to Lands End, including the Channel Islands). The number of individuals of each species of beaked whale stranded in each region was recorded by month and the data were examined for geographic and temporal variations. Chi-squared tests were used to test the null hypothesis that strandings for each species did not differ significantly from an even spread between each month of the year, while Kolmogorov-Smirnov tests were used to test the null hypothesis that there were no differences in the distributions of strandings in each month between pairs of regions. This last analysis was only conducted for regions with 10 or more stranding events.

To test whether strandings patterns are consistent with the hypothesis of spatio-temporal segregation between northern bottlenose whales and Cuvier's beaked whales, the numbers of individual whales stranding in four time periods during the year (November to January, February to April, May to July and August to October) were compared using a Chi-Squared test. This analysis was undertaken only for strandings in western regions (northern, mid and southern Atlantic and Irish Sea coasts), the area where spatio-temporal segregation has been hypothesised to occur and for the period of systematic recording of strandings (1913-2002).

RESULTS

Records of 257 beaked whale strandings which occurred between 1800 and 2002 were located. Of these, 251 were listed as identified to species level. These were 109 northern bottlenose whales, 70 Sowerby's beaked whales, 63 Cuvier's beaked whales, 7 True's beaked whales (*Mesoplodon mirus*), 1 Gervais' beaked whale (*Mesoplodon europeaus*) and 1 Blainville's beaked whale (*Mesoplodon densirostris*). For all regions, with the exception of the English Channel which only had 12 stranding records, the numbers of strandings were of the same order of magnitude (between 34 to 52 —Fig. 1). The locations of strandings of northern bottlenose whales, Sowerby's beaked whales and Cuvier's beaked whales are plotted in Fig. 2.

Northern bottlenose whales

Strandings of northern bottlenose whales were highest in September ($n = 33$) and lowest in April ($n = 0$) and the intra-annual pattern of strandings differed significantly from an even spread across all months ($\chi^2 = 114.1$, d.f. = 11, $p < 0.001$ —Fig. 3a). Fewer strandings occurred than expected between November and June (significantly fewer in December and April: $\chi^2 = 5.3$, d.f. = 1, $p = 0.022$ and $\chi^2 = 8.8$, d.f. = 1, $p = 0.030$ respectively) while there were more strandings than expected between July and October (significantly more in August, September and October: $\chi^2 = 5.8$, d.f. = 1, $p = 0.016$; $\chi^2 = 66.1$, d.f. = 1, $p < 0.001$; $\chi^2 = 14.1$, d.f. = 1, $p = 0.0002$ respectively).

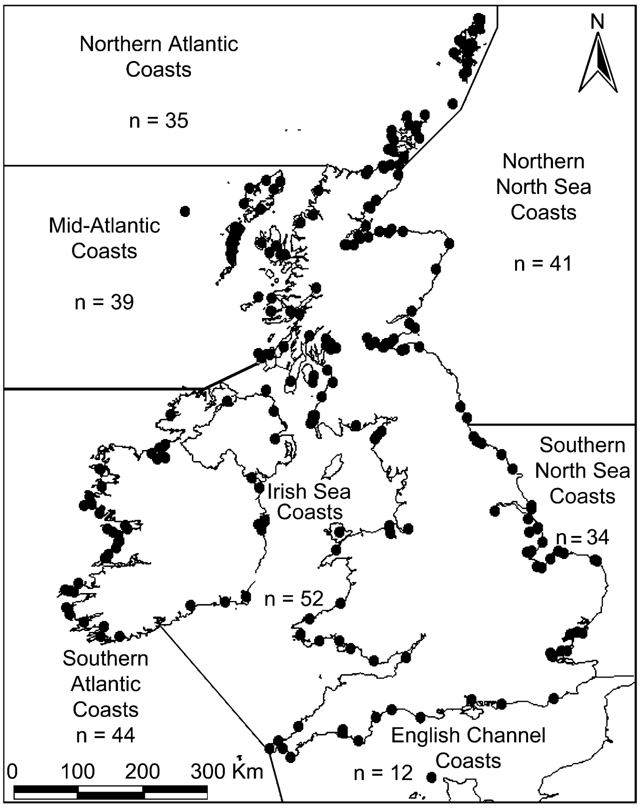

Fig. 1. Regions of the United Kingdom and the Republic of Ireland used in analysis of beaked whale strandings. Black dots represent strandings records for beaked whales between 1800 and 2002.

The number of northern bottlenose whales stranded in each region varied from four strandings on English Channel coasts to 36 on Irish Sea coasts (Fig. 2). Between the different regions, there were differences in the month of peak strandings (Fig. 3a). On northern Atlantic coasts, the peak number of strandings occurred in June, on the mid-Atlantic coasts the peak was in October, September on Irish Sea coasts and August and September on south Atlantic coasts. On North Sea coasts, strandings of northern bottlenose whales peaked in September in the northern region and July and August in the southern region. On English Channel coasts, the total number of strandings (4 events) was too low to consider seasonality. Five regions had a sufficient number of strandings to allow comparisons of the distribution of strandings with month. The distribution of strandings by month differed significantly between the northern Atlantic region and all regions which were compared to it (mid-Atlantic coasts: D = 0.627, $n_1 = 11$, $n_2 = 10$, $p < 0.05$; Irish Sea coasts: D = 0.652, $n_1 = 11$, $n_2 = 36$, $p < 0.05$; northern North Sea coasts: D = 0.527, $n_1 = 11$, $n_2 = 20$, $p < 0.05$; southern North Sea coasts: D = 0.616, $n_1 = 11$, $n_2 = 8$, $p < 0.05$). Therefore, it was concluded that there was a difference in the distribution of strandings by month between northern Atlantic coasts and the other regions and that the peak of strandings in this region occurs earlier in the year (June) than in other regions (July to October depending on region). However, there were no significant differences between mid-Atlantic coasts and Irish Sea coasts (D = 0.144, $n_1 = 10$, $n_2 = 36$, $p > 0.05$) or between the two North Sea regions (D = 0.489, $n_1 = 20$, $n_2 = 18$, $p > 0.05$).

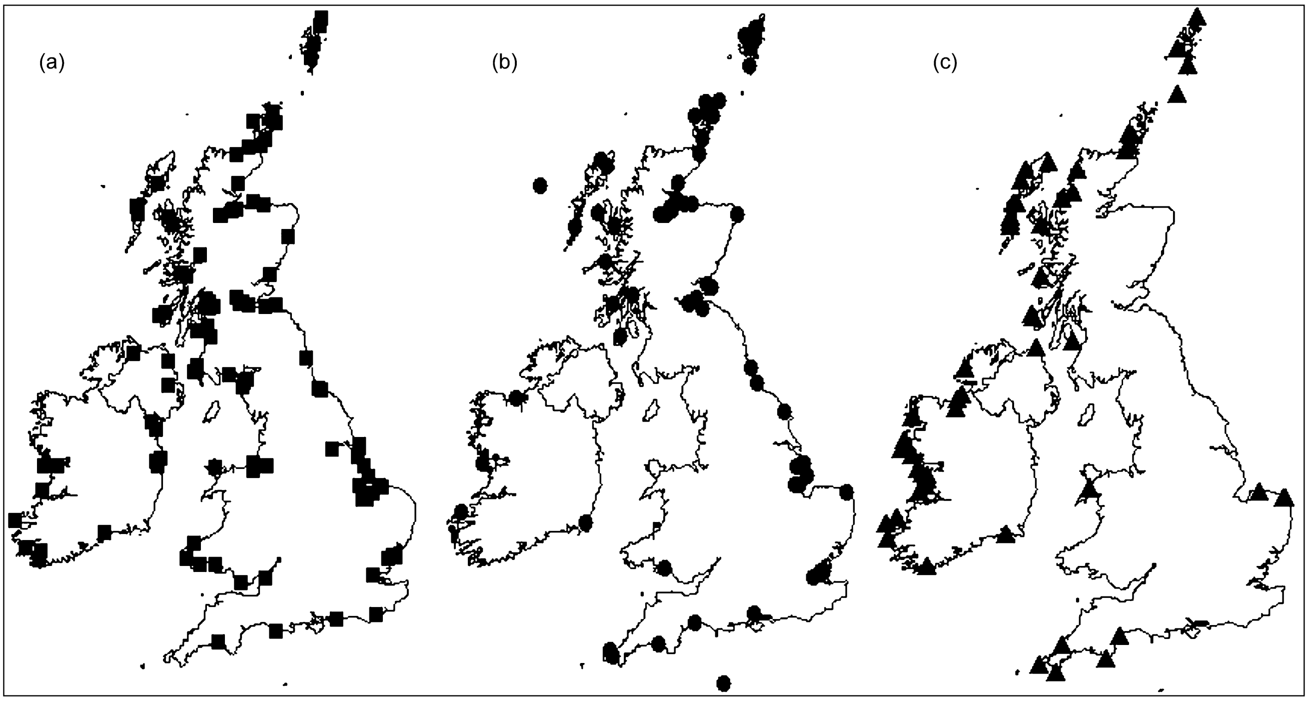

Fig. 2. Locations of strandings of: (a) northern bottlenose whales; (b) Sowerby's beaked whales; and (c) Cuvier's beaked whales on the coasts of the United Kingdom and the Republic of Ireland between 1800 and 2002.

Sowerby's beaked whales

While the number of strandings of Sowerby's beaked whales varied between months (Fig. 3b), with the highest numbers occurring between July and October, this did not differ significantly from an even spread across all months ($\chi^2 = 12.6$, d.f. = 11, $p = 0.323$). The number of strandings varied from four on south Atlantic coasts to 20 on northern North Sea coasts. Between the different regions, there were differences in the timing of the majority of strandings. On northern Atlantic coasts, most strandings occurred between March and August (11 out of 14) and on the mid-Atlantic coasts between July and September (6/8). Both the Irish Sea coasts and the southern Atlantic coasts had no obvious periods with higher numbers of strandings. On North Sea coasts, most strandings occurred between August and October (11/20) in the north and between July and October (10/13) in the south. Again, on English Channel coasts the total number of strandings (5 events) was too low to identify any definite peaks in occurrence. Three regions had a sufficient number of strandings to allow comparisons of the distribution of strandings with month. The distribution of strandings by month did not differ significantly between the northern Atlantic coasts and northern North Sea coasts ($D = 0.386$, $n_1 = 14$, $n_2 = 20$, $p > 0.05$) or southern North Sea coasts ($D = 0.489$, $n_1 = 14$, $n_2 = 13$, $p > 0.05$) or between the two North Sea regions ($D = 0.196$, $n_1 = 20$, $n_2 = 13$, $p > 0.05$). Therefore, the null hypothesis that there was no difference in the distribution of strandings in each month between these regions was accepted.

Cuvier's beaked whales

Strandings of Cuvier's beaked whales were highest in July and January ($n = 11$ in both) and lowest in October and November ($n = 1$ in both) and the pattern of strandings differed significantly from an even spread across all months ($\chi^2 = 30.6$, d.f. = 11, $p = 0.001$). Fewer strandings than expected occurred in April and May and between August and December and there were more strandings than expected in January and February and in June and July (significantly more in January and July: $\chi^2 = 8.59$, d.f. = 1, $p = 0.003$; $\chi^2 = 8.60$, d.f. = 1, $p = 0.003$ respectively). The number of Cuvier's beaked whale strandings varied from zero on the northern North Sea coasts to 21 on mid-Atlantic coasts. Only the three Atlantic regions and the Irish Sea coasts had more than three strandings per region. In all of these four regions, most strandings occurred between January and July (Fig. 3c). Only two regions, the mid and southern Atlantic coasts, had sufficient numbers of strandings to allow the frequency of strandings in each month to be compared. The difference between these regions was not significant ($D = 0.336$, $n_1 = 21$, $n_2 = 16$, $p > 0.05$) and therefore, the null hypothesis that there were no differences in the frequency of strandings in different months between the two regions was accepted.

In terms of the frequency of strandings in different months, northern bottlenose whales (NBW), Sowerby's beaked whales (SBW) and Cuvier's beaked whales (CBW) all differed significantly from each other (NBW vs SBW: $D = 0.224$, $n_1 = 106$, $n_2 = 69$, $p < 0.05$; NBW vs CBW: $D = 0.556$, $n_1 = 106$, $n_2 = 56$, $p < 0.05$ and SBW vs CBW: $D = 0.224$, $n_1 = 69$, $n_2 = 56$, $p < 0.05$). Therefore, in each case the null hypothesis was rejected and it was concluded that there were significant differences in the frequency of strandings in each month between each species. Northern bottlenose whales have a strong peak in strandings between July and October, while strandings of Sowerby's beaked whales are more evenly spread throughout the year. Cuvier's beaked whales differ from both of these species as most strandings occurred between January and July.

Other species

Of the remaining three beaked whale species recorded from the UK and Ireland, True's beaked whale has the most strandings records (7), all from southern Atlantic coasts and spread throughout the year. Gervais' beaked whale and Blainville's beaked whale were both recorded only once, Gervais' beaked whale from the southern Atlantic region in January and Blainville's beaked whale from the Irish Sea in July.

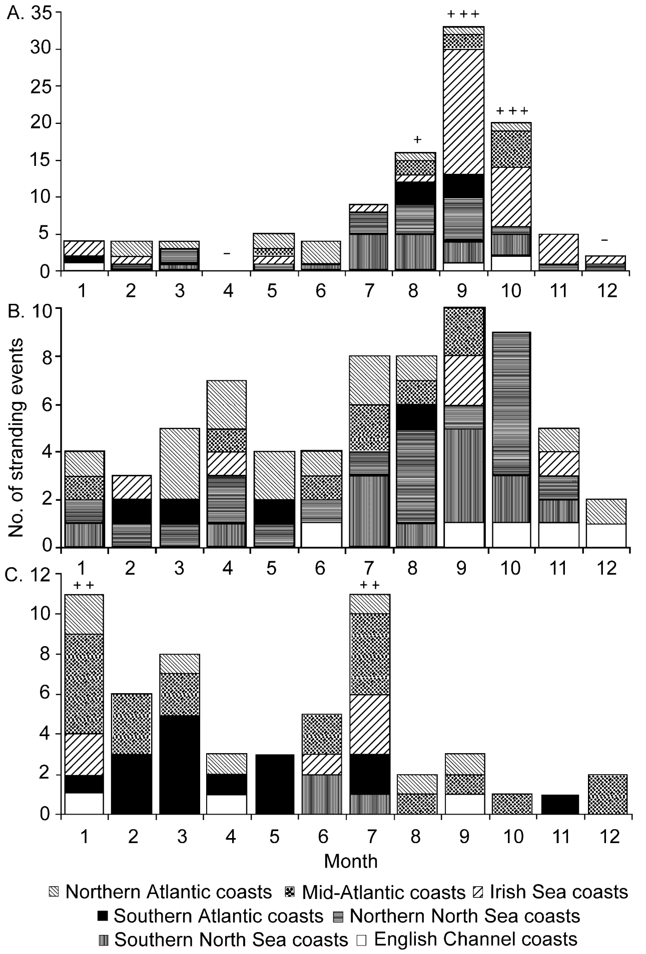

Fig. 3. Number of strandings for different beaked whale species in each month and region of the United Kingdom and the Republic of Ireland (1800-2002). Minus (-) signs indicate values which are significantly lower than expected if strandings were evenly spread throughout all months and plus (+) signs indicate significantly higher than expected (+ or —: $0.05 > p \geq 0.01$; ++ or – $0.01 > p \geq 0.001$; +++ or --- $p < 0.001$). A = Northern bottlenose whales; B = Sowerby's beaked whales; C = Cuvier's beaked whales.

Comparison of northern bottlenose and Cuvier's beaked

The number of northern bottlenose whale and Cuvier's beaked whale strandings in western regions (the three Atlantic regions and Irish Sea coasts) between 1913 and 2002 in four periods of the year (November to January, February to April, May to July and August to October) differed significantly from the expected values ($\chi^2 = 85.4$, d.f. = 3, $p < 0.001$ —Fig. 4). In this comparison, more Cuvier's beaked whale strandings occurred from November to July and significantly fewer than expected from August to October ($\chi^2 = 54.4$, d.f. = 1, $p < 0.001$). Northern bottlenose whales showed a reverse trend with fewer strandings than expected from November to July (significantly less in February to April: $\chi^2 = 13.1$, d.f. = 1, $p = 0.0003$) and significantly more than expected from August to October ($\chi^2 = 6.2$, d.f. = 1, $p = 0.013$).

DISCUSSION

Interpreting the distribution of living animals from strandings records is problematic, particularly when trying to infer seasonal variations in their distribution. This is particularly true of oceanic species, which are not usually found in coastal waters. An animal which dies far from shore may be carried long distances on surface currents or by prevailing winds. When stranding rates vary with season, this may reflect changes in species distribution, but it could also reflect changes in the likelihood of dead animals reaching the shore. In particular, seasonal changes in current speed, strength or direction, changes in wind strength and direction and changes in the frequency of storms could all affect the rate of stranding of oceanic species on the nearest coasts. Therefore, care must be taken that any seasonal changes in the distribution of a species inferred from seasonal changes in stranding patterns are both biologically plausible and are consistent with findings from other avenues of investigation.

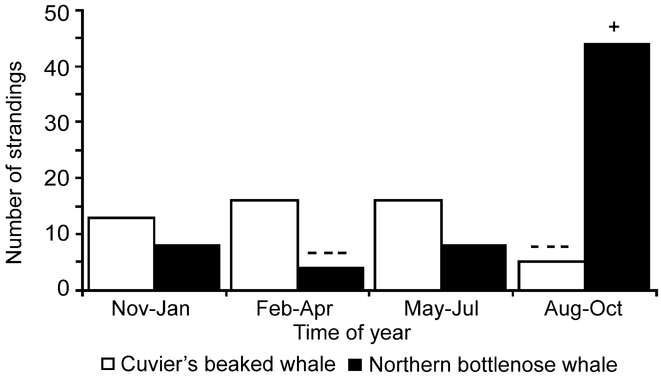

Fig. 4. Number of strandings in western regions of the United Kingdom and the Republic of Ireland (northern, mid and southern Atlantic and Irish Sea coasts) of northern bottlenose whales and Cuvier's beaked whales in three month periods between 1913 and 2002 (the period where systematic recording of strandings has occurred). Minus (-) signs indicate values which are significantly lower than expected if both species had the same seasonal pattern of strandings and plus (+) signs indicate significantly higher than expected. (+ or —: $0.05 > p \geq 0.01$; +++ or ---: $p < 0.001$).

Northern bottlenose whales

The variations in the numbers of individual beaked whales stranding in the different regions might reflect patterns of movements of beaked whales. For northern bottlenose whales, the variation in number of individuals stranding in different months suggests a period of movement around the coasts of the UK and Ireland between July and October, with a peak in September when most strandings occurred. Such movements may represent north-south migrations or offshore-nearshore movements, or a combination of the two. The significant difference in the frequency of strandings in each month between the northern Atlantic region and other regions to which it was compared, with the peak in strandings occurring earlier in the summer than more southern regions, suggests there may be a southerly component in the direction of movement of northern bottlenose whales at this time of year.

Stomach contents of stranded northern bottlenose whales in the North Sea in later summer and early winter contained mainly the hard remains (the chitinous mandibles) of the oceanic squid *Gonatus fabricii* (Clarke and Kristensen, 1980; Lick and Piatkowski, 1998; Santos *et al.*, 2001b). The prevalence of this cephalopod species in the diet, for which the southern limit of distribution is believed to be the southern Norwegian Sea (Bjorke, 1995), suggests that the whales had previously been feeding in the northern North Atlantic. Smeenk (1997) suggested that sperm whales (*Physeter macrocephalus*) which strand in the North Sea in winter are animals which have mistakenly entered this area during southward migrations in late autumn and that the

North Sea effectively functions as a trap for such oceanic species during southward movements. This hypothesis is consistent with the strandings of northern bottlenose whales in the North Sea reported here; the large number of strandings of northern bottlenose whales at this time of year may represent animals which have moved south from the northern North Atlantic and inadvertently entered the North Sea. While in the North Sea whales may try to continue their usual movement in a south-westerly direction (which would normally take them through the deep water to the west of the UK and Ireland) and as a result strand in areas which appear to offer 'outlets' to the south-west. This would explain why northern bottlenose whale strandings in the western North Sea are clustered in estuaries, such as the Moray Firth, the Firth of Forth, the Humber Estuary and the Thames estuary, which run inland in a westerly direction from the coast. This has also been suggested for sperm whales stranding in the North Sea (Santos *et al.*, 1999). The peak in strandings of northern bottlenose whales reported here for September occurs two months after reported movements of animals from the Norwegian Sea and peak catches around Iceland, and one month after the start of increased catches in the Faeroes, suggesting that the movements may be part of a larger-scale movement throughout the northern northeast Atlantic (Mitchell and Kozicki, 1975; Bloch *et al.*, 1996; Sigurjónsson and Vikingsson, 1998). This suggests that the peak in numbers of northern bottlenose whale strandings in late summer and autumn most likely reflects a southward movement of whales rather than a simple offshore-nearshore movement.

Northern bottlenose whales are generally sighted in deep waters beyond the shelf edge to the north and west of the British Isles. The southward movement in late summer and autumn suggested by the stranding pattern presumably takes place primarily in such waters. If this is the case, deep water areas, such as the Faroe-Shetland Channel, may form 'corridors' for movements of northern bottlenose whales. However, some animals may also move into shallower shelf waters during the southward movements. For example, whales are occasionally seen in the coastal waters between the Western Isles and mainland Scotland, particularly in August and September (Evans, 1992). The high numbers of strandings in the Irish Sea at this time of year suggests animals may also pass through this area as they travel south. There are at least three possible reasons for animals entering such coastal waters as they move south. These are: (1) navigational errors similar to those which take animals into the North Sea 'trap'; (2) animals following movements of prey which enter coastal waters at the time northern bottlenose whales move south; and (3) that such areas are used as 'short cuts' between deep water areas such as the Faroe-Shetland Channel and the Rockall Trough or the Bay of Biscay. Currently, there are insufficient data to test which, if any, of these possibilities is correct.

If the grouping of strandings in July to October for northern bottlenose whales represent animals which are moving south, there may be an opposite movement of animals at other times of the year. Strandings of northern bottlenose whales between January and March (Fig. 3a), may be representative of a second period of movement heading in a northward direction. This is supported by the fact that a male northern bottlenose whale which stranded in Denmark in February contained a mandible from the octopod *Vampiroteuthis infernalis* which has not been recorded further north than Spain (Santos *et al.*, 2001b) and had a more varied diet than those whales stranded later in the year. The smaller number of strandings early in the year may indicate either that animals move northward further offshore than their southward movements, as suggested by Evans (1980), or that animals are less prone to fall into navigational 'traps', such as the North Sea, when moving north and so strand in lower numbers.

Sowerby's beaked whales

Little is known about the distribution of Sowerby's beaked whales around the UK and Ireland. Only a small number of definite sightings have been recorded, all in deep waters beyond the shelf edge (e.g. Pollock *et al.*, 2000). Most strandings occur along North Sea coasts and this led some people to suggest that the North Sea area was the centre of distribution for this species (e.g. Moore, 1966). However, despite high levels of survey effort (e.g. Hammond *et al.*, 1995; 2002), there are no published definite sightings of Sowerby's beaked whales in the North Sea. In addition, most sightings of Sowerby's beaked whales which have been recorded have occurred in waters deeper than 500m (e.g. Hooker and Baird, 1999; Pollock *et al.*, 2000), compared with water depths of less than 200m available in the North Sea. Therefore, it is unlikely that the North Sea is a normal part of the range of this species (MacLeod, 2000). Although month-to-month variation in numbers of strandings was not statistically significant, most recorded strandings occurred between July and November with a possible secondary peak in April (Fig. 3b). In addition, as in northern bottlenose whales, strandings of this species on North Sea coasts in late summer and autumn were concentrated in estuaries. The similarities in the geographic distribution of strandings between this species and northern bottlenose whales suggests that the two species may have similar seasonal movements, and that Sowerby's beaked whales also move southward in late summer and autumn, with some animals getting caught in the North Sea trap, and northward in late winter and spring. However, the comparison of strandings frequency in each month suggests that although the geographic pattern may be similar between northern bottlenose whales and Sowerby's beaked whales, the extent and/or timing of these movements may differ. There is little available stomach contents information on Sowerby's beaked whales in the northeast Atlantic that could be used to infer where these animals had been feeding prior to stranding, and more data are needed.

Cuvier's beaked whales

In contrast to northern bottlenose whales and Sowerby's beaked whales, most Cuvier's beaked whale strandings occur in the first half of the year, between January and July. Within this time there are two apparent peaks, one between January and March and one in June and July (Fig. 3). In addition, there are very few strandings of Cuvier's beaked whales from North Sea coasts, where high numbers of strandings of the other two species occur. Furthermore, all the Cuvier's beaked whale strandings in the North Atlantic region have occurred between March and September. This strongly suggests that Cuvier's beaked whales have a different pattern of seasonal distribution than either northern bottlenose whales or Sowerby's beaked whales.

Firstly, these data suggest Cuvier's beaked whales do not routinely go far enough north to enter the North Sea 'trap' at any time, as indicated by the lack of strandings on North Sea coasts. Therefore, we propose that the normal northern limit of Cuvier's beaked whale in the northeast Atlantic is located somewhere around the latitude of the Shetland Islands (around 60°30'N) and that this species does not regularly occur in more northern areas of the northeast Atlantic, such as the Faroe-Shetland Channel or the Norwegian Sea. However, Cuvier's beaked whales may occur at more northern latitudes in the central North Atlantic as there have been a small number of strandings in Iceland (Petersen, pers. comm.). This difference in distributional limits between the northeast and central North Atlantic may reflect a similarity in sea surface temperatures in southern Iceland and Shetland. Secondly, the proposed northern limit in the northeast Atlantic may only be reached in spring and summer months, the only time when strandings occur on northern Atlantic coasts. This, combined with the bimodal peak in strandings suggests that Cuvier's beaked whales may move northward between January and March to the most northern waters of their range in the northeast Atlantic in spring and summer, before moving southward again from June onwards. The stomach of a Cuvier's beaked whale stranded in North Uist in the mid-Atlantic region in February contained no fresh remains of cephalopods and most of the species identified in the diet were similar to those found in the stomachs of two Cuvier's beaked whales stranded in northwestern Spain in February (Santos *et al.*, 2001a). This suggests that the Scottish individual had been feeding further south and may have recently moved into more northern waters before stranding.

Other species

The lack of strandings in any regions other than Ireland and the small total number of strandings suggests that True's beaked whales occur only in more southern waters to the south of Ireland, and possibly as far north as Irish waters on occasions. However, all the animals could have been transported from further south by the prevailing surface currents. Similarly, the strandings of Gervais' beaked whale and Blainville's beaked whale are consistent with a more southern distribution, either with a very occasional occurrence in southern UK and Irish waters or with these animals being transported by prevailing surface currents from further south.

Comparison of northern bottlenose and Cuvier's beaked

The strandings data suggest that there are differences between northern bottlenose whales and Cuvier's beaked whales in their patterns of occurrence around the UK and Ireland. Northern bottlenose whales occur further north than Cuvier's beaked whales, which do not routinely appear to occur north of Shetland in the northeast Atlantic, while northern bottlenose whales regularly enter the North Sea, presumably from waters north of Shetland (a possibility supported by stomach contents data). This is indicative of spatial segregation between the two species in the more northern regions of the northeast Atlantic. The seasonal distribution of strandings also suggests that there is some temporal segregation in the Atlantic waters to the west of the UK and Ireland. Cuvier's beaked whales appear to move northward into these waters in late winter and spring (again supported by stomach contents data) and southward out of the most northern waters in this area in summer. In contrast, the peak in strandings of northern bottlenose whales does not occur until late summer and early autumn, when animals may be moving southward into this area. These differences in spatial and temporal distribution of strandings between these two species are consistent with the hypothesis of spatio-temporal segregation between *Ziphius* and *Hyperoodon* to reduce potential competition for the same prey resources proposed by MacLeod *et al.* (2003).

ACKNOWLEDGEMENTS

This work would not have been possible without the efforts of all those who have contributed time and effort to recording strandings data from the UK and Ireland over the last two centuries and the authors would like to thank all who have participated in the collection of these data. In particular, the authors would like to thank Bob Reid and the Scottish Agricultural College, Inverness, for providing details of recent strandings in Scotland and Richard Sabin and the Natural History Museum, London, for providing similar details for recent strandings throughout the UK. Details of recent strandings in Ireland were found with the help of the online strandings database of the Irish Whale and Dolphin Group. Finally, the authors would like to thank Emer Rogan and an anonymous reviewer for their useful comments on earlier drafts of this work. The initial collation of strandings data was undertaken by CDM as part of a collaboration with Jessica Ward, Ellen Keene, Glenn Mitchell and Angela D'Amico and initiated by Dr. Robert Gisiner, Office of Naval Research. CDM is currently funded by the Office of Naval Research, Grant Number N00014-02-1-0253.

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J. CETACEAN RES. MANAGE. 6(1):87–99, 2004

Cetacean strandings in Oregon and Washington between 1930 and 2002

S.A. Norman[*], C.E. Bowlby[**], M.S. Brancato[**], J. Calambokidis[+], D. Duffield[++], P.J. Gearin[#], T.A. Gornall[##], M.E. Gosho[#], B. Hanson[#], J. Hodder[†], S.J. Jeffries[‡], B. Lagerquist[^^], D.M. Lambourn[‡], B. Mate[^^], B. Norberg[*], R.W. Osborne[^], J.A. Rash[##], S. Riemer[§] and J. Scordino[*]

Contact e-mail: Stephanie.Norman@noaa.gov

ABSTRACT

The Northwest Region (NWR) Marine Mammal Stranding Network was created in the early 1980s to provide a consistent framework in which to collect and compile data about marine mammal strandings in Oregon and Washington. The NWR includes the nearshore waters and 4,243km (2,632 n.miles) of coastline. For the years 1930-2002, there were 904 stranding events, representing 951 individual animals and 23 species: 4 species of balaenopterids, 1 eschrichtiid, 2 physeterids, 4 ziphiids, 10 delphinids and 2 phocoenids. Gender was determined for 343 males and 266 females. Only one mass stranding was recorded (sperm whales: 1979). A few species comprised the majority (71%) of stranding events in the NWR: harbour porpoise (34%), gray whales (23%), Dall's porpoise (12%) and Pacific white-sided dolphins (4%). There was a steep increase (511%) in the number of stranding reports beginning in the 1980s with over 86% of all records occurring during the last two decades (1980s and 1990s). The general trend of increased reported strandings during the last two decades corresponds to the formation of a formal stranding network and a heightened interest and dedication by the public and government agencies in reporting and documenting strandings. For all events combined, the primary stranding peak was April-July. Since stranding recoveries depend heavily on reports from the general public, most stranding records were in summer when more people are present along the coastline. Individual species or species groups showed varying levels of conformity to this overall seasonal trend. The value and limitations of the use of strandings data in a management context are discussed.

KEYWORDS: CETACEANS; STRANDINGS; DISTRIBUTION; OCEANOGRAPHY; GRAY WHALE; HARBOUR PORPOISE; DALL'S PORPOISE; WHITE-SIDED DOLPHIN; TRENDS; HABITAT; NORTH PACIFIC; SPERM WHALE; NORTH AMERICA

INTRODUCTION

A systematic effort to collect and compile data on marine mammal strandings in Oregon and Washington began with the formation of the Northwest Region (NWR) Marine Mammal Stranding Network in the early 1980s (1980-1981). The network is composed of volunteers based at academic institutions, state and federal wildlife and fisheries agencies, veterinary clinics, enforcement agencies and by individuals who respond to or provide professional advice on handling stranding events (Scordino, 1991). Stranding Network activities are coordinated by the National Marine Fisheries Service (NMFS), Marine Mammal Health and Stranding Response Program (MMHSRP)/Regional Coordinator based in Seattle, Washington, USA. Each stranding event is handled on a case-by-case basis because response capability varies between areas depending on available resources, personnel and logistics.

The NWR Stranding Network coverage area includes the nearshore waters and shoreline of Oregon and Washington north of 42°0'N and south of 49°0'N (the US/Canada border), including the inland waters of Washington State (Fig. 1). There are 3,767km (2,337 n.miles) of marine shoreline in Washington State and 476km (295 n.miles) of shoreline in Oregon.

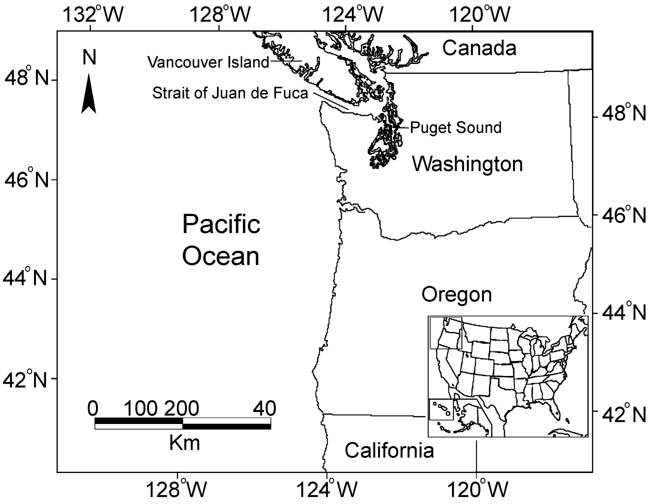

Fig. 1. Geographic area covered by the Northwest Region marine mammal stranding network.

The data collected from stranded cetaceans provide information on distribution, mortality and seasonal movements (e.g. Scheffer and Slipp, 1948; Fiscus and

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[*] *Northwest Regional Office, NMFS, NOAA, 7600 Sand Point Way NE, Bldg. 1, Seattle, Washington 98115-6349, USA.*
[**] *Olympic Coast National Marine Sanctuary, 138 W 1st Street, Port Angeles, Washington 98362, USA.*
[+] *Cascadia Research Collective, 218 ½ West 4th Ave., Olympia, Washington 98501, USA.*
[++] *Department of Biology, Portland State University, P.O. Box 751, Portland, Oregon, 97207, USA.*
[#] *National Marine Mammal Laboratory, Alaska Fisheries Science Center, NMFS NOAA, 7600 Sand Point Way NE, Seattle, Washington 98115-6349, USA.*
[##] *Marine Animal Resource Center, TAG Consulting, P.O. Box 1847, Vashon, Washington 98070, USA.*
[†] *Oregon Institute of Marine Biology, University of Oregon, P.O. Box 5389, Charleston, Oregon 97420, USA.*
[‡] *Washington Department of Fish and Wildlife, Marine Mammal Investigations, 7801 Phillips Rd. SW, Tacoma, Washington 98498, USA.*
[^^] *Oregon State University, Marine Mammal Program, Hatfield Marine Science Center, 2030 South Marine Science Drive, Newport, Oregon 97365, USA.*
[^] *The Whale Museum, PO Box 945, Friday Harbor, Washington 98250, USA.*
[§] *Oregon Department of Fish and Wildlife, P.O. Box 642, Gold Beach, Oregon 97444, USA.*

Niggol, 1965; Balcomb, 1980; Rice *et al*., 1986; Osborne and Ransom, 1988; Osborne *et al*., 1988; Ferrero and Tsunoda, 1989; Scordino, 1991; Ferrero *et al*., 1994). The entire cetacean stranding record for the NWR is summarised here, covering the years 1930 to 2002. Trends in stranding reports are analysed in relation to species composition and abundance, geographic and seasonal distribution, group size of stranded animals and gender. No attempts have been made to explain the cause of strandings except in general terms.

OCEANOGRAPHY OF THE NORTHWEST REGION

The influence of wind on carcass movement varies depending on carcass height above the water line, winds and water currents. In the NWR, winds are typically from the west/northwest during the summer and from the east/southeast during the winter. Wind transitions usually occur during April-May and October-November (Hickey, 1979). Cetacean carcass distribution can be influenced by these current and wind conditions, along with upwelling and downwelling. Coastal upwelling occurs most frequently in summer and fall when it is promoted by northerly and northwesterly winds. The upwelling season runs from April to October, with maximum intensity in July and August (Bakun, 1973), its effects extending to slope and offshore waters. Upwelling intensity is usually greatest along the southern Oregon coast and diminishes northward, although it can occur anywhere along the Oregon-Washington coast under favorable wind conditions. The Columbia River defines the coastal boundary between Oregon and Washington. Its effluent contributes to approximately 60% of the freshwater entering the Pacific Ocean between San Francisco and the Strait of Juan de Fuca in the winter, and up to greater than 90% in the remainder of the year, heavily influencing the oceanography of the area (Barnes *et al*., 1972).

The continental shelf (waters typically <200m deep) is less than 80km wide along the coast of Oregon and Washington. The continental slope (200-2,000m) is wider off Washington than Oregon (National Oceanic and Atmospheric Administration, 1988). There are a series of submarine canyons that transect the shelf and slope along the Washington coast but are absent off Oregon. Several rocky submarine banks occur off Oregon. The shelf between Washington and Vancouver Island is interrupted by the Strait of Juan de Fuca (National Oceanic and Atmospheric Administration, 1988).

McGowan (1974) describes the biogeography of the NWR as part of a transition zone, which includes the North Pacific and California currents where annual primary productivity is moderate, peaking in the late spring to early summer. Sea surface temperatures range from 13°-20°C in summer to 8°-17°C in winter (National Oceanic and Atmospheric Administration, 1988).

The oceanic current system in the NWR is comprised of the California Current, Davidson Current and California Undercurrent, and can vary interannually (Hickey, 1979). The California Current flows southerly beyond the continental shelf throughout the year, but is typically strongest during the summer (Hickey, 1979). In winter, this current moves offshore and is replaced by the northward flowing Davidson Current.

El Niño-Southern Oscillation (ENSO) events can influence sea surface temperature and current patterns in the NWR. Warm events of the equatorial Pacific Ocean generate significant sea surface temperature anomalies in North America (Aceituno, 1992; Bunkers *et al*., 1996; Hoerling and Kumar, 1997), which may lead to unusual distributions of cetacean species during years of abnormally warm water temperatures in the North Pacific (Osborne and Ransom, 1988; Ferrero and Tsunoda, 1989; Ferrero *et al*., 1994).

MATERIALS AND METHODS

Source of records and reporting effort

Records of cetaceans stranded alive or dead on the beach prior to 1989 were collected on an opportunistic basis and were not maintained in a computerised database. Records dating from 1989 have been maintained in a web-based database system at the National Marine Fisheries Service Marine Mammal Health and Stranding Response Program. Sources of records included unpublished reports provided by university and aquarium personnel, the general public, state and federal agencies and published reports. Reports prior to 1980 were accepted into this study only if the information provided allowed verification of species identification. Network volunteers responded to most ($n = 715$; 79%) of the total reported cetacean stranding events ($n = 904$) between 1930 and 2002.

The NWR stranding network often receives reports from the general public or US Coast Guard stating that a dolphin, porpoise or whale has been found stranded on the beach. A stranding network member may not have been able to respond to a report, but as much descriptive information as possible is obtained from the caller in an attempt to more specifically identify the animal. If the network member was not able to examine the animal at a later date, it was recorded as an 'unknown' odontocete, mysticete or cetacean.

Records of stranded cetaceans after 1980 came mostly from members of the NWR stranding network. Data included species, stranding date, location, length, body condition and gender. When species identification could be verified from non-network sources, it was included in this review. In instances when species could not be determined, regardless of source, the reports were tallied in one of several 'Unknown' categories based on the amount of information received. In this report, a stranding event is defined as one or more animals present on the beach at the same time and includes calves, but not foetuses. The last stranding included in this report took place on 5 November 2002. All data in this review are maintained at the NMFS Northwest Regional Office in Seattle. For all cetaceans that are physically examined, the only morphometric measurement requirement indicated on the NMFS stranding form is total straight length, which is measured from the tip of the snout to the fluke notch (if present) or centre of the trailing edge of the flukes (Norris, 1961). Individual responders do have established protocols for detailed measurements, as well as for tissue and skeletal sampling and archiving. However, these data and samples have been considered proprietary and to this point have been maintained by each individual responder. Contact information is available upon request through the NMFS Northwest Regional Office in Seattle.

RESULTS AND DISCUSSION

Factors affecting stranding frequency and distribution (other than the abundance and distribution of the animals themselves)

A number of authors have considered the possibilities and limitations of strandings information (e.g. Klinowska, 1985; IWC, 1986). It is possible that the proportions of species in the stranding record reflect the relative abundance of live animals of the species in the respective region (Sergeant, 1979; Woodhouse, 1991). For instance, most species that are relatively rare in the NWR are represented by a small number of strandings. However, strandings may also reflect nothing more than a general region of occurrence and may not be related to a specific habitat preference.

Strandings are highly dependent on physical oceanographic features that bring the carcass to shore. Currents and wind affect when and where (and if) an animal strands. Other environmental factors might influence carcass distribution: water temperature affecting decomposition rate, degree of buoyancy (e.g. some cetaceans might sink soon after death while others float) and biodegration/scavenging of the carcass before it reaches the shore. Animals may strand hundreds of kilometres from their normal range. The species that occur in the NWR frequently are either primarily cosmopolitan, or associated with the temperate/sub-Arctic, or mixed-water oceanographic regions (Rice, 1998). In the NWR, unusual distributions of cetacean species may be observed during years of abnormal influxes of warm water. This is most likely related to incursion of warm waters into this region, related to El Niño/El Niño-Southern Oscillation (ENSO) events, allowing some species to move temporarily into more northerly latitudes.

When an animal is found stranded, it must be determined whether it is a live (at least one animal alive when first observed) or dead (all animal(s) dead when first observed) stranding. It is important to try and determine if the animal arrived at the stranding location under its own power or if it died at sea and washed ashore with tides or currents (Klinowska, 1985). The vast majority of strandings in the NWR were dead strandings. Only 68 of 951 individuals were live-stranded and subsequently either died (n = 59) or were returned to the water (n = 9). In general, we conclude that the stranding of a cetacean in a certain area at a particular time does not necessarily mean that it is representative of live animal distribution or relative abundance.

Species

The total number of stranding events recorded for the NWR during 1930-2002 was 904, representing 23 species and 951 individuals (Table 1). In 7 events, more than 1 animal was involved. Although most were adequately identified, 97 animals could not be identified to species level. Four species of balaenopterids, 1 eschrichtiid, 2 physeterids, 4 ziphiids, 10 delphinids and 2 phocoenids stranded in the NWR. Four species comprised the majority (71%) of stranding events in the NWR: harbour porpoise, *Phocoena phocoena* (34% —Table 1; Fig. 2); gray whale, *Eschrichtius robustus* (23% —Table 1; Fig. 3); Dall's porpoise, *Phocoenoides dalli* (12% —Table 1; Fig. 4); and Pacific white-sided dolphin, *Lagenorhynchus obliquidens* (4% — Table 1; Fig. 5). Stranding events involving multiple animals occurred in four species: harbour porpoise, killer whale (*Orcinus orca*), rough-toothed dolphin and sperm whale (*Physeter macrocephalus*). Only the sperm whale had a mass stranding (47 individuals), whereas the remainder of the multiple strandings were of just two individuals.

Specific protocols for examinations and necropsies differ from examiner to examiner depending on the nature of the investigative inquiry, the experience of the examiner(s), the ultimate analysis envisioned for the samples collected, and the size or species involved. Measurements for total body length were recorded for 748 (79%) stranded individuals, however, 120 (16%) of these values were estimated lengths.

Table 1

Species and occurrence of cetaceans stranded in Oregon (OR) and Washington (WA) from 1930-2002.

	Stranding events			Individuals stranded		
Species	Total	OR	WA	Male	Female	Unknown
Balaenopterid						
Blue whale	1	0	1	0	1	0
Fin whake	8	2	6	3	5	0
Humpback whale	6	4	2	2	1	3
Minke whale	21	7	14	8	8	5
Eschrichtiidae						
Gray whale	200	54	146	85	63	52
Physeteridae						
Sperm whale	18	7	11	21	33	4
Pygmy sperm whale	8	4	4	6	1	1
Ziphiidae						
Baird's beaked whale	4	0	4	0	2	2
Cuvier's beaked whale	17	13	4	9	2	6
Stejneger's beaked whale	13	7	6	5	4	4
Hubbs' beaked whale	2	1	1	0	1	1
Delphinidae						
Killer whale	16	9	7	8	5	6
False killer whale	3	1	2	2	0	1
Short-finned pilot whale	7	1	6	3	1	3
Risso's dolphin	9	4	5	5	1	3
Pacific white-sided dolphin	34	20	14	18	10	6
Bottlenose dolphin	1	0	1	1	0	0
Common dolphin	5	4	1	3	1	1
Rough-toothed dolphin	3	1	2	2	2	0
Striped dolphin	12	10	2	8	4	0
Northern right whale dolphin	8	4	4	2	3	3
Phocoenidae						
Harbour porpoise	303	162	141	105	93	108
Dall's porpoise	107	14	93	44	25	38
Unknown species						
Unknown ziphiid spp.	5	5	0	3	0	2
Unknown odontocete	44	26	18	0	0	45
Unknown mysticete	1	1	0	0	0	1
Unknown cetacean	48	32	16	0	0	49
Total	**904**	**393**	**511**	**343**	**266**	**342**

The most common balaenopterid stranding was of minke whales, *Balaenoptera acutorostrata* (Table 1; Fig. 6). Four humpback whales (*Megaptera novaeangliae)* stranded in Oregon, and two stranded in Washington. All of these strandings occurred on the outer coast, and in Oregon the strandings occurred in the mid-to southern half of the state. On the other hand, 3 of the 8 fin whales (*Balaenoptera physalus*) strandings in Washington occurred inside Puget Sound. These three fin whales had been struck by ships and were presumably carried into the Sound. Only one blue whale (*Balaenoptera musculus*) has stranded in the NWR. The death of this animal may also have been caused by a ship strike, as it was draped around the bow of a freighter. The strike was theorised to have occurred off California along the freighter's route. The animal was a 16.2m female and based on lengths at sexual and physical maturity of females (22.5m and 24.8m, respectively, for the North Pacific; Omura, 1955; Ohsumi, 1979), this animal was probably a subadult.

Killer whales (*Orcinus orca*) are widely distributed in the NWR in small numbers (Leatherwood and Dahlheim, 1978) and stranded infrequently. Populations in the NWR are divided into 2 distinct 'forms' called *resident* and *transient* (Baird and Stacey, 1988; Baird *et al*., 1992; Hoelzel *et al*., 1998; Ford *et al*., 2000). The residents can be further divided into 3 geographically-based communities: northern and southern residents and offshore whales, the latter two of which are found most commonly in NWR waters (Bigg *et*

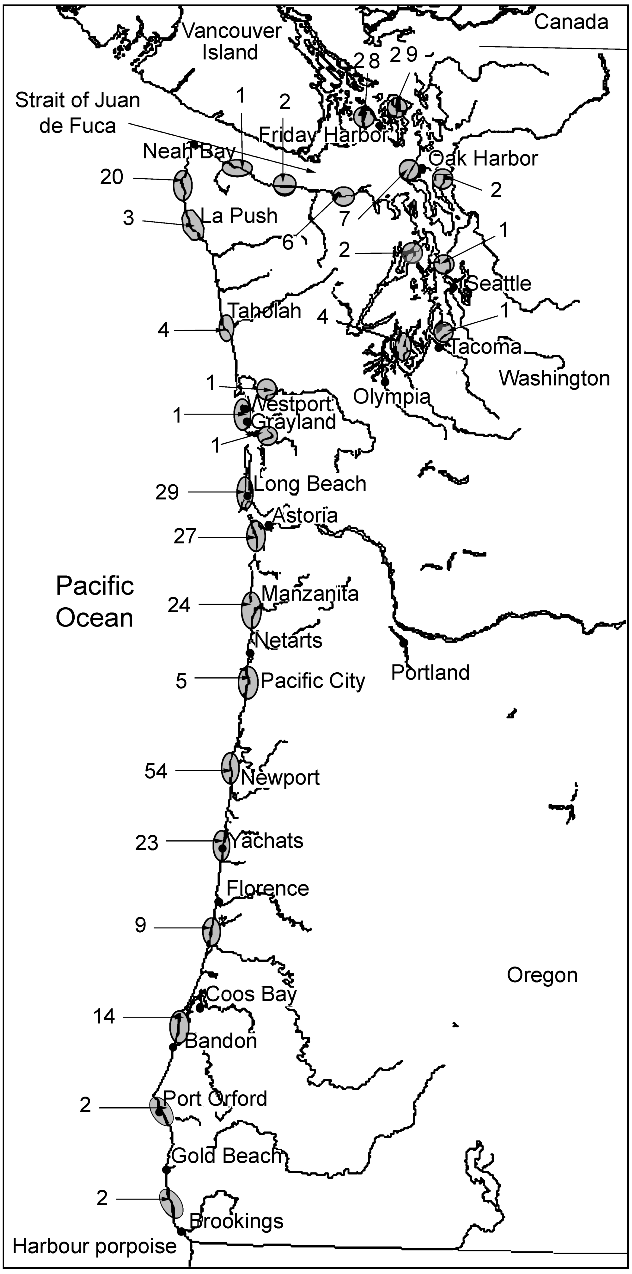


Fig. 2. Spatial distribution of stranded harbour porpoises in Oregon and Washington (1930-2002).

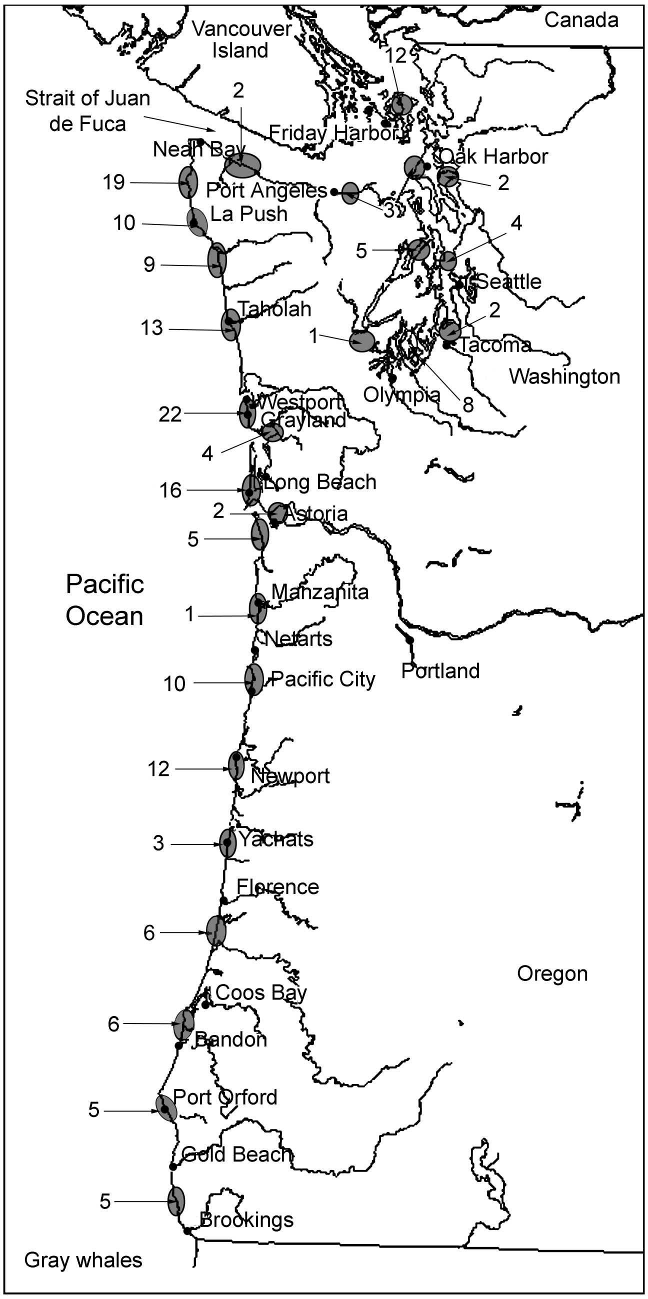


Fig. 3. Spatial distribution of stranded gray whales in Oregon and Washington (1930-2002).

al., 1987; Baird, 2001). Six of the 17 (35%) individual stranded killer whales were confirmed as southern residents (Osborne, 1999). Two of the individual stranded killer whales in Oregon were confirmed as transient (Stevens *et al.*, 1989).

Five of the species that stranded in the NWR are considered rare inhabitants due to their normal preference for warm temperate and tropical waters: short-finned pilot whale (*Globicephala macrorhynchus*), false killer whale (*Pseudorca crassidens*), bottlenose dolphin (*Tursiops truncatus*), common dolphin (*Delphinus* spp) and rough-toothed dolphin (*Steno bredanensis*). Most of their strandings events ($n = 13$) occurred during or within a year of an El Niño year(s) (Table 2). Their presence is thus considered extralimital rather than an extension of their range. Examples of unusual extralimital strandings in the NWR are bottlenose dolphins, common dolphins and rough-toothed dolphins (Osborne and Ransom, 1988; Ferrero and Tsunoda, 1989; Ferrero *et al.*, 1994).

Reporting efficiency

There are strong geographical and seasonal biases in stranding reporting efficiency and effort. Whether cetacean strandings are recorded depends upon many factors including human activity and awareness, the physical environment and climate, and seasonal animal movements. Seasonal movements of animals into the NWR do account for the rise in strandings of some species as do an increased number of visitors to coastlines during these same months, leading to more frequent reporting. Strandings in the NWR were recorded most frequently in regions with high human population or activities, particularly near towns or areas popular with vacationers, such as the San Juan Islands in northern Puget Sound and along the Oregon coastline. The general trend of increased reported strandings during the last two decades (Table 3) corresponds to the formation of a formal stranding network and a heightened interest and dedication by the public and government agencies in reporting and documenting strandings.

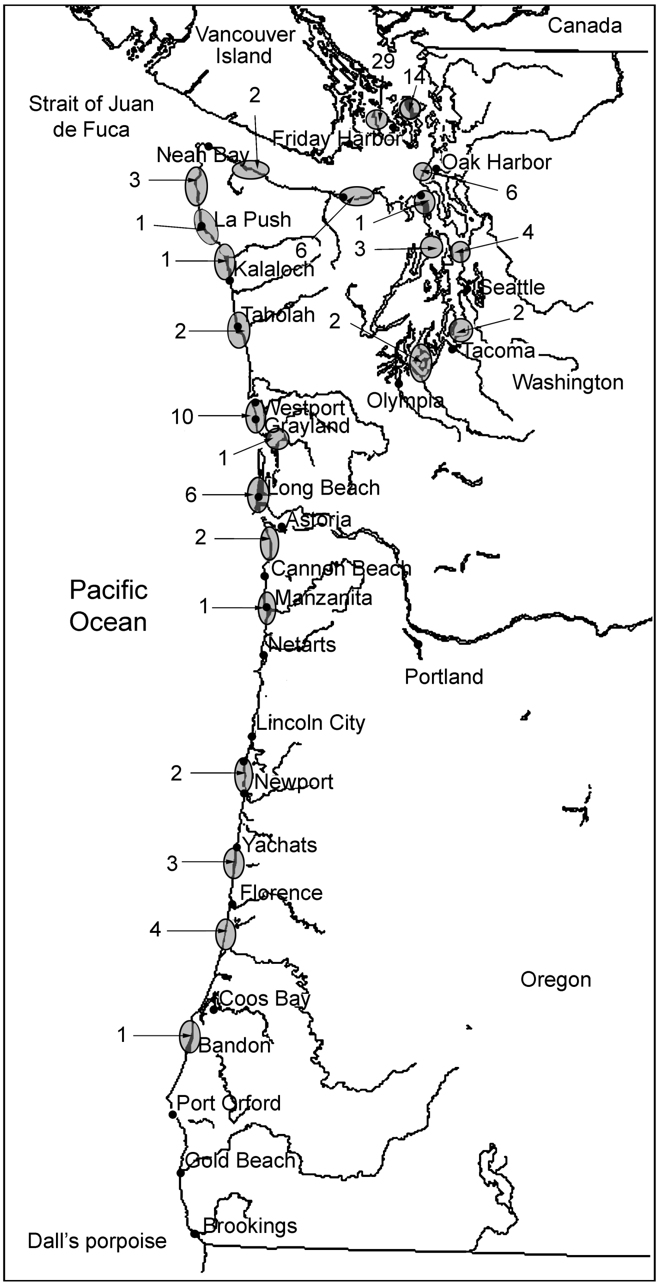


Fig. 4. Spatial distribution of stranded Dall's porpoises in Oregon and Washington (1930-2002).

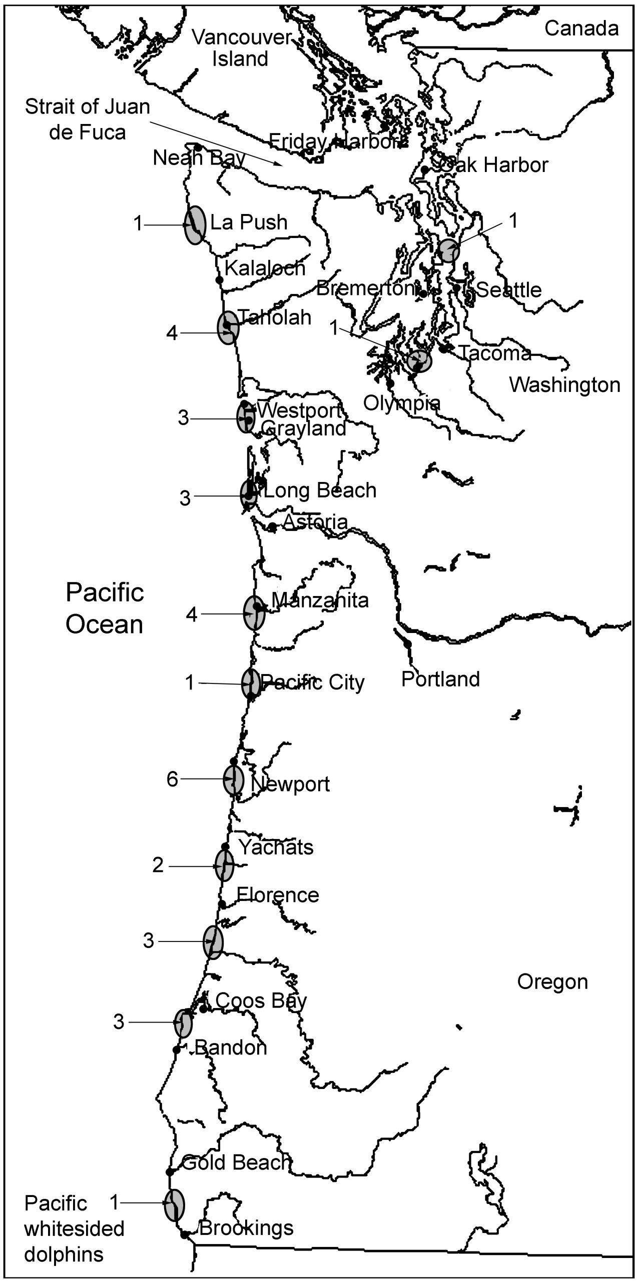


Fig. 5. Spatial distribution of stranded Pacific white-sided dolphins in Oregon and Washington (1930-2002).

Temporal distribution of strandings

Seasonality

In the NWR, cetacean strandings were recorded throughout the year, although generally there were more strandings reported from May to September (Fig. 7). This is probably due to one or more of (1) increased presence of the public at the coast; (2) increased abundance of certain species during this period; (3) oceanographic features (e.g. wind speed and direction, currents or upwelling/downwelling – see the 'Oceanography' section above). This general trend, was not applicable to all species or species groups (see below). Coastal upwelling occurs most frequently in summer and autumn when it is promoted by northerly and northwesterly winds. The upwelling season runs from April to October, with maximum intensity in July and August (Bakun, 1973). Conclusions regarding seasonality of strandings could not be drawn for species with small stranding sample sizes (<15 stranding events has been arbitarily chosen). Seasonal distribution of stranding events was analysed for species in which the total sample size was >15 over the whole period covered in this report (Table 4).

For species listed in Table 4, actual seasonal distribution was compared to an expected even distribution across all seasons using a Chi-squared test. Seasonal stranding patterns differed significantly ($P < 0.001$) from expected even seasonal distribution for harbour porpoise, gray whales and Dall's porpoise, whereas the other species showed no significant differences.

Spring (March-May)

There are several species that stranded most frequently in the spring months (Table 4). Although a small portion of the gray whale population spends the summer along the Pacific coast between Vancouver Island and central California (Flaherty, 1983; Sumich, 1984; Calambokidis and Quan, 1999), most gray whales migrate along the coast in the NWR travelling between Mexico and the Bering and Chukchi

Table 2

Strandings of extralimital species in the NWR relative to ENSO years. An ENSO year is the year leading into the winter of the mature event (Bunkers *et al.*, 1996).

Species	Stranding date	ENSO year
Short-finned pilot whale	1 September 1937	1939
	24 March 1968	1968
	12 November 1977	1977
	7 December 1980	1982
	14 March 1996	1997/1998
	2 June 1998	1998
	15 August 2002	2002
False killer whale	15 May 1937	1939
	15 May 1984	1986
	5 May 1987	1986
Bottlenose dolphin	9 March 1988	1986
Common dolphin	February 1942	1941
	6 March 1976	1977
	24 November 1983	1982
	25 November 1985	1986
	10 January 1996	1997/1998
Rough-toothed dolphin	August 1980	1982
	14 January 1991	1991
	4 October 1992	1991

Seas. However, they migrate closest to the NWR coastline during the spring months (April–June) when most of their strandings are observed (Fig. 8b). Animals located in the far north Arctic region (e.g. north central Bering Sea) during the summer months usually begin migrating south in late autumn to early winter (Rugh *et al.*, 2001). Surveys have been conducted off the Washington coast during winter to ascertain whale distribution there that time of year, as it appears whales are also present across the continental shelf during periods of non-migration (Shelden *et al.*, 1999). Subadults (n = 29; 32%) and adults (n = 27; 30%) represented over half (62%) of the gray whales that stranded in the spring, based on age classes defined in Norman *et al.* (2000).

Although killer whales are present year-round in Washington waters, they are most commonly sighted in Puget Sound during summer and early autumn (Leatherwood *et al.*, 1982). They have been reported off the Washington coast during April (Fiscus and Niggol, 1965); however, data on winter distribution are lacking (Baird, 2001). Killer whale populations in the NWR are divided into two distinct 'forms' called *residents* and *transients* (Baird and Stacey, 1988; Hoelzel *et al.*, 1998; Ford *et al.*, 2000). The residents can be further divided into three geographically-based communities: northern and southern residents and offshore whales, the latter two of which are found most commonly in NWR waters (Bigg *et al.*, 1987; Baird, 2001). Of the killer whale strandings, 41% (*n* = 7) stranded in the spring; four of which were neonates or young calves. The number of calf strandings is not surprising given this age class is especially vulnerable to disease, predation and separation from the pod.

Although Dall's porpoise strandings were reported in every month, the highest numbers were in spring (n = 47; 44%; Table 4; Fig. 8a).

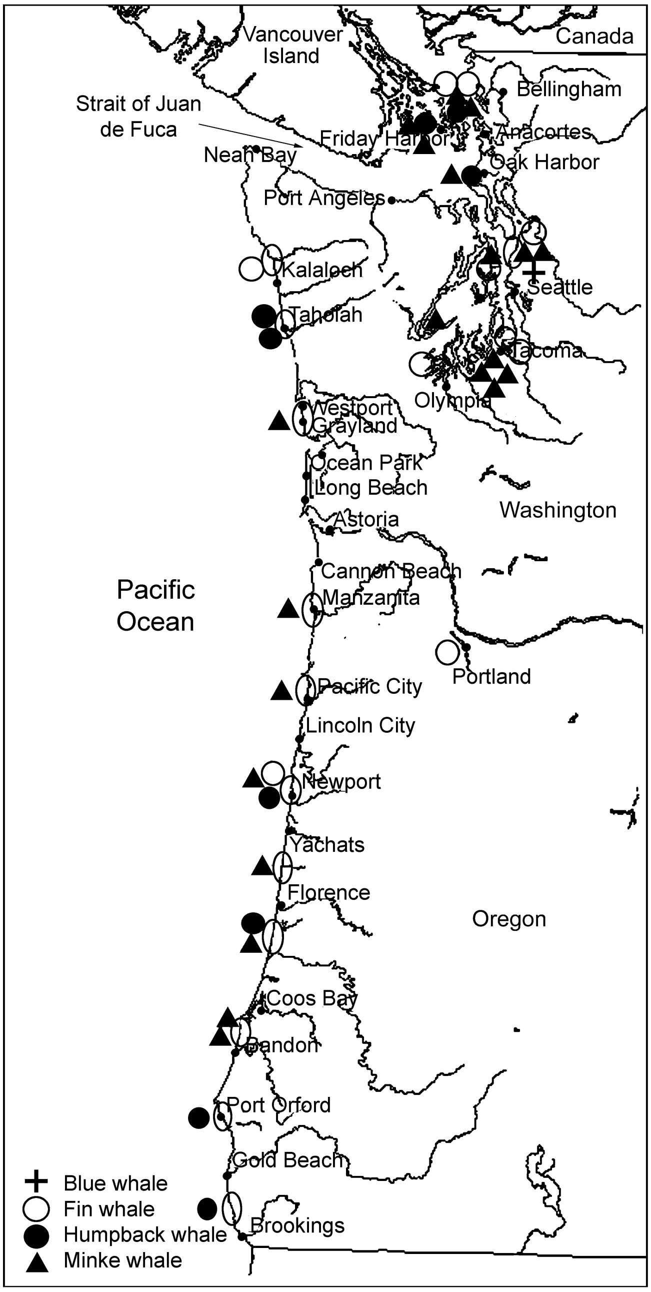


Fig. 6. Spatial distribution of stranded balaenopterid whales in Oregon and Washington (1930-2002).

Table 3

Summary of stranding events and annual strandings rate by decade (all species combined).

Period	Total strandings	Annual rate	SD
1930-39	6	0.6	0.8
1940-49	10	1.0	1.73
1959-59	1	0.1	0.3
1960-69	4	0.4	0.66
1970-79	52	5.2	5.67
1980-89	325	32.5	12.7
1990-99	395	39.5	7.65
2000-2002	111	37.0	9.09

Summer (June-August)

The number of harbour porpoise strandings is highest in July and August (Fig. 8c) and January (see below). This may be partially due to the summer gillnet fishery 1 May – 15 September, with peak landings of chinook salmon in July and August in north Washington and along the southwest coast of the Strait of Juan de Fuca (Gearin *et al.*, 1994). The

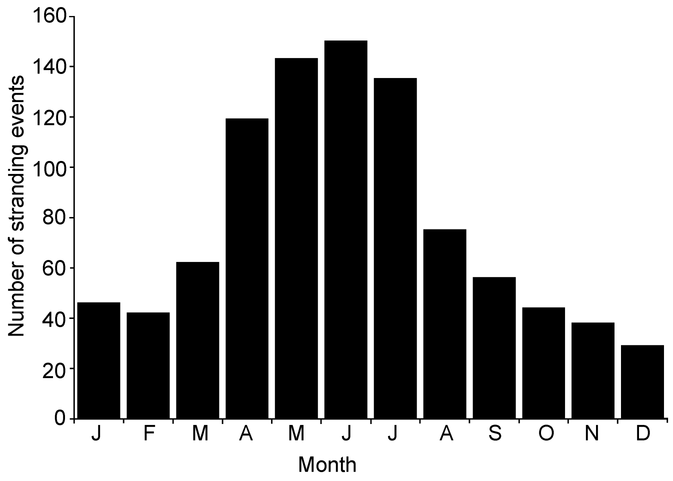


Fig. 7. Monthly distribution of cetacean stranding events in Oregon and Washington (1930-2002).

Table 4

Seasonal distribution of stranding events for species where the total sample size is 15+ over the whole period.

Species	Total	Spring (Mar-May)	Summer (Jun-Aug)	Autumn (Sep-Nov)	Winter (Dec-Feb)
Minke whale	21	11 (52%)	6 (29%)	3 (14%)	1 (5%)
Gray whale	200	105 (53%)	60 (30%)	12 (6%)	22 (11%)
Sperm whale	18	6 (33%)	3 (17%)	6 (33%)	3 (17%)
Cuvier's beaked whale	17	7 (41%)	4 (24%)	2 (11%)	4 (24%)
Killer whale	16	7 (44%)	1 (6%)	3 (19%)	5 (31%)
White-sided dolphin	34	9 (26%)	8 (24%)	8 (24%)	9 (26%)
Harbour porpoise	303	76 (25%)	150 (50%)	46 (15%)	31 (10%)
Dall's porpoise	107	47 (44%)	34 (32%)	16 (15%)	10 (9%)

seasonal distribution of harbour porpoises is unknown, but Barlow (1987) observed higher densities of harbour porpoises in northern Oregon and Washington in a September survey compared to surveys completed in January and February. In a year-long survey conducted by Calambokidis *et al.* (1987) in the Strait of Juan de Fuca, harbour porpoises were the most commonly sighted cetacean with the most numerous sightings recorded in autumn (specifically September). Based on the latter survey, one would expect to see more harbour porpoise strandings in September, but this may not be the case due to the fact that reporting effort is more efficient in the summer months due to increased numbers of individuals inhabiting the coastlines and encountering stranded animals at this time of year.

Dall's porpoise also show higher numbers of stranding events in the spring and summer ($n = 81$, 75%; Fig. 8a); although at least in Puget Sound they occur year-round (Miller, 1989; 1990). In Calambokidis *et al.* (1987), an insufficient number of Dall's porpoise sightings were made to make inferences about seasonal distribution. However, Everitt *et al.* (1980) noted that although this species has been sighted throughout the inland waters of Washington State year-round, it was more abundant during the spring and summer months.

The seasonal distribution for most ziphiids is not well defined. Therefore, no reliable inferences could made from the stranding data for these species other than that more beaked whales were reported stranded in the spring and summer months, presumably due to better weather and increased human presence along the coastline during these times of year.

Autumn/Winter (September-February)

Pygmy sperm whale (*Kogia breviceps*), common dolphin and Risso's dolphin (*Grampus griseus*) stranded primarily in the autumn and winter months ($n = 5$, 4, 5 events, respectively). These strandings most likely represent extralimital occurrences of these species that usually inhabit warm temperate and tropical water rather than representing populations found in the NWR. Of the killer whale strandings, 50% ($n = 8$) occurred during these months.

No seasonality

Sperm whale strandings occur throughout the year. During the summer months, this species can be found anywhere in the North Pacific. They were seen in every season except winter (Dec.- Feb.) in Washington and Oregon (Green *et al.*, 1992). Mate (1981) has found sperm whales to be relatively common off the coast of Oregon between June and September. This observation was not based on formal surveys, but rather on sighting information gathered while at sea for other projects.

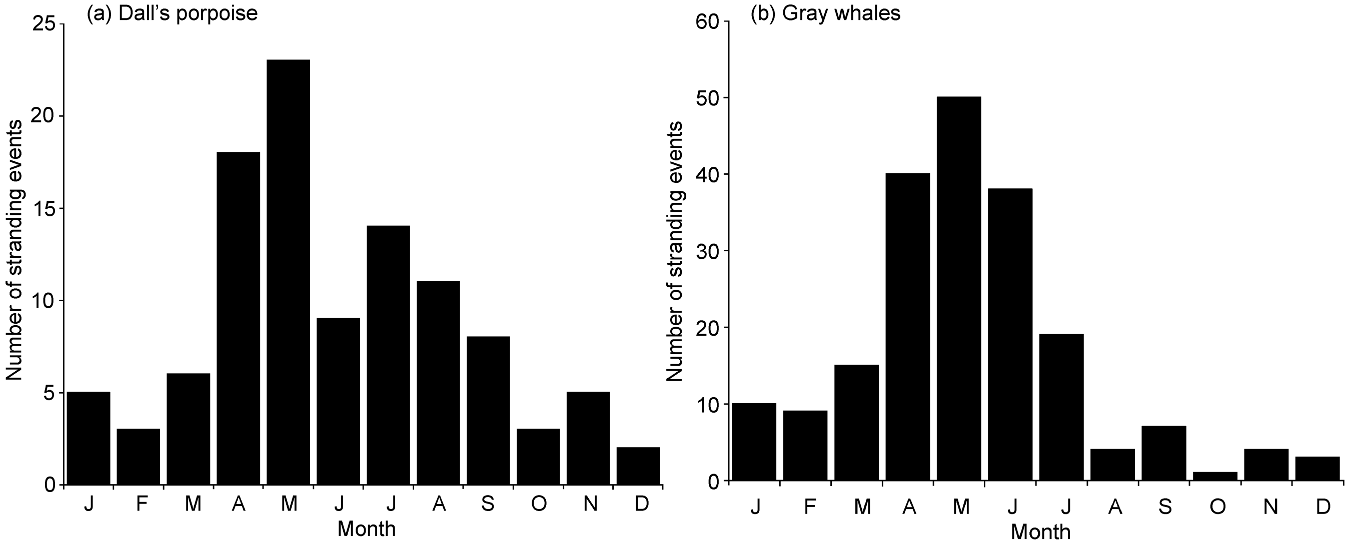
Minke whales stranded in almost every month of the year in Washington, which seems to support a year-round presence of this species in the region. In survey efforts by Everitt *et al.* (1980), most observations of this species were made during the spring and summer months, although sightings did occur in all months except February and November. The reduction in number of autumn and winter sightings may reflect a reduction in sighting effort and efficiency rather than a seasonal reduction in numbers.

Pacific white-sided dolphins were the most abundant cetacean sighted in slope and offshore waters of Oregon and Washington during aerial surveys conducted in 1992 by Green *et al.* (1993) during the months of March-May. Pike and MacAskie (1969) noted this species annually moves inshore in winter and offshore in summer, with inshore densities highest in autumn. Strandings have occurred in every month except April, which may be an anomaly (Fig. 8d).

Four of the fin whale strandings are noteworthy since they occurred in the autumn months (Sep — Nov), outside the usual period of sighting this species in coastal northwest waters (Leatherwood *et al.*, 1982). Three of the four were animals struck by ships (a fourth fin whale was struck by a ship in summer — August 2002). Two of the ships originated from the Alaskan Peninsula (Dutch Harbor) and the third from Japan. Both of these ships crossed the Gulf of Alaska and arrived in Puget Sound waters with the whale draped over the bow of the ship. It was presumed the whales were struck somewhere in the Gulf of Alaska or near the entrance to the Strait of Juan de Fuca. The circumstances of the fourth autumn stranding were not described (Scheffer and Slipp, 1948). Likewise, of the 4 humpback whales that stranded in Oregon, 1 stranded in December which is also outside the usual season in which this species is observed in this area (Calambokidis *et al.*, 1996).

No inference possible

No inferences about seasonality could be made for species with small sample sizes (e.g. < 15 stranding events). Some of the species such as the false killer whale, short-finned pilot whale, bottlenose dolphin, common dolphin and rough-toothed dolphin are considered rare inhabitants and usually prefer warm temperate and tropical waters.



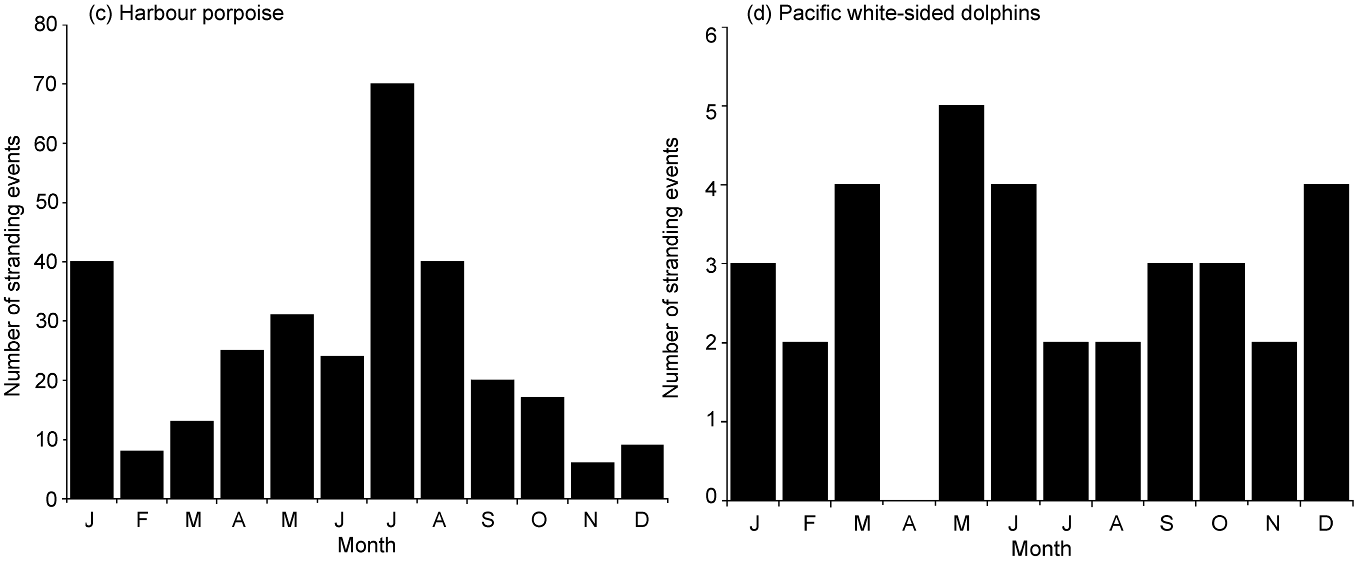


Fig. 8. Temporal distribution of the four most commonly stranded cetacean species in Oregon and Washington (1930-2002).

Spatial distribution of strandings

The spatial distribution of stranding events differed between the two states as well as within each state. The majority of stranding events took place in Washington ($n = 511$; 56%) compared to Oregon ($n = 393$; 44%). Within Washington (Fig. 9), three areas showed a higher percentage of total stranding events: (1) Willapa Bay/Long Beach peninsula (28%); (2) San Juan Islands archipelago (25%); and (3) the far northwest coast of the state near (11%). In Oregon (Fig. 9), the areas with the highest percentage of total stranding events were: (1) northern and central Lincoln County (28%); (2) Clatsop/northern Tillamook Counties (24 %); (3) southern Lincoln County/northern Lane County (11%); and (4) Coos County (9%). In Washington, areas (1) and (2) have high percentages of strandings due to: increased numbers of certain marine mammal species moving inshore in the summer (e.g. Pacific white-sided dolphins, killer whales), resulting in higher stranding numbers, and due to the increased human population in the same months, leading to increased reporting efficiency and effort. In area (3), there has been seasonal stranding coverage due to the presence of NMFS biologists in that area on a yearly basis every month of the year. In Oregon, areas (1-3) are the most populated areas of the coastline, with increased reporting effort during the summer months and increased numbers of marine mammals moving inshore at this time of year. Area (4) in the southern half of Oregon receives many stranding reports presumably due to the proximity of the slope waters to the coastline compared to the rest of the region, and greater upwelling intensity in this area, both of which may bring cetaceans closer to the coastline (Bakun, 1973). Fifty of the stranding events did not have specific enough geographic locale information for determination of stranding location.

Trends in the geographic distribution of stranding events are evident for some species or species groups. For instance, gray whales stranded along the coastline of both states, but most occurred on Washington's outer coast (Fig. 3). This species experienced an unusual mortality event during 1999 and 2000, when 32 and 25 animals, respectively, stranded in the NWR relative to annual averages of 6/year (SD = 32.2). This standard deviation incorporates upward bias since many years with possible zero stranding rates are not included. It is unknown whether those years had a true zero stranding rate or lack of reporting. The role of ship strikes, disease and biotoxins as factors in this mortality event could not be assessed as too few carcasses were sampled adequately to assess these factors. Intensive gray whale foraging may have caused localised prey depletion, or

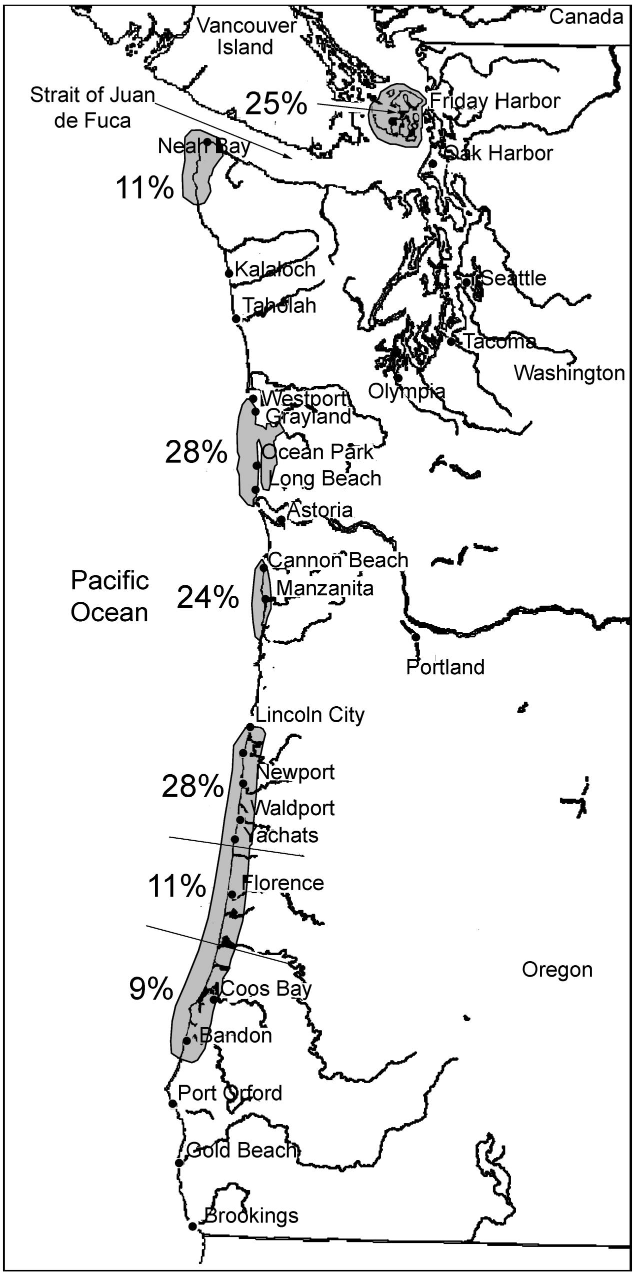


Fig. 9. Regions with the highest percentages of reported cetacean stranding events in Oregon and Washington (1930-2002).

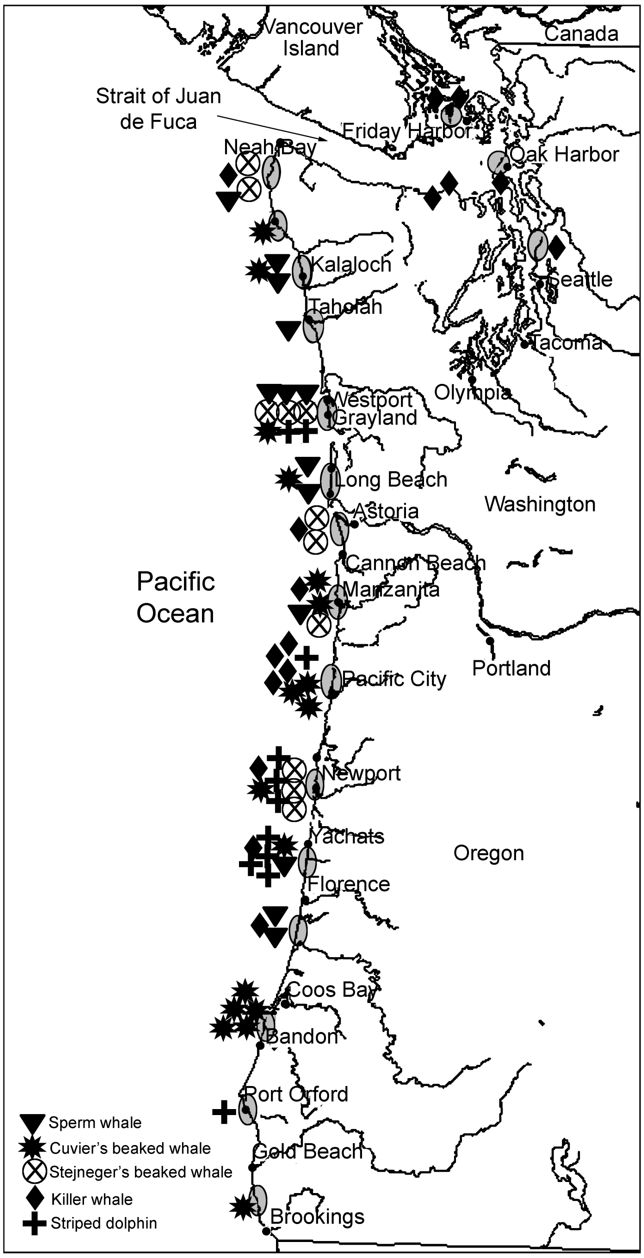


Fig. 10. Spatial distribution of the more commonly stranded delphinids and ziphiids in Oregon and Washington (1930-2002).

environmental changes such as the El Niño event in 1998, or longer-term climatic changes, could have resulted in shifts in prey availability in the summer feeding grounds (Le Boeuf *et al.*, 2000; Moore *et al.*, 2001; 2003).

The deep-diving species (Families Physeteridae, Kogiidae and Ziphiidae) were recorded along the entire coast of Oregon and outer Washington, but stranded more commonly in Oregon (Table 1; Fig. 10). All strandings of Baird's beaked whale (*Berardius bairdii*) occurred in Washington, while 75% of Cuvier's beaked whale strandings took place in Oregon. Strandings of Stejneger's beaked whale were evenly distributed between the two states Although sample sizes in these species are very small, reasons for their spatial distribution may be: (1) the close proximity of the slope waters (suitable habitat for beaked whales) to the shore in Oregon, versus Washington; (2) in Washington State, the continental shelf is furrowed by at least seven submarine canyons which may also be suitable habitat for beaked whales. Fourteen out of 15 (93%) strandings in Washington State were located on a beach across from a submarine canyon; and (3) winds and currents may affect distribution of carcasses onto the shore.

Of delphinid strandings, Pacific white-sided dolphins were the most numerous (Table 1; Fig. 5). They were the most abundant cetacean sighted off of Oregon and Washington in a survey conducted in April-May (Green *et al.*, 1993), with greater numbers sighted off Oregon than Washington. Strandings occurred with a greater frequency in Oregon versus Washington. This may be due to their preference for shelf and slope waters (Stacey and Baird, 1994), which tend to occur closer to shore in Oregon. Killer whale numbers were fairly well distributed between Oregon and Washington (Fig. 10).

No inferences could be drawn on spatial stranding distribution of species with very small sample sizes. Species such as the bottlenose dolphin, rough-toothed dolphin, common dolphin, false killer whale, short-finned pilot whale and striped dolphin (*Stenella coeruleoalba*) are more likely to strand in the NWR when stretches of warm water reach northward.

Despite the small sample size for Risso's dolphins some inferences can be made about regional sightings of the species. They occur in the slope and offshore waters of Oregon and Washington (Green *et al.*, 1992) and are represented by a fairly even distribution of strandings between the two states.

Harbour porpoises were the most numerous stranded cetacean in Oregon and the second most common in Washington. In Oregon, harbour porpoises stranded most commonly in the northern and central parts of the state (Fig. 2). In Washington, Dall's porpoise strandings were concentrated within Puget Sound (Fig. 4). The large number of harbour and Dall's porpoise strandings in the NWR supports what is known about their abundance and distribution in this region (Leatherwood *et al*., 1982; Barlow, 1987; Miller, 1989; 1990; Calambokidis and Barlow, 1991; Calambokidis and Quan, 1999). Prior to 1975, there was only one harbour porpoise stranding record from the NWR (in 1943). However, since the mid-1970s, stranding numbers for this species have remained fairly consistent (~ 10-25/year).

It is possible that the proportions of species in the stranding record reflect the relative abundance of live animals of the species in the respective region (Sergeant, 1979; Woodhouse, 1991). For instance, most species that are relatively rare in the NWR are represented by a small number of strandings. However, strandings may also reflect nothing more than a general region of occurrence and may not be related to specific habitat preference. We conclude that the stranding of a cetacean in a certain area at a particular time does not mean that it is representative of live animal distribution.

Stranding event numbers were compared to neighbouring regions: California and Alaska. In California, 1,800 cetaceans stranded from 1983-2000, representing 25 species and in Alaska 1,390 cetaceans stranded from 1975-2000, consisting of 15 species (US Department of Commerce, 1975-2000). In terms of species composition, the most commonly stranded species in California were common dolphin (*Delphinus* spp.; $n = 435$), gray whale ($n = 309$), harbour porpoise ($n = 306$) and Pacific white-sided dolphin ($n = 70$). The most commonly stranded species in Alaska were gray whale ($n = 275$), harbour porpoise ($n = 75$), killer whale ($n = 69$) and humpback whale ($n = 65$). Stranding summaries from the late 1970s and early 1980s were not consistently broken down by species so these numbers may be artificially low. Three of the four most commonly stranded cetaceans in the NWR (harbour porpoise, gray whale and Pacific white-sided dolphin) were also in the top four stranded species for California. This is not surprising as these species are distributed in both regions and would therefore be expected to have similar stranding frequencies and occurrences. Common dolphins did not contribute to a significant portion of the NWR stranding numbers, however, as they are rarely found in the NWR. In Alaska, gray whales and harbour porpoises comprised the two most stranded species as they did in the NWR, but in reverse order. This again is not unexpected since large aggregations of gray whales migrate to their summer feeding grounds in Alaska. Although Dall's porpoises were sighted more often then harbour porpoises during summer ship and aerial surveys in Alaska (Waite and Hobbs, 1998; Waite *et al.*, 2001; Moore *et al.*, 2002), they do not seem to strand as frequently as harbour porpoises presumably due to their preference for generally deeper waters than harbour porpoises, therefore their carcasses may sink before reaching shore. Greater killer and humpback whale stranding numbers may occur in Alaska due to larger populations of these species inhabiting the waters of this state.

Sex of stranded animals

Sex was determined for 609 stranded individuals (Table 1). Sex could not be determined in 342 individuals (36%) due to advanced decomposition of the carcass, examiner's inexperience in sexing animals or carcass position. Sex ratios were not significantly different from 1:1 for the most commonly stranded species: harbour porpoise (105 males:93 females, $\chi^2 = 2.48$, $P = 0.115$), gray whales (85 males:63 females, $\chi^2 = 2.78$, $P = 0.095$), Dall's porpoise (44 males:25 females, $\chi^2 = 3.06$, $P = 0.080$) and Pacific white-sided dolphin (18 males:10 females, $\chi^2 = 2.29$, $P = 0.131$).

Mass strandings

The only mass stranding in the database involved sperm whales. On 16 June 1979 near Florence, Oregon, a group of 41 animals (28 females and 13 males) live-stranded (Rice *et al.*, 1986). All of the males were subadults, of the adult females, 3 of the 9 were pregnant and none were lactating. The oldest female was 58 years old. One of the females was sexually immature and the remaining were sexually mature. There were neither calves nor animals under 10 years of age. The low number of mass strandings in the NWR may reflect the lack of relative coastline features which may make cetaceans vulnerable (e.g. sloping beaches, geomagnetic disturbances).

Uses of stranding data for management

Data gathered from stranding events can help facilitate management in several ways. It provides an overview of distribution and stranding trends usually observed in the NWR which can provide an early warning system in the event of an unusual stranding event. Monitoring of stranding patterns (spatial and temporal) helps identify unusual mortality events. For instance, an extraordinarily high number of strandings of gray whales in 1999 and 2000 warranted further attention (Le Boeuf *et al*., 2000; Norman *et al.*, 2000). In addition, stranding data may supplement existing knowledge on distribution of cetaceans in the NWR already obtained from aerial and ship surveys of the region (e.g. Fiscus and Niggol, 1965; Everitt *et al.*, 1979; Barlow, 1987; Brueggeman, 1990; 1992; Green *et al.*, 1992; Green *et al.*, 1993; Calambokidis *et al.*, 1997). For some species of cetaceans, little is known beyond what is learned from strandings. For example, in the NWR little is known about northern right whale dolphin (*Lissodelphis borealis*) distribution and ecology except from stranded specimens ($n = 8$). Few specimens of Hubb's beaked whale (*Mesoplodon carlhubbsi*) have been recovered in the NWR. They are very cryptic and difficult to identify at sea. Their presence in the NWR would be unknown if not for two stranded animals. Stranded specimens provide an invaluable source of information on anatomy and taxonomy (particularly through genetic analysis), since access to live animals is limited and expensive and there are few direct hunts (or bycatch schemes) that can provide specimen material.

Stranded marine mammals do not constitute an ideal sentinel system for population health as they do not represent the entire population (Aguilar and Borrell, 1994). In addition, samples from stranded animals are infrequently age and sex structured. Biological data such as life history, reproductive success, feeding habit, and disease progression are not typically available. Nonetheless, contaminant analysis of tissues collected over a stock's range may identify patterns of exposure (Varanasi *et al*., 1993; 1994; Krahn *et al*., 2001) There are limitations, however, to the use of stranded specimens for contaminant analysis. The effect of disease and nutritive condition may affect lipid content of the tissues (Aguilar *et al*., 1999). Most often the time of death of a stranded animal is unknown, hence samples collected may not adequately reflect tissue pollutant concentrations. Changes in the levels of contaminants occur post-mortem due to the inevitable physiological changes and breakdown of tissues associated with autolysis (Reijnders *et al*., 1999). The effect of weather (e.g. wind and direct sun) on a carcass may also cause loss of the more volatile organic compounds present in tissues (Aguilar *et al*., 1999).

Examination for evidence of human interaction in strandings may point to a need for closer monitoring of a specific geographic area or for development of appropriate mitigation measures to reduce take levels in certain fisheries (Gearin *et al*., 1994; 2000) as well as threats from ship strikes, shooting or other direct mortality. Fishery interactions with gray whales and ship strikes have been reported. There were six mortalities due to fisheries interactions reported in 1999 and eight in 2000, and two fatal ship strikes, one in 1999 and one in 2000 (Angliss and Lodge, 2002). In 2000, the Center for Coastal Studies (Provincetown, MA) and NMFS cosponsored a large whale disentanglement training workshop in Seattle, WA for primary network responders in the NWR. The discovery of stranded animals bearing evidence of ship strike (e.g. four ship-struck fin whales reported in the NWR in 2002) may prompt future management measures such as reduction of vessel speed through areas of known large whale aggregations or sensitive habitat (Laist *et al*., 2001). In cases of suspected shooting (which are often a result of fishery interaction), involvement of state (e.g. Oregon and Washington Departments of Fish and Wildlife) and federal (e.g. NMFS) enforcement agencies will help mitigate marine mammal-fishery interaction problems. Cetaceans may be affected by oil spills such as in a primary feeding area by contaminating prey items (Moore and Clarke, 2002).

Stranded animals may also provide information on population movement patterns or residency of a given species. It may be possible to draw correlations between beached species and their parent populations in the region (Woodhouse, 1991). For instance, the location of a NWR resident killer whale stranding during the winter provides information on residency of the population that otherwise is little known during that time of year (Olesiuk *et al*., 1990). Likewise, the stranding of a seasonal 'resident' gray whale in Puget Sound during the summer confirms that some gray whales do not complete the migration to the usual summer feeding grounds in the Bering and Chukchi Seas, but rather remain in NWR waters to feed (Sumich and Gilmore, 1977; Calambokidis and Quan, 1999).

The existence of a parent population in the region may not necessarily be reflected by the presence of strandings. For example, summer and autumn feeding aggregations of humpback whales have been reported off the Washington coast (Scheffer and Slipp, 1948; Calambokidis *et al*., 2000), however, they are underrepresented in the stranding records ($n = 6$). One explanation may be that the whales remain far enough offshore that when they die, the carcass sinks before reaching shore or is swept away by currents.

Analyses of stranded animals may lead to the identification of novel diseases or patterns of antibiotic resistance not previously known in cetaceans (Foster *et al*., 1996; Fox *et al*., 2000). Health trends of free-ranging populations of marine mammals may be assessed through investigation of stranded animals, particularly those that have live-stranded. Necropsy investigations of stranded animals provides data on pathogens that could possibly cause disease in humans or domestic animals that come in contact with these animals.

Since the implementation of a coordinated stranding network in Oregon and Washington, a greater number of strandings have been recorded and a significant amount of data has been collected. For example, contaminant levels in stranded NWR gray whales have been compared to harvested animals in Russia (Krahn *et al*., 2001). Identification of infectious diseases in stranded cetaceans can serve as a basis for developing a standardised necropsy and disease testing protocol (Gaydos *et al*., 2004) for stranded southern resident killer whales which were recently listed as depleted under the Marine Mammal Protection Act (NMFS, 2003). Future participation of the network will continue to further understanding and insight into the mortality, life history, disease processes and stock structure of cetaceans within the waters of the NWR.

ACKNOWLEDGEMENTS

The authors wish to acknowledge the Northwest Region Stranding Network and the many volunteers who reported strandings. Many of the Network members responded to strandings with their own funds. The collection and handling of specimens was carried out under permit of NOAA's National Marine Fisheries Service's Protected Species Management Branch. The authors would also like to thank the following for their assistance in responding to strandings and collecting data: Robin Brown (Oregon Department of Fish and Wildlife, Corvallis, Oregon); Keith Chandler (Seaside Aquarium, Seaside, Oregon); San Juan County Marine Mammal Stranding Network/Whale Museum (Friday Harbor, Washington); Hatfield Marine Science Center, Oregon State University, Newport, Oregon and the Oregon Institute of Marine Biology, University of Oregon, Charleston, Oregon and all other network volunteers. Glen Watabayashi, of the Hazardous Materials Response Division, Office of Response and Restoration, National Ocean Service (NOS), NOAA provided wind and current data for the Northwest Region. David Rugh and Janice Waite, both biologists at the National Marine Mammal Laboratory (NMML), kindly reviewed this manuscript. Robert DeLong, Leader of the California Current Ecosystems Program at NMML also provided helpful comments and suggestions. Comments by the reviewers and the Editor greatly improved the manuscript. Richard Morse of the Damage Assessment and Restoration Center, NOS, NOAA developed the maps.

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Winter distribution and abundance of humpback whales (*Megaptera novaeangliae*) off Northeastern Brazil

ALEXANDRE N. ZERBINI[*], ARTUR ANDRIOLO[+], JESUINA M. DA ROCHA[#], PAULO CÉSAR SIMÕES-LOPES[++], SALVATORE SICILIANO[**], JOSÉ LUIZ PIZZORNO[##], JANICE M. WAITE[¥], DOUGLAS P. DEMASTER[¥] AND GLENN R. VANBLARICOM[*]

Contact e-mail: azerbini@u.washington.edu

ABSTRACT

The Brazilian coast is recognised as a Southern Hemisphere humpback whale (*Megaptera novaeangliae*) wintering ground (IWC breeding stock 'A'). The northeastern coast of Brazil was an important whaling ground in the 20th century. Shipboard sighting surveys were conducted in this area to evaluate large whales' distribution and density in 1999 and 2000. Humpback whale sightings ($n = 81$, 153 individuals) were recorded using line transect methodology. Data from the 2000 survey were used to estimate abundance over the continental shelf from 5 to 12°S (20,040km^2). A total of 872.1km were surveyed on effort. Humpback whales were distributed from nearshore to the 800m isobath, but 93.5% of sightings were recorded shoreward of the 300m isobath. The relatively high density off northeastern Brazil suggests that the species is reoccupying historical areas of distribution and the presence of newborn individuals indicates that calving and nursing occur in the area. The hazard rate model best fit perpendicular distance data. Abundance was estimated at 628 individuals (CV = 0.335, 95% CI = 327-1,157). This estimate probably corresponds to only a portion of the breeding population. Therefore, additional studies must be conducted to estimate the total size of the humpback whale population wintering off Brazil.

KEYWORDS: HUMPBACK WHALE; DISTRIBUTION; BREEDING GROUNDS; ABUNDANCE ESTIMATE; SURVEY-VESSEL; SOUTHERN HEMISPHERE; SOUTH AMERICA

INTRODUCTION

Humpback whales (*Megaptera novaeangliae*) occur in all major oceans from polar and sub-polar regions to the Equator. In the Southern Hemisphere they migrate from summer feeding grounds in the Antarctic to mating and calving grounds in tropical and subtropical regions (e.g. Dawbin, 1956; Chittleborough, 1965; Mackintosh, 1965), where they tend to concentrate near islands and coral reef systems (e.g. Clapham and Mead, 1999). The species was heavily exploited in the Southern Hemisphere from both coastal stations and pelagic waters in all major ocean basins (e.g. Chittleborough, 1965; Gambell, 1973; Williamson, 1975; Tønnessen and Johnsen, 1982; Best, 1994). About 200,000 whales were taken both in the Antarctic and the breeding grounds after 1900 (Findlay, 2001), causing declines of populations to small percentages of their pre-exploitation levels (Gambell, 1973).

The International Whaling Commission (IWC) currently recognises seven humpback whale breeding populations in the Southern Hemisphere (IWC, 1998). Breeding stock 'A' is one of the least known and corresponds to whales wintering off Brazil. Historically, this population was believed (e.g. Slijper and Utrecht, 1959; Slijper, 1962; 1965; IWC, 1998) to migrate to feeding grounds in IWC Management Areas I (the Antarctic Peninsula) and II (the South Georgia Islands). Recent studies, however, have not provided clear evidence that whales breeding off Brazil indeed migrate to these areas. Moore *et al.* (1999) recorded a notably small number of humpback whales around the South Georgia Islands within a period of 10 years and suggested that this population has not recovered after being exploited in the region. This contrasts with the apparent growth of the humpback whale population off the coast of Brazil (Siciliano, 1997) and may suggest that these whales migrate somewhere else in the Antarctic Ocean. In addition, photo-identification studies, analysis of fluke colouration patterns and molecular genetic data indicated that whales feeding near the Antarctic Peninsula were linked to breeding grounds in the eastern South Pacific (breeding stock 'G') and that whales from the coast of Brazil are possibly linked to populations in Africa and Oceania (e.g. Rosenbaum *et al.*, 1995; 2000; Muñoz *et al.*, 1998; Engel *et al.*, 1999; Olavarría *et al.*, 2000; Dalla-Rosa *et al.*, 2001).

Humpback whales were harvested by open boat whalers off the southern and central coast of Brazil between the 16th and the early 20th centuries (e.g. Ellis, 1969; Lodi, 1992). Modern whaling operations took humpback whales off the coast of Cabo Frio (~23°S) from 1960 to 1963 and off the coast of Costinha (~7°S) from 1910 to 1964 (Paiva and Grangeiro, 1965; 1970; Williamson, 1975). Additional whales may have been taken illegally off the central coast of Brazil by the former Soviet Union fleet, after the species was protected in the middle 1960s (see Yablokov *et al.*, 1998). The total number of whales caught is unknown. Bureau of International Whaling Statistics (BIWS) catch data account

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[*] *Washington Cooperative Fish and Wildlife Research Unit, School of Aquatic and Fishery Sciences, Box 355020, University of Washington, Seattle, WA, 98195-5020, USA.*

[+] *Departamento de Zoologia, Instituto de Ciências Biológicas, Universidade Federal de Juiz de Fora, Campus Universitário, Juiz de Fora, MG, 36036-330, Brazil.*

[#] *Departamento de Vida Silvestre, Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA). Ed. Sede do IBAMA. Av. L4 Norte, Brasília, DF, 70800-200, Brazil.*

[++] *Laboratório de Mamíferos Aquáticos, Departamento de Ecologia e Zoologia, Universidade Federal de Santa Catarina, CP 5102, Florianópolis, SC, 88040-970, Brazil.*

[**] *Museu Nacional, Departamento de Vertebrados, Secção de Mamíferos, Universidade Federal do Rio de Janeiro, Rio de Janeiro, RJ, 20940-040. Brazil.*

[##] *Biodinâmica Engenharia e Meio Ambiente Ltda, Av. Marechal Câmara, 186, 4° andar, Rio de Janeiro, RJ, 20020-080, Brazil.*

[¥] *National Marine Mammal Laboratory, Alaska Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, 7600 Sand Point Way, NE, Seattle, WA, 98115, USA.*

for about 1,600 individuals, but this figure does not consider individuals taken by open boat fishers or by the modern whaling industry from 1929 to 1946.

Current information on the distribution of humpback whales shows that the species is abundant at Abrolhos Bank, 15°-18°S (e.g. Siciliano, 1995; Bethlem *et al.*, 1996; Martins *et al.*, 2001). Occasional sightings and strandings have been reported for the Fernando de Noronha Archipelago (~ 3°S) and in southern and southeastern Brazil (e.g. Lodi, 1994; Siciliano, 1997; Pizzorno *et al.*, 1998). However, distribution and density are still poorly known for a great portion of the Brazilian coast. Population size estimates are available only for the Abrolhos Bank area (Kinas and Bethlem, 1998).

From 1998 to 2001, cetacean surveys were conducted off the northeastern coast of Brazil. The survey area included the former whaling ground off Costinha (~ 7°S), Paraíba State, where a substantial number of large whales were captured in the past. The objective of this study was to verify the winter distribution and density of large whales in the former whaling area. In this paper, only data on humpback whales collected during the 1999 and 2000 cruises are presented. Information on other species is summarised in da Rocha *et al.* (1999), Siciliano *et al.* (2000), Zerbini *et al.* (2000) and Andriolo *et al.* (2001).

METHODS

Planning the surveys, research area and track design

Surveys were planned to take place at the peak of abundance of large whales off northeastern (NE) Brazil (August-November, Paiva and Grangeiro, 1965; 1970; Williamson, 1975) and were scheduled according to ship availability. Cruises were conducted on board the Brazilian Navy ship *Almirante Graça Aranha* on 6-27 September 1999 and from 14 August to 1 September 2000. Planning meetings to discuss survey design and protocols were held three days before the cruises started.

The study area included the former whaling grounds off NE Brazil and is illustrated in Fig. 1. The 1999 study was limited to the 5° and 10°S parallels and the coastline and the 33°W meridian. Given the spatial differences in large whale distribution observed in 1999, the 2000 cruise was stratified into two blocks: an oceanic block that covered the same study area as the 1999 survey, except that its western boundary was moved to the 500m isobath; and a coastal block that comprised the continental shelf (as far offshore as the 500m isobath[1]) that was extended further south to 12°S parallel (Fig. 1). This block was relatively long (740km) and narrow (22-50km) compared to the oceanic stratum. A saw-tooth transect design was adopted in all surveys (Fig. 2).

Survey protocol

The cruises were divided into two phases: training and actual line transect survey. The former took place at the beginning of each cruise and the objectives were to train observers and simulate the sampling routine adopted during the surveys. The 'flying deck' was used as the observation platform and was located 13.8m above the surface. Cetaceans were continuously searched from 05:30hs to 17:00-17:15hs. Eight

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[1] The offshore limit of the continental shelf was considered the 500m isobath to assure that sightings recorded near the shelf break (usually the 200m isobath) were included in the analysis.

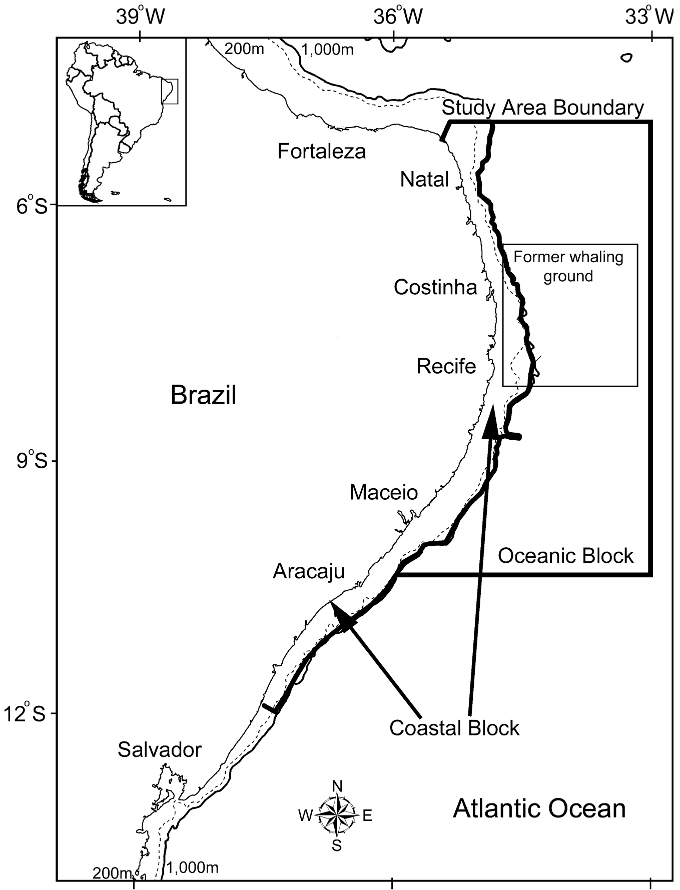

Fig. 1. Study area and blocks of the cetacean sighting surveys conducted off NE Brazil in 1999 and 2000.

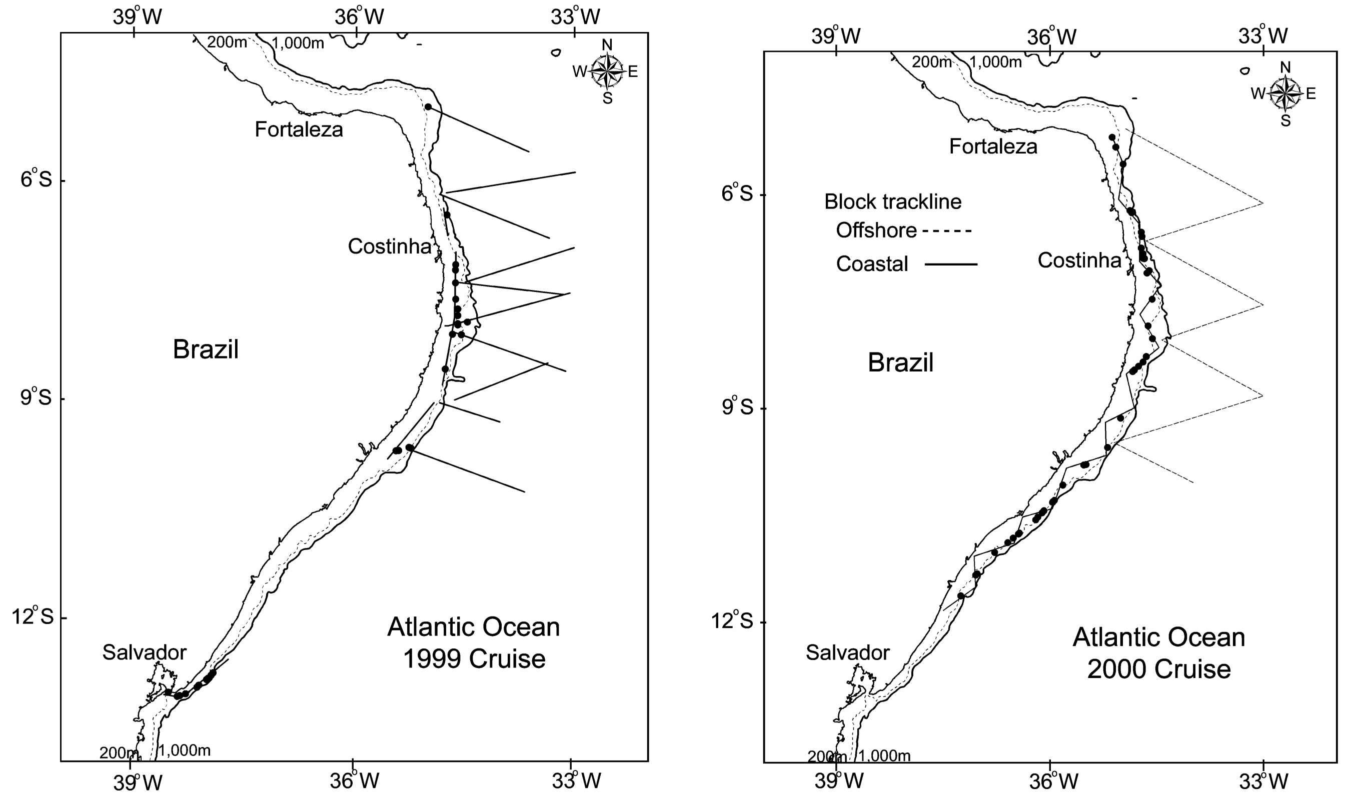

Fig. 2. Trackline design and humpback whale sightings during the 1999 and 2000 cruises conducted off NE Brazil.

(1999) or nine (2000) scientists rotated through four observation positions. A full observation period lasted two hours (30 minutes in each position) and was followed by a two-hour resting period. One scientist at port and another at starboard searched from 0° (the bow) to 90° while the other two observers searched a narrower sector, between 40° port and 40° starboard of the trackline. A fifth observer acted as data recorder and was not involved in searching, but aided the observer team in identifying species, tracking detected groups and estimating group size and composition. The team of observers was randomly selected for each day of survey.

Cetaceans were searched for using reticuled binoculars (80-90% of the time) and by naked eye (10-20% of the time). Immediately after a sighting was detected, the number of reticules between the horizon and the sighting and the radial angle between the group sighted and the ship's track were recorded on a standard data form. Environmental variables such as cloud cover, wind strength and direction, sea state (Beaufort scale) and sea surface temperature were also recorded. Sightings made while the observer team was on-watch were considered 'on effort'. Those recorded by the ship's crew, during training days, during the night or during off-watch periods were considered 'off effort'. Search was abandoned when the weather and visibility conditions were poor and sea state was above Beaufort 5. The 1999 and 2000 surveys were conducted in closing and passing mode, respectively (e.g. see Matsuoka *et al.*, 2003, p.179).

Data analysis

Distribution and group characteristics

The distribution of humpback whales was studied by pooling data from the two years. Sightings collected during training and actual survey, as well as *en route* to and from the ports were included in this analysis. Differences in latitudinal distribution were investigated only with data collected during the 2000 cruise. The sighting rate of whales seen within 1.5km of the trackline was considered an index of density. The study area was then divided into four equally spaced latitudinal intervals and the expected and observed number of whales in each interval was compared. The expected number of whales was assumed to be uniform and was calculated by multiplying the overall encounter rate by the survey effort in that interval. A chi-square test was used to investigate significant differences in relative density.

Abundance

Abundance was derived using line transect methods (Buckland *et al.*, 2001) with data from the coastal block (area = 20,040km^2) of the 2000 cruise because it was designed to uniformly cover the continental shelf. Nineteen transects were surveyed in this block at a speed of 9-11 knots, resulting in a total trackline of 872.1km.

Radial distance of each sighting was calculated using the 'approximation 2' suggested by Lerczak and Hobbs (1998, erratum).

Sightings recorded as 'confirmed humpback whales' and 'probable humpback whales' were included in the analysis. It is reasonable to pool these records since the only other large whale species sighted in the coastal block, the Antarctic minke whale (*Balaenoptera bonaerensis*) and the dwarf minke whale (*B. acutorostrata*), present behaviour and blow characteristics distinctive from humpback whales.

Abundance was estimated as:

$$\hat{N} = \frac{A \cdot n \cdot \bar{s}}{2 \cdot L \cdot ESW \cdot \hat{g}(0)}$$

where:

A is the survey area;
n is the number of sightings recorded 'on effort';

$\bar{s}$ is the mean group size;
L is the total trackline length surveyed;
ESW is the effective strip width;
$\hat{g}(0)$ is the estimated probability of detection on the trackline (assumed to be equal to 1 here).

Data analysis was undertaken with the software *Distance 3.5* (Thomas *et al.*, 1998). Perpendicular distance data were truncated at 3km and *ESW* was estimated by modelling ungrouped data using the half normal and hazard rate functions. Cosine and hermite polynomial series expansions (for half normal function) and cosine and simple polynomial adjustments (for hazard rate) were also considered in the set of candidate models. Model selection uncertainty was incorporated in the analysis by running 999 bootstrap replicates and letting the Akaike Information Criterion (AIC) (Akaike, 1973) select the best model for each bootstrap replicate. Variance and confidence intervals were also obtained from the bootstrap replicates.

RESULTS

The total number of humpback whale sightings and individuals observed in 1999 and 2000 is summarised in Table 1.

Table 1

Humpback whales sighted off NE Brazil in 1999 and 2000.

	On effort		Off effort		Total	
Year	No. of sightings	No. of indiv.	No. of sightings	No. of indiv.	No. of sightings	No. of indiv.
1999	13	25	18	31	31	56
2000	42	80	8	17	50	97
Total	55	105	26	48	81	153

Distribution

Whales were regularly found along the survey area (from 5° to 12°S, Fig. 2) from close to shore to depths of up to 750m. Most groups sighted were observed within the 200m isobath and those with calves tended to occur in shallower waters than groups without calves (Fig. 3). No sightings in deeper, oceanic waters were observed. There was no significant difference in the relative density of whales across the study area (Table 2, $\chi^2 = 2.71$, df = 3, $p = 0.439$).

Group size and composition

Group sizes of humpback whales ranged from 1-4 in 1999 and 1-6 in 2000; the modal group size was two in both years. Group composition is presented in Table 3. Overall, calves were observed in 29.5% of the humpback whale groups sighted. The proportion of calves/group was smaller in 1999 (20.7%, $n = 31$) than in 2000 (35.6%, $n = 45$), but this difference was not statistically significant ($\chi^2 = 1.66$, df = 1, $p = 0.198$). Newborn individuals were observed in 80% of the triads, 50% of the groups with four individuals and 37.5% of the groups with two whales.

Abundance

The hazard rate model with no adjustment best fitted perpendicular distance data. Abundance was estimated at 628 individuals (CV = 0.335, 95% CI = 327-1,157). Table 4 summarises encounter rate, average group size and model parameters. Fig. 4 presents the distribution of perpendicular distances and fitted detection function.

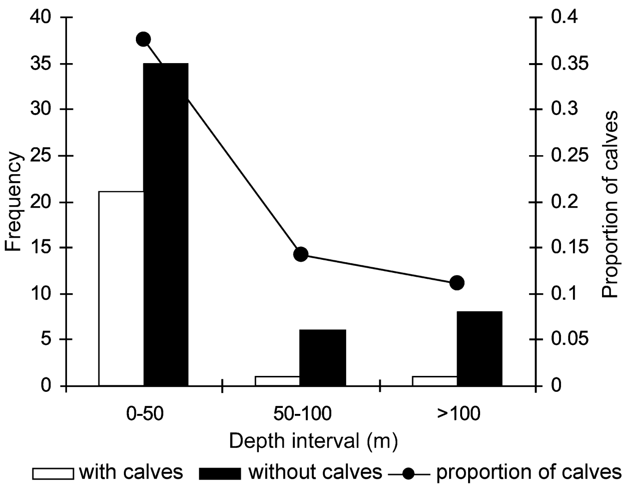

Fig. 3. Depth distribution of humpback whale groups off NE Brazil, and proportion of calves per depth interval.

Table 2

Observed and expected number of humpback whales per latitudinal interval off NE Brazil.

Interval	Observed	Expected	χ^2 value
5-6°45'S	5	6.4	0.30
6°45'-8°30'S	6	6.9	0.12
8°30'-10°15'S	7	8.7	0.33
10°15'-12°S	12	8.0	1.96
Total	30		$\chi^2 = 2.71$

Table 3

Group composition of humpback whales recorded off NE Brazil in 1999 and 2000.

Group composition	1999	2000	Total
Group of one	11	16	27
Mother/calf	6	9	15
Group of two (no calf)	10	12	22
Mother/calf/escort	0	6	6
Competitive group	2	1	3
Competitive group with a calf	0	1	1
Undefined	2	5	7
Total	31	50	81

DISCUSSION

Distribution

The current winter distribution of humpback whales in the southwestern Atlantic Ocean has been better understood in recent years. Whales have long been regularly found at Abrolhos Bank, ~15-18°S (Siciliano, 1995; Freitas *et al.*, 1998), but more recent records have shown that the species is more widely distributed along the South American coast. Sightings have been reported from São Paulo (24°S), southeastern Brazil, to the northern coast of Bahia, 12°S (Siciliano *et al.*, 1999) and strandings were reported as far south as Rio Grande do Sul, 34°S (Siciliano, 1997). Some records have been reported for oceanic islands such as the Archipelagos of Fernando de Noronha (3°51'S) and Trindade and Martin Vaz (20°30'S) (Lodi, 1994; Siciliano *et al.*, 1999). The present study shows that humpback whales

Table 4

Estimated model parameters, density and abundance of humpback whales off NE Brazil.

Parameter	Point estimate	% CV	95% CI
f(0)	0.719	22.8	0.456-1.135
p	0.464	22.8	0.294-0.731
ESW (km)	1.39	22.8	0.881-2.194
Encounter rate (n/L)	0.045	23.31	0.024-0.066
Density of groups (groups/km^2)	0.016	32.60	0.008-0.029
Average group size (s)	1.95	8.83	1.63-2.32
Density of individuals (ind/km^2)	0.031	32.78	0.016-0.059
Abundance	628	33.46	327-1,157

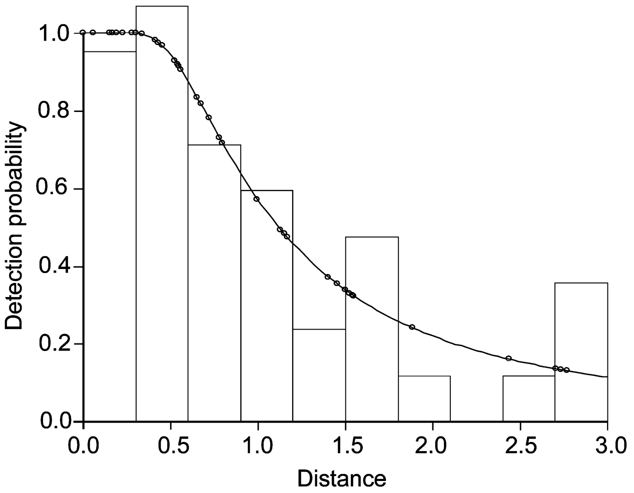

Fig. 4. Hazard rate model fit to humpback whale perpendicular distance (dots represent expected detection probability for individual sightings).

are regularly found in coastal waters as far north as 5°S along the northeastern coast of Brazil. In addition, a whale stranded in Ceará (3°43'S, 38°30'W), west of the northwestern tip of South America (Furtado-Neto *et al.*, 1998), suggests that humpback whales may be moving west along the northern coast of Brazil.

Although the distribution of humpback whales is better known, the extent of the calving grounds is not yet clear. Given its shallow and relatively protected waters, the Abrolhos Bank has been recognised as a major calving/nursing area. Siciliano (1997) reported that females with calves corresponded to 33 and 49% of the groups in the area during the breeding seasons of 1989 and 1990 respectively. The present study shows that newborn individuals are also present in a high portion of the groups observed off NE Brazil, indicating that this region has also been used as a calving ground. The proportion of groups containing newborns (20.7-35.6%) was not as high as observed for the Abrolhos Bank, but was higher than other Southern Hemisphere humpback whale breeding grounds such as Mozambique (14.8%; Findlay *et al.*, 1994) and Ecuador (17%; Scheidat *et al.*, 2000). The high proportion of newborn whales off NE Brazil, contrasts with data collected during the whaling period (1914-1985). They indicate that few whales taken in Costinha were pregnant or were accompanied by calves. In addition, biological inspection of the whales captured from 1947 to 1963 found only one pregnant individual among the 76 females taken (data from Bureau of International Whaling Statistics [BIWS]/IWC). In addition, calves were observed in only three out of 46 groups sighted during whaling operations from 1979 to 1984. The proportion of females with calves was not as high or is unknown for other areas of the Brazilian coast. Lodi (1994) reported that two out of 11 groups (18%) observed in Fernando de Noronha from 1989 to 1993 included calves. However, humpback whales are not regularly seen in the area (F. Camargo, pers. comm.). Newborn individuals have also been recorded south of Abrolhos Bank suggesting that births also take place there. Sightings of females with calves have been regularly reported by local inhabitants or boat operators cruising the coast of Espírito Santo and northern Rio de Janeiro (Pizzorno, unpublished data). In addition, a relatively high proportion of the incidental catches recorded in this area corresponded to calves (Siciliano, 1997; Zerbini and Kotas, 1998). Recently recorded distributional data therefore indicate that the humpback whale calving grounds range from about 5 to 21°S in the western South Atlantic Ocean.

Humpback whales were captured off NE Brazil since early in the 20th century. Whaling statistics revealed that an annual average of 150 whales was taken in the period 1911-1914/1924-1928 (BIWS/IWC; Williamson, 1975). This number dropped to about 12 individuals per year from 1947 to 1963, clearly reflecting the overexploitation of the stock (see also Pinedo, 1985). Whaling operations continued in the area up to 1985 and sightings recorded by the catcher boat from 1979 to 1984 (Antonelli *et al.*, 1987; Siciliano, 1997) revealed that humpback whales were rare in the area by the end of the whaling period. Despite the high observation effort (the season was five months long and the catcher operated almost every day), an average of only eight sightings per season was recorded at that time. Current data show that abundance is greater off NE Brazil, indicating the species has reoccupied this historical area of distribution.

It is most likely that whales moving to the northeastern coast of Brazil are passing through the Abrolhos Bank on their way north. Freitas *et al.* (1998) reported that a whale photographed off Salvador (12°S) was previously recorded off Abrolhos in the same season. This suggests that there is some degree of movement between the two areas. However, alternative migration routes (e.g. whales migrating through offshore waters) cannot be ruled out. The lack of photo-identification effort precludes any conclusion regarding movements of whales visiting NE Brazil. Such studies should be initiated in the area.

Abundance estimates

The abundance estimate reported in this paper corresponds to a fraction of the stock size of whales wintering off the coast of Brazil. The surveys covered the northern portion of the humpback whale area of distribution (5-12°S) during the breeding season. During this period, whales are known to be found as far south as about 21°S (Pizzorno, unpublished data). The 2000 cruise, from which estimated abundance was obtained, was carried out in late August to early September, approximately the peak of abundance of humpback whales off Brazil. Whaling statistics indicate that catches off Costinha peaked in August (data from the BIWS/IWC) while sighting frequency off Abrolhos Bank peaked in early September (Siciliano, 1997). This suggests that the cruise covered the area when density was expected to be the greatest.

In the present study, probability of detecting whales on the trackline [$g(0)$] was considered to be unity (as assumed by line transect theory), but this assumption could lead to a slight downward bias in the abundance estimation because some whales might have been undetected. This problem was possibly minimised by allocating more effort to the trackline. In addition, it is believed that in general, $g(0)$ for humpback whales is very close to one. Barlow and Gerrodette (1996) and Barlow (1997) estimated that $g(0)$ for groups ranging from 1-3 individuals was 0.9 and for groups with more than four whales was one.

The total size of the stock breeding along the coast of Brazil is unknown. Kinas and Bethlem (1998) estimated that about 1,600 individuals occurred in the Abrolhos Bank in the mid 1990s using an empirical Bayes closed mark-recapture model. The present work provides an estimate for a previously unsurveyed area, but the two estimates cannot be combined because sampling periods were relatively far apart and because sampling conditions and assumptions are different. In addition, it is possible that an unknown proportion of whales heading to or from NE Brazil may move through the Abrolhos Bank and hence may have been captured in the photo-identification surveys.

Total stock size is necessary for any population assessment work, and it is recommended that both line transect and mark recapture survey efforts be expanded to estimate population size and trends of humpback whales off Brazil.

ACKNOWLEDGEMENTS

The authors are greatly indebted to M. Bassoi, D. Danilewicz, A. Lucena and I.B. Moreno for their help during field activities. We also acknowledge the IBAMA's Directors for Ecosystems, Mr R. Soavinski and Mr L.M. dos Santos, the Chief of the IBAMA's Wildlife Division, Mrs M.I. Bampi and the Secretaries of CIRM, Rear-Admirals A.C. da Câmara Brandão and L.A. Monclaro de Malafaia. Without their effort, encouragement and support, cruises would not have been conducted. We would also like to thank Commander F. Barbosa de Oilveira (SECIRM), Commander A.P. Miguens (Programa REVIZEE, Ministério do Meio Ambiente), Commander J.C. Feijó (EMA) and Mr R.P. Lima (Centro Mamíferos Aquáticos, IBAMA) for their interest and help during the preparation of the cruises. The III Distrito Naval de Natal, Companhia Naval do Nordeste and Diretoria de Hidrografia e Navegação (DHN) provided logistical support. The hospitality and interest of the ship's crew, in particular Captains José Renato de Oliveira (CC), Wésley Cavalheiro (CF) and Olavo Barroca Jr. (CF), executive officer Luiz Torres (CC) and Antônio José Mesquita (CC), were greatly appreciated. This manuscript was greatly improved by comments provided by Paul G. Kinas and two anonymous reviewers. We also welcome the support from the Washington Cooperative Fish and Wildlife Research Unit (WCFWRU), the CIRM's staff and from the IBAMA's staff at Pernambuco, Rio Grande do Norte and São Paulo. Financial support was provided by IBAMA, Ministry of the Environment, Brazilian Navy, National Marine Mammal Laboratory and Cetacean Society International. A.N. Zerbini was supported by Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq) (processo 200285/98-0).

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The International Whaling Commission
The Red House, 135 Station Road, Impington, Cambridge CB4 9NP UK
Tel: +44 (0)1223 233971 Fax: +44 (0)1223 232876
Subscriptions e-mail: *subscriptions@iwcoffice.org*
Web page: http://www.iwcoffice.org