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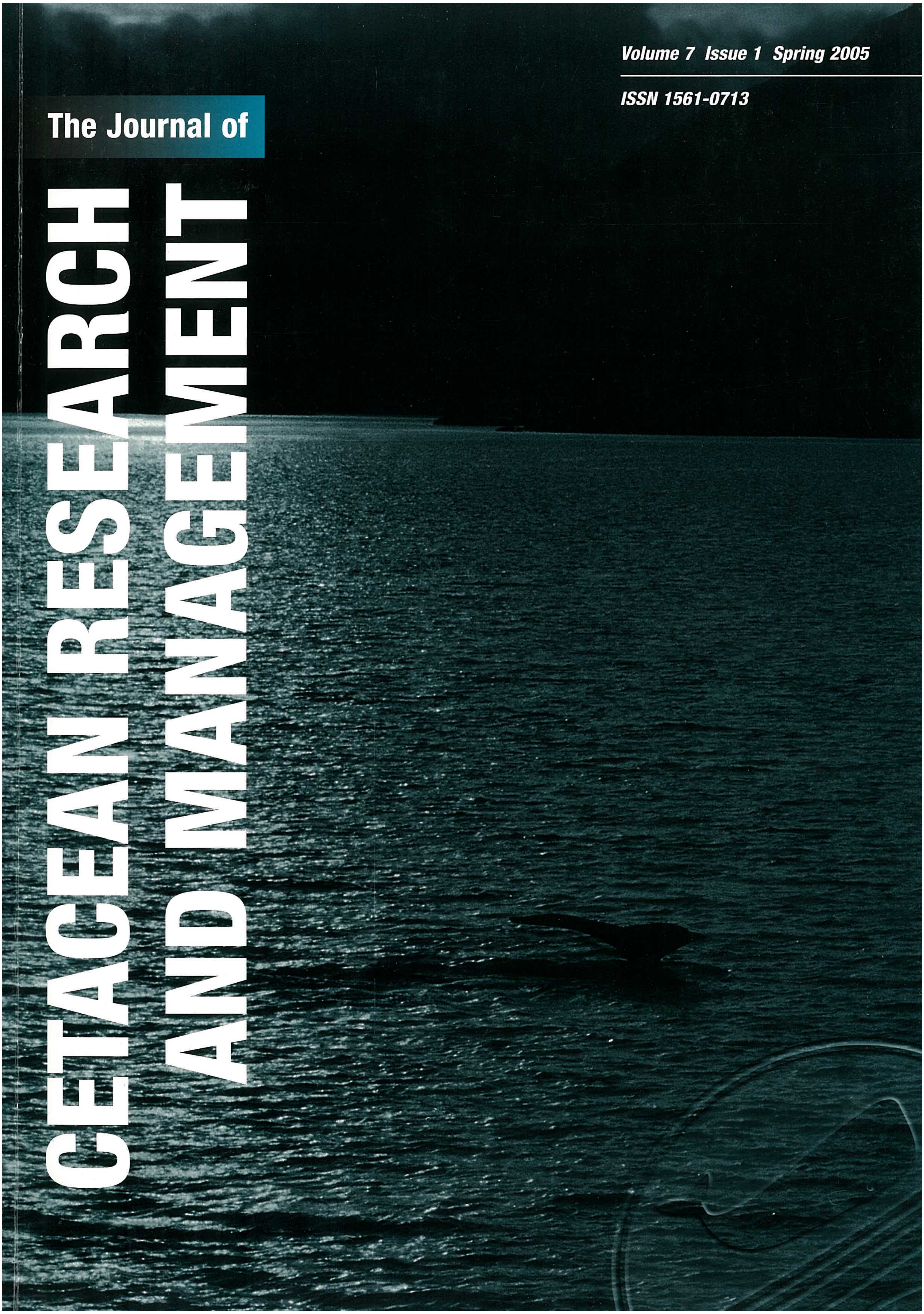

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Editorial

Welcome to this the first issue of the seventh volume of the *Journal of Cetacean Research and Management*. This volume contains eleven papers covering a wide range of management issues.

Wise management decisions require certain fundamental information and one of the most basic parameters is population abundance. One of the most encouraging success stories of cetacean conservation is the recovery of the eastern North Pacific stock of gray whales. The population was reduced to very low levels by the turn of the 20th century by whaling but has subsequently recovered since protection from commercial whaling (limited aboriginal subsistence whaling has continued throughout the period) and is now thought to be back to its pre-exploitation abundance. The paper by Rugh *et al.* reports on abundance estimates made during the period 1997-2002. This is a particularly interesting period because it includes a period of high natural mortality in 1999 and 2000. Management of aboriginal subsistence whaling requires not only information on the stock structure and abundance of the whales but also information on the catches themselves. Koski *et al.* provide information on a long time series (1973-2000) of catches from the community of Kaktovik in the extreme northeast of Alaska, where bowhead whales are taken.

One of the most important threats to cetacean populations is their incidental capture in fisheries. There are three papers relevant to this issue. The first paper, by Cavortata *et al.*, attempts to look at one aspect of finding a solution to the entanglement problem for the critically endangered North Atlantic right whale. The second, by Dawson and Slooten, concerns the management of gillnet bycatches in New Zealand where there is particular concern over the situation of the Hector's dolphin. The final paper, by Gillespie *et al.*, reports on abundance estimates for the endangered Baltic Sea population of harbour porpoises, whose survival is threatened by incidental captures.

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 along the coast of Ecuador (Felix and Haase) and the pygmy right whale in the southwestern Pacific Ocean (Matsuoka *et al.*).

A relatively recently identified threat to cetaceans concerns the impact of noise. Clark and Norman provide results from an experiment conducted by the US Navy in 2001 and discuss the nature of proposed mitigation measures and assess their performance. The issue of noise will be also addressed in forthcoming papers in volume 7, particularly with respect to beaked whales.

Good management is dependent on knowledge of population structure. There are two relevant papers in this issue. The first, by Rankin *et al.*, examines acoustic data collected by IWC-sponsored cruises in the Southern Hemisphere to examine the feasibility of using acoustics to distinguish between two sub-species of blue whales in the Southern Ocean. The second, by Sanino *et al.*, uses the more traditional approach of examining mitochondrial DNA to ascertain stock structure of common bottlenose dolphins in Chilean and Peruvian waters.

Local management requires knowledge of the distribution, movements and site fidelity of animals. Photo-identification studies can be a powerful tool in obtaining such knowledge as well as monitoring abundance and the final paper in this issue, by Bearzi, examines the bottlenose dolphins found in Santa Monica Bay, California – an area subject to considerable anthropogenic disturbance as well as potentially being affected by El Niño.

Finally can I draw your attention to the new *Guide for Authors* included at the end of this issue. There are a number of modifications so please read them carefully before submitting manuscripts. Not following the guidelines may result in considerable delay in the consideration of your manuscript!

G.P. DONOVAN
Editor

Estimates of abundance of the eastern North Pacific stock of gray whales (*Eschrichtius robustus*) 1997-2002

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ABSTRACT

The southbound migration of the eastern North Pacific stock of gray whales (*Eschrichtius robustus*) was documented by the National Marine Fisheries Service from 13 December 1997 to 24 February 1998, 13 December 2000 to 5 March 2001 and from 12 December 2001 to 5 March 2002. Research protocol was essentially identical to that used in previous surveys. This involved single observers independently searching for whales and recording data on effort and sighting time, location, count and direction-headed. In 1997/98, there were 2,346 pods (3,643 whales) counted during 435.0h of standard observational effort when visibility was recorded as fair to excellent. In 2000/01, a total of 1,694 pods (2,754 whales) were counted during 592.4h, and in 2001/02, there were 1,712 pods (2,800 whales) during 531.5h. The southbound migrations in 1997/98 and 2001/02 were normal, beginning in mid-December, centred on mid-January (mean dates=18 January 1998 and 15 January 2002 respectively) and ending by mid-February. However, in 2000/01 (mean date=25 January 2001) the migration was more protracted than any other migration observed in the past 25 years, with many whales still travelling south three weeks after the typical end date. Data analysis procedures were comparable to those used in previous years, with the exception of a new correction factor for night-time travel rates. Abundance estimates were 29,758 whales in 1997/98 (CV=10.49%; 95% log-normal confidence interval=24,241 to 36,531), 19,448 in 2000/01 (CV=9.67%; 95% log-normal confidence interval=16,096 to 23,498) and 18,178 in 2001/02 (CV=9.79%; 95% log-normal confidence interval=15,010 to 22,015). The abundance in 1997/98 was the highest estimate made since this project began in 1967/68. It was followed by two much lower estimates – probably related to the high mortality rates observed in 1999 and 2000. This whale population appears to be approaching the carrying-capacity of its environment.

KEYWORDS: GRAY WHALE; MONITORING; SURVEY – SHORE-BASED; ABUNDANCE ESTIMATE; TRENDS; MIGRATION; PACIFIC OCEAN; NORTHERN HEMISPHERE

INTRODUCTION

The National Marine Fisheries Service (NMFS) has conducted shore-based counts of the eastern North Pacific stock of gray whales most years since 1967 (Table 1) at Granite Canyon (or Yankee Pt), 13km south of Carmel, in central California. Access to this site is convenient, and the narrowness of the whales' migratory corridor in this area has permitted an efficient counting process that has been repeated through many seasons. All of these counts were done during the two-month southbound migration (from mid-December to mid-February), which is less protracted than the three-month northbound migration (from mid-February to late May). The predictability of the migration and routine nature of these counts contribute to inter-annual trend analyses. For example, Buckland and Breiwick (2002) showed there has been an increase of 2.5% per annum (SE=0.3%) between 1967/68 and 1995/96, and Wade and DeMaster (1996) have shown how this population may be approaching its carrying-capacity.

Tests of the counting procedure used in this study have included: (1) aerial surveys to document the distribution of whales relative to shore near Granite Canyon (Shelden and Laake, 2002); (2) high-power binoculars to monitor trends in offshore distribution (Rugh *et al.*, 2002); (3) corrections for estimates in pod size (Laake *et al.*, 1994); (4) paired, independent counting effort to estimate whales missed within the viewing area (Rugh *et al.*, 1990; 1993); (5) estimates of night travel rates via thermal sensor imaging (Perryman *et al.*, 1999); and (6) a study of migratory timing relative to this site (Rugh *et al.*, 2001).

The analytical techniques developed by Reilly (1981) to assess gray whale populations have been modified as more sophisticated algorithms have become available, such as Hermite polynomials to interpolate for unwatched periods (Buckland *et al.*, 1993), and improved estimates of variance (Hobbs *et al.*, 2004). For trend analyses, these improved techniques can be applied to all years so that analytical methods are consistent.

The primary objective of the field studies presented here has been to continue the standardised counts for purposes of extending the trend analyses, relying on single observers doing independent counts with minimal optical aids, as in the past. Of particular interest is that this may be the first large whale stock that has been monitored through the recovery process as it approaches its carrying-capacity. An additional incentive to conduct the study in 2000/01 and 2001/02 was to assess the abundance after two years (1999 and 2000) in which unusually high counts of dead gray whales had been reported (Le Boeuf *et al.*, 2000; Norman *et al.*, 2000; Gulland *et al.*, 2005). This monitoring is part of management recommendations following the removal of this stock from the list of endangered or threatened wildlife (Rugh *et al.*, 1999).

METHODS

Field methods

Systematic counts of gray whales were conducted throughout most daylight hours, covering most of the duration of the southbound migration past the Granite Canyon research station (Table 1). Three 3hr standard effort periods covered the nine daylight hours from 07:30 to 16:30. Observers were rotated to keep a balance of effort in each of the three shifts. A total of 10 people took part in the counts in 1997/98 and 10 in 2000/01, while 15 were involved in 2001/02 (see Acknowledgments). Observation sheds provided a writing platform with some protection from the elements. Average eye height above sea level was 22.5m.

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Table 1

Survey dates and gray whale abundance estimates from counts conducted by NMFS during the whales' southbound migration past Granite Canyon, California. The most recent estimates have a better assessment of standard error than was available in data collected during previous years. New abundance estimates incorporate a night-time correction factor adapted from Perryman *et al.* (1999).

Start dates	End dates	Cited abundance	SE	Source	New abundance	New SE	CV
18 Dec. 1967*	3 Feb. 1968	12,921	964	1	13,776	1,082	0.0785
10 Dec. 1968*	6 Feb. 1969	12,070	594	1	12,869	708	0.0550
8 Dec. 1969*	8 Feb. 1970	12,597	640	1	13,431	758	0.0564
9 Dec. 1970*	12 Feb. 1971	10,707	487	1	11,416	590	0.0517
18 Dec. 1971*	7 Feb. 1972	9,760	524	1	10,406	614	0.0590
16 Dec. 1972*	16 Feb. 1973	15,099	688	1	16,098	834	0.0518
14 Dec. 1973*	8 Feb. 1974	14,696	731	1	15,960	872	0.0546
10 Dec. 1974	7 Feb. 1975	12,955	659	1	13,812	781	0.0565
10 Dec. 1975	3 Feb. 1976	14,520	796	1	15,481	930	0.0600
10 Dec. 1976	6 Feb. 1977	15,304	669	1	16,317	818	0.0501
10 Dec. 1977	5 Feb. 1978	16,879	1,095	1	17,996	1,249	0.0694
10 Dec. 1978	8 Feb. 1979	13,104	629	1	13,971	753	0.0539
10 Dec. 1979	6 Feb. 1980	16,364	832	1	17,447	984	0.0564
27 Dec. 1984	31 Jan. 1985	21,443	1,182	1	22,862	1,379	0.0603
10 Dec. 1985	7 Feb. 1986	20,113	927	1	21,444	1,120	0.0522
10 Dec. 1987	7 Feb. 1988	20,869	913	1	22,250	1,115	0.0501
10 Dec. 1992	7 Feb. 1993	17,674	1,029	2	18,844	1,190	0.0632
10 Dec. 1993	18 Feb. 1994	23,109	1,262	2	24,638	1,475	0.0599
13 Dec. 1995	23 Feb. 1996	22,571	1,182	3	24,065	1,393	0.0579
13 Dec. 1997	24 Feb. 1998	27,958	2,853	4	29,758	3,122	0.1049
13 Dec. 2000	5 Mar. 2001	18,246	1,707	4	19,448	1,882	0.0967
12 Dec. 2001	5 Mar. 2002	16,848	1,599	4	18,178	1,780	0.0979

Source: 1=Buckland *et al.* (1993); 2=Laake *et al.* (1994); 3=Hobbs *et al.* (2004); 4=Rugh *et al.* (2003).
*Observation site was at Yankee Pt., 5km north of Granite Canyon.

Although the field of view covered >150°, observers generally searched through an arc of only 40-50° near the standard azimuth, a line perpendicular to the coastline (241° magnetic) intersecting the survey site. Standard search efforts were the same as in previous surveys (Rugh *et al.*, 1993). Each observer searched for whales independently and hand-recorded entries onto a data form. When a gray whale pod was first sighted within the primary viewing range, the time, horizontal bearing and vertical angle were recorded as a 'north sighting'. Magnetic compasses in *Fujinon* 7 × 50 binoculars provided the horizontal bearings, and 14 reticle marks in the binoculars provided vertical angles relative to the horizon (detailed in Rugh *et al.*, 1993; Kinzey and Gerrodette, 2001). A chart was available to help predict the time and vertical angle at which the pod would cross the standard azimuth. If possible, another sighting (the 'south sighting') was recorded when the whale(s) were close to the standard azimuth. Entries included time, horizontal bearing, vertical angle, and a pod size estimate, as well as any unusual behaviours and calf sightings. During periods of routine search effort, observers recorded the number of times each pod was sighted within the viewing area ('cue counts'). These counts were treated in the analysis as cues per pod and compared between seasons as a quantifiable index of relative visibility. Also, observers recorded start and end times of systematic search effort and times of environmental changes, which included visibility (subjectively categorised from 1 to 6 for excellent to unacceptable), sea state (Beaufort scale) and wind direction. Visibility was recorded as a sightability index, that is, a record of how well observers thought they could see whales, not the visibility of the horizon. Primary considerations in establishing visibility were: (1) observer attentiveness; (2) light level and direction; (3) rain or fog; and (4) sea state. During shift changes, observers conferred and agreed on visibility and Beaufort conditions.

In addition to the primary effort, a second, independent effort was conducted once or twice daily during January (when sighting rates were high[1]) for each of the three seasons reported here. The paired effort had a field of view and station conditions nearly identical to those of the primary effort. This provided an independent sighting record, allowing for comparisons between observers, and an estimation of the number of whales missed within the viewing area. The methods applied were as described in Rugh *et al.* (1993), which have been used since 1986 (Rugh *et al.*, 1990) during much or all of these shore-based studies.

The offshore distribution of whale sightings was documented through a shore-based 25 power binocular on a fixed-mount, as per Rugh *et al.* (2002). No correction factor, other than for probability of detection by distance, was applied for whales passing the site beyond 5.6km (3 n.miles) because aerial surveys conducted in the past have estimated that only 1.28% of the whale population travels beyond this distance (Shelden and Laake, 2002), considered to be the outer limit of the typical viewing area for shore-based observers.

Abundance analysis

Population abundance calculations from the observer counts followed the analytical procedures described in Hobbs *et al.* (2004). These methods account for: (1) whales that passed during periods when there was no observational effort (prior to and after the census season, at night or when visibility was poor); (2) whales missed within the viewing range during on-effort periods; (3) differential sightability by observer, pod size, distance offshore and various environmental conditions; (4) errors in pod size estimation; (5) covariance within the corrections due to variable

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[1] It has not proved cost effective to maintain two simultaneous efforts throughout the season, and the abundance algorithm includes a density dependent factor.

sightability by pod size; and (6) differential diel travel rates of whales. Although the methods used here are essentially the same as used in the past, a new correction factor for night travel rate has been included (see below) based on a study conducted by Perryman *et al.* (1999). Previous abundance analyses (e.g. Hobbs *et al.*, 2004) have used several different programs for synthesising the observational records. In order to streamline the analysis process, a new program was written (Lerczak, 2003) providing a common language (Visual Basic) and convenient outputs for use in analyses carried out in S-plus or R statistical programs. The same analysis routine was applied to each of the three seasons reported here.

Calculation of crossing times

The recorded sighting time and location closest to the standard azimuth (usually within a few degrees of 241°) were used to estimate the time and offshore distance at which each pod crossed this line. This was based on the assumption that southbound migrating gray whales travel at 6km/h (3kt) and maintain a course parallel to shore (c.f. Swartz *et al.*, 1987). The time from the beginning to the end of the survey season was partitioned into effort periods (time between 07:30 and 16:30 with visibility 4 or better and an observer on effort) and non-effort periods. Each sighting was assigned to the effort or non-effort period into which it fell as a function of the calculated time it crossed the standard azimuth. Whale sightings were eliminated from the analysis if they crossed this line prior to the start of an effort period or if they had not crossed the line by the end of an effort period.

Correction for missed pods

Corrections for whale pods missed within the viewing area during a systematic effort were estimated from the paired, independent observation records. These paired records provide capture-recapture data that were used to estimate the total number of pods passing the station while observations were underway. A scoring algorithm established by Rugh *et al.* (1993) defined matches between records based on time, offshore distance and pod size. Iterative logistic regression (Buckland *et al.*, 1993) was used to identify significant covariates to the probability of detecting a pod and to estimate the detection probability associated with each recorded pod. Possible covariates were observation site (north or south), effort period (1, 2 or 3), day, observer, distance offshore, pod size, sea state (Beaufort scale), wind direction and whales per hour averaged over each day. After establishing the matching record, all covariates were examined individually as binned categorical data. All covariates were then entered into the model, and a backward step-wise model selection was followed until no step decreased the Akaike Information Criterion (AIC). Once the best linear model fit was determined, interactions between each possible pair of the retained covariates were considered. The logistic regression model was used to estimate p_{ei}, the detection probability of the *i*th pod of size *e* passing during the effort periods of the survey. The total number of pods of size *e* passing during the effort periods of the survey, $\hat{M}_e$, and its variance were estimated as:

$$\hat{M}_e = \sum_{i=1}^{m_e} \frac{1}{p_{ei}}$$

$$Var\left(\hat{M}_e\right) = \sum_{i=1}^{m_e} \left[\frac{1-p_{ei}}{p_{ei}^2} \right] + D_\beta\left(\hat{M}_e\right)^T \hat{\Sigma}_\beta \, D_\beta\left(\hat{M}_e\right)$$

where m_e is the number of pods of size *e* sighted from the primary site, D_β ($\hat{M}_e$) is the vector of partial derivatives of $\hat{M}_e$ with respect to the vector of parameters, β, estimated in the logistic regression evaluated at $\hat{\beta}$, the vector of parameter estimates, and $\hat{\Sigma}_\beta$ is the estimated variance-covariance matrix of $\hat{\beta}$ (c.f. Borchers, 1996). The estimated total number of pods passing the field site while systematic efforts were underway, $\hat{M}$, is then:

$$\hat{M} = \sum_{e=1}^{E} \hat{M}_e \qquad Var\left(\hat{M}\right) = \sum_{e=1}^{E} Var\left(\hat{M}_e\right) + 2\sum_{j=1}^{E-1} \sum_{k=j+1}^{E} D_\beta\left(\hat{M}_j\right)^T \hat{\Sigma}_\beta \, D_\beta\left(\hat{M}_k\right)$$

where *E* is the largest observed pod size.

Bias in recorded pod sizes

Bias in the recorded pod size resulting from underestimation by observers is removed by an additive correction which has been estimated for each pod size, *e*, from data collected during earlier surveys (Laake *et al.*, 1994), with the variances and covariances calculated as in Hobbs *et al.* (2004). Corrected pod sizes were then summed by effort period with the sum rounded to the nearest integer so they could be used in the FORTRAN program *gwnorm*. In earlier gray whale analyses, observed pod sizes were used with *gwnorm*; however, in the present analyses, distributions of the estimated number of whales passing during an effort period were analysed via *gwnorm* so that the variance inflation factor was based on variation in the passage rate of whales rather than the passage of pods.

The total number of whales, W_e, passing the observation site during effort periods represented by pods of size *e*, was estimated as:

$$\hat{W}_e = \hat{M}_e\left(e + b_e\right) \qquad Var\left(\hat{W}_e\right) = Var\left(\hat{M}_e\right)\left(e + b_e\right)^2 + \hat{M}_e^2 \hat{\sigma}_{b_e}^2$$

where b_e is the estimated additive bias correction for *e* from Laake *et al.* (1994) and $\hat{\sigma}_{b_e}$ is the bootstrap estimate of the variance of b_e. The variance consists of two summands representing the estimation errors in $\hat{M}_e$ and b_e.

The total number of whales, *W*, passing the site during usable effort periods was estimated as:

$$\hat{W} = \sum_{e=1}^{E} \hat{W}_e,$$

$$CV\left(\hat{W}\right) = \frac{1}{\hat{W}} \sqrt{\sum_{e=1}^{E} Var\left(\hat{W}_e\right) + 2\sum_{j=1}^{E-1} \sum_{k=j+1}^{E} \left[\left(j + b_j\right) D_\beta\left(\hat{M}_j\right)^T \hat{\Sigma}_\beta \, D_\beta\left(\hat{M}_k\right)\left(k + b_k\right) + \hat{M}_j \hat{M}_k \hat{\sigma}_{b_{jk}} \right]}$$

where E is the maximum observed pod size and $\hat{\sigma}_{b_{jk}}$ is the bootstrap estimated covariance of b_j and b_k.

Correction for whales passing during off-effort periods (f_t)

The rate of whales passing the site was modelled by a normal distribution with Hermite polynomials added to adjust for skewness, kurtosis and higher moments (Buckland *et al.*, 1993). The model defines a bell-shaped rate function, *q*(*t*), of expected whales per day that was integrated to correct for periods when no search effort was underway. The correction factor, f_t, was defined as the ratio of the area under *q*(*t*) integrated over the entire survey period, *Q*, to the area under *q*(*t*) integrated only over effort periods. Although the histograms used to portray the seasonal distribution of sighting rates averaged data through each day, the model used to interpolate the generalised distribution was based on each effort period no matter how small. No corrections were applied for whales passing prior to or after the apparent start and end of the migrations based on the distribution of sighting rates for the respective season (Figs 1-3), and no correction was included for whales travelling beyond the viewing range of the shore-based observers because these factors appear to involve very few whales without satisfactorily quantifiable estimates.

Correction for nocturnal travel rates (f_n^*)
The correction for night travel rate, f_n = 1.020 (SE=0.023), used by Buckland *et al.* (1993), was based on data from three radio-tagged gray whales recorded by Swartz *et al.* (1987) during both day and night hours near Granite Canyon, excluding six other whales followed either during the day or the night. To further study diurnal variations in gray whale travel rates, Perryman *et al.* (1999) recorded thermal images of whales at Granite Canyon, California, while the census of the southbound migration was underway in January 1994, 1995 and 1996 (total sample size=116h by day; 146h by night). As with the tagging results, the imagery showed elevated travel rates at night, or put more accurately, depressed rates during the day, perhaps related to increases in non-migratory behaviour in daylight hours after 15 January (Perryman *et al.*, 1999)[2]. For calculations of abundance, median sighting dates were used instead of 15 January (which, on most years are virtually the same), because the median date may be more representative of the whales' behaviour than a calendar date. Accordingly, an additive correction factor $f_n^* = 1 + 0.28\, f\,(15/24)$ from Perryman *et al.* (1999) was applied, where f is the fraction of total whales migrating after the median date. Because this fraction is 0.5, the correction can be simplified to f_n^* = 1.0875 with SE=0.116 $f\,(15/24)$=0.0363. This SE term has been changed from the one in Perryman *et al.* (1999) in that the amount of night hours is 15/24 instead of 14/24, and the f term has been included (J. Laake, pers. comm.).

Synthesis
The total number of whales passing through the viewing area at Granite Canyon during effort periods, W, was multiplied by corrections for whales passing when no search effort was in effect (including periods with poor visibility), f_t, and differences in diurnal/nocturnal travel rates, f_n^*. Accordingly, the total abundance estimate, N is calculated as:

$$\hat{N} = W \cdot f_t \cdot f_n^*$$

The coefficient of variation, CV, is estimated by:

$$CV\left(\hat{N}\right) = \sqrt{\frac{\chi^2/df}{W} + CV^2\left(f_t\right) + CV^2\left(f_n^*\right) + CV^2\left(W\right)}$$

where χ^2/df is a variance inflation factor from fitting a Hermite polynomial to the sighting rates.

RESULTS

Sample size

Shore-based observations were conducted during most daylight hours from 13 December 1997 to 24 February

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[2] To confirm that there was a change in whale behaviour midway through the migration, the primary observational records were examined for milling whales and whales seen going north before 13 February 1998, 15 February 2001 and 18 February 2002, dates on which it appeared the northbound migration was underway. Of 37 gray whales seen deviating from their migration south, 30 (81%) of the deviations were after 15 January.

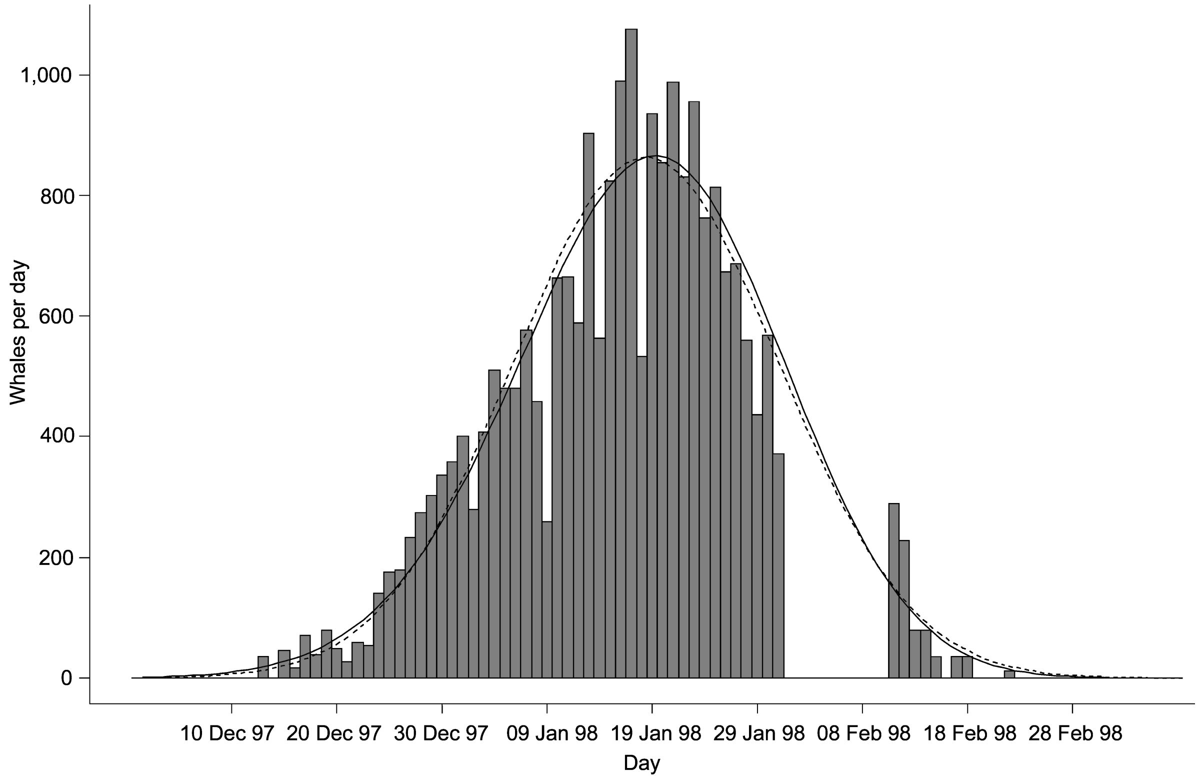

Fig. 1. Histogram of estimated number of whales per day for 1997/98 with Hermite polynomial (solid line) and normal distribution (dashed line) fitted to whales per effort period.

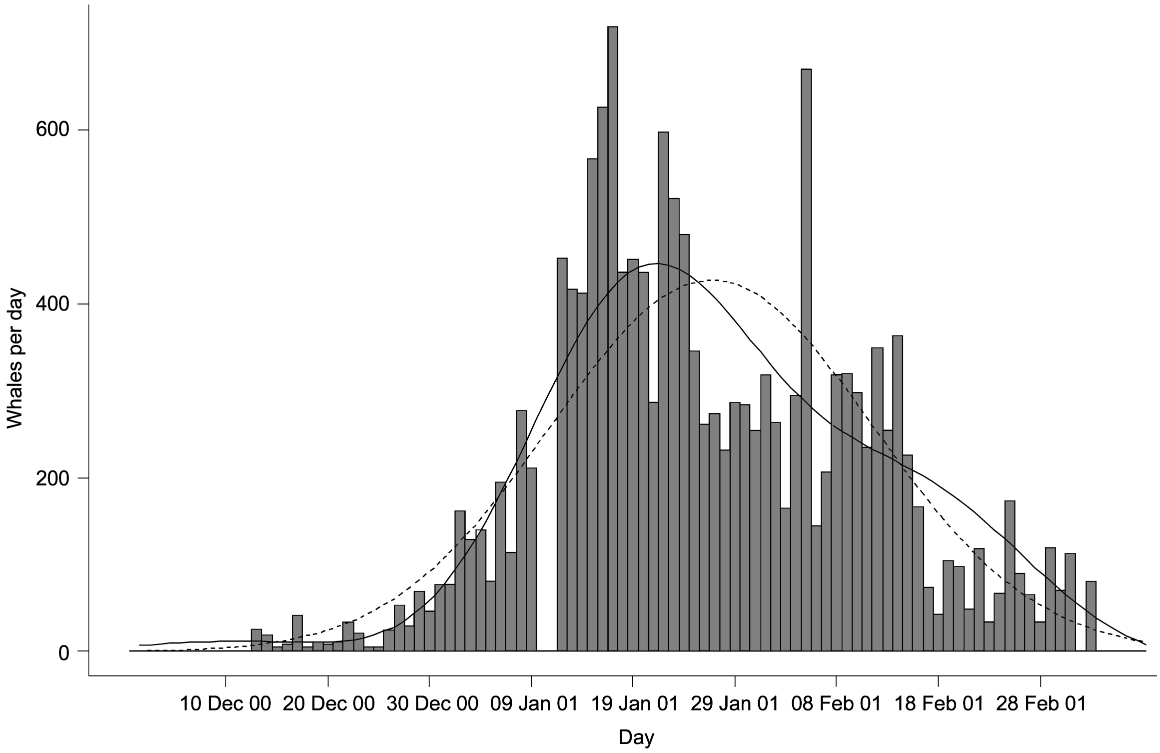

Fig. 2. Histogram of estimated number of whales per day for 2000/01 with Hermite polynomial (solid line) and normal distribution (dashed line) fitted to whales per effort period.

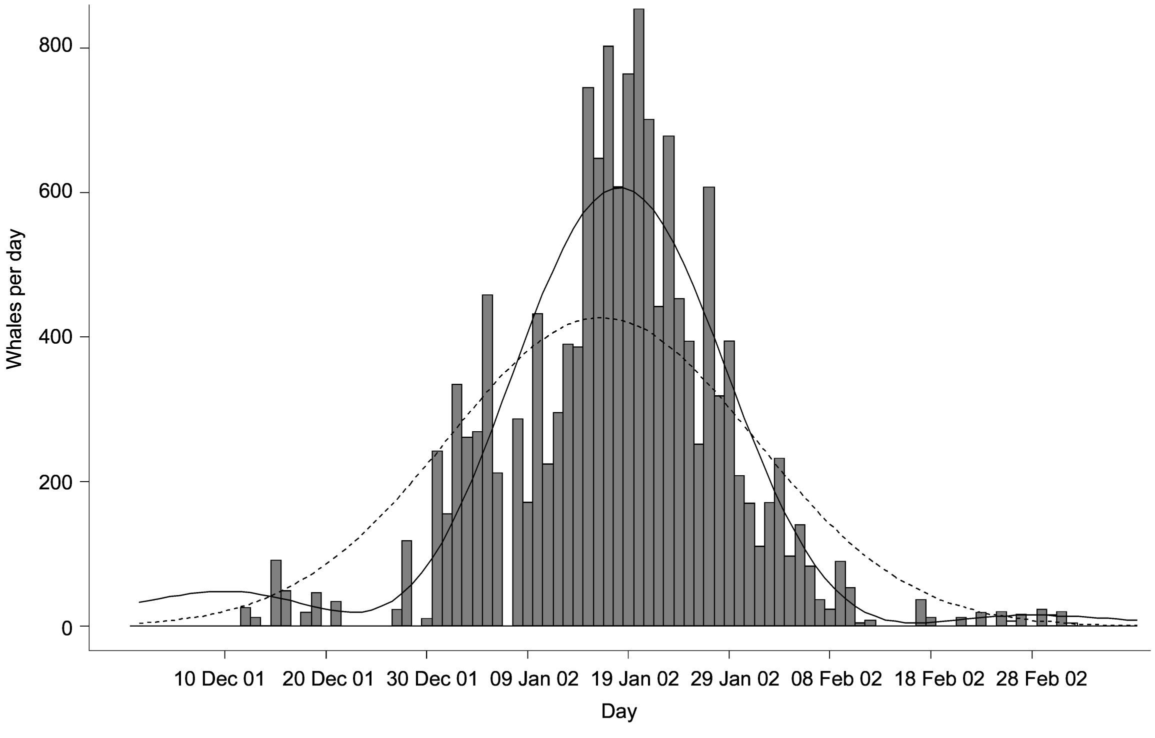

Fig. 3. Histogram of estimated number of whales per day for 2001/02 with Hermite polynomial (solid line) and normal distribution (dashed line) fitted to whales per effort period.

1998[3] (507.2h of effort), 13 December 2000 to 5 March 2001 (698.5h) and 12 December 2001 to 5 March 2002 (621.1h; Table 1). Southbound whales were seen throughout almost all of these days. During the 1997/98 study, a total of 2,346 pods of gray whales was recorded from the primary observation shed, compared to 1,694 in 2000/01 and 1,712 in 2001/02, despite the longer seasons in the latter two years. Searches were maintained from the secondary shed 3-26 January 1998 (173.9h and 1,325 pods), 29 December 2000 to 11 February 2001 (300.6h and 1,169 pods) and 2 January to 7 February 2002 (174.0h and 945 pods). In each of these years, there were respectively, 107.4h, 55.6h and 53.1h on the fixed, high-power binoculars.

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[3] No effort was conducted from 3 to 10 February 1998 due to unusually violent storm activity in the area. The road to the Granite Canyon study site was washed out, preventing any further survey work at that site for the remainder of the southbound migration period. On 11 February, the weather improved sufficiently to allow the establishment of an alternate site at Point Lobos State Park where the final two weeks of survey effort was conducted.

Visibility

Of the six subjective visibility categories, very little time was spent in excellent conditions (2.6h in 1997/98; 5.4h in 2000/01; 10.9h in 2001/02; Table 2). Accordingly, the small sample sizes in excellent conditions were not considered representative sighting rates. Larger sample sizes in the other categories indicated there were no real differences between visibilities 2-4, but sighting rates dropped in visibilities 5 and 6 (Fig. 4). As has been done in previous seasons (e.g. Hobbs *et al.*, 2004), categories 5 and 6 (72.3h in 1997/98; 106.1h in 2000/01; 89.6h in 2001/02) were deleted from further analyses and were treated as unwatched periods. The remaining categories (approximately 85% of the total effort) did not need to have any corrections applied as a function of visibility.

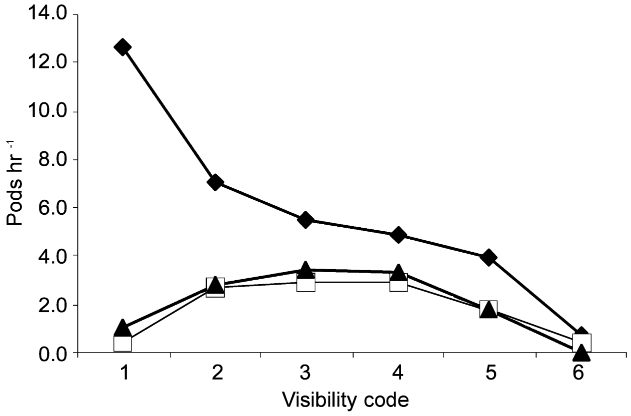

Fig. 4. Annual averages of gray whale pods seen in different visibilities (from excellent [1] to useless [6]; 1997/98 = diamonds; 2000/01 = squares; 2001/02 = triangles).

Table 2

Rates of sightings of gray whale pods as a function of visibility as recorded in the primary observation shed in 1997/98, 2000/01 and 2001/02.

Visibilities	Visibility code	Effort (h)	Number of pods	Pods h^{-1}	SE	Mean pod size	SE
1997/98							
Excellent	1	2.6	33	12.69	0.08	1.667	0.161
Very good	2	40.0	283	7.08	1.10	1.710	0.072
Good	3	162.6	901	5.54	0.42	1.594	0.036
Fair	4	230.0	1,129	4.91	0.32	1.546	0.030
Poor	5	61.0	240	3.94	0.45	1.500	0.061
Unacceptable	6	11.3	8	0.71	0.39	1.000	0.000
All effort	1-6	507.2	2,594	5.11	0.23	1.576	0.021
Usable effort	1-4	435.0	2,346	5.39	0.26	1.586	0.022
2000/01							
Excellent	1	5.4	2	0.37	0.25	1.500	0.500
Very good	2	83.1	223	2.68	0.43	1.874	0.075
Good	3	237.8	686	2.88	0.24	1.723	0.037
Fair	4	266.2	783	2.94	0.19	1.489	0.032
Poor	5	99.6	177	1.78	0.18	1.395	0.048
Unacceptable	6	6.5	3	0.46	0.27	1.000	0.000
All effort	1-6	698.5	1,874	2.68	0.12	1.611	0.022
Usable effort	1-4	592.4	1,694	2.86	0.14	1.635	0.024
2001/02							
Excellent	1	10.9	11	1.01	0.83	2.000	0.357
Very good	2	98.4	279	2.83	0.42	1.767	0.085
Good	3	238.5	814	3.41	0.36	1.677	0.035
Fair	4	183.7	608	3.31	0.34	1.515	0.036
Poor	5	85.8	151	1.76	0.35	1.325	0.053
Unacceptable	6	3.8	0	0.00			
All effort	1-6	621.1	1,863	3.00	0.19	1.611	0.024
Usable effort	1-4	531.5	1,712	3.22	0.21	1.636	0.025

The six visibility categories are subjective and might not have been consistently determined between seasons, therefore observers were asked to record the number of times each pod was seen (see Methods). These 'cue counts' provide an empirical indicator of relative visibility of whales. Accordingly, results show that cues/pod were closely correlated to visibility ($R^2 = 0.98$; $p<0.01$; Fig. 5). There were significant differences between years (mean ($\bar{x}$)=1.91 for 1997/98; 1.84 for 2000/01; 1.73 for 2001/02; $p<0.01$, ANOVA). This apparent decrease in annual averages suggests that sighting rates were generally better in 1997/98. However, this might instead be a reflection of differences between observers, many of whom were not available for more than one season, and many of the observers were new in the latter two years (see 'Observer Performance'). Since individual observers could have varying abilities or styles in recording sighting cues, the analysis of each observer's data between years is a more accurate comparison than pooling each year's results. Accordingly, cues pod^{-1} were compared between 1997/98 and 2000/01 and/or 2001/02 for each observer that participated in two or more of these three seasons. In all but 2 of 7 pair-wise ANOVA comparisons, there were significant differences ($p<0.05$ in each case), and among the five observers who did have inter-year differences, four had higher sighting rates in the latter two years. Therefore, visibility was probably better in 2000/01 and 2001/02 relative to 1997/98, so visibility changes do not explain the low counts made in the most recent seasons.

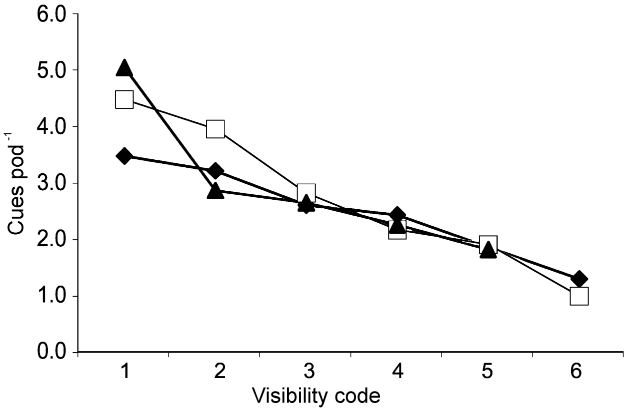

Fig. 5. Annual averages of cues/pod seen in different visibilities (from excellent [1] to useless [6]; 1997/98 = diamonds; 2000/01 = squares; 2001/02 = triangles).

Offshore distances

Several tests were run to establish whether or not inter-year differences in distance from shore were affecting changes in abundance estimates. Kendall's distribution-free test for independence (Hollander and Wolfe, 1973) showed no correlation ($p=0.36$) between average offshore distances (2.19, 2.09, 2.04, 2.33, 2.17, 2.20km, respectively) and abundance estimates (Table 1) for the years 1992-2002. This was the period when distances were established through binocular reticles (with given distances corrected to estimate the location of each whale pod when it crossed the standard azimuth at 241°) instead of uncalibrated estimates as recorded prior to 1992. ANOVA showed significant differences ($p<0.01$) in offshore distances within this period (1992-2002), but the largest mean distance of 2.33km occurred in 1997/98, the year with the highest abundance estimate, and distances in the most recent years ($\bar{x}$ =2.17 and 2.20km, respectively) were close to the average for all years ($\bar{x}$ =2.17km). Pooling distances from 1995-98 ($\bar{x}$ =2.17km; SE=0.012; *n*=5,946) and from 2000-02 ($\bar{x}$ =2.19km; SE=0.014; *n*=3,194) showed there were no significant differences (*t*-test; *p*=0.38). Therefore, the low abundance estimates in the latter two years cannot be described as a function of change in average distance from shore.

Perhaps average distances do not fully reflect variations in the proportion of the population missed as a function of distance because shore-based observers rarely could see whales beyond approximately 6km[4] (as estimated by aerial transects; Shelden and Laake, 2002). Accordingly, high-power binoculars were used to document whales passing beyond the perimeter of the standard search (Rugh *et al.*, 2002), often with good visibility as far as the horizon, 17km away. Although search effort ranged from 53 to 137h per season, this analysis was limited to only those periods when visibility was good throughout the viewing area (28.5, 48.5, 60.0, 22.8, 24.2hr for 1995, 1996, 1998, 2001, 2002 respectively). ANOVA of sightings per reticle showed no differences between these years (*p*=1.0), and χ^2 tests of sightings in 1.4km (0.75 n.miles) bins showed that the only significant differences (*p*<0.05) were between 2002 and both 1996 and 1998. In a *t*-test restricted to sightings beyond 5.6km, there were no significant differences (*p*=0.33) between pooled years (1995-98; $\bar{x}$ =7.45km; SE=0.39; *n*=20; and 2000-02, $\bar{x}$ = 8.98km; SE=1.44; *n*=8). It seems, then, that there is no evidence that the whale migrations in 2000/01 and 2001/02 were farther from shore than in other years, removing this as an explanation for the recent low abundance estimates.

Observer performance

When observers were compared through the paired sighting effort, it was evident that all missed a few whale sightings relative to the other observer. The paired records provided a means to compare many variables that may affect sighting rates. Individual categorical parameter fits of all covariates are shown in Table 3, indicating the respective correction factors. In a test of observer performance, averages in number of sightings recorded for each whale group (cues pod^{-1}) were compared among observers as a function of how many previous seasons of experience they had had with this project. In 1997/98, all of the observers were considered experienced, having had two or more seasons at Granite Canyon. In 2000/01, 5 observers were new, and 5 were experienced. In 2001/02, 6 were new, 3 had one season of experience, and 5 had several seasons. There was a direct correlation between experience and mean cues pod^{-1} (*p*<<0.01; ANOVA): first-time observers averaged 1.70 cues pod^{-1} (SE=0.02; *n*=3,019); during their second season observers averaged 1.77 cues pod^{-1} (SE=0.06; *n*=486); those with many seasons of experience averaged 2.08 cues pod^{-1} (SE=0.05; *n*=1,079). Furthermore, throughout their first season, observers showed an increase in cues pod^{-1} (*n*=11 observers; 3,019 observations; R^2=0.03; *p*<0.01), starting at 1.66 cues pod^{-1} and increasing to 1.84 by the 300th observation (most observers had at least 300 observations in a season).

Table 3

Covariates and fitted parameters estimated for the logistic regression of the probability of detection of whale pods by individual observers in 1997/98, 2000/01 and 2001/02.

Coefficient	Estimate	SE
1997/98		
Intercept	2.079	0.415
Site (N Vs S)	-0.360	0.165
Period 2	-1.958	0.157
Period 3	-2.313	0.297
Observer A	-0.523	0.406
Observer B	0.033	0.313
Observer C	-0.309	0.315
Observer D	-0.590	0.305
Observer E	-1.514	0.212
Observer F	-1.118	0.368
Observer G	-0.695	0.310
Observer H	-1.660	0.250
Observer I	-0.015	0.406
Pod size	0.290	0.073
Pods hr^{-1}	0.034	0.014
Distance	0.001	2.000 x 10^{-4}
Distance^2	-1.743 x 10^{-7}	3.008 x 10^{-8}
Cos (wind dir.)	0.227	0.113
2000/01		
Intercept	1.512	0.282
Period 2	-0.804	0.176
Period 3	-1.397	0.163
Observer A	0.784	0.334
Observer B	-0.296	0.283
Observer C	0.151	0.296
Observer D	-0.255	0.214
Observer J	-0.333	0.377
Observer K	-0.837	0.201
Observer L	0.597	0.262
Observer M	-0.544	0.192
Observer N	-0.703	0.207
Pod size	0.509	0.079
Beaufort	0.080	0.054
Pods hr^{-1}	0.047	0.017
Distance^2	-3.83 x 10^{-8}	1.452 x 10^{-8}
Cos (wind dir.)	1.701	0.462
Beaufort x cos (wind dir.)	-0.613	0.165
2001/02		
Intercept	3.863	0.604
Site (N vs S)	0.869	0.369
Period 2	-0.309	0.207
Period 3	-1.143	0.177
Observer A	-1.400	0.526
Observer B	-1.230	0.526
Observer C	-0.462	0.557
Observer F	-0.994	0.557
Observer H	-2.531	0.482
Observer L	-2.697	0.619
Observer N	-3.112	0.624
Observer O	-1.567	0.630
Observer P	-3.179	0.604
Observer Q	-1.543	0.516
Observer R	-2.787	0.600
Observer S	-1.858	0.491
Observer T	-1.945	0.662
Pod size	0.337	0.079
Beaufort	-0.041	0.071
Visibility	-0.189	0.092
Pods hr^{-1}	0.050	0.016
Distance^2	-4.886 x 10^{-8}	1.248 x 10^{-8}
Cos (wind dir.)	-1.204	0.576
Cos^2 (wind dir.)	-0.861	0.258
Beaufort x cos (wind dir.)	0.325	0.185

Migratory timing

The passing rate of the 1997/98 migration was nearly symmetrical around the peak on 18 January 1998 ($\bar{x}$ =day 49.4; SE=0.18, with day 1=1 December; Fig. 1). A Hermite polynomial with added terms up to order 3 was hardly different from the normal distribution for this year.

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[4] During the past three seasons, 0.37% of the observers' sightings were beyond 5.6km (34 sightings, or 1.51%, in 1997/98; 4 sightings, or 0.25%, in 2000/01; 3 sightings, or 0.19% in 2001/02). Maximum distances were 15.9, 7.2 and 9.3km for the respective years. These sightings, when applied to the corrections for missed whales, may in part compensate for the calculated 1.28% of the population estimated to be beyond 5.6km (Shelden and Laake, 2002).

The mean sighting date in 2000/01 was 25 January (day 55.9; SE=0.14), 10 days after the expected date of 15 January (Rugh *et al.*, 2001). However, a ‘peak’ in sighting rates occurred on 17 January, which is within the expected time frame (Fig. 2). Sighting rates were lower than expected (relative to 1997/98) through most of this migration, but rates were much higher than expected after 15 February, when the migration usually ends. A Hermite polynomial of order 6 was fitted to the temporal distribution of the 2000/01 sighting data. Unlike in previous years, when the sighting rates closely approximated a normal distribution, in 2000/01 there was a nearly exponential rise in sighting rates from the start of the census until the peak in mid-January, followed by a disordered period until rates dropped in early March. Prior to 2001, these gray whale surveys were usually terminated by mid-February (Table 1); however, in 2001 the survey was extended an additional three weeks because whales continued to pass the site in significant numbers through February and into March.

In 2001/02, the mean sighting date was 16 January (day 47.3; SE=0.16), which was virtually the same as most dates observed in the 1980s and 1990s (Rugh *et al.,* 2001). An apparent peak in sightings occurred on 20 January. A Hermite polynomial distribution (of order 6) had a normal, bell-shaped curve appearance and was approximately symmetrical around the mean date. In 2002 the survey was again conducted until 5 March to better compare with the survey effort in 2000/01; however, the migration ended in 2002 as it typically had in the past, on or about 15 February (Rugh *et al.*, 2001).

Pod size

The mean recorded pod sizes during periods when visibility was adequate (1-4) was 1.586 (SE=0.022), 1.635 (SE=0.024) and 1.636 (SE=0.025) for 1997/98, 2000/01 and 2001/02, respectively. Sighting rates relative to each pod size are shown in Table 4. Observers tend to underestimate pod size, therefore bias corrections were applied as per Laake *et al.* (1994), based on aerial studies of previous years. These corrected pod size estimates are shown in Table 4 without rounding (values used in the abundance estimates are slightly different because they were based on whole integers for the respective effort periods). Average pod sizes after bias correction were 2.40, 2.43 and 2.43.

Corrections for using Point Lobos State Park in 1998

During a severe winter storm in February 1998 (during an El Niño year), part of the road to Granite Canyon was washed out and was not repaired until 7 May. The storm’s duration meant that eight days went by without any search effort. By 11 February the weather abated enough to allow two observers a chance to resume the search, but without access to Granite Canyon. The observations were made at *ad hoc* sites in Point Lobos State Park, 9km north of Granite Canyon and 7km south of Carmel. Two sites were used during the final two weeks of the survey (11-24 February) until the southbound migration appeared to be over. One site, in a car park at approximately 6m altitude, was used when there was rain because observers could retreat into a parked car. The other site, at 25m altitude, was accessed by a footpath used by many tourists. This was considered the primary site but could only be used in mild weather because of the lack of protection from the elements.

It was unclear how comparable the results between Pt Lobos and Granite Canyon were, so in January 2002 two observers returned to Pt Lobos to conduct counts while counts were ongoing at Granite Canyon. Because the two sites are 9km apart, the average whale swimming at 6km h^{-1} takes 1.5h to reach Granite Canyon after passing Pt Lobos. Accordingly, data collected at Pt Lobos were compared to sightings made 1.5h later at Granite Canyon. During 8.8h of systematic searches on three days, 69 pods were sighted at Pt Lobos and 62 at the primary site at Granite Canyon. Recorded pod sizes were higher at Pt Lobos ($\overline{x}$ =2.09 whales pod^{-1}; SE=0.15) than at Granite Canyon ($\overline{x}$ =1.45; SE=0.09; $p<0.001$, $Z=3.61$), which provided a correction of 0.70 used to adjust the Pt Lobos counts relative to those of Granite Canyon. This correction is nearly the same value (0.67) as the average correction for pod sizes at Granite Canyon (Table 5). Therefore, it appears that the pod size estimates made at Pt Lobos were fairly accurate. The higher counts at Pt Lobos are probably because the whales were concentrated closer to shore ($\overline{x}$ =1.65km; SE=0.099; $n=76$) than at Granite Canyon ($\overline{x}$ =2.13km; SE=0.096; $n=62$; $p<0.001$; Z=3.51), where the continental shelf is somewhat wider. Whales on the southbound migration arrive at Pt Lobos after crossing Monterey Bay (which cuts eastward as much as 30km from a straight-line course across the mouth of the bay), Carmel Bay (which cuts 3km eastward) and Carmel Canyon (which is as much as 360m deep on a line connecting the outermost points of land). If gray whales use bathymetry to navigate, then these marine canyons cause them to move closer to shore.[5]

Abundance estimates

Uncorrected counts (m) of southbound gray whale pods seen during periods with good visibility (<5) during the primary effort are shown in Table 5 for 1997/98, 2000/01 and 2001/02 (2,347; 1,694 and 1,712, respectively). These counts of pods were multiplied by corrected pod sizes to estimate the number of whales (W =7,299; 5,053 and 5,103, respectively). These estimates were then corrected for whales passing between effort periods (f_t) and a differential night travel rate ($f_n^* = 1.0875$). In addition, the abundance estimate in 1997/98 has been corrected for counts conducted at Pt Lobos instead of Granite Canyon. This correction and the new program for matching sightings meant the previously circulated estimate for 1997/98 (26,635; CV=10.06%; 95% log-normal confidence interval = 21,878 to 32,427; Hobbs and Rugh, 1999) has been changed. Accordingly, the abundance estimate for 1997/98 is 29,758 whales (CV=10.49%; 95%; log-normal confidence interval (CI) = 24,241 to 36,531), the estimate for 2000/01 is 19,448 whales (CV=9.67%; 95%; CI = 16,096 to 23,498) and the estimate for 2001/02 is 18,178 whales (CV=9.79%; 95%; CI = 15,010 to 22,015). The lower bound of the 95% CI for the 1997/98 estimate (24,241) does not overlap with the upper bounds for the 2000/01 and 2001/02 estimates (23,498 and 22,015, respectively), indicating this to be a statistically significant drop.

Table 1 and Fig. 6 summarise estimates of gray whale abundance including standard errors (Table 1) and 95% log-normal confidence intervals (Fig. 6). Two regressions were run on these data, one from 1967/68 to 1997/98 and the other from 1967/68 to 2001/02. Assuming a Poisson error distribution with over-dispersion and a logarithmic link function, estimates of the average annual increase were 2.59% (SE=0.28%) and 1.86% (SE=0.32%), respectively.

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[5] Although it appears that gray whales pass closer to shore at Pt Lobos than at Granite Canyon, only the latter site has an unobstructed view from a sea cliff with vehicle access, nearby accommodations, restricted access for tourists, options for constructing observation sheds and a research facility appropriate for the gray whale census.

Table 4

Estimations of total numbers of whales passing during systematic observational periods (visibility ≤ 4) in 1997/98, 2000/01 and 2001/02.

Pod size	Number of recorded pods	Average correction for missed pods	Bias-corrected pod size	$\hat{M}_e$	$\hat{W}_e$	$CV(\hat{W}_e)$
1997/98						
1	1,535	1.364	1.941	2,094	4,065.2	14.48%
2	502	1.275	2.646	640	1,693.9	10.68%
3	177	1.197	3.607	212	764.0	12.12%
4	77	1.165	4.25	90	381.1	17.31%
5	23	1.204	5.25	28	145.4	19.31%
6	15	1.069	6.25	16	100.2	14.94%
7	8	1.046	7.25	8	60.7	15.00%
8	7	1.026	8.25	7	59.3	13.30%
9	2	1.018	9.25	2	18.8	18.75%
10	1	1.050	10.25	1	9.9	29.29%
All	2,347	1.320	2.40	3,098	7,298.5	9.45%
2000/01						
1	998	1.302	1.941	1,300	2,522.6	14.05%
2	459	1.211	2.646	556	1,470.6	10.12%
3	151	1.124	3.607	170	612.4	11.70%
4	49	1.081	4.25	53	225.1	17.21%
5	26	1.051	5.25	27	143.4	15.05%
6	7	1.037	6.25	7	45.4	17.14%
7	2	1.027	7.25	2	14.9	23.55%
8	1	1.013	8.25	1	8.4	26.84%
10	1	1.001	10.25	1	10.3	20.09%
All	1,694	1.250	2.43	2,115	5,032.5	8.46%
2001/02						
1	1,033	1.258	1.941	1,299	2,522.2	14.07%
2	432	1.247	2.646	540	1,425.5	10.30%
3	150	1.188	3.607	178	642.9	11.98%
4	65	1.122	4.250	73	309.9	17.04%
5	15	1.054	5.250	16	83.0	16.71%
6	11	1.049	6.250	12	72.1	15.46%
7	3	1.030	7.250	3	22.4	20.18%
8	1	1.043	8.250	1	8.6	31.53%
16	1	1.006	16.250	1	16.3	14.49%
All	1,711	1.240	2.43	2,123	5,103.0	8.46%

Table 5

Estimated abundance and intermediate parameters for the eastern North Pacific stock of gray whales counted at Granite Canyon.

	1997/98			2000/01			2001/02		
Parameter	Est.	SE	CV (%)	Est.	SE	CV (%)	Est.	SE	CV (%)
Total number of pods recorded by primary observers during effort periods with visibility ≤ 4 (*m*):	2,347			1,694			1,712		
Mean recorded pod size:	1.59	0.022	1.39	1.635	0.024	1.45	1.636	0.025	1.54
Corrected mean pod size:	2.40	0.018	0.77	2.43	0.019	0.80	2.43	0.021	0.86
Estimated number of whales passing during effort periods (*W*):	7,299	690	9.45	5,053	427	8.46	5,103	432	8.46
Correction for pods passing outside effort periods (f_t):	3.749	0.015	0.39	3.539	0.005	0.15	3.276	0.009	0.27
Estimated total number of whales without night travel correction (*Q*):	27,364			17,883			16,715		
Correction for night travel (f^*_n):	1.0875	0.036	3.33	1.0875	0.036	3.33	1.0875	0.036	3.33
Estimated number of whales passing Granite Canyon ($\hat{N}$):	**29,758**	3,122	10.49	**19,448**	1,882	9.67	**18,178**	1,780	9.79
95% CI	24,241-36,531			16,096-23,498			15,010-22,015		

A discrete, logistic model was also fit to the data:

$$N_{t+1} = N_t + R_{\max} N_t (1 - N_t / K) - C_t$$

where N_t is the abundance in year *t*, $R_{\max}$ is the maximum growth rate, *K* is the carrying-capacity and C_t is the catch in year *t*. The parameters of the model ($N_0 = N_{1967}$, $R_{\max}$ and *K*) were estimated by maximising the log-normal likelihood function. The estimated asymptote, *K*, was 26,290 (SE=1,562).

DISCUSSION

Gray whale abundance estimates made from data collected at or near Granite Canyon during southbound migrations showed an upward trend of 2.5% from 1967 to 1995

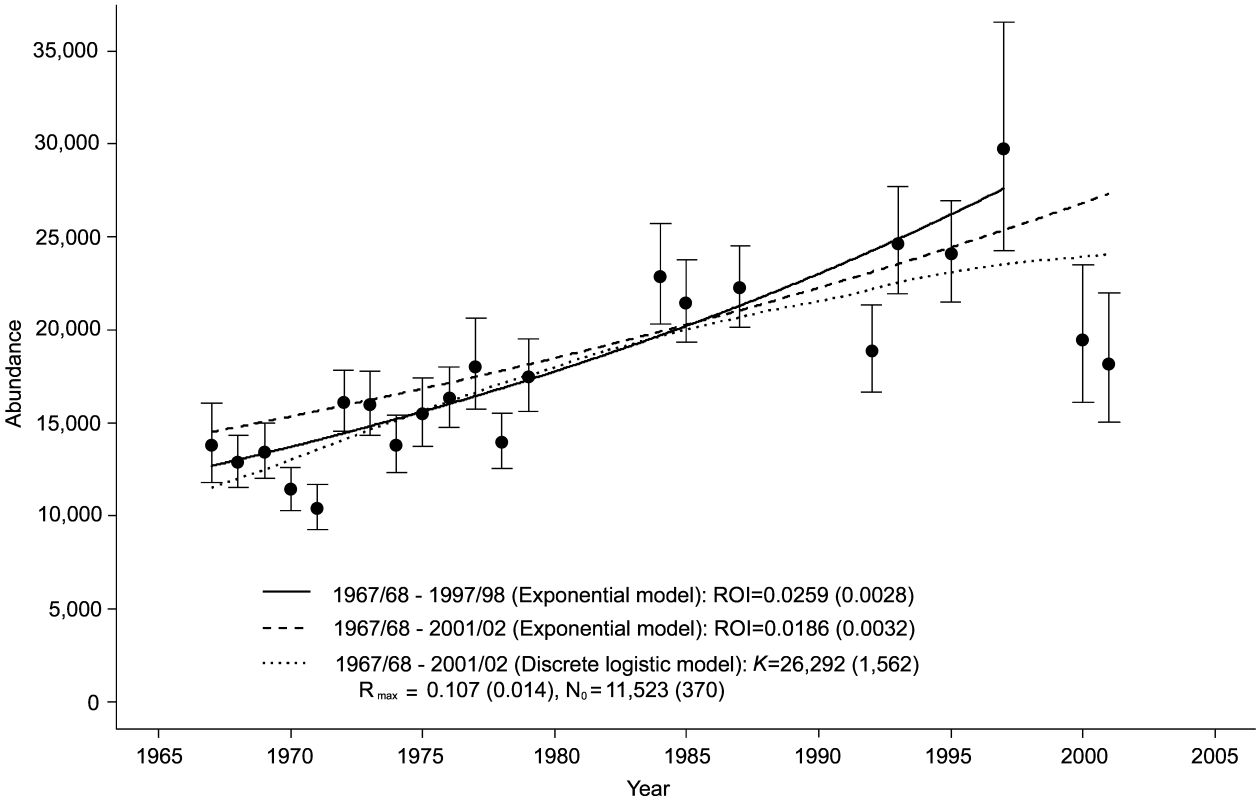

Fig. 6. Gray whale abundance estimates and 95% log-normal confidence intervals. The regression of abundance on time (1967/68 to 1997/98 and 2001/02), assuming a Poisson error distribution with over-dispersion and a logarithmic link function, gave estimates of average annual rate of increase (ROI) of 2.59% (SE=0.28%) and 1.86% (SE=0.32%). A discrete, logistic model was also fit to the abundance data (dotted curve), including parameter estimates and their standard errors (in parentheses) where K is the carrying-capacity, $R_{\max}$ is the maximum growth rate and N_0 is the abundance in the first year 1967/68).

(Buckland and Breiwick, 2002). This trend appeared to continue through 1997/98, but in 2000/01 and 2001/02, abundance estimates were well below this trend line. Although at first the low counts in 2000/01 were thought to be related to an unusual migration (see ‘Migratory timing’), with whales continuing to go south long after the usual timeframe, the migration timing in 2001/02 appeared to be quite typical, and yet the abundance was still low. Both of these years (2000/01 and 2001/02) had estimates that were 65% and 61%, respectively, of the estimate made in 1997/98. Several possible explanations for the low estimates are presented here.

Visibility

If visibility was persistently lower in the 2000/01 and 2001/02 seasons relative to 1997/98, the year of the highest counts, then the recent counts might have been downwardly biased. Yet, there was no real difference in the percentage of time spent in adequate visibility (conditions 1-4) in 1997/98 (86%) and 2000/01 (85%) or 2001/02 (86%). Also, the number of sightings recorded per pod (cue counts) for observers who were available for multiple seasons, suggested that visibility was better in the more recent years than in 1997/98. Therefore, visibility does not explain the low encounter rates recorded in 2000/01 and 2001/02.

Change in offshore distribution

Data from the standard effort (using reticles in 7 × 50 binoculars) and from dedicated effort on fixed, high-power binoculars (25 × 150) showed there was no apparent offshore shift in the migration that could explain low encounter rates in 2000/01 and 2001/02 relative to previous years.

Observers

Approximately half of the observers were new in each of 2000/01 and 2001/02, and therefore it may be argued that their lack of experience led to lower sighting rates, explaining in part the low counts from these two years. Indeed, cue counts indicate that new observers made fewer sightings than experienced observers: the overall mean cues pod^{-1} showed a 4% drop in 2000/01 and 9% drop in 2001/02 relative to 1997/98, but this was far less than the observed drop in abundance. Although new observers had lower sighting rates than experienced observers, inter-observer differences were compensated for in the corrections for missed pods, to minimise bias. With sufficient overlap and testing among observers between seasons, it is not likely that changes in performance would explain the low counts recorded in the final two seasons.

Migratory change

The timing of the gray whale southbound migration past Granite Canyon has been phenomenally regular, with median dates consistently near 15 January in recent years, and generally ending in mid-February as the northbound migration begins (Rugh *et al.*, 2001). In 2000/01, however, the median migration date appeared to be 10 days late, and whales continued passing the station until the effort was terminated on 5 March, when counts of southbound whales had dropped to 0.7h^{-1}, and northbound counts had risen to 1.3h^{-1}. Small numbers of gray whales continued to travel south long after this date as was evident from shore-based surveys at Piedras Blancas, 130km south of Granite Canyon (W. Perryman, pers. comm.) and Pt Vicente, 485km south of Granite Canyon, near Los Angeles in southern California

(A. Schulman-Janiger, pers. comm.). Although the migratory timing in 2000/01 was unusual, the timing appeared normal in 2001/02, yet the abundance estimate was still low, and so the delayed migration in 2000/01 does not explain the low numbers. Of course, it is possible that in both years a significant portion of the population did not migrate as far south as Granite Canyon. Unexpectedly low abundance estimates also occurred in 1970/71, 1971/72, 1978/79 and 1992/93, yet each (except the first) was followed by several seasons with much higher estimates (Fig. 6). One of the explanations for the low estimate in 1992/93 was that varying proportions of the gray whale population remain north of Granite Canyon each year (Laake *et al.*, 1994). Perhaps in some years, such as in 2000/01 and 2001/02, many whales did not migrate as far south as Granite Canyon.

Abundance decline

If none of the other theories fully explain the low counts recorded recently, then the change may be attributed to being a true drop in the population size. This may have been indicated by a high mortality rate between the 1997/98 and 2000/01 censuses: 274 dead gray whales were reported in 1999 (Le Boeuf *et al.*, 2000; Norman *et al.*, 2000) and 368 in 2000 (Gulland *et al.*, 2005), significantly above the average rate of 38yr^{-1} from 1995-98 (Norman *et al.*, 2000). Of course, these stranding reports reflect only a small proportion of the total mortality rate. Visibly emaciated whales (Le Boeuf *et al.*, 2000; Moore *et al.*, 2001) and low calf production (Perryman *et al.*, 2002) are suggestive of a deterioration in available resources, such as benthic amphipods in the Bering and Chukchi seas (Le Boeuf *et al.*, 2000), perhaps associated with unusually high sea temperatures in 1997 (Minobe, 2002). However, several factors indicate this was an acute event, not a chronic situation or trend, because since then: (1) counts of dead gray whales (21 in 2001 and 26 in 2002; Gulland *et al.*, 2005) have dropped to levels below those seen prior to this event; (2) living whales no longer looked emaciated in 2001 (W. Perryman, pers. comm.); and (3) calf counts in 2002, a year after the event ended (gestation=13 months; Rice and Wolman, 1971), and in subsequent years were near or higher than averages for previous years (Perryman *et al.*, 2004; A. Schulman-Janiger, pers. comm.).

The drop in abundance following many years of increasing numbers invites speculation on this population's carrying-capacity. Gray whale abundance prior to commercial takes in the 19th century has been estimated at 30,000-40,000 (Scammon, 1874) or 15,000-20,000 (Henderson, 1972). Models projecting into the future have produced point estimates of carrying-capacity (K) based on the abundance data through 1995/96 ranging from 24,000 to 35,000 (Wade and DeMaster, 1996; 1998; Wade, 1997; 2002), but with broad credibility intervals. Wade and Perryman (2002) obtained more precise interval estimates of K by incorporating the abundance data through 2001/02, as well as data from surveys for calves during the northbound migration. Their 90% credibility interval incorporating the calf estimates through 2001 was 19,830 to 28,470, suggesting that currently the population is essentially at K.

After the heavy exploitation of gray whales, especially from 1855-74, the abundance may have dropped to only a few thousand animals (Henderson, 1972). This low abundance lowered the efficiency of the hunt, reducing further takes, but it has also led to conservation measures, which began in 1937 under the International Agreement for the Regulation of Whaling[6] (Reeves, 1984). Since that time, this stock of whales has demonstrated a remarkable recovery. During the documented period from 1967/68 to 1995/96, there was a 2.5% per annum increase in abundance estimates (Buckland and Breiwick, 2002). A plateau in this increase has been anticipated (Reilly, 1992; Wade, 1997), but through 1997/98[7], abundance estimates continued to rise almost linearly. Until 2000/01, there was only a suggestion of density-dependence beginning to occur (Wade and DeMaster, 1998), though it has been proposed that this whale stock was close to its equilibrium level (Wade, 2002; Wade and Perryman, 2002). Possibly, then, the abundance estimates from 2000/01 and 2001/02 were the first clear indication that the abundance was responding to environmental limitations, albeit temporarily exaggerated by unusual conditions in 1998 and 1999. It is anticipated that in the future, abundance estimates will rise and fall as the population finds a balance with the carrying-capacity of its environment.

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[6] International Convention for the Regulation of Whaling. 1946. (Dec 2, 1946), art. XI, 62 Stat. 1716, T.I.A.S. No. 1849, 161 U.N.T.S. 72, 4 Bevans 248, 249.

[7] The abundance estimate from 1997/98, which in Fig. 6 appears to be well above the others, may have been biased upwardly if interpolations overcompensated for missed observational periods that year (8 days were lost due to a severe storm, and on 14 days effort was conducted at an alternate site; i.e. Granite Canyon was used during only 70% of the season).

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Vocalisations of Antarctic blue whales, *Balaenoptera musculus intermedia*, recorded during the 2001/2002 and 2002/2003 IWC/SOWER circumpolar cruises, Area V, Antarctica

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ABSTRACT

Blue whale vocalisations recorded during the 2001/2002 and 2002/2003 International Whaling Commission-Southern Ocean Whale and Ecosystem Research (IWC/SOWER) cruises were analysed to determine the feasibility of using acoustic recordings for sub-species identification of the Antarctic blue whale (*Balaenoptera musculus intermedia*) and the pygmy blue whale (*B.m. brevicauda*). The research was conducted in IWC Area V, from latitude 60°S to the ice edge and between longitudes 130°E and 150°E on the *Shonan Maru* (2001/2002), and between 150°E and 170°W on the *Shonan Maru No.2* (2002/2003). Data including 15 groups consisting of 42 animals, as well as opportunistic recordings of an unknown number of animals during evening sonobuoy stations were examined for this study. Vocalisations included long-duration 28Hz tonal sounds and relatively short-duration frequency-modulated sounds. The short-duration calls were similar to vocalisations recorded in the presence of blue whales in other locations worldwide. Not all recordings contained the long-duration 28Hz call, considered to be a species-specific vocalisation of Antarctic blue whales. None of the sounds that have previously been attributed to pygmy blue whales were detected. The long-duration 28Hz tonal vocalisations included 3-unit calls, considered to be song phrases, as well as simple 28Hz sounds and 28Hz sounds followed by a downsweep. The centre and peak frequencies of the 28Hz tone for these three sound types were stable regardless of signal strength; however, for the 3-unit vocalisation, the presence and characteristics of their 2nd and 3rd units were variable. Examination of two distinct groups of simultaneously vocalising blue whales showed no evidence of temporally repeated patterns of vocalisations (song phrases). The results of this study suggest that the peak frequency of the 28Hz vocalisations may be used as a diagnostic feature to aid in discriminating between Antarctic blue whales and pygmy blue whales in the field; however, examination of vocalisations in relation to group size and behaviour are necessary to understand the circumstances in which the 28Hz vocalisations are produced.

KEYWORDS: BLUE WHALE; COMMUNICATION; VOCALISATION; ANTARCTIC; MANAGEMENT PROCEDURE; SURVEY-ACOUSTIC; POPULATION ASSESSMENT; ACOUSTICS; DISTRIBUTION; SOWER

INTRODUCTION

Two sub-species of blue whale are found in the Southern Ocean, the Antarctic blue whale, *Balaenoptera musculus intermedia*, and the pygmy blue whale, *B.m. brevicauda*. There appears to be a general geographic segregation of the sub-species in the austral mid-summer, with pygmy blue whales occurring primarily north of 60°S and Antarctic blue whales south of this latitude (Kato *et al.*, 1995). Sub-species discrimination in the field is problematic because it relies on experienced observers noting relative body proportions and details of the head shape. Population research related to the conservation of large baleen whales requires accurate species identification (IWC, 1995). Recent studies indicate that monitoring of blue whale vocalisations may provide a means of determining sub-species in the field (Ljungblad *et al.*, 1997; 1998; Stafford *et al.*, 1999; 2001).

Sounds recorded in the presence of blue whales can be divided into two categories: short-duration or long-duration (Thompson and Cummings, 1996; Norris and Barlow, 2000). The short-duration vocalisations consist of individual pulses and frequency-modulated (FM, typically downswept) sounds of less than five seconds duration. These vocalisations vary in frequency and duration and have been recorded in the presence of blue whales in many locations (Thompson and Cummings, 1996; Ljungblad *et al.*, 1997; Stafford *et al.*, 2001). Short-duration sounds appear to be common; however, they are underrepresented in the literature.

Long-duration vocalisations are composed of one or more units that are FM or amplitude-modulated (AM) sounds and longer than five seconds (McDonald *et al.*, Submitted). An individual unit is defined as a continuous sound having consistent characteristics; these vocalisation units are often repeated in patterned sequences, or songs (Payne and McVay, 1971; McDonald *et al.*, Submitted). These song units have been shown to vary geographically (Cummings and Thompson, 1971; Edds, 1982; Thompson and Friedl, 1982; Alling *et al.*, 1991; Thompson and Cummings, 1996; Stafford *et al.*, 1999; 2001). Preliminary examination of sounds recorded in the presence of Antarctic blue whales and pygmy blue whales in the Southern Hemisphere indicate a similar geographic distribution of long-duration, low-frequency song units (Clark and Fowler, 2001; Ljungblad *et al.*, 1997).

Recordings of pygmy blue whales off Madagascar show repetitive sequences of 10-20s tonal sounds in the 25-45Hz band (Ljungblad *et al.*, 1998). Pygmy blue whale vocalisations recorded off Australia consist of three separate long tonal units in the 18-26Hz band (McCauley *et al.*, 2000). The long-duration sounds recorded in the presence of Antarctic blue whales in the Antarctic consist of patterned sequences of tonal sounds composed of three distinct units. The first tone is centred at 28-29Hz with a duration of 8-12s. A short 2s downsweep from 28-20Hz connects the first tonal unit to the third, a slightly modulated tone (20-18Hz), that is approximately 8-12s in duration (Ljungblad *et al.*, 1998). The three-unit vocalisation, or phrase, is usually repeated

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every 70-80s, at intervals between 40-50s (Ljungblad *et al.*, 1998). There is a high degree of variability in the presence and intensity of the three individual units, and therefore we use the terms '3-unit vocalisation' to describe vocalisations with all three units intact, '28Hz downsweep' to describe vocalisations with the first two units intact, and '28Hz tone' to describe vocalisations where only the first unit is intact.

In addition to studying the distinct variation in vocalisations of the two sub-species of blue whale for accurate species identification in the field, knowledge of the behavioural contexts of these sounds is needed for long-term vocal and population studies. The blue whale component of the International Whaling Commission's SOWER (Southern Ocean Whale and Ecosystem Research) programme obtains videos, photographs, biopsies and acoustic recordings of blue whales in the field. This study examines recordings and behavioural information obtained in the presence of blue whales during two seasons of SOWER cruises to provide a preliminary examination of the variability associated with the 3-unit vocalisation, and its effect on blue whale population studies in the Southern Ocean.

METHODS

Data collected by the authors during the 2001/2002 SOWER cruise from the vessel *Shonan Maru* and the 2002/2003 SOWER cruise from the vessel *Shonan Maru No.2* were used for this study. The research area surveyed was in IWC Area V[1] between 130°E-150°E (2001/2002) and 150°E-170°W (2002/2003), and extending from 60°S to the ice edge (Fig. 1). Line-transect visual observations of cetaceans were conducted between 06:00 and 18:00 local time, weather permitting, using a visual observation team consisting of three tiers of experienced observers (Anonymous, 2002b). Summaries of the methods are given in Ensor *et al.* (2002 and 2003). Briefly, the visual observation team was responsible for sighting and positively identifying whales, estimating group sizes and obtaining biopsy samples, video tapes, and photo-identification photographs. An acoustician was responsible for obtaining recordings in the vicinity of blue whales and collecting opportunistic evening recordings.

The primary acoustic recording method used expendable DIFAR (Direction Finding and Ranging) AN/SSQ 53B sonobuoys. These were deployed in close proximity to sighted blue whales and monitored for as long as time permitted for a minimum of 30 minutes. Opportunistic recordings were also made while drifting in the evenings. The sonobuoy radio signal was received *via* the ship antenna, which was coupled to an *ICOM* IC-R100 single channel receiver. This output was connected to a *Sony* DAT TCD-D7 recorder (flat frequency response from 5Hz to 24kHz) or a *Sony* mini-disk MZ-R700 recorder (frequency response 20Hz-20kHz ± 3dB). Recordings were later digitised to a *Sony* PCG-FX120 computer (sample rate 48kHz) using the software program ISHMAEL (Mellinger, 2001) and analysis was performed using Spectra-Plus software. All vocalisations attributed to blue whales with a strong signal-to-noise ratio (SNR) from the 2001/2002 season were measured (44.1kHz sample rate, 32768 point FFT size, 90% overlap, Hamming window). Only long-duration calls with a strong SNR from the 2002/2003 season were measured (5512Hz sample rate, 8192 point FFT size, 90% overlap, Hanning window).

All high-quality vocalisations attributed to blue whales were categorised according to their frequency and duration characteristics. The short-duration FM calls included amplitude-modulated downsweeps, high-frequency downsweeps, low-frequency downsweeps, high-frequency upsweeps, low-frequency upsweeps and complex calls. The long-duration sounds included the 3-unit vocalisations, 28Hz downsweeps and 28Hz tonal vocalisations. Measurements were made of lowest frequency, highest frequency, centre frequency (for tonal sounds), start frequency, end frequency, frequency shift, peak frequency, and duration for all vocalisations and vocalisation units. Measurement of the time between the deployment of the sonobuoy and detection of the first 3-unit vocalisation was made to examine feasibility of using these vocalisations for *in situ* species identification.

Temporal patterns of vocalisations were examined for a series of recordings (10.5 hours total) associated with blue whale sightings on 23 January 2003. All times are given as local times at sea. Bearings to each vocalisation were obtained using DIFAR signal processing. This was performed using an automatic *MATLAB* function within Ishmael that executes a series of commands for de-multiplexing the DIFAR signal (software developed by Greenridge Sciences, Inc.), and determines the bearing to a sound source (software designed by M. McDonald). Bearings of individual vocalisations allowed the detection of distinct groups of vocalising whales, so that patterns of vocalisations could be examined within and between groups. It was not possible to use DIFAR to distinguish individuals within groups due to the close association and variable movement patterns of animals. The paucity of recording tapes available during the 2001/2002 season necessitated recording at the lowest possible sampling rate to maximise the recording time (with a sample rate of 32kHz, the frequency response of the *Sony* TCD-D7 was 20-14,500Hz ± 1dB). This eliminated the multiplexed DIFAR signal and so bearings could not be obtained for these data.

RESULTS

Recordings were made in the vicinity of 12 blue whale groups (31 animals total) during the 2001/2002 season and in the vicinity of three blue whale groups (11 animals total) for the 2002/2003 season (Table 1). Blue whale sounds were detected during 14 of these 15 groups.

Blue whale encounters

2001/2002

Between 6 and 8 January 2002, a total of nine sightings of blue whales were observed within an area bounded by 64°18'S and 64°29'S and 136°29'E and 137°24'E, near the northern margin of belts of the pack ice. Seven of the groups (totalling 14 animals) were determined to be Antarctic blue whales; photo and/or video and biopsy attempts were made for these groups (Table 1). The eighth was a group of three animals observed at night that was not approached and was classified as undetermined blue whales. A distant group of two animals determined to be 'like' blue whales were sighted outside of a larger congregation of blue whales, but these animals were not approached. During all encounters, sounds attributed to Antarctic blue whales were recorded, although for most encounters these sounds were not detected within the first hour of recording (Table 1).

From 21-31 January 2002, three groups of Antarctic blue whales (totalling 12 animals) and three groups of unidentified blue whales (six animals total) were sighted.

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[1] For a description of IWC Areas, see Donovan (1991).

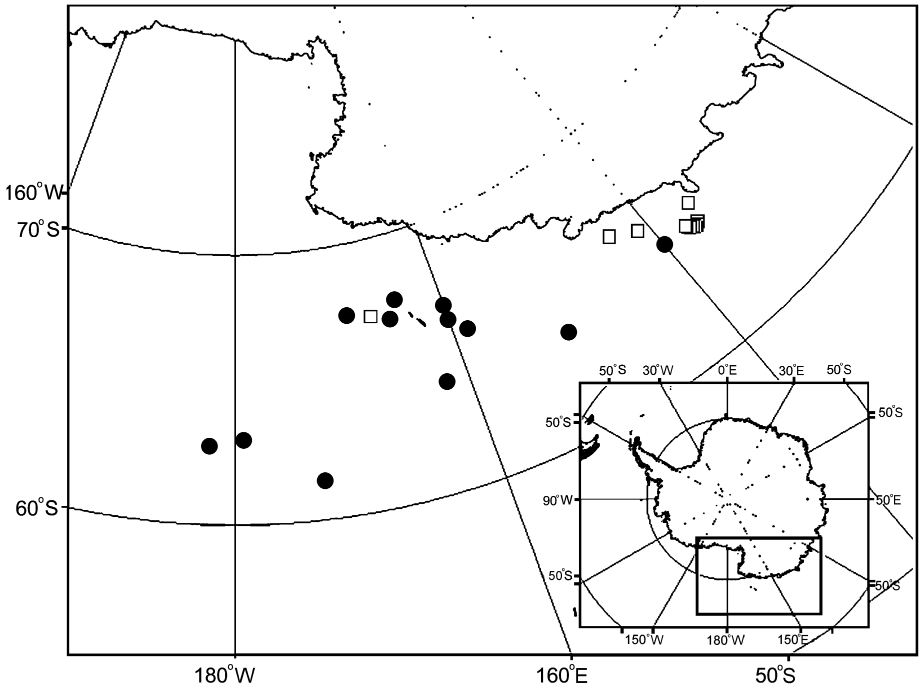

Fig. 1. Locations of acoustic detections of blue whales within the study area. The open squares represent recordings from blue whale sightings; the closed circles represent opportunistic evening sonobuoy stations with blue whale acoustic detections.

Table 1

Blue whale sightings with habitat, presence of vocalisation types and acoustic identification (ID) time. DIFAR confirmation of the location of the sound source was performed for 23/01/03 only; all others are speculative based on sound intensity and presence of other cetaceans. Detection of 'faint' vocalisations suggests that vocalisations were not produced from the sighted blue whales. Operations conducted for each sighting include: P = photo-identification; B = biopsy attempt; V = video; A = acoustic recording. Whale identification includes: True = Antarctic or true blue whale; Und = undetermined blue whale; Like = like-blue whale. Acoustic ID time is the time necessary for acoustic identification of species and is based on the time delay between the deployment of the sonobuoy and the first recorded 3-unit vocalisation.

Date	Sighting number	Group size	Habitat	Operations	3-unit vocalisations	Long tonal vocalisations	Short FM vocalisations	Whale ID	Acoustic ID time
06/01/02	1	2	Pack ice	P, B, V, A	Yes	Yes	Yes	True	>1hr
06/01/02	2	2	Pack ice	P, B, V, A	Yes	Yes	Yes	True	>1hr
06/01/02	3	1	Pack ice	P, B, V, A	Yes	Yes	Yes	True	>1hr
06/01/02	4	2	Pack ice	P, B, V, A	Yes	Yes	Yes	True	>1hr
06/01/02	5	1	Pack ice	P, B, V, A	Yes	Yes	Yes	True	>1hr
07/01/02	2	1	Pack ice	B, V, A	Yes	Yes	Yes	True	50min.
07/01/02	4	3	Pack ice	A	Yes	Yes	Yes	Und	45min.
08/01/02	2	5	Pack ice	P, B, A	None	Faint	Faint	True	-
08/01/02	5	2	Pack ice	A	Faint	Faint	Faint	Like	-
21/01/02	9	3	Ice edge	P, A	None	Yes	None	True	-
29/01/02	1	5	Open water	P, B, V, A	Yes	Faint	Faint	True	50min.
31/01/02	14	4	Open water	P, B, V, A	None	None	None	True	-
23/01/03	11	3	Pack ice	P, B, A	Yes	Yes	Yes	True	30min.
23/01/03	12	5	Pack ice	A	Yes	Yes	Yes	True	30min.
24/01/03	1	3	Pack ice	A	None	None	Yes	True	-

Photos and/or video and acoustic recordings were obtained for all but the undetermined groups of whales, and biopsy attempts were made for the groups identified on 29 and 31 January (Table 1). Very few vocalisations were recorded during these encounters.

2002/2003

On 23 January 2003, two groups of Antarctic blue whales (totalling 8 animals) were sighted in the outer margin of the pack ice, in the vicinity of 67°07'S and 166°54'E (Table 1). All animals appeared to be feeding on krill patches. Photography, biopsy attempts and acoustic recordings were undertaken during this sighting. Acoustic recordings began at 17:10 and continued throughout the night in the location of the scattered blue whale sightings. An additional sighting of three animals was detected in the middle of the night (24 January) and confirmed the continued presence of blue whales in the area. A detailed examination of DIFAR processing of the acoustic behaviour of these groups is described below.

Characteristics of blue whale vocalisations

A total of 85 hours of recordings were made from the *Shonan Maru* during the 2001/2002 cruise, with over 33 hours of recordings in areas of blue whale sightings. All recordings were monitored for the presence of sounds that could be attributed to blue whales, and over 42 hours of recordings contained blue whale vocalisations. A total of 193 short FM vocalisations and 261 long-duration tonal vocalisations (including 3-unit, 28Hz downsweep and 28Hz tonal vocalisations) were measured from the recordings for this survey.

A total of 38.7 hours of recordings were made from the *Shonan Maru No.2* during the 2002/2003 survey, including 11 hours in the vicinity of blue whale sightings. Sounds that could be attributed to blue whales were detected in nearly 26 hours of recordings, however only recordings associated with sightings were examined. A total of 92 long-duration tonal vocalisations with high SNRs were measured from this survey; short FM vocalisations were recorded, but not measured.

The most common short-duration FM vocalisation was the simple high-frequency downsweep from 76.3-40.0Hz, with a mean signal duration of 2.7 seconds (*n*=132, Table 2, Fig. 2(a)). The amplitude-modulation found in the pulsed downsweep appeared to be caused by propagation, and the basic frequency and duration characteristics closely resemble those of the high-frequency downsweep (Fig. 2(b)). The low-frequency downsweep (*n*=4), low-frequency upsweep (*n*=7) and high-frequency upsweep sounds (*n*=4) were relatively uncommon compared to the high-frequency downsweep vocalisations (Table 2).

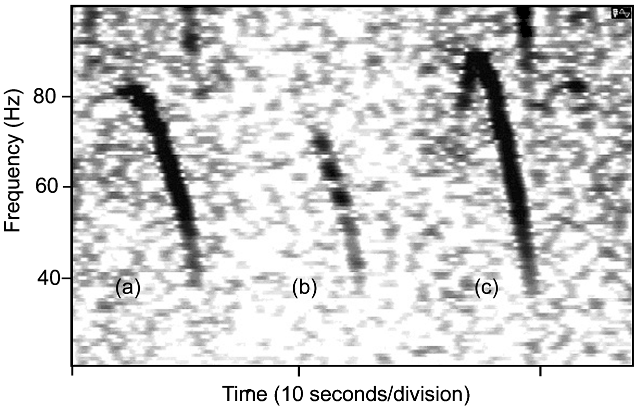

Fig. 2. Spectrogram of the most common short-duration FM vocalisations (5kHz sample rate, 2048 point FFT): (a) high-frequency downsweep; (b) amplitude-modulated downsweep; and (c) a complex variation of the high-frequency downsweep.

Although the complex sounds were variable in nature, several similar types were frequently observed. The most common complex vocalisations were variations on the high-frequency downsweep, with one or more inflection points (Fig. 2(c)). Other less common complex vocalisations are short, high-frequency downsweeps, variable FM sounds and ‘concave’ sounds (Fig. 3).

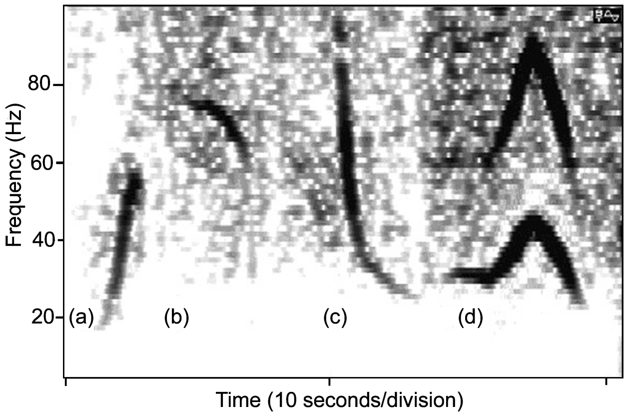

Fig. 3. Spectrogram of uncommon short-duration FM vocalisations (5kHz sample rate, 2048 point FFT): (a) high frequency upsweep; (b) short high frequency downsweep; (c) variable high frequency downsweep; and (d) concave vocalisation.

The long-duration calls were divided into three categories as described in the Introduction: the 3-unit vocalisation; the 28Hz downsweep; and the 28Hz tone (Fig. 4). The 3-unit vocalisation consisted of a tone at 27.7Hz lasting an average of 8.3 seconds, occasionally followed by a brief downsweep of variable duration, to a typically FM moan from 19.5-19.1Hz with an average duration of 6.9 seconds (Table 3). The 28Hz downsweep consisted of a moan at 27.7Hz of a variable duration followed by a downsweep to approximately 19.1Hz. Measurements of the entire sample of vocalisations (‘All’) were compared with a sub-sample of high-quality vocalisations (‘Best’); the centre and peak frequency of the 28Hz tone varied little, regardless of the vocalisation type or signal quality. For vocal animals, at least 30 minutes elapsed between sonobuoy deployment and initial detection of vocalisations associated with the Antarctic blue whales (defined as the acoustic identification time) (Table 1).

DIFAR analyses of vocalisations

DIFAR analyses of recordings from 23 January 2003 allowed differentiation between the calls from several groups of blue whales (sightings 11 and 12; Fig. 5). Determination of the bearing angles to the sound source using DIFAR analysis was performed on 1,069 vocalisations (208 long-duration, 861 short-duration) during the 11 hours of recordings. Close association of several animals within a group prevented identification of the vocalising animal in most cases; sound source for all sightings is for the group and not an individual animal (where group size >1). The similar DIFAR bearing angles of the blue whales (160°) and the ship (167°) at 17:45 indicate that animals biopsied at this time were vocalising (Fig. 5(a)). The discontinuity in the ship’s course at 18:15 (Fig. 5(b)) occurred as the ship returned to course and speed after biopsy sampling;

Table 2

Summary of frequency measurements and signal duration of simple short-duration FM vocalisations measured in the presence of blue whales during the 2001-2002 season (48kHz sample rate, 32,768 point FFT). Due to the variation inherent in the complex FM vocalisations, frequency characteristics and signal duration where not measured. AM downsweep calls are considered here for comparison with the high-frequency downsweep calls.

	Start frequency (Hz)			End frequency (Hz)			Frequency change (Hz)			Signal duration (s)		
Sound type (sample size)	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range	Mean	SD	Range
High-frequency downsweep (132)	76.3	11.5	48.5-106.9	40.0	7.5	22.1-58.4	36.2	10.6	11.7-65.6	2.7	0.9	0.8-7.5
AM downsweep (46)	73.7	5.9	60.1-85.4	41.0	6.0	23.3-52	32.7	6.5	22.1-48.6	2.5	0.7	1.1-5.9
Low-frequency downsweep (4)	29.8	0.6	28.9-30.3	19.1	0.6	18.1-19.5	10.7	1.1	9.4-12.2	3.0	0.8	2.4-4.3
High-frequency upsweep (4)	23.2	9.4	16.7-36.9	57.4	6.5	49-64.9	34.2	5.2	28-40.1	1.6	0.2	1.4-1.9
Low-frequency upsweep (7)	14.7	1.0	12.9-15.4	24.5	2.0	20.8-27.6	9.7	1.9	7.9-13.5	3.8	1.2	1.6-5.5

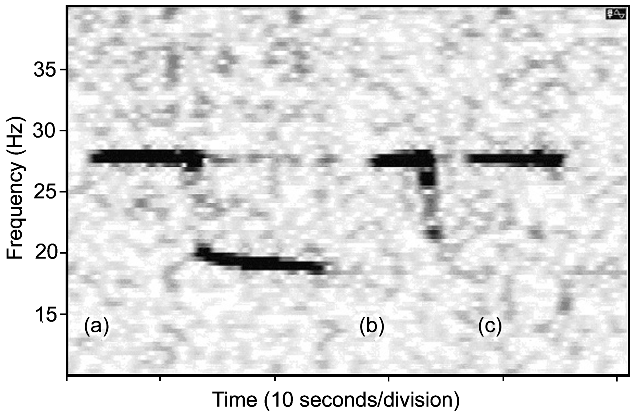

Fig. 4. Spectrogram of three long-duration blue whale vocalisations associated with different Antarctic blue whales (48kHz sample rate, decimation ratio 4:1, 32768 point FFT): (a) 3-unit vocalisation including 28Hz tone followed by an inter-tone interval and a 19Hz tone; (b) 28Hz tone plus downsweep; and (c) a 28Hz tone. All were sufficiently intense to suggest detection of the entire signal.

excessive noise during this manoeuvre temporarily precluded DIFAR processing. At 18:15 the ship's true course of 5° closely matched that determined by the sonobuoy. At 18:45 the ship's course crossed 0° in front of the sonobuoy.

Movement of the ship and the subsequent loss of the sonobuoy signal led to a gap in the recordings from 19:00 until 23:00. Recordings continued while the vessel was drifting for the remainder of the night, and DIFAR bearing angles suggest that a large congregation of vocalising blue whales separated into two distinct groups at about 00:30 (Fig. 6(a)). At 01:20 on 24 January a group of three blue whales (Fig. 6(b), 350°) were seen feeding next to the ship. DIFAR angles show that this group may have produced occasional short-duration FM vocalisations, but was not a part of either consistently vocalising group (40° and 150°).

Between 00:39 and 02:49 the sounds from the two simultaneously vocalising blue whale groups (denoted A and B) had a high SNR and well-defined DIFAR bearing angles. Both short-duration FM and long-duration tonal vocalisations from the two groups were plotted over time to identify patterns in temporal variation (Fig. 7). However, there was no apparent temporal pattern for the short-duration or long-duration 28Hz vocalisations. An expanded view of the common high-frequency downsweeps (HFDN) does not suggest countercalling between groups A and B.

DISCUSSION

This study shows that the 3-unit vocalisation is a geographically distinct call associated with Antarctic blue whales in the Southern Ocean south of 60°S. The 3-unit vocalisations recorded during 2001/2002 are consistent with previous results for sounds attributed to Antarctic blue whales in the Antarctic (Anonymous, 2002a; Clark and Fowler, 2001; Ljungblad *et al.*, 1998). The dataset presented here represents the largest analysis to date of these calls. None of the characteristic sounds attributed to pygmy blue whales in Madagascar (Ljungblad *et al.*, 1998) or Chile (Cummings and Thompson, 1971) were detected in the 247 hours of recordings. During the two cruises, all whales visually identified at the sub-species level were considered to be Antarctic blue whales. Genetic analysis of biopsies obtained on these cruises is underway and will hopefully confirm that these sounds were indeed produced by Antarctic blue whales. The association of specific calls exclusively to Antarctic blue whales provides a step towards *in situ* acoustic sub-species identification. As noted earlier, real-time identification currently relies on visual inspection by experienced observers. However, whales do not vocalise continuously, which limits the value of the technique. In addition, should time be limited, identifications in real-time may not be feasible due to the processing time required for a single identification. Nonetheless, the technique is a valuable tool, particularly when used in conjunction with genetic analysis and visual identification methods.

To use vocalisation for species identification one must be able to positively detect the call. There appears to be variability in the presence and characteristics of the 2nd (inter-tone downsweep) and 3rd (19Hz tone) unit of this vocalisation (Table 3). It is clear that blue whales produce both 28Hz tonal and 28Hz downsweep vocalisations, in addition to the 3-unit calls previously examined. Even with a high SNR, an overlap of the multi-path signals of long-

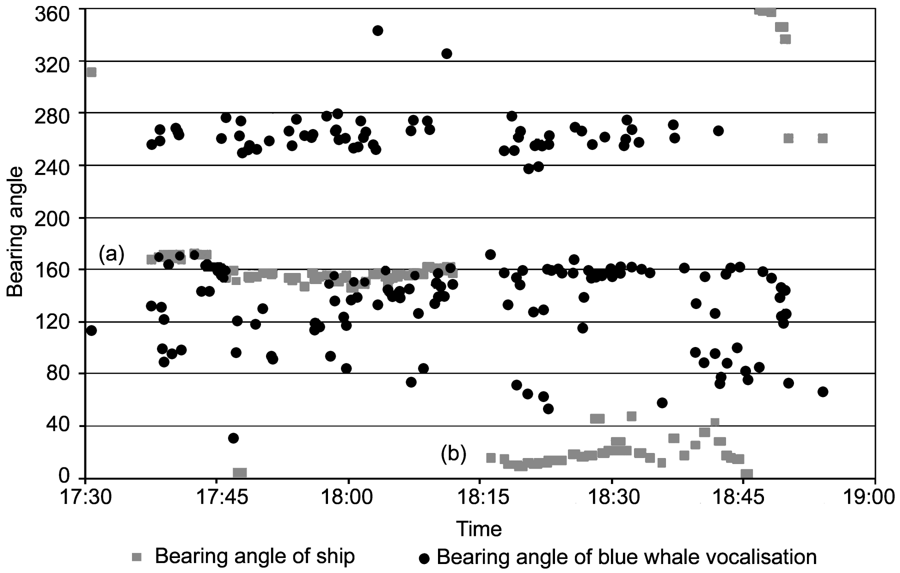

Fig. 5. DIFAR bearings for blue whales during biopsy attempts for sighting number 11 on 23 January 2003. Two distinct groups could be identified, one at 260° and the other at 160°. All magnetic bearing angles were converted to true angles for comparison.

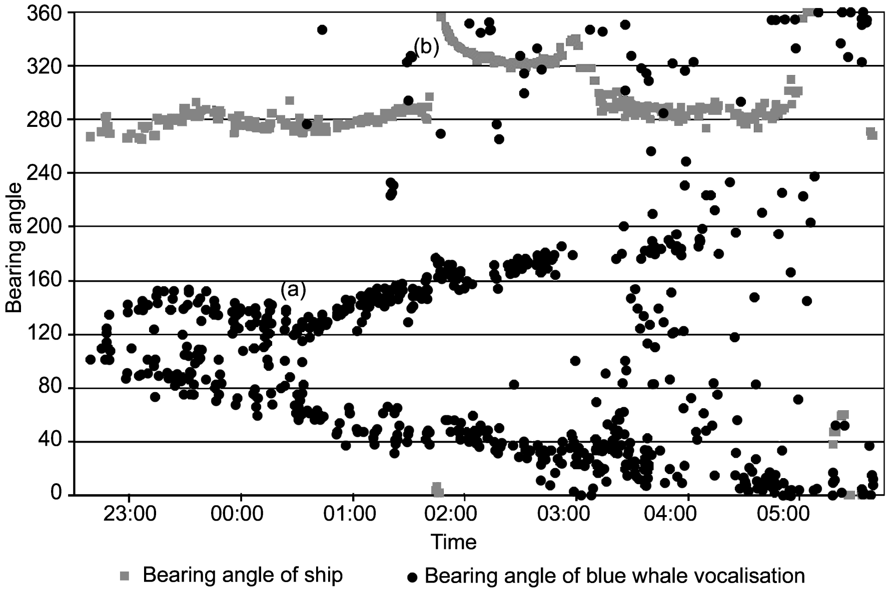

Fig. 6. DIFAR bearings for blue whales during evening recording in location of sighting numbers 11 and 12 for 23 January 2003 and sighting number 1 on 24 January 2003. All magnetic bearing angles were converted to true angles for comparison. The discontinuities in the ships' bearing angle are associated with repositioning.

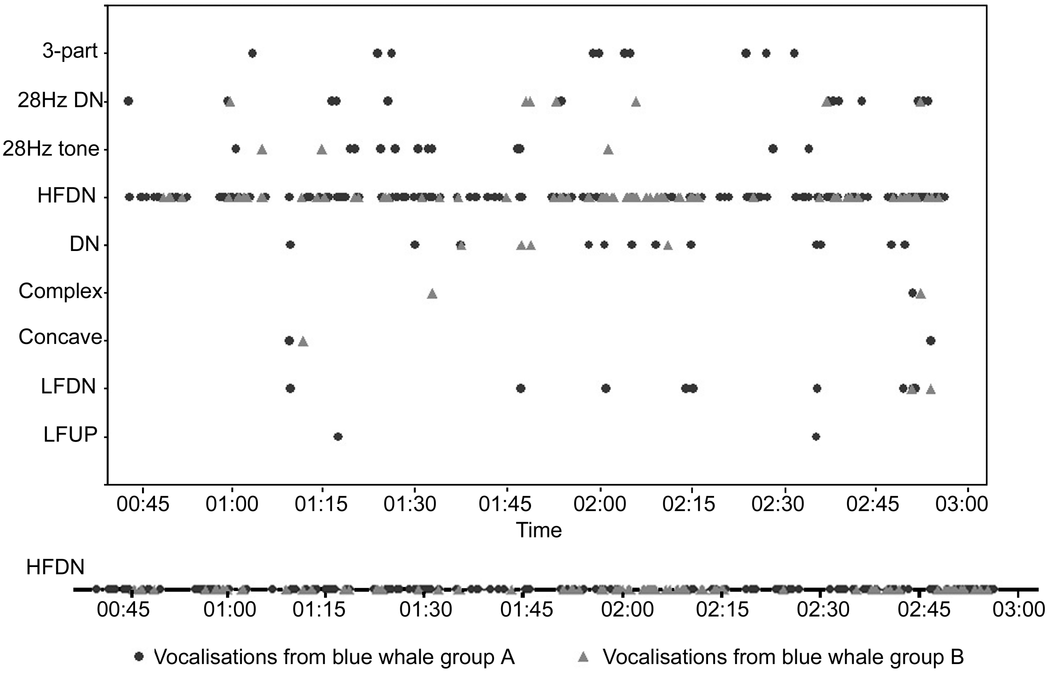

Fig. 7. Temporal variation of vocalisation types for the two blue whale groups from 00:30 until 3:00, 24 January 2003. An expanded view of HF DN vocalisations is provided for clarity. HF=high frequency; LF=low frequency; DN=downsweep; UP=upsweep.

duration calls can make it difficult to determine the characteristics of individual sounds. With increasingly faint vocalisations, it may be difficult to distinguish the 3-unit vocalisations from the 28Hz tonal and 28Hz downsweeps. The primary consistent feature is the tone centred at 27.7Hz; peak frequency varies little among these three vocalisations regardless of the signal intensity. If all three long-duration 28Hz vocalisations can be positively, and exclusively, linked to Antarctic blue whales, then this research suggests that it may be possible to attribute any long (6-12s) 28Hz tonal vocalisation south of 60°S to Antarctic blue whales. Research to date suggests this to be the case. Sub-species identification based on detection of the 28Hz tonal vocalisation is feasible for most groups of Antarctic blue whales, assuming a minimum one hour recording time. Future efforts should include deployment of sonobuoy arrays to localise calling animals so that comparisons with visual detection and genetic sampling of individual calling animals can be conducted.

Previous studies suggested that the long-duration, low-frequency sounds produced by blue whales are songs (Anonymous, 2002a), or patterned series of repeated vocalisations. The 2.5 hour sample of two vocalising groups within the larger congregation of scattered blue whales, recorded on 23 January 2003, does not suggest that the vocalisations were repeated in patterned series within a given group, or between the two groups (Fig. 7). The comparisons of vocalisations with the various blue whale

Table 3

Summary statistics of frequency measurements and signal duration for long-duration low-frequency vocalisations (5,512Hz sample rate, 8,192 point FFT). For each vocalisation type, all measured vocalisations are presented in the left column, while a sub-sample of the clearest vocalisations are presented for comparison in the column on the right. The sample sizes for all vocalisation types are given in parentheses.

		3-unit		28Hz down		28Hz tone	
		All (66)	Best (22)	All (70)	Best (19)	All (217)	Best (16)
First unit							
Centre frequency (Hz)	Mean	27.7	27.7	27.7	27.8	27.6	27.8
	SD	0.3	0.2	0.3	0.3	0.2	0.3
	Range	25.5-28.3	27.2-28.3	27.2-29.2	27.2-28.8	26.7-28.6	27.5-28.6
Signal duration (s)	Mean	8.7	8.3	5.9	6.3	7.9	7.2
	SD	1.7	1.5	1.9	1.9	2.2	2.2
	Range	4.1-13	5.3-11.2	2.2-10.2	3.1-9.7	2.7-18.8	3.6-11.5
Spectral peak (Hz)	Mean	27.9	27.8	28.0	28.0	28.0	27.8
	SD	0.4	0.2	0.4	0.2	0.3	0.4
	Range	26.8-29.2	27.4-28.3	26.8-29.5	27.7-28.3	26.8-28.3	26.8-28.3
Downsweep							
Low frequency (Hz)	Mean	19.8	20.0	20.0	19.1	-	-
	SD	0.9	0.8	1.7	1.6	-	-
	Range	18.1-22.5	18.3-21.8	16.9-23.8	16.9-23.5	-	-
Frequency shift (Hz)	Mean	7.3	6.9	7.5	8.4	-	-
	SD	1.3	1.3	1.7	1.5	-	-
	Range	4.4-9.5	4.4-8.8	4.0-11.5	4.5-11.5	-	-
Signal duration (s)	Mean	0.7	0.5	1.2	1.3	-	-
	SD	0.3	0.1	0.4	0.5	-	-
	Range	0.2-1.5	0.2-0.9	0.3-2.2	0.3-2.1	-	-
Third unit							
Start frequency (Hz)	Mean	19.4	19.5	-	-	-	-
	SD	0.4	0.5	-	-	-	-
	Range	18.6-21.2	18.8-21.2	-	-	-	-
End frequency (Hz)	Mean	19.1	19.1	-	-	-	-
	SD	0.5	0.6	-	-	-	-
	Range	17.9-21.2	18.2-21.2	-	-	-	-
Frequency shift (Hz)	Mean	0.3	0.4	-	-	-	-
	SD	0.5	0.5	-	-	-	-
	Range	0-2.0	0-2.0	-	-	-	-
Signal duration (s)	Mean	7.3	6.9	-	-	-	-
	SD	3.4	3.9	-	-	-	-
	Range	1.2-13.5	1.2-12.3	-	-	-	-
Spectral peak (Hz)	Mean	19.1	19.1	-	-	-	-
	SD	0.4	0.4	-	-	-	-
	Range	18.7-20.5	18.7-20.5	-	-	-	-

sightings (Table 3) suggest that there is considerable variation in the vocal behaviour of different groups. The sightings in the 2001/2002 study differed in group size, behaviour and habitat. Unfortunately, the locations of vocalising animals in relation to the sonobuoy could not be determined due to problems in recording the DIFAR signal. This severely limited our ability to examine temporal patterns, or the presence of song during the 2001/2002 cruise. Further research combining visual and acoustic studies (with functional DIFAR) on different blue whale groups is necessary to understand the circumstances in which the temporal patterns considered to be ‘song’ are produced.

Successful use of bottom-mounted hydrophones to monitor whale song in the Pacific (McDonald *et al.*, 1995; Stafford *et al.*, 1999) and Atlantic Oceans (Clark, 1995) led to deployment of similar hydrophones in the Southern Ocean to monitor the blue whale population year-round (Sirovic *et al.*, 2004). Minimum abundance can be estimated through noting the ranges of individual singing whales; however, our results suggest that only a small proportion of the blue whale population may be singing. The ability to relate geographically distinct vocalisations (song units) to an index of abundance relies heavily on their behavioural contexts. These concerns are essentially the same as those for using acoustics as a method for species identification.

The short FM vocalisations recorded here are similar to sounds associated with blue whales in other regions. With the exception of the high-frequency downsweep, most short FM vocalisations are uncommon. Groups of vocalising whales were noted to produce both short-duration FM and long-duration 28Hz vocalisations. During extended biopsy attempts during the 2001/2002 survey there was an apparent overall increase in vocalisations. The inability to confirm the vocalising group using DIFAR software limits this to a simple speculation. This should be examined for other close approaches, as this may influence the ability of acoustics to determine species identification.

Clearly there is great variation in the vocal behaviour of different blue whale groups; however, we cannot yet explain these differences. The structure of the 3-unit vocalisation appears to be highly variable, but 27.7Hz peak frequency is stable even over great distances. The 3-unit vocalisations, and the other 28Hz vocalisations do not always occur in patterned series or ‘songs’, and some whale groups are not vocal. More information must be gathered on the variations in vocalisations by age, sex, season, time of day, group composition and behaviour. These data can only be obtained by integrated *in situ* studies of blue whales.

The IWC has stated that there is a need for a dedicated blue whale study in the Southern Ocean to combine visual and acoustics surveys with biopsy, photo-identification and satellite tagging of individuals in order to determine the winter breeding grounds. Blue whales have been known to frequent the ice edge between 150°E and 165°E (Kato *et al.*, 1995). The relatively high populations of blue whales in this area during the 2001/2002 and 2002/2003 confirm that this is an ideal location for deployment of a series of bottom-mounted hydrophones for recording of vocalisations, coinciding with future shipboard populations surveys.

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Distribution of humpback whales along the coast of Ecuador and management implications

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ABSTRACT

As part of a long-term population study of humpback whales breeding on the coast of Ecuador (2°S, 81°W), four sites on the central coast were surveyed: Puerto Cayo, Puerto López, La Plata Island and Salinas. The spatial, temporal and age class distributions of 322 groups positioned during the period of 1996-2003 were analysed regarding their distance from the shore and water depth with two statistical methods: one-way ANOVA and linear modelling. The average sighting distance from shore varied between 5.31km in Salinas and 10.16km in Puerto Cayo with mid values in Puerto López and La Plata Island. Average water depth was similar in Puerto López, La Plata Island and Salinas (36-39m) but lower in Puerto Cayo (19.43m). Differences were highly significant in both cases ($p<0.01$). A progressive but not significant increase in the average distance from shore was found (6.2km in June to 7.17km in September). Sighting depth was constant between June and August (average 35-36m) but decreased significantly in September to 27m ($p<0.01$). This difference was attributed to the presence of mother-calf pairs in shallower water by the end of the season. Age class analyses using ANOVA showed highly significant differences between groups of adults, and adults with subadults with respect to singleton subadults, and groups containing a mother-calf pair for both distance from shore and depth ($p<0.01$); however, linear modelling analyses showed only depth was significant ($p=0.026$). This suggests that depth is a more important determinant of differences in distribution between these age classes than proximity to shore. The sightings distribution showed segregation of both mother-calf pairs (towards shallow waters) and of singleton subadults (towards the boundaries of the surveyed area). Since only eight sightings (2.5%) were in waters deeper than 60m, we propose that depth is a major feature determining humpback whale distribution in these waters. Implications of this coastal distribution are discussed, particularly with respect to bycatch in fishing gear and whalewatching. A review of recent southeast Pacific sightings showed that humpback whales are also abundant in coastal waters to the southwest of Ecuador (3°S) and confirmed that they are scarce offshore. However, whales are more widely distributed in the north of Peru (4°-6°S) where they make the transition between deeper oceanic and shallower coastal waters when arriving at and leaving the breeding area.

KEYWORDS: HUMPBACK WHALE; SOUTH AMERICA; DISTRIBUTION; BREEDING GROUNDS; CONSERVATION; MODELLING; STATISTICS

INTRODUCTION

During the winter months in both hemispheres, humpback whales concentrate in tropical and subtropical zones along continental margins, coastal islands or archipelagos for breeding and calving (e.g. Clapham and Mead, 1999). In most cases, whales are distributed in waters less than 100 fathoms (183m) deep (Winn *et al.*, 1975; Herman and Antinoja, 1977; Urban and Aguayo, 1987; Ersts and Rosenbaum, 2003), although the reasons for this remain unknown. The distribution of humpback whales in open waters during breeding is less well understood. Acoustic studies have demonstrated that some male singers occur in waters deeper than 3,000m and up to 57km from the coast in the Caribbean (Swartz *et al.*, 2003). In Hawaii, Frankel *et al.* (1995) found the concentration of singing males to be 3.6 times higher in coastal waters than in waters of more than 100 fathoms in depth. This suggests that, although they are more widely spaced than in coastal waters, many humpback whales may use deep waters during the breeding season.

A southern humpback whale stock (Group G – see IWC, 1998) migrates along the southeast Pacific between the Antarctic Peninsula and south of Chile where they feed, (Gibbons *et al.*, 2003; Stevick *et al.*, 2004) and the coasts of Ecuador, Colombia and Panama where they breed (Clarke, 1962; Flórez-Gonzàlez, 1991; Scheidat *et al.*, 2000; Félix and Haase, 2001a). Humpback whales are found in Ecuador from May-November, with the greatest numbers occurring in July and August (Félix and Haase, 2001a). During the breeding period in Ecuador, groups of humpback whales appear to show a heterogeneous distribution according to their age and class composition; for example, groups containing mother-calf pairs prefer waters of 20m or less in depth, singleton subadults also prefer shallow waters, whereas groups of adults occur in the deeper waters further from shore (Félix and Haase, 1997; 2001a).

Their coastal habit renders humpback whales vulnerable to certain human activities such as chemical pollution, vessel traffic noise, industrial activities and particularly interactions with fishing gear (Reeves *et al.*, 2003). Reports of humpback whales entangled in artisanal gillnets in Ecuadorian waters are a cause for concern, and evidence suggests that the problem is increasing (e.g. Félix *et al.*, 1997; Alava *et al.*, 2002). This artisanal fishery is directed toward demersal resources (crustaceans, reef and bottom fish) and large pelagic fish (billfish, tuna, sharks, etc.) and limited to 40 n.miles from the coast over the continental shelf (Martínez, 1987). These waters are also used by humpback whales during their breeding season. It was estimated that there were 15,000 artisanal boats by the end of the 1990s in the country (Ormaza and Ochoa, 1999), 50% of which used gillnets up to 3km in length and 15m high with variable mesh sizes (Martínez *et al.*, 1991). Entanglement of large whales occurs more frequently in gillnets directed to large pelagic fishes, with mesh size of 7.3-13cm (Félix *et al.*, 1997). The importance of marine mammal bycatches to their conservation has been highlighted by a number of international organisations (e.g. see Northridge, 1984; Perrin *et al.*, 1994; Reeves *et al.*, 2003).

The development of whalewatching programmes along the coast of Ecuador constitutes another potential source of disturbance for whales. Changes in movement and activity patterns during encounters with tourist boats have been reported in several sites including Ecuador (e.g. Corkeron, 1995; Brtnik, 2001; Félix, 2001; Scheidat *et al.*, 2004). There has been a steady growth of whalewatching activity since the mid 1990s due to increased 'ecotourism' along the coast of Ecuador. Nowadays, the activity is carried out from at least six different sites in the country (Félix, 2003).

This paper investigates the relationship between the distribution of humpback whale groups and some of the physical and geographic features that may be related to or determine whale distribution; specifically water depth and distance from shore. These relationships may be used to predict other regions of high humpback whale population density along the coast of Ecuador and ultimately help to minimise conflict with human activities.

Previous records in Ecuador and in other parts of the Southeast Pacific

It has been known for a long time that the coasts of Ecuador, Colombia and Panamá are breeding sites for humpback whales (e.g. Townsend, 1935; Clarke, 1962). However, recent studies have more accurately identified humpback whale breeding habitat along the northwestern coast of South America. This area of around 2,000km in length includes the coasts of Peru (7°S), Ecuador, Colombia, Panama (Flórez-González *et al.*, 1998) and as far north as 8°N and the Dulce Gulf in Costa Rica (Acevedo and Smultea, 1995). Humpback whales have occasionally been reported around the Galapagos Islands, 1000km off Ecuador, although they are considered uncommon there (Day, 1994; Merlen, 1995; Palacios and Salazar, 2002).

Several expeditions have sighted humpback whales between Ecuador and the Galápagos Islands in the past four decades (e.g. Clarke, 1962; Lyrholm *et al.*, 1992; Clarke *et al.*, 2002). Details of the sightings including date, position and number of animals are provided in Appendix Table 1. With the exception of Clarke (1962), who reported a group of humpback whales 50 n.miles off Isla Santa Clara in the southern part of Ecuador (number 1) and another 25 n.miles further away (without position), humpback whales were only reported in coastal waters, and not in the archipelago. Sightings were also reported during cetacean surveys in the Eastern Tropical Pacific conducted by the United States National Oceanographic and Atmospheric Agency (NOAA) from the late 1980s until 2003. These data include thirteen sightings off the northern coast of Peru between 3°30'S and 6°25'S, seven in coastal waters of the central and southern part of Ecuador, two 120 n.miles off the south of Ecuador, two around the Galapagos Islands, 10 in coastal waters of Colombia and two off (120 and 250 n.miles) the central coast of Colombia (Fig. 1).

Other records for the coastal waters of southern Ecuador were obtained around Santa Clara Island between 1 and 5 July 1998 during a period of seismic prospecting in the Gulf of Guayaquil (Yturralde and Suárez, 1998). The positions, depths and distances to the largest island of the Gulf (Puná Island) of these 35 sightings are shown in Appendix Table 2. Whales were recorded in shallow waters (mean depth 45.7m; SD=18.6) and an average of 25.42km (SD=6.79) offshore showing that humpback whales are also present in the southern coastal waters of Ecuador (Fig. 1).

Excluding the two sightings around the Galápagos Islands, only three of the sightings plotted in Fig. 1 were in deep waters off the south of Ecuador, and the remainder were recorded in coastal waters. The sightings suggest a continuous distribution of humpback whales along the coastal waters of the entire region (4°N-6°S). In contrast, offshore sightings were absent between 84°W and 90°W (Galápagos Islands) and between 2°S (central-south coast of Ecuador) and 4°N (central coast of Colombia). The small number of sightings in deep waters in this area infers that humpback whales are rarely found offshore during the breeding season, although the lack of survey effort is acknowledged.

Off southern Ecuador, however, humpback whales seem to be distributed further offshore. Most catches made between May and November in the period 1961-1966 from land stations located in Paita (5°S, 81°14'W) and Chimbote (7°S, 78°30'W) occurred within 100 n.miles of the coast, with the greatest concentration between 81°30'W and 82°W, although some whales were caught as far as 200 n.miles offshore (Ramírez, 1988). Sightings from the period 1975-1985 show a similar distribution pattern (Ramírez, 1988). More recently, Sánchez and Arias (1998) stated that humpback whales were the most abundant large cetacean observed during a cruise along the northern coast of Peru between August and September 1998, with the highest concentration at 5°S and between 4 and 99 n.miles offshore.

MATERIAL AND METHODS

A population study of the humpback whale on the coast of Ecuador has been carried out aboard whalewatching vessels since 1990 (Félix and Haase, 1997; 2001a). Information is recorded on group size, group composition, behaviour, oceanographic conditions and the geographic position. Position is determined using a portable global positioning system (GPS). This study uses data for a total of 322 groups recorded during 159 trips between 1996 and 2003.

Study area

The study was carried out at four sites along the central coast of Ecuador: Puerto López, Puerto Cayo, La Plata Island and Salinas (Fig. 2). The Puerto López (1°30'S, 80°50'W) region is fairly homogeneous with water depths of 30-50m extending some 40km offshore. Near Puerto Cayo (1°20'S, 80°50'W), the coast forms a wide, shallow embayment of less than 30m depth that extends some 30km along the coast in a northwest direction. La Plata Island (1°15'S, 81°W) is located 24km off the mainland at its nearest point. The water depth on the east side varies between 15 and 50m in a relatively small area, but on the west side the depth changes abruptly to 100m in the first kilometer. Salinas is located on the outermost tip of the Santa Elena peninsula (2°10'S, 81°W), 80km south of Puerto López. Here, the continental shelf is narrow and the depth increases rapidly westward, reaching depths of 200m just 13.5km offshore.

Sites and surveys

Boats departed from three sites: Puerto López, Puerto Cayo and Salinas (Fig. 2).

Puerto López

Trips were conducted from this port in 1996, 1997 and 2000. Boats headed northwest towards La Plata Island 40km offshore, along the 50m isobath. After 3 hours, the boats returned to port.

Puerto Cayo

Trips were conducted in 1996 and 1997. Boats headed westwards up to 20km offshore and then returned to port. The operation was carried out close to shore in waters averaging 20m in depth.

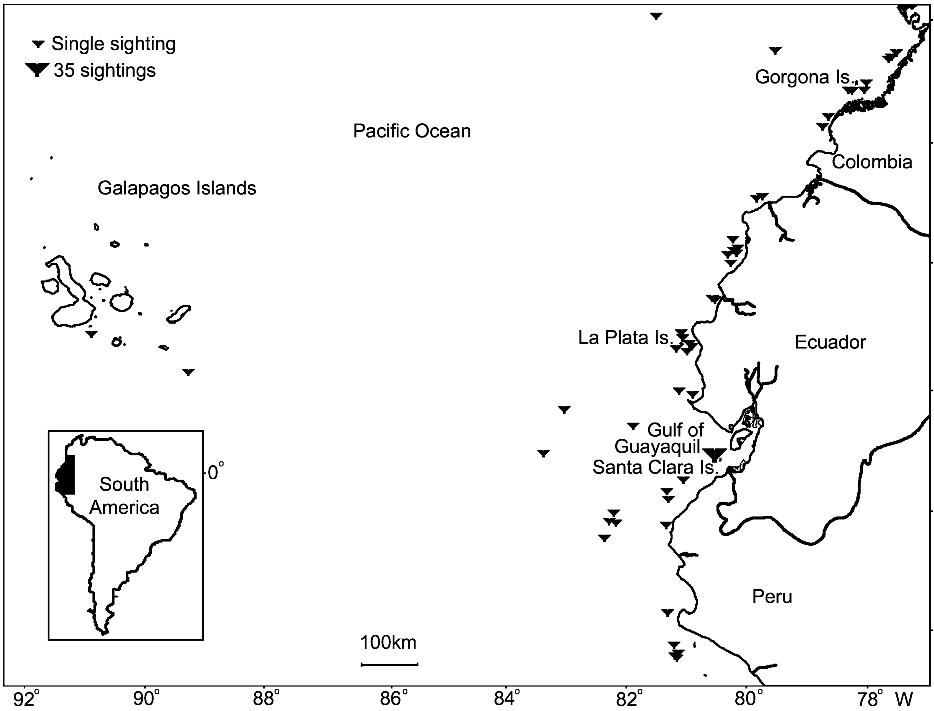

Fig. 1. Sightings of humpback whales along the coast of Ecuador and in other parts of the southeast Pacific by expeditions between 1959 and 2003 (see text).

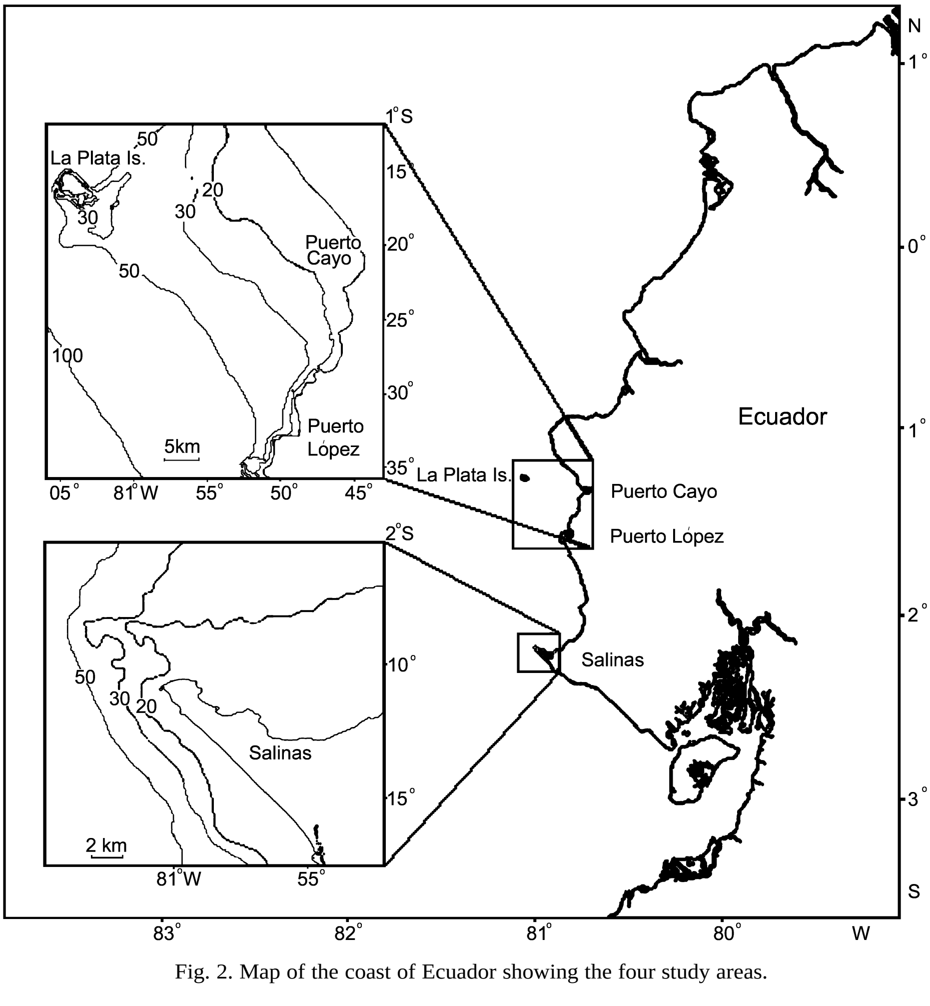

Fig. 2. Map of the coast of Ecuador showing the four study areas.

Salinas

Trips were conducted between 2001 and 2003. Surveys extended 6-10km west and water depths of 50-60m were quickly reached.

Survey effort

Information regarding the number of sightings recorded in each year is shown in Table 1. The effort deployed by month in each site is shown in Table 2. The effort varied during the four months as follows: June, 5.7%; July, 39.5%, August, 38.2%; and September, 16.6%. In general, effort was more homogeneous by month in Puerto López, La Plata Island and Salinas, whereas in Puerto Cayo there was a lower proportion of surveys in July and a higher proportion in September. However, these differences were not statistically different (χ^2_{11}=9.55, p<0.05). June and October were excluded from this comparison because of the low number of surveys.

Table 1

Number of groups sighted in each site and year (period 1996-2003) (n = 322).

	Number of groups					
Site	1996	1997	2000	2001	2002	2003
Puerto López	11	23	9			
Puerto Cayo	23	28				
La Plata Island	24	27	14			
Salinas				39	57	66
Total	58	78	23	39	57	67

Table 2

Number of trips conducted by month in each site (period 1996-2003) (n = 159).

	Puerto López		Puerto Cayo		La Plata Island		Salinas	
Month	*n*	%	*n*	%	*n*	%	*n*	%
Jun.	3	11.1			1	3.2	5	6.8
Jul.	11	40.8	6	22.2	16	51.6	29	39.2
Aug.	9	33.3	12	44.5	11	35.5	28	37.8
Sept.	4	14.8	9	33.3	3	9.7	10	13.5
Oct.							2	2.7
Total	27	100	27	100	31	100	74	100

Group composition

Three different age classes were distinguished based on size: (1) adults, animals estimated to be larger than 10m in length; (2) subadults, estimated to be 6-10m in length; and (3) calves, estimated to be less than 6m in length and always accompanied by an adult animal, presumably the mother (Félix and Haase, 2001a). These results must be treated with caution since lengths were estimated by eye. However, given this proviso, groups were categorised as either: A=all adults, AS=adults with subadults, S=single subadults, MC=mother with calf alone or accompanied by one or more escorts.

Depths and distances

For each group for which a position was obtained, the depth and distance to shore were estimated using the following navigation charts[1]: I.O.A. 104 (Punta Jaramijó to Salango Island), I.O.A. 105 (Santa Elena Bay, Salango Island-Chanduy) and I.O.A. 10 (Cabo Manglares-Punta Malpelo). Sightings were marked on the chart and then distances were measured in a straight line to the nearest point on either the mainland or La Plata Island coast. Depth was recorded as either the nearest known point on the chart with a value or the value of the isobath line if this was the nearer point. If several values marked on the chart were equidistant, then an average value was used.

Statistical analysis

Distance from shore and depth of sightings were analysed using two statistical methods: (1) one-way ANOVA for areas, month and group class; and (2) linear modelling to test combinations of variables and interactions. Linear modelling was conducted using R analysis software (version 1.3.1; *http://www.r-project.org*). Date (days subsequent to May 1), distance and depth were used as response variables. For each, model selection was based upon a fully saturated model including the remaining two variables, as well as group category and site. All terms that were significant at the p=0.05 level were included in the final model.

RESULTS

ANOVA

Spatial distribution

The distributions of whales with respect to both depth and distance from shore were related to the topographic characteristics of each study site (Fig. 3). Off Puerto Cayo, groups were found mainly between 2.5 and 10km offshore in waters of 10-25m in depth. Off Puerto López, groups were more spread out and on average further away from the coast, with most sightings being recorded between 8 and 15km offshore in waters 30-50m deep. Near La Plata Island, around 50% of groups were found within 5km of shore, in waters of 25-40m in depth. Except for one sighting (1.5%) made in waters deeper than 60m, the remaining groups were found up to 17km from shore, in waters of 30-50m in depth. In Salinas, there was also a concentration of sightings within 5km of shore in waters of 20-50m in depth. As with the other sites, water depths at which sightings were made did not increase with distance from shore, and remained between 30 and 60m. Thus despite the narrow shelf off Salinas, only 7 sightings (4.3%) occurred in waters deeper than 60m.

Mean sighting distances to shore varied from 5.31km in Salinas to 10.16km in Puerto Cayo, with moderate values for Puerto López and La Plata Island (Table 3). Sighting distances were significantly different between study sites (ANOVA, $F_{3,318}$=11.08, p<0.01). Sighting depths were more uniform, with similar values for Puerto López, La Plata Island and Salinas (mean 36-39.03m), but significantly lower off Puerto Cayo (19.49m) (ANOVA, $F_{3,318}$=33.47, p<0.01) (Table 3).

Table 3

Comparison of the average values regarding the distance to the shore and depth of the sightings in each study site (n = 322).

	Distance to shore (km)			Depth (m)			
Site	Mean	SD	Range	Mean	SD	Range	*n*
Puerto López	8.05	4.34	0.1-19.7	36	11.45	15-50	43
Puerto Cayo	10.16	4.17	1.2-20.5	19.43	6.7	10-41	51
La Plata Is.	7.04	5.16	0.1-17.1	36.65	10.99	5-56	65
Salinas	5.31	3.1	0.1-14.2	39.03	15.16	10-104	163

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[1] Instituto Oceanográfico de la Armada de Ecuador INOCAR (Oceanographic Institute of the Navy).

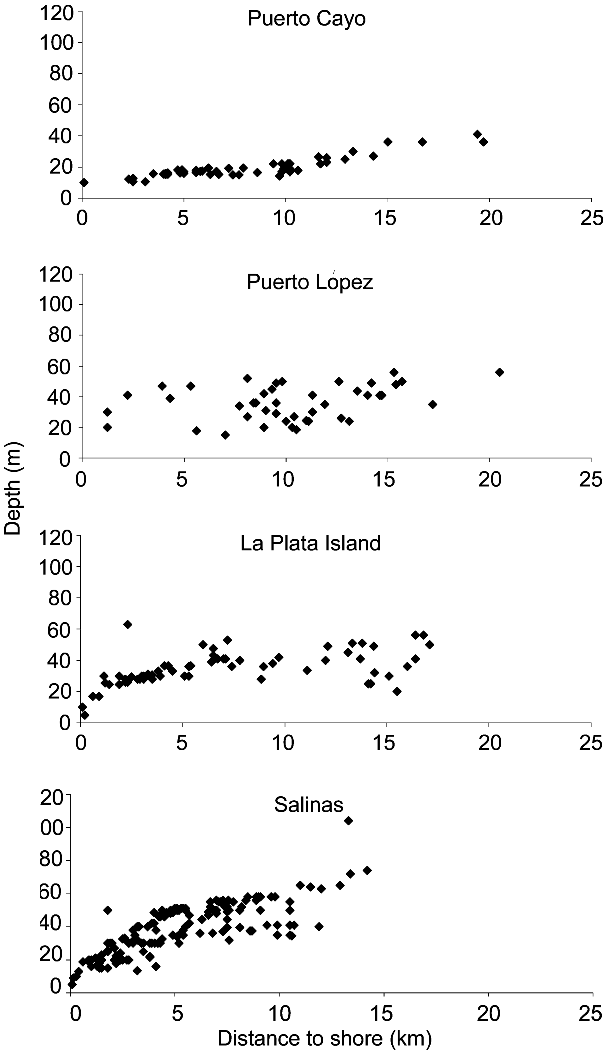

Fig. 3. Distribution of humpback whale groups with respect to their distance from shore and depth at each study site.

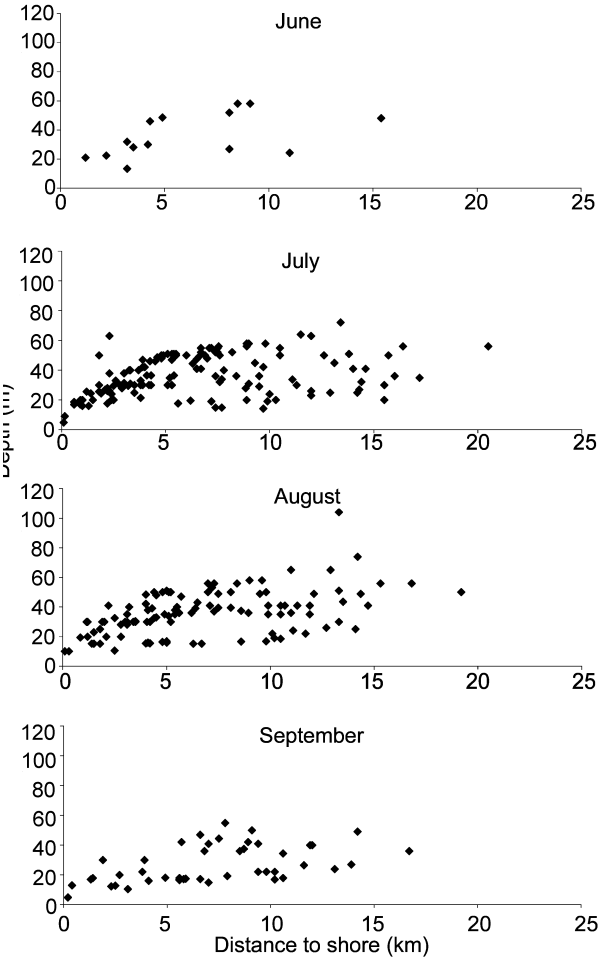

Fig. 4. Monthly distribution of humpback whale groups with respect to their distance from shore and depth.

Table 4

Comparison of the average sighting distance to the shore and depth by month (period 1996-2003) ($n = 319$).

Month	Distance to shore (km)			Depth (m)			n
	Mean	SD	Range	Mean	SD	Range	
Jun.	6.2	3.9	1.2-15.4	36.33	14.88	13.3-52	14
Jul.	6.54	4.28	0.1-20.5	36.60	13.65	5-72	141
Aug.	6.68	4.27	0.1-19.4	35.13	15.48	10-104	119
Sept.	7.17	3.39	0.2-16.7	26.93	12.59	5-55	45

Temporal distribution

The monthly distribution of whale groups is shown in Fig. 4. As shown in Table 4, the average distance to the shore increased during the season (but not significantly) from 6.2km in June to 7.17km in September (ANOVA $F_{3,315}$=0.2, p>0.05).

In June, sightings showed a bimodal distribution with respect to distance from shore, with concentrations within 2-5km and 8-11km from shore in waters of 20-60m in depth. In July and August, however, sightings were concentrated within 5km of shore followed by a decrease in numbers with distance. Water depth increased up to 5km offshore and thereafter it maintained a relatively constant level between 20 and 60m. By September, the distribution was again bimodal with one area of concentration in shallow waters of 20m or less extending up to 14km offshore and another one in deeper waters (30-60m) starting at 6km offshore. As shown in Table 4, the average depth of the sightings was constant between June and August (average 35.13-36.60m) but in September decreased significantly by 25% (ANOVA, $F_{3,315}$=6.37, p<0.01).

The apparent contradiction of a higher average sighting distance from shore with a significantly lower depth found in September seems to be the product of a sampling artifact, since in September both the sighting distance range and the depth range are smaller than in July and August (Table 4); because whales were less abundant by the end of the season in September, boats probably had to make longer trips, although these were not necessarily to further offshore.

Age class distribution

Only those groups for which an age class was assigned to every member were considered for analysis (196 out of 322 or 61%) and distribution by age is shown in Fig. 5. A and AS groups showed similar distribution patterns despite A groups being almost three times as abundant as AS groups. These classes were found on average between 7 and 8km offshore in waters of 36m in depth (Table 5). In contrast, S and MC groups showed a more coastal, shallower distribution with an average distance to shore of 5km in waters of 23-28m in depth. Although sightings of S groups were not as abundant as for other classes, the data suggest segregation of subadults toward the edges of the area used by A and AS groups. For MC groups, segregation toward coastal areas is evident. The comparison among the four age class categories shows a highly significant difference between distances from shore (ANOVA, $F_{3,192}$=5.93, p<0.01), as well as water depth (ANOVA, $F_{3,192}$=12.51, p<0.01). Separate analyses of sighting depths of A and AS groups compared to S and MC groups were performed, and showed that in both cases the difference was significant for S groups (ANOVA, $F_{2,161}$= 4.27, p<0.05) and highly significant for MC groups ($F_{2,180}$=18.26, p<0.01).

Table 5

Comparison of the average sighting distance to shore and depth by age/class groups (*n*=196).

Age/class group	Distance to shore (km)			Depth (m)			
	Mean	SD	Range	Mean	SD	Range	*n*
A	7.1	4.47	0.1-20.5	35.94	14.59	10-104	110
AS	7.88	4.39	1-19.4	36.78	14.2	15.2-72	41
S	5.1	3.88	0.1-10.6	28.48	18.15	5-56	13
MC	4.8	3.53	0.2-13.1	22.73	11.78	5-44.5	32

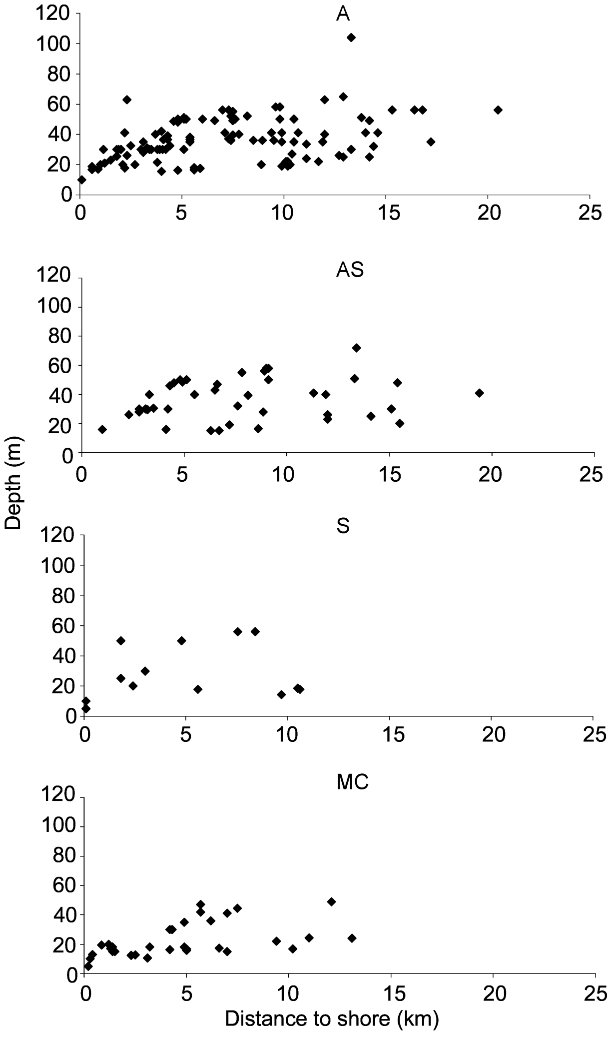

Fig. 5. Distribution of humpback whale groups with respect to their distance from shore and depth by age-class groups.

Linear modelling analyses

In addition to ANOVA, linear modelling was conducted to examine the relationships between the temporal, spatial and age variables detailed earlier. The date of the sighting showed significant relationships with site, depth and age class (F=6.6, F=9.1 and F=14.67 respectively, p<0.01), but not with distance from shore. Similar results were obtained using ANOVA (see above). Distance from shore showed significant relationships with respect to both site and depth (F=25 and F=126.9 respectively, p<0.01), but not with age class and this conflicts with the ANOVA results. A significant relationship (F=7.5, p<0.01) between site and depth was revealed, and also between site and distance from shore (F=17.12, p<0.01) reflecting the different topographies of the areas. Depth was significantly related to site (F=37.2, p<0.01), distance (F=146.5, p<0.01) and age class (F=3.14, p=0.026).

The modelling showed no relationship with distance from shore and age class when site and depth were taken into account. This suggests that depth (or a factor related to depth) is a more important determinant of differences in distribution between the different age classes than proximity to shore, and that the relationship between age class and distance observed using ANOVA may be in part an artifact of the correlation between depth and distance from shore.

DISCUSSION

Despite the different topography of the four study sites, the humpback whales appeared to maintain a common pattern of distribution in waters of 20-60m in depth. Only eight sightings (2.5%) were found in waters deeper than 60m. This was confirmed by both data analyses presented here which indicated that depth is the critical factor in determining distribution in breeding areas off the Ecudorian coast. Irrespective of topography whales remained most abundant in waters between 20 and 60m in depth. Due to this preference for shallow waters, the population densities in these sites are correlated to the width of the adjacent continental shelf; higher in places with narrow shelves (e.g. Salinas and La Plata Island) and lower where the shelf is wider (e.g. Puerto Lopéz). For Salinas, as the peninsula projects 15km westward, the corridor narrows to just 10 kilometres in width. Similar funnels can also be found in other parts off the north Ecuadorian coast.

Our data are insufficient to determine the offshore distribution of humpback whales, since they were collected aboard whalewatching boats whose operations were limited to coastal areas. However, as noted in the Introduction the limited records available for Ecuador indicate that humpback whales are uncommon offshore. Clarke *et al.* (2002), who followed a relatively coastal course along the meridian 81°10’W, only recorded humpback whales along the coasts at Salinas (2°10’S) and Manta (1°S). Sightings in 1959 by Clarke (1962) 50 n.miles west of southern Ecuador contrasted with the coastal findings in the central and north records of other expeditions, but were concordant with whaling records and sightings from the north of Peru by Ramírez (1988) and by the NOAA surveys (Fig. 1). It seems that humpback whales are more widely distributed in the north (and perhaps south) of Peru. Although the migratory route of humpbacks in the Southeast Pacific is unknown, Clarke (1962) suggested that it must be off the coast of Chile and Peru to avoid the cold waters of the Humboldt Current running northward along the west South American coast. When the Humboldt Current reaches the northern part of Peru and meets the warmer southerly current at the Equatorial Front[2] around 5°S, it turns westward and joins with the South Equatorial Current in the Galapagos Islands

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[2] The Equatorial Front is a transition zone between warm and low saline southward waters from the Panama bright and the subtropical colder and more saline waters from the Humboldt Current extending westward.

(Cucalón, 1996). Since whales are abundant in the coastal waters within the Gulf of Guayaquil, southwest of Ecuador (see Fig 1), this suggests that the north of Peru (4°-6°S) is where humpback whales from the south begin the transition from oceanic deep-water to a coastal tropical breeding environment. In fact, it may be that the Equatorial Front is the feature that causes the humpback whales to move towards the coast. This may also explain the absence of offshore records between 2°S (central-south of Ecuador) and 4°N (central Colombia). When leaving the breeding area, whales may be expected to follow a similar pattern but in the opposite direction, except when oceanographic events such as El Niño occur. These seem to influence the distribution of humpback whales in this part of the migratory route because of southward displacement of the Equatorial Front (Félix and Haase, 2001b).

Humpback whales show temporal variation in migration related to their reproductive condition and physical maturity (Dawbin, 1966). This is reflected in sighting distributions along the coast of Ecuador in September, when, for example mothers nurse calves in shallow waters (Félix and Haase, 1997; 2001a). Preference for shallow waters and sheltered places is typical for this species and has also been observed in other breeding areas such as Hawaii (Smultea, 1994), the Caribbean (Scott and Winn, 1979; Whitehead and Moore, 1982), Australia (Vang, 2002) and Madagascar (Ersts and Rosenbaum, 2003).

Our data reveal a segregation of singleton subadults toward the edge of the A and AS group distributions. Félix and Haase (1997) noted a concentration of singleton subadults in shallow waters, but the larger series of data presented here shows that they also distribute in deeper waters. Although the sex of these animals is unknown, it is possible that this type of segregation occurs mainly in immature males, who do not participate in the reproduction cycle. Although such segregation of immature individuals is frequently seen in odontocetes (e.g. Wells *et al.*, 1980; Caldwell *et al.*, 1966), it has not previously been reported for humpback whales. However, Scott and Winn (1979) reported a different distribution pattern for immature individuals breeding in the Caribbean. They found a cluster distribution at Silver Bank, and a more uniform distribution at Navidad Bank, attributing this difference, among other reasons, to the presence of a large number of non-calling whales, especially immature animals, at Silver Bank.

Management implications

Knowledge of the explanatory variables affecting humpback whale distribution during the breeding season in Ecuadorian waters can be valuable for the conservation of this population at both local and regional levels. If humpbacks distribute in the northern part of the country in the same way as they do in central and southern parts, it may be possible to predict their distribution along the entire coast of Ecuador, and possibly in the rest of the breeding area further north. Using this analysis as a baseline for whale distribution during the breeding season, will hep in developing measures to reduce potentially harmful interactions with human activities.

Incidental catches in fishing gear

The bycatch of small and large cetaceans in artisanal gillnets is a problem that has been known of for more than a decade in Ecuador (Félix and Samaniego, 1994; Haase and Félix, 1994; Félix *et al.*, 1997; Alava *et al.*, 2002) but it has received little attention from local authorities. The problem has worsened because the artisanal fishing effort doubled over 10 years (Campbell *et al.*, 1991; Ormaza and Ochoa, 1999). The lack of population studies on the distribution, abundance and other ecological parameters of cetaceans makes evaluation and management of the problem difficult. However, bycatch of humpback whales in fishing gear has been identified as their main non-natural mortality along the Colombian coast (Capella *et al.*, 2001) and it is possible that the current level of bycatch in the breeding grounds may affect the recovery of this population.

There have been no studies comparing the distribution of artisanal fishing areas with whale distribution since information on artisanal fishing effort is sparse given the informal nature of the fishery. Most of the studies that are conducted are socio-economic assessments and focus on censuses of boats and fishing gear at port (Ormaza and Ochoa, 1999). Artisanal fishermen do not usually use navigation instruments, charts or GPS devices; rather, details of fishing sites are passed on to the next generation of fishermen by word-of-mouth. To establish effective management measures, it is necessary to identify fishing areas and determine under which conditions interactions with humpback whales and other cetaceans take place.

In order to reduce conflicts with fishing activities we recommend the following:

1. comprehensive documentation of the locations, areas and times of operation of the fishing areas used by artisanal fishing fleets;
2. a reduction in the fishing effort using gillnets in areas associated with high population densities of humpback whales, either through closures during the breeding season or by use of alternative fishing gear (e.g. long lines);
3. evaluation of the use of acoustic devices attached to gillnets[3].

Whalewatching

The present study also has implications for the management of whalewatching. The data presented allow the prediction of sites on the Ecuadorian coast where whales may congregate, and thus where new commercial operations may be established and where protection of important calving and nursery areas should occur. Although whalewatching has become one of the most popular activities on the coast of Ecuador, and is helping to promote other ecotourism activities in an area where traditionally natural resources were previously exploited only consumptively, it is important that it is properly regulated.

Other activities

The information presented here also has implications for other activities currently implemented or planned for the country including: (1) maritime traffic – commercial routes pass through the near coastal waters off Salinas and around La Plata Island that have been shown to support a high population density of whales; (2) military manoeuvres – UNITAS, operates every year off the coast of Ecuador during the humpback whale breeding season; (3) seismic prospecting and offshore oil-drilling – carried out in the Gulf of Guayaquil, an area identified as part of the whale migration corridor; (4) mariculture farms, planned in central and southern parts (La Plata Island and Santa Clara Island) to raise tuna; and (5) marinas and artisanal ports. Future environmental studies for these activities must take into

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[3] This technology has been shown to reduce humpback whale bycatch in other regions by either alerting them to the presence of nets, or detering them from the area (Todd *et al.*, 1992).

account the use of coastal areas by humpback whales, and their impact on the species. This has not been the case for previous studies.

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Appendix Table 1

Sightings of humpback whales recorded in coastal and offshore waters of Ecuador, north of Peru, south of Colombia and around Galapagos Islands, during marine mammal surveys between 1959 and 2003.

Number	Date	Position	Group size	Reference
1	9 Oct 1959	2º37’S, 81º51’W	7	Clarke (1962)
2	1 Nov 1987	4º6’N, 81º39’W	2	Holt and Jackson (1987)
3	20 Sep 1988	1º6’N, 79º50′W	2	Lyrholm *et al.* (1989)
4	20 Sep 1988	1º4’N, 79º44’W	3	Lyrholm *et al.* (1989)
5	21 Sep 1988	0º12’N, 80º11’W	1	Lyrholm *et al.* (1989)
6	21 Sep 1988	0º11’N, 80º11’W	2	Lyrholm *et al.* (1989)
7	21 Sep 1988	0º11’N, 80º11’W	3	Lyrholm *et al.* (1989)
8	21 Sep 1988	0º1’N, 80º14’W	4	Lyrholm *et al.* (1989)
9	22 Sep 1988	1º19’S, 80º56’W	1	Lyrholm *et al.* (1989)
10	22 Sep 1988	1º19’S, 80º56’W	2	Lyrholm *et al.* (1989)
11	22 Sep 1988	1º23’S, 81º7’W	3	Lyrholm *et al.* (1989)
12	3 Oct 1989	2°11.65’S, 80°54.63’W	2	Hill *et al.* (1990)
13	6 Oct 1989	3°1.45’S, 83°21.60’W	1	Hill *et al.* (1990)
14	23 Sep 1990	1°49.12’S, 89°14.84’W	1	Hill *et al.* (1991)
15	31 Oct 1990	4°17.63’S, 82°25.74’W	2	Hill *et al.* (1991)
16	31 Oct 1990	4°45.90’S, 82°35.76’W	4	Hill *et al.* (1991)
17	31 Oct 1990	4°1.73’S, 82°21.41’W	1	Hill *et al.* (1991)
18	7 Oct 1992	3°30.80’N, 79°30.98’W	3	Mangels and Gerrodette (1994)
19	25 Oct 1992	3°24.41’N, 77°36.75’W	2	Mangels and Gerrodette (1994)
20	25 Oct 1992	3°30.23’N, 77°30.71’W	2	Mangels and Gerrodette (1994)
21	25 Oct 1992	3°23.01’N, 77°38.29’W	2	Mangels and Gerrodette (1994)
22	25 Oct 1992	2°58.93’N, 78°1.43’W	1	Mangels and Gerrodette (1994)
23	26 Oct 1992	2°50.61’N, 78°16.22’W	5	Mangels and Gerrodette (1994)
24	26 Oct 1992	2°52.96’N, 78°20.01’W	2	Mangels and Gerrodette (1994)
25	26 Oct 1992	2°53.45’N, 78°3.70’W	2	Mangels and Gerrodette (1994)
26	26 Oct 1992	2°25.94’N, 78°39.62’W	2	Mangels and Gerrodette (1994)
27	27 Oct 1992	2°18.67’N, 78°45.15’W	1	Mangels and Gerrodette (1994)
28	7 Nov 1998	1°0.10’S, 90°53.56’W	2	Kinzey *et al.* (1999)
29	10 Nov 1998	2°30.86’S, 83°1.83’W	1	Kinzey *et al.* (1999)
30	29 Nov 1998	4°18.57’S, 82°20.30’W	1	Kinzey *et al.* (1999)
31	3 Nov 2000	6°39.05’S, 80°49.26’W	3	Kinzey *et al.* (2001)
32	4 Nov 2000	3°44.85’S, 81°0.30’W	1	Kinzey *et al.* (2001)
33	4 Nov 2000	4°20.46’S, 81°24.14’W	1	Kinzey *et al.* (2001)
34	10 Nov 2000	3°4.92’N, 77°56.38’W	2	Kinzey *et al.* (2001)
35	18 Sep 2001	1º59.95’S, 81º9.35’W	3	Clarke *et al.* (2002)
36	18 Sep 2001	0º55.01’S, 80º48.77’W	1	Clarke *et al.* (2002)
37	19 Sep 2001	0º55.52’S, 80º43’13’W	2	Clarke *et al.* (2002)
38	5 Nov 2003	6°13.95’S, 81°10.90’W	4	NOAA database
39	5 Nov 2003	6°22.21’S, 81°7.40’W	4	NOAA database
40	5 Nov 2003	6°24.51’S, 81°10.37’W	2	NOAA database
41	5 Nov 2003	6°24.75’S, 81°10.60’W	6	NOAA database
42	5 Nov 2003	5°41.60’S, 81°17.40’W	2	NOAA database
43	6 Nov 2003	3°43.39’S, 81°17.66’W	1	NOAA database
44	6 Nov 2003	3°50.60’S, 81°19.72’W	2	NOAA database
45	7 Nov 2003	1°6.87’S, 81°3.51’W	2	NOAA database
46	7 Nov 2003	1°16.42’S, 81°2.46’W	1	NOAA database
47	7 Nov 2003	1°45.90’S, 80°58.70’W	2	NOAA database
48	8 Nov 2003	0°24.03’N, 80°13.69’W	3	NOAA database
49	8 Nov 2003	0°9.98’N, 80°18.32’W	2	NOAA database

Appendix Table 2

Sightings of humpback whales at Santa Clara Island, south of Ecuador (from Yturralde and Suárez, 1998).

Date	Position	Group size	Depth (m)	Distance to shore (km)
1 Jul 1998	3°5.6’S, 80°25.1’W	??	55	18
1 Jul 1998	3°5.7’S, 80°27.7’W	2	67	22.8
2 Jul 1998	3°1’S, 80°25’W	2	12	15
2 Jul 1998	3°4.3’S, 80°22.6’W	1	60	13.8
2 Jul 1998	3°4.1’S, 80°23.1’W	2	50	14.4
2 Jul 1998	3°5.7’S, 80°28.2’W	1	55	21
3 Jul 1998	3°5.9’S,80°21.6’W	2	50	15
3 Jul 1998	3°5.9’S,80°21.6’W	2	50	15
3 Jul 1998	3°5.9’S,80°21.6’W	>2	50	15
3 Jul 1998	3°6.2’S, 80°24.9’W	1	52.5	18
3 Jul 1998	3°6.2’S, 80°25.6’W	4	76	22.2
3 Jul 1998	3°6.3’S, 80°27.1’W	1	76	25.2
3 Jul 1998	3°3.6’S, 80°29.3’W	>1??	58	24.6
3 Jul 1998	3°2.9’S, 80°28.7’W	4??	50	22.8
3 Jul 1998	3°1.5’S, 80°27.5’W	1	40	19.2
3 Jul 1998	3°S, 80°32’W	2	50	30
3 Jul 1998	3°1.7’S, 80°33.1’W	4	62	31.8
3 Jul 1998	3°2.1’S, 80°33.7’W	1	60	31.2
4 Jul 1998	3°3.8S, 80°34.1’W	2	60	33.6
4 Jul 1998	3°6.8’S, 80°30.8’W	1	60	31.2
4 Jul 1998	3°7.’S, 80.31’W	3	69	31.8
4 Jul 1998	3°8.6’S, 80°29.6’W	1	50	29.4
4 Jul 1998	3°9.3’S, 80°29.1’W	1	40	30
4 Jul 1998	3°9.5’S, 80°28.9’W	??	40	30
4 Jul 1998	3°9.9’S, 80°28.6’W	1	30	27
4 Jul 1998	3°10.3’S, 80°28.3’W	2	30	27
4 Jul 1998	3°10.5’S, 80°28.2’W	2??	30	26.4
4 Jul 1998	3°10.8’S, 80°28’W	1	20	27.6
4 Jul 1998	3°12.2’S, 80°28’W	2	15	30
4 Jul 1998	3°13’S, 80°28.5’W	1	10	30.6
4 Jul 1998	3°13’S, 80°28.5’W	1	10	30.6
4 Jul 1998	3°14.9’S, 80°28.8’W	1	30	30.6
4 Jul 1998	3°11.2’S, 80°27.7’W	1	20	28.2
5 Jul 1998	3°9’S, 80°38’W	4	63	41.4
5 Jul 1998	2°59’S, 80°32’W	4	41	29.4

Subsistence harvests of bowhead whales (*Balaena*[1] *mysticetus*) at Kaktovik, Alaska (1973-2000)

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ABSTRACT

Kaktovik is a small community located on Barter Island in the extreme northeast of Alaska. The bowhead whale hunt is important to the community from both an economic and cultural perspective. Harvests were generally 1-2 bowheads per year in 1973-1988 and 2-4 bowheads per year in 1989-2000. The hunt normally begins on the first Monday in September and historically 83% of harvested whales have been taken in September. In recent years, typical harvest dates have been significantly earlier even though the quota and number of whales taken have increased. The core whaling area extends from 15km west to 25km east of Kaktovik, and offshore as far as 32km. Most whales have been taken within 30km of the village and the mean distance of harvest locations from Kaktovik has not changed from the 1970s to present. Whaling captains select small whales over large whales and there has been a marginally significant decrease in the average size of whales harvested from the 1970s to the present. The size of whales harvested does not increase with date, although other data show that smaller whales become less common in the area as the season progresses. Male and female bowhead whales are harvested in very similar numbers, but females make up 67% of whales harvested early in the season and 32% late in the season.

KEYWORDS: WHALING–ABORIGINAL; BOWHEAD WHALE; ARCTIC; BEAUFORT SEA; MIGRATION; NORTHERN HEMISPHERE; SEX RATIO; NORTH AMERICA

INTRODUCTION

Kaktovik, also referred to as Barter Island, is a small community located on Barter Island in the extreme northeast of Alaska, within the boundaries of the Arctic National Wildlife Refuge (ANWR) (Fig. 1). The 2000 US Census enumerated 293 people, most of whom (247, or 84%) are native. Household economies rely upon both wage labour (and other income sources) and subsistence activities as vital components of an integrated system. Subsistence whaling is of high importance to the Kaktovikmiut, the 'people of Kaktovik', from both economic and cultural perspectives (Kaktovikmiut and Francis, n.d.). Subsistence activities in Kaktovik make use of a unique set of resources. Due to Kaktovik's location, hunters have access to terrestrial, riparian and marine resources, and make substantial use of all three. Jacobson and Wentworth (1982) summarised literature indicating that a prehistoric village existed at Kaktovik where 'many whale bones could be found'. Thus, the prehistoric people of the area, the 'Qanmaliurat', were certainly whale hunters, which suggests that bowhead whale (*Balaena mysticetus*) migratory patterns in the area have been similar for centuries. Of the marine mammals, the bowhead whale is the primary subsistence resource, but seals and polar bears are also taken (Jacobsen and Wentworth, 1982; Impact Assessment Inc, 1990). Subsistence activities, and especially activities surrounding the bowhead whale hunt, are central to the structural organisation and cultural identity of Kaktovik residents.

People from Kaktovik hunt whales only in the autumn, as the spring migration of bowheads past Kaktovik occurs far offshore, beyond the landfast ice zone. At Kaktovik, whaling is done from powerboats. These boats vary in characteristics, from an 18ft open Lund skiff to a 24 or 25ft cabin-cruiser type vessel. As speed is a much desired characteristic, motor size has tended to increase through time. Depending on the year, there are up to 11 whaling crews in Kaktovik. With a minimum of four or five people to a crew, most adult men are involved with whaling. Most other people in the village are involved in some support or processing capability. Whaling is an important community-wide activity.

Information from bowhead whales captured during subsistence harvests has been investigated as input to population models but it was concluded that the availability of whales to the hunters was not uniform (Punt *et al.*, 2003). If harvested whales are not a random sample of the population as a whole, then allowance must be made for the biases. Hunters from villages in northern Alaska prefer small whales to larger whales because they are easier to handle and the meat and blubber is said to be softer and better tasting (Braham *et al.*, 1980; McCartney, 1995). Thus, harvested whales do not represent a random sample of the population. This paper describes the bowhead whale harvest at Kaktovik and examines the size, sex, timing information and locations of bowhead whales harvested to assess whether they are a random sample of the population, and if not, to describe the biases.

METHODS

The data on the bowhead whale harvest at Kaktovik have been collected by the North Slope Borough (NSB) Department of Wildlife Management (Suydam *et al.*, 1995), Alaska Department of Fish and Game (ADF&G) (Lowry *et al.*, 2004), and the National Marine Fisheries Service (NMFS) (Marquette, 1977; Braham, 1987; Withrow *et al.*, 1992). The data are archived in a database maintained by the North Slope Borough. Postmortem examinations at Kaktovik are sometimes conducted by biologists, unlike many of the other villages along the Alaskan coast. The postmortem examinations include data on sex, body length,

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[1] Spelling corrected from the published version.

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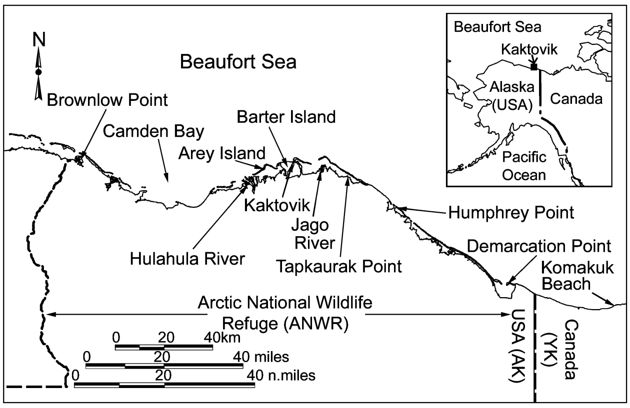


Fig. 1. Kaktovik and vicinity. Place names are according to USGS Geographic Names Information System (*http://geonames.usgs.gov*).

harvest date, a series of morphometric measurements, prey items in the stomach, reproductive status and scarring. Numerous tissue samples are also collected.

The size distribution of whales near Kaktovik was obtained by taking calibrated vertical aerial photographs of bowhead whales during studies based at Kaktovik during 1985-1986 and 1998-2000. Details of the photography methods are found in Koski *et al.* (1993) and Koski and Miller (2002).

RESULTS AND DISCUSSION

Numbers taken by year

Recent bowhead whale harvests at Kaktovik commenced in 1964, when two whales were harvested. NMFS records of harvests in 1964-72 are incomplete, and do not list any whales harvested at Kaktovik in 1965-72 (Marquette, 1977). However, a map in Oil/Whalers Working Group (1986) indicates that single whales were harvested there in 1968 and 1969. It is unlikely that many additional whales were harvested during this period because Kaktovik residents would remember an event as rare as a bowhead whale harvest during that period.

There was no quota on the number of bowhead whales that could be harvested before 1978, but rapid increases in bowhead harvest levels in Alaska during the mid-1970s caused concern that harvest levels were not sustainable. The International Whaling Commission (IWC) decided to impose quotas on the number of bowhead whales that could be taken by Alaskan native hunters, starting in 1978. The IWC quota is administered and monitored by the Alaska Eskimo Whaling Commission (AEWC). From 1978-1991, no more than two bowhead whale strikes or kills were allocated per year to Kaktovik. From 1992-2001, Kaktovik has been allocated three strikes or kills per year. In most years when Kaktovik reached its quota, the Kaktovik Whaling Captains Association (KWCA) could have applied for additional strikes because some strikes were not used by spring whaling villages. The most recent year when a 4th strike was transferred to Kaktovik was 2001 (and before that, 1997). During 1998-2000, additional strikes were available, but the KWCA decided not to request additional strikes because village requirements were met by the three whales landed in each of those years.

Since 1973, data on bowhead harvests have been collected by NMFS, ADF&G and the NSB, including information on numbers of whales landed, dates when whales were landed, and the sizes and sex of those whales. Fig. 2 summarises harvests at Kaktovik from 1973-2000. From 1973-1988, one or two whales were generally harvested, reflecting the village quota. In 1979 and 1981, whaling crews from Nuiqsut joined the Kaktovik whalers and the higher catches of five and three, respectively, in those years reflect the quotas from both villages. From 1989-2000, generally 2-4 whales were harvested per year.

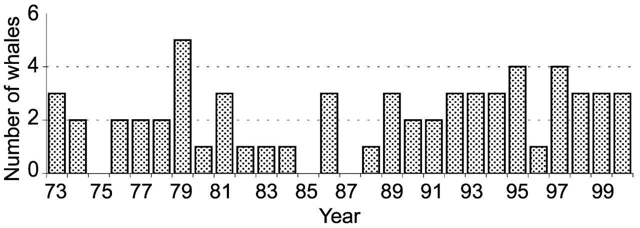


Fig. 2. Numbers of bowhead whales landed at Kaktovik each year (1973-2000). Main sources: Marquette (1977); Braham (1987); Withrow *et al.* (1992); Lowry *et al.* (2004); J.C. George (unpublished data).

Timing of the harvest

Each year, the KWCA decides at a meeting shortly before the start of the whaling season, the date at which the hunt will begin (weather permitting). In recent years, the bowhead whale hunt at Kaktovik has normally begun on or after the first weekend in September. The scheduled start dates for the 1997-2001 hunts were 3, 4, 11, 2 and 2 September respectively. The start date of the 1999 hunt was delayed by a local emergency – a fatal boating accident. However, the hunt has started earlier in some years (e.g. by 22 August in 1992). In most years, relatively few bowhead whales are present near Kaktovik until the beginning of the westward migration of whales from summering areas east of Kaktovik (see Miller *et al.*, 2002). In addition, in recent years the KWCA has voted to delay the hunt until September, when temperatures are cooler and so the whale meat is less likely to deteriorate. Thus, the start of the hunt is usually timed to coincide with the early part of the main westward migration in early September. The whales accessible then tend to consist primarily of the small sub-adult whales that are preferred by hunters (Koski and Miller, 2002).

The dates when whales landed at Kaktovik during 1976-2000 were struck are shown in Fig. 3, organised by year (panel A) and 10-day period (panel B). The date is unknown for one of the 61 whales landed during these years. The majority (83%) of the whales landed during this period were struck during September. Thirty-two percent of the whales were struck in each of the 1-10 and 11-20 September periods, 20% were struck 21-30 September, 10% were struck 1-10 October and 3% were struck in each of 22-31 August and 11-20 October (Fig. 3B).

In recent years, the typical harvest dates have become earlier although the quota and the number of whales taken have increased. The trend for an earlier harvest is significant ($r = -0.46$, df = 58, $p<0.001$). Based on personal observations, this change is at least partially due to an increase in the efficiency of the Kaktovik hunters in harvesting whales due to improvements in hunting techniques and equipment (boats, global positioning system (GPS), etc.). Another contributing factor may be the increase in the bowhead whale population (e.g. George *et al.*, 2004b). As a result, whales presumably are now more numerous near Kaktovik early in the hunting season than they were during the 1970s and 1980s. Changes in whale utilisation of the general Kaktovik area (see Miller *et al.*, 2002) may also be involved. Average sighting rates during aerial surveys increased markedly in the 1990s relative to the 1980s (Miller *et al.*, 2002).

Harvest locations

Since 1973, all bowhead whales harvested by residents of Kaktovik for which the harvest locations have been reported were struck within 43km of the village. Most of these whales were struck within 30km (Fig. 4). The core area where whalers search for whales is from Hulahula River in the west to Tapkaurak Point in the east and offshore as far as 32km (Fig. 1). Although a few of the most distant harvest locations were during the 1970s (Fig. 4), the mean distance of reported harvest locations from Kaktovik was not significantly different between the 1970s (17.0km, *n*=16), 1980s (17.9km, *n*=14) and 1990-2000 (15.2km, *n*=21) (Kruskal-Wallis test, p>0.05). It should be noted that the locations where 10 bowheads were struck are not known, and some reported locations, especially for years before GPS units were widely used, are approximate.

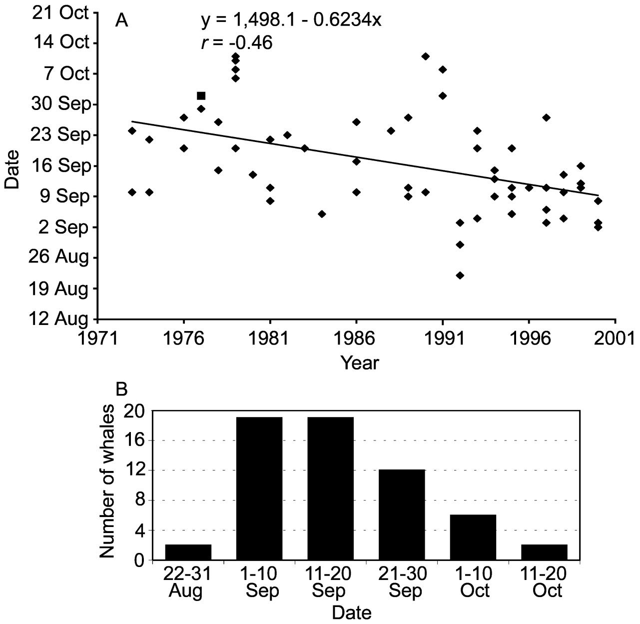


Fig. 3. Strike dates for bowhead whales landed at Kaktovik (A) by year and (B) by 10-day period, 1973-2000. The square symbol near the upper left side of panel A represents a young-of-the-year calf (approx. 5 months old). Main sources same as Fig. 2.

Sizes of harvested bowhead whales

As noted earlier, the Kaktovik whalers attempt to harvest small whales because they are easier to handle and are considered to taste better. Thus, although the lengths of the harvested whales partly reflect the length distribution of the whales near Kaktovik at the dates in question, they are strongly influenced by hunter selectivity. The frequency distribution for the lengths of whales landed at Kaktovik is shown in Fig. 5. The reported lengths of harvested whales in Fig. 5 and subsequent figures have been reduced by 8.2% to account for the stretching that occurs when the whale is dragged onto land (see George *et al.*, 2004a). The overall range of the whales landed at Kaktovik is similar to that of the living whales whose lengths have been measured in the Arey Island to Humphrey Point area during September (Fig. 5B vs 5A).

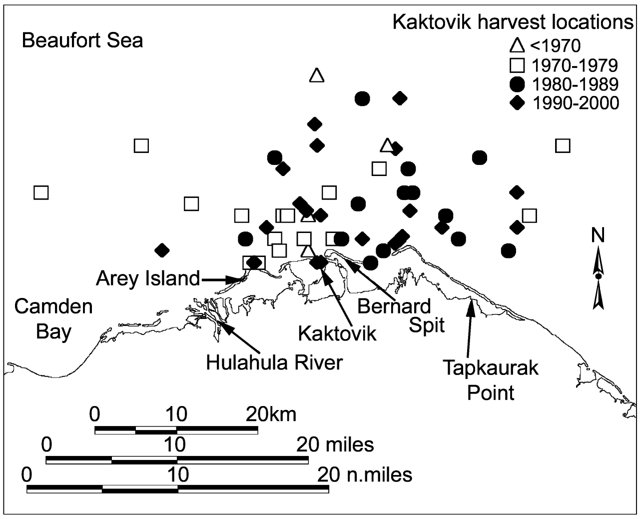


Fig. 4. Locations where bowhead whales were reported to have been harvested by residents of Kaktovik (1976-2000). Not shown are two whales taken ~170km west of Kaktovik in 1937 and 1940. Major Sources: Oil/Whalers Working Group (1986), Kaleak (1996), Lowry *et al.* (2004), J.C. George (unpublished data).

However, small whales constitute a higher percentage of the landed whales than of the whales photographed in the area. Of the bowhead whales harvested by Kaktovik whalers, 23.6% were longer than 13m (after allowance for stretching), and therefore were considered to be adults (Koski *et al.*, 1993). This compares to 43.4% adults in the overall bowhead whale population, if calves are excluded (Angliss *et al.*, 1995), and 50% adults among whales photographed near Kaktovik, calves excluded (Koski and Miller, 2002). The proportion of adults was significantly lower among the harvested whales than among the population as a whole during 1973-2000, regardless of which abundance estimate was used in the analysis, (χ^2 test, p <0.01) and among the whales that were photographed near Kaktovik (χ^2 test, p <0.001). These data confirm that the whales landed by Kaktovik whalers tend to be smaller than those in the population as a whole.

The autumn migration is partially segregated according to size, with the smaller whales tending to occur earlier in the autumn (Braham *et al.*, 1984; Moore and Reeves, 1993; Koski and Miller, 2002). However, there was no significant correlation between date and the size of a whale harvested (r = –0.064, df=53, p >0.05; Fig. 6). This indicates that whalers were able to select small whales throughout the whaling season even though the small whales become proportionally scarcer as the season progresses.

Fig. 7 shows the lengths of the harvested whales by the year harvested. There has been a marginally significant decline in the average size of whales harvested over the 1976-2000 period (r = –0.33, df=53, p <0.05). This suggests that the whalers have become more selective about the sizes of whales that they have harvested in recent years. This increased selectivity has probably been possible through some combination of two factors: increased availability of whales associated with the bowhead whale population increase, and increased efficiency of the hunters in capturing whales (allowing them to be selective while still filling their quota).

Sex of harvested bowheads

The sex of a whale cannot generally be determined by whalers before they strike it unless it is a female accompanied by a calf, which hunters avoid taking. The sex of 55 bowhead whales harvested at Kaktovik has been recorded. Twenty-eight were males and 27 were females, which is not significantly different than 1:1 (χ^2 test, $p > 0.05$). However, during the first half of the harvest (22 August-13 September), 67% of the harvested whales were female, and during the last half of the harvest (14 September-11 October) only 32% were female (Figs 6 and 8). This difference is significant (χ^2 test, $p<0.05$). From 1990 to the present, females have been more common among the harvested whales (18 females and 13 males), but before 1990 more males than females were harvested (15 males and 8 females). This difference was not significant (χ^2 test, $p>0.05$) and is due to the tendency for earlier harvests in recent years, and the preponderance of females early in the season (Figs 6 and 7). The reason for segregation by sex near Kaktovik is not known.

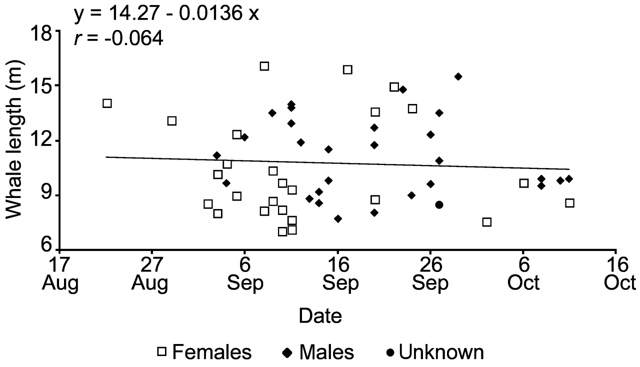


Fig. 6. Whale length vs date for whales harvested at Kaktovik (1976-2000); females and males are distinguished. A 6.2m calf harvested on 2 October 1977 is excluded. Whale lengths are adjusted downward to allow for stretching (see text).

Conclusions

The subsistence harvest at Kaktovik during recent years is not a random sample of the bowhead population. The autumn migration of bowhead whales past Kaktovik is segregated by age and sex. The harvest, especially in recent years, has been primarily during the early part of the migration. Hunters have purposely and successfully selected small whales from among those present near Kaktovik even during periods when primarily larger whales were present. There were also sex related biases in the harvest because females appear to be more common early in the season, and males more common later in the season. Why this pattern might occur is unknown.

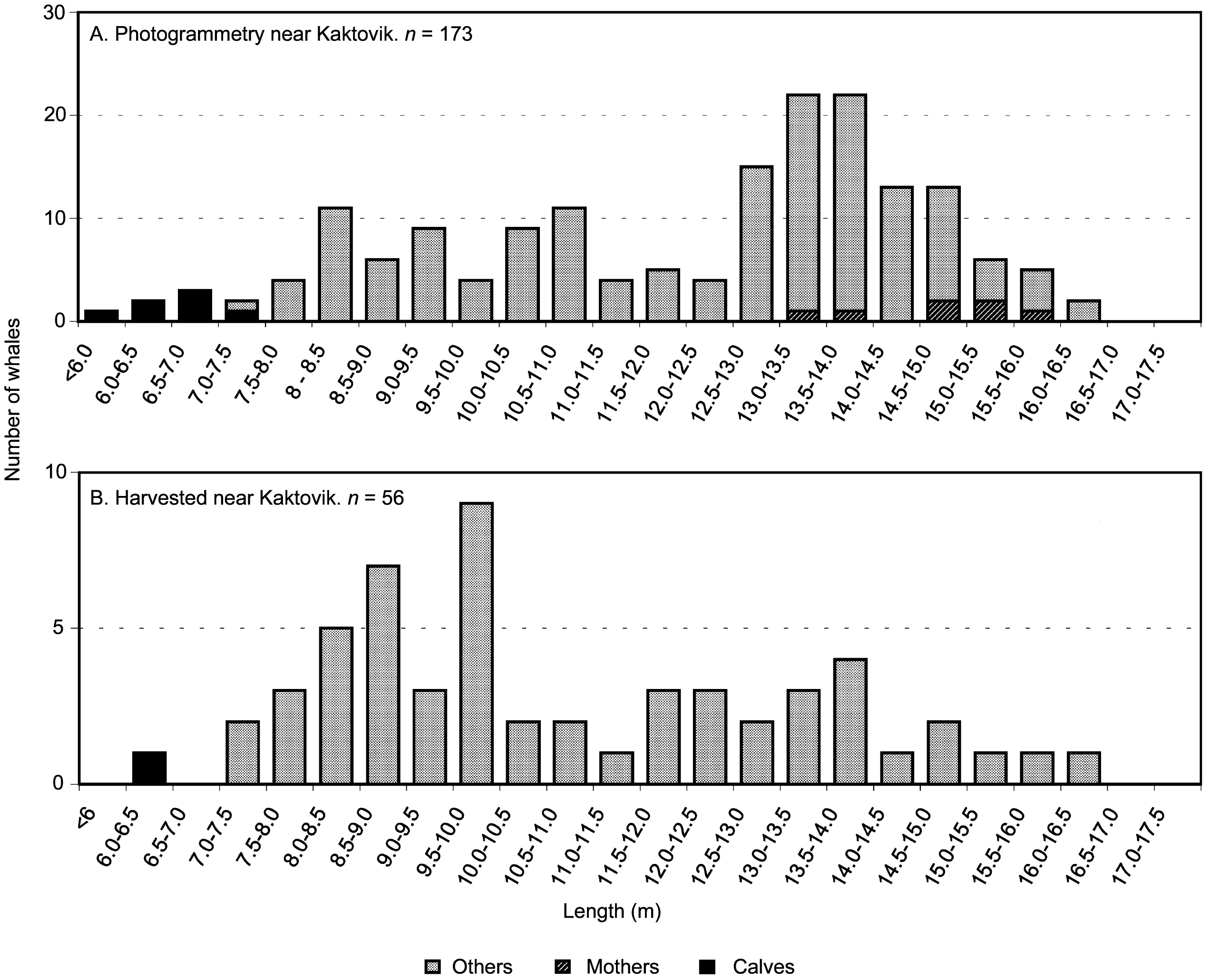
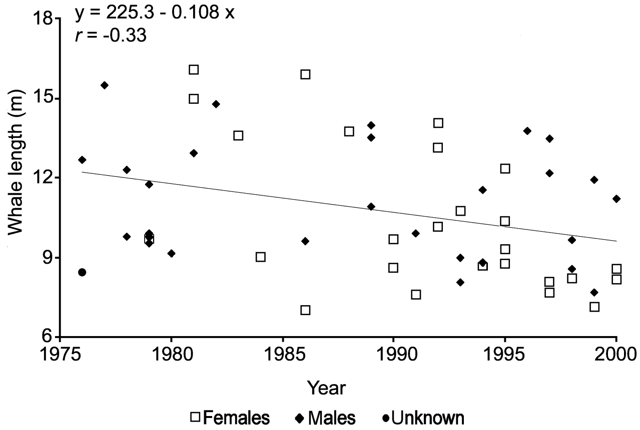


Fig. 5. Length frequency distributions of bowheads (A) photographed near Kaktovik 1982-2000 (from Koski and Miller, 2002) and (B) harvested near Kaktovik 1976-2000 (same sources as Fig. 2). In (B), lengths have been adjusted downward by 8.2% to account for stretching (see text and George *et al.*, 2004a).



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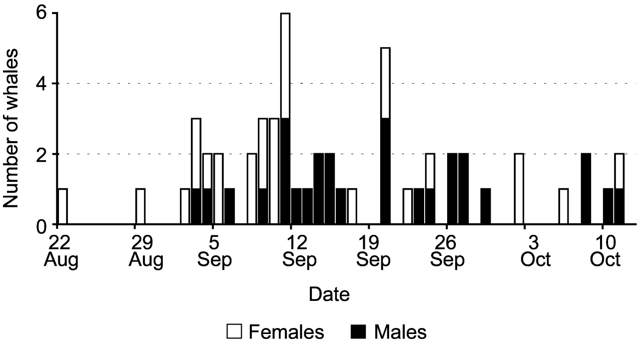

Fig. 8. Sexes of whales vs date for whales harvested at Kaktovik (1976-2000).

ACKNOWLEDGEMENTS

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A note on the friction of different ropes in right whale (*Eubalaena glacialis*) baleen: an entanglement model

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ABSTRACT

Entanglement in fishing gear, particularly fixed trap, constitutes a significant source of North Atlantic right whale (*Eubalaena glacialis*) mortality. Entanglements may initiate with rope fouling baleen plates before snagging other appendages. Low friction between rope and baleen may minimise the risk of a sustained, progressive entanglement. The friction of eight different rope types against right whale baleen was examined by measuring the tension as each rope was pulled through two baleen plates held underwater. Polypropylene rope generated less friction with the baleen than all other fibres tested, including nylon, polyester, and commercial sinking line (a polypropylene/polyester blend). Thus, new commercial floating line (3-strand polypropylene) generates less friction than new commercial sinking line, both of which are commonly used in the fixed gear industry. Therefore, minimising rope friction should be one of the design parameters for whale-safe fixed fishing gear. Further study is required on the impact of rope aging, mouth closing and operator safety before recommendations can be made to industry.

KEYWORDS: FISHERIES; CONSERVATION; INCIDENTAL CAPTURE; NORTH ATLANTIC RIGHT WHALE

INTRODUCTION

A study of 54 known North Atlantic right whale (*Eubalaena glacialis*) deaths from 1970 to 2001 revealed that ship collisions were responsible for 35% of these deaths and 9% were the result of gear entanglement (Knowlton and Kraus, 2001; McKiernan *et al.*, 2002). More than 70% of the right whale population bears scars from entanglement, and the numbers of fatal and potentially fatal entanglements have increased significantly in the last few years (Knowlton *et al.*, 2001). Many whales that die from entanglement are extremely emaciated at the time of death: the carcass will sink if negatively buoyant. Since these carcasses often cannot be recovered, it appears likely that the impact of gear entanglements has been underestimated in the past (Knowlton and Kraus, 2001).

North Atlantic right whales filter feed by swimming with their mouth open, gathering large numbers of copepods as the water flows in passively. Right whale entanglements regularly involve the mouth, with fishing gear being entangled in the baleen. Kozuck *et al.* (2003) found that 74% (29/39 entanglement events which involved 35 individuals) involved the head/mouth region as the point of gear attachment (Figs 1 and 2), along with other body parts. A total of 54% (21/39 entanglement events) involved only the head/mouth region as the point of gear attachment. In the same study 8/15 dead or presumed dead right whales were entangled in the head/mouth region. Therefore, it seems reasonable to predict that using fishing gear with reduced friction against right whale baleen may help increase the rate of self disentanglement. This study was conducted to compare common types of line used in the commercial fishing industry. Eight ropes of various fibres and constructions were pulled through the baleen, and the resulting tension was measured. It was discovered that polypropylene generates the least friction with baleen.

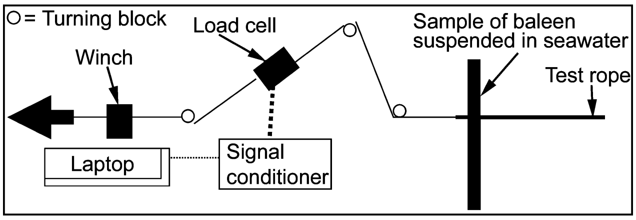

Fig. 1. Experimental design for pulling rope through right whale baleen suspended in sea water.

It has been predicted that preventing the deaths of two females per year may allow the right whale population to begin to recover (Fujiwara and Caswell, 2001). Gear modifications that prevent entanglements are critical for the survival of the species.

METHODS

Samples of baleen were removed during necropsy from three North Atlantic right whales, comprising New England Aquarium Catalogue Numbers 1014, 1504 and 1238 (Moore *et al.*, 2005). The samples, which contained blocks of 9 to 24 plates, were clamped between two timbers that were fixed to overlie a tank (Fig. 1). The gum line was situated at the underside of the timbers, which coincided with the surface of sea water filling the tank into which the baleen plates projected. The test line was pulled between two plates of baleen in the sample with a *Lewmar 44* 2-speed manual self-tailing winch via turning blocks. The same pair of plates was used for each sample. The order in which the rope types were tested was randomised for each baleen sample. Preliminary tests showed that friction was independent of

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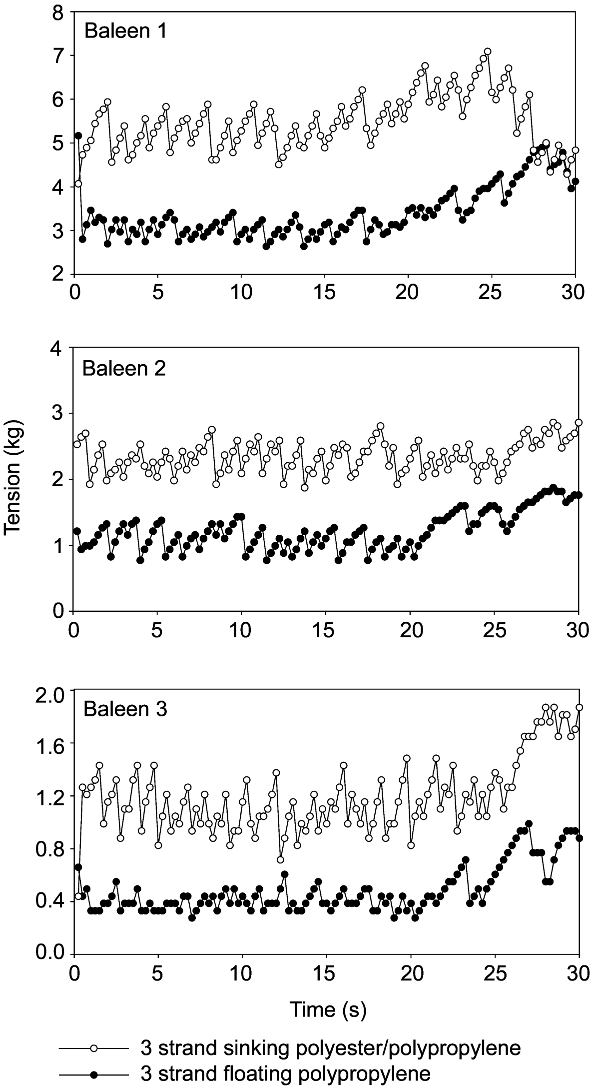

Fig. 2. Examples of tension records for two rope types pulled between three different samples of right whale baleen as examples of the raw data obtained.

the speed at which the rope was pulled, confirming that friction is independent of speed at low pulling velocities (Mclean and Nelson, 1952).

Eight different rope types were tested. All ropes had a diameter of 9.5mm (3/8 inch). Rope types were: (1) 3-strand polyester; (2) 3-strand nylon; (3) 3-strand polypropylene (commercial floating line); (4) 3-strand commercial sinking line (polypropylene and polyester blend); (5) braided polyester; (6) braided nylon; (7) hollow braid polypropylene; and (8) braided polypropylene. Test ropes were pulled through the baleen just below the gum line (40cm below the crown of baleen sample 1, 26.5cm below the crown of baleen sample 2, and 30cm below the crown of baleen sample 3). Three samples of each rope type were used, one for each sample of baleen. Rope samples were obtained from 3 different spools for all rope types except the 3-strand sinking line, 3-strand polyester, and hollow braid polypropylene. All ropes were manufactured by New England Ropes, New Bedford, Mass. (USA) except for the commercial sinking (Super Hyliner, Cape Fishermen’s Supply, Chatham, USA) and floating (Wellington Puritan, Madison, USA) lines.

An MLP-100 load cell tensiometer (Transducer Techniques, Temecula, USA), measured the tension as the test rope passed through the baleen (see Fig. 1). The signal from the load cell was modified by a TMO-1 amplifier/conditioner module with optional ATM-1 enclosure and APD-12 AC Power Adaptor (Transducer Techniques), a domino-2 coprocessor (Micromint Inc., Lake Mary, USA), and an 8-channel multi-range A to D converter (Digi-Key Corporation, Thief River Falls, USA). The modified signal was recorded using software developed by Upper Cape Systems (East Falmouth, USA). This system generates an accuracy of 0.03% of the maximum capacity of the load cell (45kg capacity, 14g accuracy for this model: Transducer Techniques, product literature). The tension was measured four times every second during the test pull. The initial start up phase was then deleted from each pull data file. Tension, over a standard amount of time of each pull for each baleen sample, was then averaged in a series of comparisons.

To test whether rope types differed in the amount of friction generated, a randomised blocks design was utilised with rope type as the fixed factor and the three pieces of baleen as blocks. Four *a priori* contrasts tested whether friction differed between the two types of rope construction (3-strand versus braided) and whether friction differed between ropes of different fibre (polypropylene versus nylon, polypropylene versus polyester and polyester versus nylon). The per comparison error rate ($\propto = 0.0125$) was set using the Bonferroni method. A Tukey’s multiple comparison test was performed to determine whether the mean tension of 3-strand sinking line and hollow braided polypropylene line differed from the other rope types. Data were log transformed to homogenise the variances for the statistical tests.

RESULTS

For a given rope type and baleen sample, mean tension varied over time and between baleen samples, but the differences in tension between the rope types for a specific baleen sample were relatively consistent. Examples of the raw data plots obtained are shown in Fig. 2. The tension showed a phasic pattern as the rope went through successive cycles of sticking and releasing, producing a saw tooth tension plot. The difference in the mean tension of the two rope constructions, 3-strand and braided, was not significant (Table 1). Different rope fibres, however, resulted in some significantly different mean tensions. The mean tension of polypropylene line was significantly lower than that of nylon or polyester line (Table 1; Fig. 3), whilst there was no significant difference between tensions of polyester and nylon.

Table 1

A priori contrast results comparing mean tensions of different rope constructions and fibre types.

Test	Df	SS	*F*	*P*
3-strand vs. braided	1	0.019	0.661	0.4297
Polypropylene vs. nylon	1	0.708	24.168	0.0002*
Polypropylene vs. polyester	1	0.729	24.894	0.0002*
Polyester vs. nylon	1	0.000	0.0326	0.8592

*Statistically significant ($P < 0.01$).

Commercial sinking line had the highest mean tension of all the lines tested, but mean tension of this line was only significantly different from those of the three different polypropylene lines tested, Fig. 4). Mean tension did not differ significantly between the three polypropylene lines.

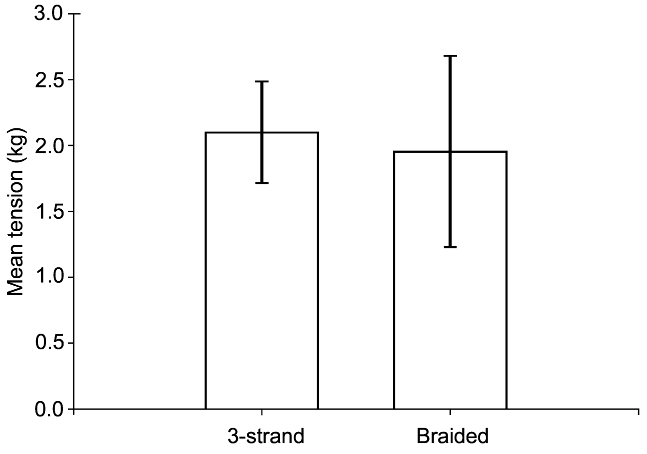

Fig. 3. Mean tension and ±1 standard deviation (*n*=3) for the two types of rope construction. The mean was calculated from the average of the three rope materials, nylon, polyester and polypropylene.

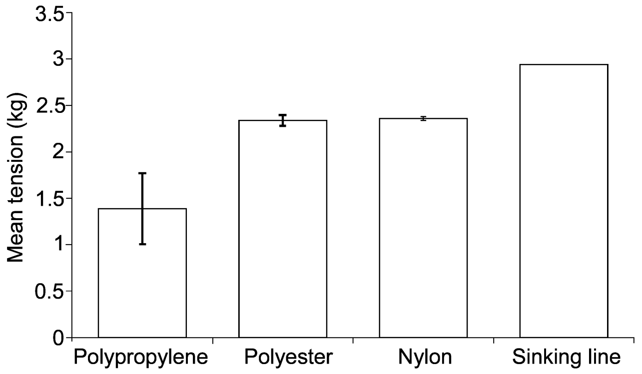

Fig. 4. Mean tension and 1 standard deviation (*n*=2) for the 4 rope materials. The mean was calculated from the average of the braided and 3 strand values. There is no error bar for sinking line because no braided line was available for this rope type.

DISCUSSION

This study was designed to compare the behaviour of different kinds of rope in right whale baleen in order to develop a better understanding of rope-baleen interactions during gear entanglement with the mouth. Of all the rope fibres examined, polypropylene line (commercial floating line) generated the least friction with right whale baleen.

The fixed trap industries generally prefer to use floating ground-line between traps because it is less expensive, avoids chafing on rocky substrates, and allows fishermen to retrieve trap gear more easily when buoy lines are severed (McKiernan *et al.*, 2002). However, the use of floating lines for ground-line has been shown to produce arcs of line between traps extending 3-6m above the substrate. These arcs of line in the water column increase the risk of entanglement. For this reason, regulations exist to encourage the use of sinking line in parts of the United States east coast lobster industry.

Commercial sinking line is typically composed of about 60-65% polyester and 35-40% polypropylene fibres (pers. comm. with Hy-Liner Rope Inc., Rockland, Maine, USA). The polyester is required to counter the buoyancy of polypropylene so the line will sink. Polypropylene is about a third of the cost of polyester, so manufacturers generally use as much polypropylene as possible.

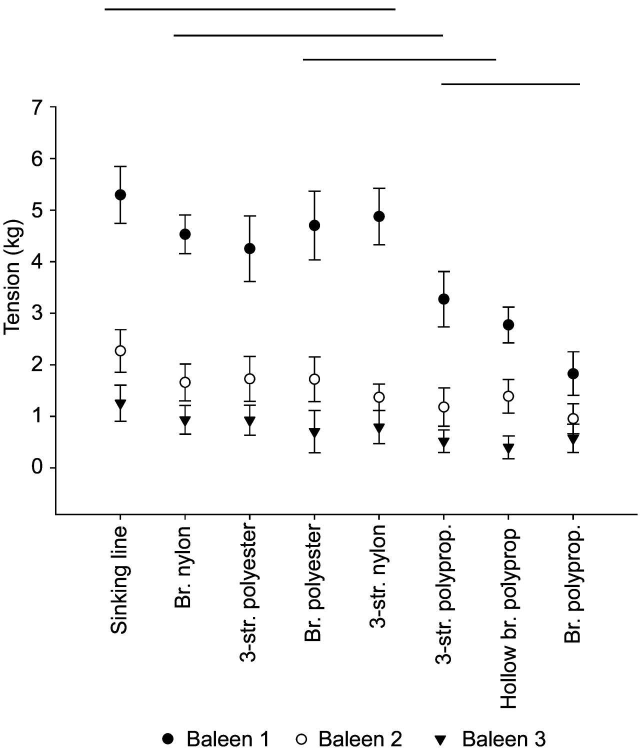

Fig. 5. Mean tension and 1 standard deviation for each rope type and baleen sample tested. The mean was calculated by taking the average of the tension values over a standard portion of the test run. Horizontal lines above the graph connect means of rope types that are not statistically significant. (Br.=Braided, Str.=Stranded).

Given the regular appearance of mud on their heads (unpublished data), right whales must sometimes feed on and in bottom mud. If entanglements do occur there, this may be fishery dependent – lobster gear is typically set on harder substrates. The results presented in this paper suggest value in the development of a sinking ground-line that has less friction with right whale baleen. A relatively low-friction sinking line could be made by creating a polypropylene sheath around the line, while maintaining the property of negative buoyancy. Weighting polypropylene line with lead to make it sink is also a possible alternative.

The relative tensions on baleen of the different rope fibres do not relate directly to the coefficients of friction reported by Samson Rope Technologies (Table 2). A coefficient of friction is defined as the ratio of force of friction to the normal (perpendicular) force. These values are dependent on the surface that it is being pulled against. Coefficients of friction for one rope type can change greatly when measured against different surfaces and in wet versus dry conditions (Brown, 1977). Therefore, rope-baleen tensions cannot be predicted by established friction coefficients. Furthermore, this study focused only on new rope samples. Polypropylene in particular undergoes significant changes in surface properties as it ages. Future work should focus on comparing the friction generated by rope between samples of different age. In addition further studies should also compare the operational practicalities of slippery rope in the field setting, and extend to include the behaviour of different rope types on whale skin, as any rope that is significantly entangled in baleen will also be rubbing over one or more skin covered body parts, especially the gum, axilla and caudal peduncle. When the mouth is closed, a significant increase in friction is caused by the bite of the lower lip against the gum; however this is reversible, as right whales swim for substantial periods with their mouths open when filter feeding. It is also important to try to determine whether low friction rope has a positive or negative effect on other types of entanglements (e.g. flippers, tail).

Table 2

Coefficient of friction vs. mean tension for selected commercial ropes.

Rope fibre	Coefficient of friction	Mean tension (lbs)
Nylon	0.89	5.244
Polyester	0.33	5.196
Polypropylene	0.37	3.084

Finally, before any significant reduction in rope friction is attempted in an industrial setting, the impact of such a reduction on the safety of operators will also need to be evaluated.

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Results and evaluation of US Navy shock trial environmental mitigation of marine mammals and sea turtles

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ABSTRACT

A shock trial of a US Navy Destroyer, the USS *Winston S. Churchill*, was conducted offshore of northern Florida in May and June 2001. The shock trial consisted of three underwater detonation tests, spaced approximately one week apart. Environmental mitigation to minimise the impact of the shock trial on marine mammals and sea turtles was based on a Safety Range of 3.7km (2 n.miles) radius around the detonation site, and a Buffer Zone of an additional 1.85km (1 n.mile) radius beyond the Safety Range. Mitigation included site selection surveys, pre-detonation aerial, vessel and bio-acoustic monitoring, and post-detonation aerial and vessel monitoring. Six species of odontocete and two species of sea turtle were identified during mitigation monitoring, as well as several sightings that could not be identified by species. Site selection aerial surveys were implemented to select a test site with the lowest abundance of marine mammals and sea turtles. Nearly 300 animals were seen during site selection surveys. Pre-detonation aerial and vessel monitoring was implemented to sight any marine mammal and sea turtle within the Safety Range on designated test days, and track the animals until they could be verified to be outside the Safety Range. Approximately 1,200 marine mammals and 32 sea turtles were sighted during pre-detonation monitoring. Pre-detonation bio-acoustic monitoring was implemented to detect large cetaceans within the Safety Range and Buffer Zone; the only calls heard were from dolphins that could not be localised. Post-detonation monitoring was implemented to determine the effectiveness of mitigation procedures. No injured or dead marine mammals or turtles were detected during approximately 185 hours of post-detonation aerial and vessel visual monitoring. Post-detonation monitoring resulted in observations of 767 marine mammals and 42 sea turtles. With only two exceptions, the same marine mammal and sea turtle species were observed prior to, during and after the shock trial test time period. Factors leading to the success of this environmental mitigation effort are summarised, and recommendations for improvements to mitigation efforts of this size and scope are suggested. These recommendations include the use of a simultaneous second aircraft for improved coverage during pre-detonation surveys, increased post-detonation aerial monitoring, equitable survey data for all test sites under consideration during planning stages, and reassessment of bio-acoustic monitoring need and purpose.

KEYWORDS: ACOUSTICS; ATLANTIC OCEAN; SURVEY-AERIAL; ODONTOCETE; SEA TURTLE; NOISE

INTRODUCTION

Each new class of ship (or major upgrade) constructed for the United States Navy must undergo US Live Fire Test and Evaluation (LFT&E) based on congressional legislation (10 USC 2366). A shock trial is one means of addressing the requirements of LFT&E and the Navy's ship shock hardening requirements. A shock trial is a series of tests consisting of underwater detonations, each separated by approximately one week, that send a shock wave through the ship's hull to simulate explosive near-misses during combat. A shock trial allows the Navy to assess the survivability of the hull, the ship's systems, and the capability of the ship to protect the crew. The USS *Winston S. Churchill* (DDG 81), a Flight IIA guided missile destroyer of the *Arleigh Burke* class, was selected as the shock trial ship for this class.

Operational requirements, including water depth, ship traffic, proximity to Naval Stations for ship and air support, and proximity to ship repair facilities and an ordnance loading station, were used to identify potential test areas. Three test areas met the operational requirements: Norfolk, Virginia; Pascagoula, Mississippi; and Mayport, Florida. The Navy prepared an Environmental Impact Statement (EIS) to assess the potential environmental impacts associated with conducting the shock trial in accordance with the National Environmental Policy Act and Executive Order 12114 (Department of the Navy, 2001). The EIS was prepared with the cooperation of the US National Marine Fisheries Service (NMFS). The EIS assessed the three potential test areas and concluded that the Mayport test area represented the area least likely to negatively impact (via mortality, injury and harassment) marine mammals and sea turtles. The Mayport test area was 120km (65 n.miles) offshore of Jacksonville, Florida (Fig. 1). The overall configuration of the test area was based on operational restrictions, including minimum 600ft depth, within 100 n.miles of shore support and avoidance of offshore ordnance dumping areas. The Mayport test area was within the Gulf Stream current that runs offshore of the east coast of North America (Schmitz *et al.*, 1987). The Navy requested and received a Letter of Authorisation (LOA) from the NMFS for the incidental taking of a small number of marine mammals during the shock trial, in accordance with the US Marine Mammal Protection Act (MMPA). Additionally, incidental sea turtle takes under the US Endangered Species Act (ESA) were specified in the terms and conditions of a Biological Opinion written by NMFS.

A critical element of the EIS and the request for the LOA was the planning of environmental mitigation procedures to minimise the impact of the shock trial on marine mammals and sea turtles. The mitigation plan provided the overall approach to environmental mitigation and monitoring procedures, mitigation team and individual responsibilities and support requirements. The overall objectives of the Environmental Mitigation and Monitoring Plan were to:

(1) assist in the selection of a test site within the Mayport test area that posed the least risk to the marine environment, specifically marine mammals and sea turtles, by conducting site selection surveys;

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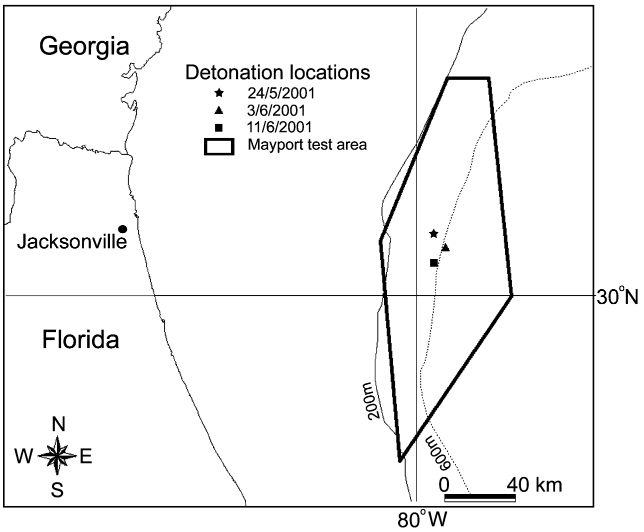

Fig. 1. Mayport, Florida, test area and locations of three shock trial tests, Spring 2001.

(2) conduct pre-detonation monitoring on the day of the test prior to each detonation in an effort to ensure it was free of detectable marine mammals, sea turtles, large *Sargassum* rafts, large schools of fish and flocks of seabirds (the latter three were considered potential indicators of marine mammal or sea turtle occurrence); and
(3) conduct post-detonation monitoring of the test site to measure the effectiveness of mitigation procedures and, when appropriate, recommend changes to the plan prior to the next test.

Environmental mitigation was based on a Safety Range of 3.7km (2 n.miles) radius around the detonation site and a Buffer Zone of an additional 1.85km (1 n.mile) radius beyond the Safety Range. The Safety Range was based on conservative calculations of the distance at which marine mammals and sea turtles would be killed or injured (Level A harassment under the MMPA) by the shock wave (Department of the Navy, 2001). The actual maximum distances at which mortality (1.35km [0.73 n.miles]) and injury (2.25km [1.22 n.miles]) were estimated to occur were well within the Safety Range. Beyond this range, impacts to marine mammals and sea turtles were expected to result only in Level B harassment, or temporary disruption of behavioural patterns, which were not mitigated because any observable disruptions were expected to be short-lived and not repetitive. The Safety Range and Buffer Zone provided one of the bases for test postponement, such that any marine mammal or sea turtle detected within the Safety Range would automatically lead to a postponement of the detonation until that animal could be verified clear of the Safety Range. Any animal seen within the Buffer Zone swimming towards the Safety Range also led to a postponement, until that animal could be verified to be swimming away from the Safety Range. Postponement would also occur if sea state conditions were unacceptable (greater than sea state 3 or Beaufort scale 4) or if visibility was not adequate for observations.

MATERIALS AND METHODS

Environmental mitigation components consisted of site selection surveys (aerial only), pre-detonation visual monitoring (aerial and shipboard), pre-detonation bio-acoustic monitoring and post-detonation visual monitoring (aerial and shipboard). The mitigation team leader coordinated all aspects of the mitigation effort and served as liaison to the rest of the shock trial operations team. The lead scientist directed the activities of the mitigation team, tracked all marine mammal and sea turtle sightings and provided recommendations to the shock trial test director. All observers involved had a minimum of two years of experience as a marine mammal/sea turtle observer, veterinary assistant and/or bio-acoustician. On average observers had greater than 10 years experience. A representative from NMFS was part of the pre-detonation shipboard monitoring team, to verify that the mitigation plan was implemented adequately.

All aerial surveys/monitoring were flown in high wing aircraft (O2 SkyMaster) at an altitude of 229m (750ft) and a speed of 110 knots. Observed marine mammals and sea turtles were identified by species where possible, and location, group size and swim direction were noted.

Site selection surveys were conducted 1-2 days prior to the planned test day, and were designed to select a test site within the test area, with the lowest relative abundance of marine mammals and sea turtles. Survey lines were spaced 9.3km (5 n.miles) apart and the entire test area could be surveyed during a 5.5 hour flight. The lead scientist assessed the site selection survey results and recommended the best area in which to conduct the test to the shock trial test director. Site selection surveys were also used to monitor environmental conditions in the test area, such as sea state and visibility, and provided the shock trial test director with valuable information on the feasibility of successfully conducting the test. If site selection monitoring of the test area could not be conducted adequately due to inclement weather or high sea states, the test was postponed for at least a day or until an adequate site selection survey could be completed. Site selection surveys occurring after the first test also served as supplementary post-detonation monitoring because the area surveyed often overlapped with the previous test site(s). However, effort and sightings recorded during site selection surveys were not double-counted as post-detonation data.

All monitoring conducted prior to the detonation on designated test days was ‘pre-detonation’ even if the test was delayed and no detonation occurred on that day. Pre-detonation aerial visual monitoring was designed to locate any animals within the Safety Range and track the animals until they could be verified to be outside the Safety Range. Aerial monitoring consisted of broad scale survey lines that were spaced 1.85km (1 n.mile) apart, followed by finer scale survey lines in the immediate area of the detonation site spaced at 0.93km (0.5 n.miles). All animal positions and swim directions were plotted relative to the detonation site using a marine animal tracking and sighting software program (MATS). The MATS software also depicted the Safety Range and Buffer Zone, which allowed the lead scientist to immediately determine whether or not a sighting was within the Safety Range. Visual monitoring from the aircraft commenced approximately 1.5 hours prior to the planned detonation and continued until detonation occurred. A second aircraft was on-call if the original aircraft had to leave the test area to refuel so that aerial monitoring was continuous.

Pre-detonation shipboard visual monitoring occurred on designated test days and was designed to locate marine mammals and sea turtles within the Safety Range and Buffer Zone, and to track the animals until they could be verified to be outside the Safety Range. Observers were based onboard the *Winston S. Churchill*, located in the centre of the Safety Range, and positioned on each bridge wing. Two observers monitored the test site with mounted 25× power binoculars (*Bigeyes*), while four additional observers monitored the site with handheld binoculars. This allowed for 360° overlapping coverage. Sighting locations were based on bearing and distance; bearing was measured relative to the bow of the vessel using a calibrated collar at the base of the yoke of the *Bigeyes*; distance was measured using a calibrated reticle in the oculars of the *Bigeyes*. All marine mammal and sea turtle positions and swim directions were immediately plotted relative to the detonation site using the MATS software. Shipboard monitoring commenced at least one hour prior to the planned detonation. Monitoring via the *Bigeyes* continued until three minutes prior to detonation, when the *Bigeyes* were stowed for safety. Monitoring via handheld binoculars continued until one minute prior to detonation, when all observers were positioned inside the bridge for safety considerations. Pre-detonation shipboard visual monitoring on designated test days also took place from the Marine Animal Recovery Team (MART) vessel. Sighting information reported by the MART was included in the pre-detonation sighting database and entered via the MATS software. The MART vessel was generally positioned about 5 n.miles from the detonation point during pre-detonation monitoring, so most sightings were well outside of the Safety Range and Buffer Zone. However, MART pre-detonation sighting information did provide an indication of animals moving towards the detonation point that could then be tracked by vessel observers onboard the *Winston S. Churchill*, or by aerial observers.

Aerial and shipboard observers tracked any marine mammals located within the Safety Range (often in tandem with each other) until the animals were verified clear of and swimming away from the Safety Range. Sea turtles were assumed to be moving north with the Gulf Stream current, which could be measured on a daily basis from the *Winston S. Churchill*; clearance of the Safety Range by sea turtles was calculated based on the original position and time of the sighting. For example, on a day when the current was measured as 3 knots, a sea turtle observed three miles from the northern edge of the Safety Range would be estimated to be clear of the area in one hour.

Pre-detonation bio-acoustic monitoring occurred on designated test days, and was designed to acoustically detect mysticetes and sperm whales (*Physeter macrocephalus*) within the Safety Range and Buffer Zone via localisation of lower-frequency (10Hz – 4kHz) calls. These species were monitored bio-acoustically in addition to visually because they are generally more difficult to detect at the surface than other marine mammals, due to their relatively solitary nature (mysticetes), or their tendency to dive deeply and remain submerged (sperm whales). A DIFAR-based Acoustic Monitoring System (D-AMS) was specifically designed for this test (Department of the Navy, 2002). The D-AMS was also capable of detecting some higher frequency calls from other marine mammals, but was not able to determine the call location. Prior to deploying the D-AMS, an ambient noise buoy and an AXBT (airborne expendable bathythermograph) were deployed for obtaining a sound-speed profile. Passive DIFAR sonobuoys, with 8 hour duration, were deployed from an Orion P-3 aircraft at least one hour prior to the planned detonation time, and continuously monitored. DIFAR buoys allowed for a radio frequency (RF) channel to be selected from 99 available, extending from 136.000 to 173.125MHz with 375kHz between channels. The radio frequency power was 1W. The acoustic coverage was from 5Hz to almost 5kHz, with maximum sensitivity at about 1.5kHz.

The sonobuoy array was a regular hexagon, formed by two rows of three buoys, each sandwiching a row of four buoys. The hexagon had 2 n.mile sides and one sonobuoy at the centre. The array spanned 4 n.miles. Based on the known ranges of source levels for large whales and expected background sounds for sea state three or less, animals could be expected to be heard from at least 5 n.miles, assuring coverage of the Safety Range and Buffer Zone. The centre of the hexagonal array was selected so that the centre buoy would cross the planned shot point at the expected detonation time. For localisation, a minimum of two buoys were required to detect the whale call, thus providing two bearings whose intersection defined the whale location. Additional buoys were deployed to replace those lost to drift over time; buoy location was monitored via frequent over-flights of the P-3, and calculation of drift based on current. Acoustic data received from the buoys was monitored by four bio-acousticians based on the P-3. Data were displayed as strip spectrograms showing frequency and duration of the call and as polar displays of bearing-time and bearing-frequency. Processing of mysticete and sperm whale call location was based on bearing and range, and locations could be ascertained in real-time. Calling animals could also be tracked acoustically until they were outside of the Safety Range. Because the bio-acoustic component was designed to provide real-time detections, the data were recorded in a continuous 30 minute loop only, such that data older than 30 minutes were replaced by new data. There was no long-term archiving of bio-acoustic data. All bio-acoustic data were immediately radioed to the lead scientist and entered into the MATS software. Bio-acoustic monitoring was continuous until the time of detonation.

Monitoring that occurred after each detonation was termed ‘post-detonation’, until the next site selection survey commenced. Each test was separated by at least one week to allow enough time to prepare the ship, and so there was usually a 2-3 day gap when no surveys or monitoring were conducted. The objective of post-detonation monitoring was to detect any marine mammals or sea turtles killed or injured by the test. Animals killed by the blast would likely suffer lung rupture, which would cause them to float to the surface due to air in the blood stream (Department of the Navy, 2001). Animals that were mortally wounded and whose carcasses sunk, would likely resurface within a few days although this would depend on size and type of animal, fat stores, currents, depth and temperature of the water and other variables. Post-detonation visual monitoring commenced immediately following each detonation. Aerial monitoring was assisted by the MART. Aerial and vessel monitoring continued in the area of the detonation and progressively down current for two days after the first two tests and seven days after the final test. Aerial monitoring via transect lines spaced 0.93km (0.5 n.miles) apart were flown centred around and down current of the detonation site immediately following the test. During aerial monitoring on the days following the test, aerial observers first monitored the area immediately surrounding the detonation site, then focused on areas down current from the test site; the exact area monitored depended on the speed and direction of the Gulf Stream.

The primary responsibility of the MART was to be on-site should any injured or dead marine mammals or sea turtles be detected in the area after the detonation and, secondarily, provide continuous on-site visual post-detonation monitoring. The MART included a marine mammal veterinarian, sea turtle veterinarian, marine mammal observer and two sea turtle specialists. The MART was based onboard the R/V *Athena*, a 50.3m (165ft) research vessel owned and operated by NSWC (Naval Systems Weapon Center), based in Panama City, Florida. This vessel was chosen specifically for the MART because it was: (1) capable of lifting injured or dead marine mammals or sea turtles up to 4,536kg (10,000lb) onto a rear deck work area; (2) outfitted with an aft dive platform for investigating larger animals; (3) capable of high speeds to transport injured animals quickly to shore; and (4) fitted with adequate deck work space, storage and freezer space. Primary visual searching aboard the MART vessel was by naked eye, and handheld 7× binoculars were used to confirm initial sightings, and to determine species identification, group size and swimming direction. Post-detonation MART monitoring commenced immediately after detonation, with a search of the detonation site, and then progressed down current. Track lines were in a zigzag pattern, with lines spaced approximately 1.85km (1 n.mile) apart and 9.3-11.1km (5-6 n.miles) long. Vessel speed ranged from 1.5-3.5 knots, depending on the speed of the Gulf Stream current pushing the vessel. A long-line high flyer was fashioned out of a buoy with a strobe light attached as a way to help the vessel maintain consistent speed with the Gulf Stream and allow the vessel to pick up a track line after diverting away for animal sightings. Environmental conditions such as sea state, wind speed and direction, swells and sea surface temperature were recorded every hour by the ship's captain. Visual observations started in the early morning when sunlight was sufficient for viewing, and ended 30 minutes prior to sunset, or when lighting conditions precluded adequate visualisation of the horizon.

RESULTS

The shock trial, consisting of three tests, was conducted in May and June 2001 (Department of the Navy, 2002). Mitigation activities in support of the test commenced 29 April and ended 18 June. For each test, a 4,536kg (10,000lb) charge was detonated, one on each of 24 May, 3 June and 11 June; there was also one mis-fire on 10 May. Environmental mitigation led to the postponement of tests on six occasions:

(1) 10 May: Sea turtle sighting and subsequent Risso's dolphin (*Grampus griseus*) sighting delayed testing by approximately 1.3 hours until both were confirmed clear of the Safety Range. No detonation occurred due to mis-fire.
(2) 31 May: Numerous marine mammal and sea turtle sightings delayed testing by one day.
(3) 1 June: Lack of bio-acoustic monitoring support delayed testing by approximately three hours, until a waiver of use was received from NMFS; thunderstorms in the area and a subsequent lack of aerial monitoring support delayed testing by one day.
(4) 2 June: Unacceptable sea state (4 and above) delayed testing by one day.
(5) 3 June: Dolphin sighting delayed testing by approximately 1.5 hours, until it was confirmed to be clear of Safety Range. Detonation occurred after the Safety Range was confirmed clear of detectable marine mammals and sea turtles.
(6) 11 June: As (5) above.

Approximately 54 hours were flown during site selection surveys (Department of the Navy, 2002). A total of 231 marine mammals and 67 sea turtles were seen (Table 1, Fig. 2). Several surveys were flown in less than optimal conditions (sea state >3, low visibility), which precluded effective surveying. Under those circumstances, an additional day of site selection surveying was completed before the test was undertaken. Marine mammals identified by species included bottlenose dolphins (*Tursiops truncatus*), pilot whales (most likely short-finned pilot whales, *Globicephala macrorhynchus*), Risso's dolphins and *Stenella* spp., in addition to loggerhead sea turtles (*Caretta caretta*). Several sightings were recorded that could not be identified by species due to the greater emphasis placed on determining relative abundance of all animals in the test area.

Approximately 45 hours of aerial surveys were flown during pre-detonation activities. A total of 694 marine mammals and 24 sea turtles were seen (Table 1, Fig. 3). Marine mammals identified by species during pre-detonation aerial monitoring included bottlenose dolphins, pilot whales, Risso's dolphins and *Stenella* spp., in addition to loggerhead sea turtles. Several sightings were recorded that could not be identified by species due to the greater emphasis placed on detection of animals relative to the detonation point during pre-detonation monitoring. This limited the effort (e.g. aerial circling) available for identifying sightings by species, and obtaining detailed information on group sizes and behaviour. The majority of sightings (448 marine mammals and 4 turtles) were observed on 31 May, when nearly 800 animals (794 marine mammals and 4 turtles) were observed by aerial and vessel observers. The number and consistency of sightings resulted in the postponement of testing on that day.

Approximately 24 hours of pre-detonation shipboard observations were carried out from the *Winston S. Churchill*, and a total of 200 marine mammals and three sea turtles were observed (Table 1, Fig. 3). The same species of marine mammals and sea turtles identified during site selection and pre-detonation aerial monitoring were identified during shipboard monitoring. Several sightings could not be identified by species due to their distance from the vessel.

The MART vessel was present just outside the Safety Range and Buffer Zone prior to detonation on several designated test days: 10 May (misfire), 11 May (on-site escort to the operations vessel), 24 May (prior to detonation), 31 May-3 June (prior to detonation) and 11 June (prior to detonation). MART observers were on-station for approximately 28 hours, and a total of 308 marine mammals and 5 sea turtles were observed (Table 1, Fig. 3). The MART was outside the immediate area of the detonation, and therefore observers were able to spend greater time on species identification and behaviour. Most sightings were of bottlenose dolphins, with the largest group estimated at 30-35 individuals. Several groups bow rode for several minutes. It was very difficult to visually distinguish whether the animals were of the shallow, warm water ecotype, or the deep, cold-water ecotype (Duffield *et al.*, 1983; Duffield, 1987; Mead and Potter, 1995). Three groups of pilot whales and one group of seven Atlantic spotted dolphins (*Stenella frontalis*) were also seen during pre-detonation monitoring. MART observers also identified one group of two false killer whales (*Pseudorca crassidens*) that approached the vessel and one group of 13 pygmy killer whales (*Feresa attenuata*) that approached the vessel and bow rode for several minutes allowing for positive identification. There were some sightings of unidentified odontocetes from the MART vessel. Most were not identified due to distance, weather conditions, or inability

Table 1

Summary of marine mammal and sea turtle sightings and observer effort (h) during site selection, pre- and post-detonation monitoring, Spring 2001.

	Monitoring platform					
	Site selection	Pre-detonation aerial	Pre-detonation *Winston S. Churchill*	Pre-detonation MART	Post-detonation aerial	Post-detonation MART
Total observation time (h)	54	45	24	28	59	125
Species						
Odontocetes						
Tursiops sp.	38	109	17	180	292	75
Short-finned pilot whale	95	62	58	72	39	38
Risso's dolphin	25	46	39	0	95	23
Stenella sp.	24	15	5	7	72	0
False killer whale	0	0	0	2	0	0
Pygmy killer whale	0	0	0	13	0	0
Unidentified dolphin	49	462	81	34	131	2
Total odontocetes	231	694	200	308	629	138
Sea turtles						
Leatherback turtle	0	0	0	0	1	1
Loggerhead turtle	30	17	2	3	36	0
Unidentified turtle	37	7	1	2	4	0
Total sea turtles	67	24	3	5	41	1
Total animals	298	718	203	313	670	139

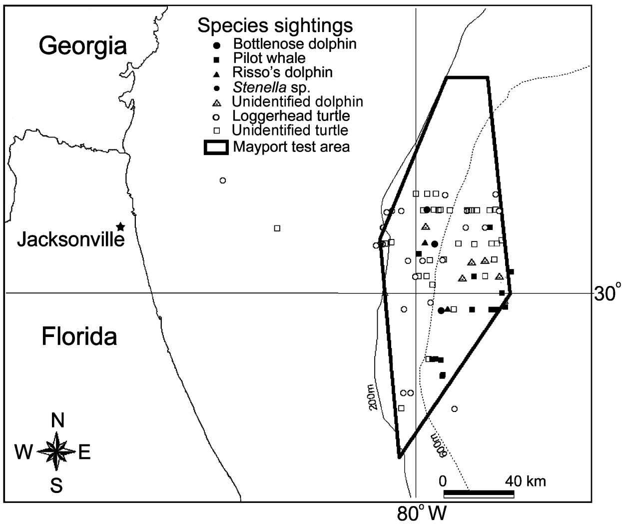

Fig. 2. Marine mammal and sea turtle sightings collected during site selection aerial surveys in support of shock trial environmental mitigation, Spring 2001.

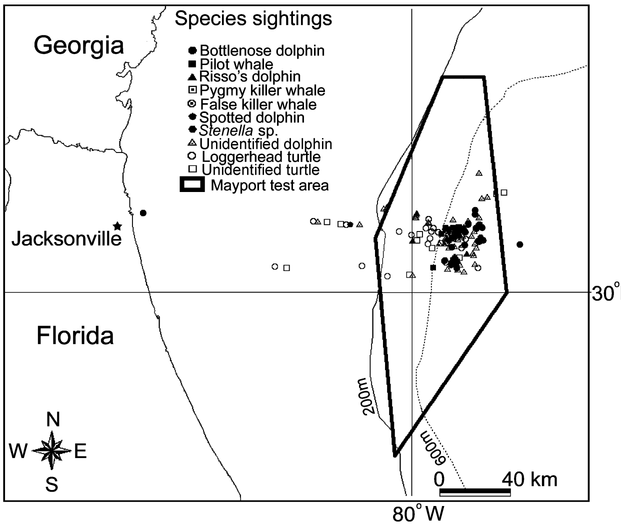

Fig. 3. Marine mammal and sea turtle sightings collected during pre-detonation aerial and vessel monitoring in support of shock trial environmental mitigation, Spring 2001.

to investigate due to monitoring protocol. Over half (159 marine mammals) of the sightings were made on 31 May.

Pre-detonation bio-acoustic monitoring on test days totalled approximately 14.5 hours. Bio-acoustic monitoring was in place for tests occurring on 24 May and 11 June; monitoring was also present on 10 May when the mis-fire occurred, and on 31 May when the test was postponed due to the high occurrence of animals within the Safety Range. Bio-acoustic monitoring was not present on 3 June when the second test occurred, due to aircraft mechanical problems. A total of 68 DIFAR buoys were deployed during acoustic monitoring. On 24 May, the initial sonobuoy field was deployed too far north and east to be useful so a second field of buoys was deployed south and west that would drift towards the planned detonation site (Fig. 4). The sonobuoy field on 11 June was initially positioned somewhat east of the planned detonation site, so an additional line of buoys was deployed farther west to improve detection and localisation capabilities. No low-frequency marine mammal calls were detected at any time (Department of the Navy, 2002). Dolphin calls were detected five times over a 1 hour, 12 minute time period on 31 May only. These calls could not be localised and were only identified based on the buoy that they were closest to.

Post-detonation aerial monitoring totalled 59 hours, and no dead or injured marine mammals or sea turtles were detected at any time. A total of 629 marine mammals and 41 sea turtles were observed during post-detonation aerial monitoring (Table 1, Fig. 5). The emphasis for post-detonation aerial monitoring was to cover as much area as possible searching for marine mammals and sea turtles that appeared dead or injured, so the data collected with each live animal sighting were minimal. The same species of marine mammals and sea turtles identified during site selection and pre-detonation monitoring were identified during post-detonation aerial monitoring.

The MART was on-site during and after the three tests continuously for a total of 14 days and approximately 125 hours: 24-26 May, 3-5 June and 11-18 June. No dead or injured marine mammals or sea turtles were seen by MART observers either at the site of each test nor down current during subsequent monitoring. A total of 138 marine mammals and one sea turtle was recorded from the MART vessel, representing four species (Table 1, Fig. 5). The most commonly encountered species were bottlenose dolphins. One large (35-45) group of pilot whales was also seen as well as two groups of Risso's dolphins.

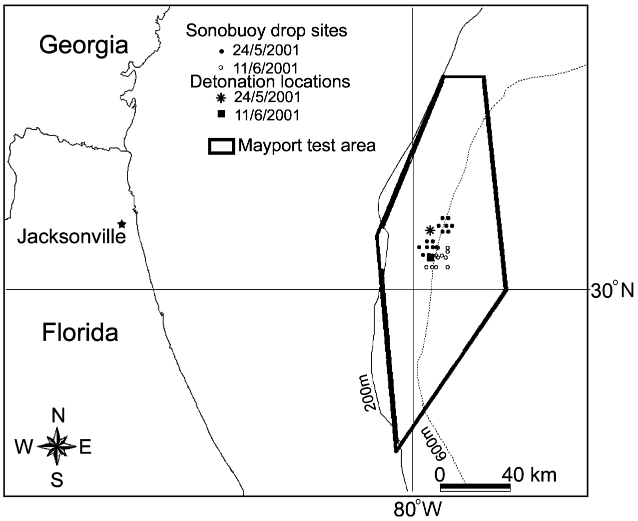

Fig. 4. Pre-detonation DIFAR sonobuoy drop locations in relation to test locations on 24 May and 11 June 2001, in support of shock trial environmental mitigation.

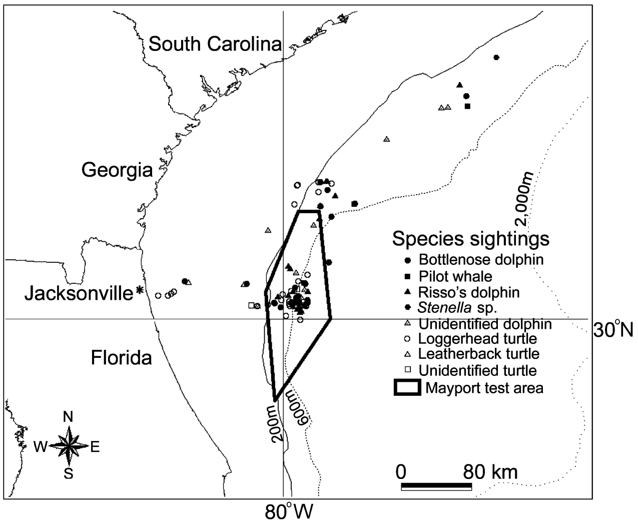

Fig. 5. Marine mammal and sea turtle sightings collected during post-detonation aerial and vessel monitoring in support of shock trial environmental mitigation, Spring 2001.

DISCUSSION

Environmental mitigation was designed to lesson the impact(s) of the shock trial on marine mammals and sea turtles (Department of the Navy, 2001; Reeves and Brown, 1994). The primary objective was to ensure, to the best of our ability, that there were no detectable marine mammals or sea turtles within the Safety Range (3.7km radius) and thereby prevent death or injury. Mitigation objectives did not include research or data collection for any purpose other than to keep the Safety Range clear of detectable marine mammals and sea turtles. Consequently, several sightings were not identified to species because the emphasis was on detection rather than identification. Group size was estimated only for purposes of re-identifying the same group to confirm it was clear of the Safety Range. Line transect methodology was not used during most monitoring surveys, as the aircraft was often re-directed to confirm that sightings were out of the Safety Range. The resulting data collected during shock trial mitigation are not equivalent to data collected during research surveys, and cannot be used for density or abundance calculations. Despite these limitations, environmental mitigation of the *Winston S. Churchill* ship shock trial represents one of the most intensive monitoring efforts for this geographic area.

Sightings

The species sighted are in agreement with those species sighted in this area during previous survey efforts that took place during the same time of year (Department of the Navy, 1995; Department of the Navy, 1998; Department of the Navy, 1999), with the exception of sperm whales and rough-toothed dolphins (*Steno bredanensis*), which were not seen during the shock trial. Bottlenose dolphins are often found along the continental shelf break (waters >25m), extending into continental slope waters (Kenney, 1990). They were also the most commonly sighted species (number of groups) during a ship-based, line-transect survey conducted between Maryland and central Florida to the boundary of the US Exclusive Economic Zone (EEZ) (Mullin and Fulling, 2003). Sightings of pilot whales within the US Atlantic EEZ are usually associated with the Gulf Stream (Waring *et al.*, 2001), so their presence in the test area was not unexpected. Risso's dolphins are a widely distributed, cosmopolitan species inhabiting deep pelagic and continental slope waters throughout tropical and temperate regions. They occur along the Atlantic coast of North America, therefore their presence in the Gulf Stream region was also expected. They have been sighted associated with the Gulf Stream in the northeastern US along the continental shelf (Waring *et al.*, 1992). Spotted dolphins are commonly found along the southeastern US (Waring *et al.*, 2000). Pygmy killer whales are distributed in tropical and subtropical waters worldwide (Ross and Leatherwood, 1994), overlapping in range with false killer whales, which are distributed in tropical and warm-temperate oceans (Odell and McClune, 1999). Nearly all species observed prior to shock trial tests during site selection and pre-detonation monitoring were also sighted after the tests occurred (see Table 1).

Mitigation effectiveness and evaluation

Overall, the environmental mitigation effort for the shock trial was effective, since no dead or injured marine mammals or sea turtles were detected after the detonations, despite several days of dedicated searching. The success of this mitigation effort emphasises the need for future shock trials and other activities of this size and scope to employ similar procedures. It is important to note that this mitigation was designed specifically for this shock trial, and would not necessarily be appropriate for other types of activities. Generally, individual activities each have unique factors (location, time of year, type of activity, potential effects) that need to be addressed, and environmental mitigation should be specifically planned and implemented separately for each activity.

Nonetheless, some elements of this mitigation effort would be essential for nearly all marine mammal and sea turtle mitigation activities. For example, the mitigation team leader should be familiar both with marine mammal and sea turtle survey techniques, as well as with the activity to be mitigated, to be able to quickly and appropriately respond to unforeseen situations. Experienced and trained observers are crucial for detecting and tracking marine mammals in the Safety Range; if those animals had not been detected, they likely would have been killed or mortally injured. Animals killed or injured as a result of the tests likely would have resulted in delays and postponements of the shock trial effort, at significant additional cost. Monitoring the Safety Range absolutely requires aerial and vessel observers working in tandem. Aerial coverage provides the means to monitor fairly large areas in a short amount of time, as well as the means to quickly investigate possible sightings at the outer limits of the Safety Range. Vessel observers are able to monitor all of the Safety Range from their central position near the detonation site, and track mammals and turtles as they leave the area. Incorporating only one of these platforms would not provide the coverage needed for adequate monitoring of a test of this magnitude. Cooperation, assistance and support from the shock trial team are also essential to the successful implementation of environmental mitigation, and allowed for smooth handling of the small problems that inevitably arose. Flexibility is critical in mitigation planning and implementation. All participating organisations, including the shock trial team, mitigation team, ship's force and sponsoring and regulatory agencies, must be keenly aware that adjustments to plans may be required as the trial proceeds, and be willing to work closely and effectively to expedite any changes. The mitigation of the *Winston S. Churchill* shock trial was greatly enhanced by including a NMFS representative within the on-site monitoring team. As unforeseen circumstances presented themselves (e.g. bio-acoustic monitoring system unavailable; redesigning pre-detonation aerial monitoring to account for Gulf Stream currents), they were immediately discussed and alternative courses of action quickly implemented.

Despite the success of the mitigation effort, there are some elements of mitigation planning and implementation that should potentially be reassessed as to their usefulness and effectiveness.

(1) Bio-acoustic monitoring of the shock trial was the most expensive mitigation component to design and implement, however no large whales (e.g. mysticete or sperm whale) were heard or seen during the entire shock trial period. Tests that are conducted during time periods and in geographic areas where, based on previously collected data, large whales are not likely to occur probably do not require bio-acoustic monitoring. Bio-acoustic monitoring that focuses on detecting marine mammal species that are difficult to detect visually (e.g. cryptic species, deep divers) would likely be more useful, but is logistically and methodologically difficult to implement (Barlow *et al.*, 1997). Therefore, bio-acoustic monitoring should be reconsidered as to need, purpose, cost and benefit.

(2) Pre-detonation aerial monitoring was possible with a single aircraft, but a second aircraft on-site would have improved overall coverage. With two aircraft available for monitoring immediately prior to the planned detonation, monitoring of both the detonation site and the outer areas of the Safety Range would have been more effective. This would likely have reduced the number of test delays as well as provided better coverage of the area, reducing potential risks even further. Two survey aircraft in a fairly confined airspace (~25km^2) would require greatly improved air traffic control to ensure flight safety. Flight safety risks could be significantly reduced with the addition of detailed briefings of all shock trial aerial support (mitigation and operational) conducted immediately prior to each test.

(3) Operational requirements will probably dictate that future shock trials should continue to be conducted in the Gulf Stream. Post-detonation aerial monitoring should be increased (e.g. more survey days and more hours surveyed per day) because it is the main search platform for detecting dead or injured animals, which can move rapidly with the Gulf Stream current, and may not surface for a few days. If aerial post-detonation monitoring is increased, MART responsibilities could be limited to recovery of dead or injured animals only. If aerial post-detonation monitoring is not increased, the MART vessel should be better equipped for marine mammal and sea turtle observations (e.g. additional observers, *Bigeyes* binoculars, laptop computer with tracking program, etc) because without the aerial support, the MART becomes the principal post-detonation search platform. Weather and sea state conditions can hinder post-detonation aerial and vessel monitoring, which should be adjusted to ensure that monitoring is adequate enough to detect dead or injured animals.

(4) The EIS process included selecting the test area that was least likely to negatively impact marine mammals and sea turtles. The three test areas considered for the shock trial (Norfolk, VA; Mayport, FL and Pascagoula, MS) were evaluated based on the best available data (Department of the Navy, 2001), and the Mayport test area was eventually selected. However, marine mammal and sea turtle data for the three test areas were collected during survey efforts that were not similar in timing, design or scope. Marine mammal and sea turtle data collected prior to the planning process in all potential test areas during concurrent time periods and using the same methodology should provide better and more equal data. This would enhance the test area selection process, whereby marine mammal and sea turtle abundance in all potential test areas would be more easily evaluated and the test area least likely to be negatively impacted more definitively identified. Although dedicated surveys can be expensive, the costs incurred would be offset by more accurate and reliable data on abundance estimates, distribution, and seasonality of marine mammals and sea turtles in all proposed test areas. This would ultimately decrease the potential for impacts to these animals during the shock trial, as well as potentially decrease the number of costly shock trial delays and postponements caused by animals in the area.

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Relative abundance of harbour porpoises (*Phocoena phocoena*) from acoustic and visual surveys of the Baltic Sea and adjacent waters during 2001 and 2002

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ABSTRACT

Boat-based acoustic and visual surveys for harbour porpoises (*Phocoena phocoena*) were conducted during the summers of 2001 and 2002 in order to investigate their distribution and relative abundance in the Baltic Sea, and to compare the results with the adjacent Kiel and Mecklenburg Bights and the Little Belt. Harbour porpoises are subject to year-round bycatch in gillnets and other fishing gear in these waters. This is of particular concern in the Baltic Sea where a survey carried out in 1995 indicated that the population is low and current levels of anthropogenic mortality are believed to be unsustainable. Polish coastal waters were not included in the 1995 survey and it has been hypothesised that these unsurveyed waters may contain a significant uncounted part of the Baltic Sea population. Results show that the porpoise detection rate was two orders of magnitude lower in the Baltic Sea than in other waters surveyed. No evidence was found that Polish waters contain a significant, previously uncounted part of the Baltic Sea population. The results confirm the endangered status of the Baltic Sea population, and stress the urgency of preventing future anthropogenic mortalities that threaten the survival of the population.

KEYWORDS: HARBOUR PORPOISE; SURVEY–ACOUSTIC; INDEX OF ABUNDANCE; CONSERVATION; EUROPE; BALTIC SEA; ACOUSTICS

INTRODUCTION

Harbour porpoises (*Phocoena phocoena*) are subject to bycatch in gillnets and other fishing gear throughout their distribution range in the Northern Hemisphere. This has led to increased concern over the population of this species in recent years (e.g. Berggren, 1994; Perrin *et al.*, 1994; HELCOM, 1996; ICES, 1997; ASCOBANS, 2000; IWC, 2000). Several studies in European waters have shown that bycatch levels in gillnet fisheries may not be sustainable, e.g. in the Celtic Sea (Tregenza *et al.*, 1997), the central North Sea (Vinther, 1999), the Skagerrak and Kattegat Seas (Harwood *et al.*, 1999; Carlström, 2003) and the Baltic Sea (Berggren *et al.*, 2002).

This issue is of particular concern in the Baltic Sea[1], where action is urgently needed to reduce bycatch to conserve Europe's most threatened population of harbour porpoises (ASCOBANS, 2000; 2002). No independent, scientific observer programmes on board fishing-vessels to estimate bycatch have been conducted in the Baltic Sea, but estimated levels cannot be sustained indefinitely by the population (Berggren *et al.*, 2002). It is further known that bycatch in the Baltic Sea occurs year-round (Berggren, 1994).

Porpoises are believed to have been common in parts of the Baltic up until the late 19th and early 20th centuries, and were distributed all the way up into the Bothnian Sea (Berggren, 1995; Berggren and Arrhenius, 1995b; Berggren and Arrhenius, 1995a; Koschinski, 2002). However, in Swedish waters of the Baltic Sea, harbour porpoise abundance appears to have declined drastically between the 1960s and 1980s (Berggren and Arrhenius, 1995b) with no subsequent recovery (Berggren and Arrhenius, 1995a). Porpoises have also become less common during recent decades in other areas of the Baltic Sea, including Danish (Andersen, S.H., 1982) and Polish (Skora *et al.*, 1988) waters. Very occasional sightings and bycaught porpoises have been recorded in Finnish and Estonian waters (Määttänen, 1990; Mattsson, 1995).

Studies of skull morphology, mitochondrial DNA and contaminants show that the Baltic Sea population should be regarded as a separate management unit. Population-level differences have been found between harbour porpoises from the Baltic Sea, the Kiel and Mecklenburger Bights, and the North Sea (Andersen, L.W., 1993; Tiedemann *et al.*, 1996; Huggenberger *et al.*, 2002). In addition, differences have been found among the Baltic Sea, the Skagerrak/Kattegat Seas and the west coast of Norway (Börjesson and Berggren, 1997; Wang and Berggren, 1997; Berggren *et al.*, 1999) and between the Kattegat/Danish Belt Seas and the Skagerrak Sea (Kinze, 1985; 1990; Andersen, L. *et al.*, 2001).

Current information on the number of porpoises in Danish, German, Swedish and international waters of the Baltic Sea (ICES – International Council for the Exploration of the Seas —rectangles 24 and 25) derives from an aerial survey conducted in 1995 (Hiby and Lovell, 1996). The abundance estimate for the area surveyed was 599 (CV=0.57) animals. Polish coastal waters were not included in the survey and it has been hypothesised that these waters may contain a significant uncounted part of the Baltic Sea population. This is based on information from incidental sightings and bycatch that has indicated that Puck Bay in the east of Poland may have a relatively high density of

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[1] By the Baltic Sea, we specifically refer to waters East of the Dars Sill. A bank running between Gedser, Denmark and Darsser Ort, Germany (Blocks 4 and 5 in Fig. 1).

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porpoises (ICES, 1997; Kuklik and Skóra, 2003). The Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS) Baltic Discussion Group (ASCOBANS, 2001) accepted the 1995 survey estimate, but noted that this was:

(a) downwardly biased, because it did not cover an area of Polish waters where harbour porpoises are known to occur; and
(b) an estimate with poor precision, due to low numbers of detected animals in the survey.

Further surveys were recommended to address these issues.

This paper presents the results of boat-based acoustic and visual surveys for porpoises carried out during the summers of 2001 and 2002. The primary aim of these surveys was to further investigate the distribution and relative abundance of porpoises in the Baltic, and particularly in 2001, to examine the hypothesis of a 'reservoir' of porpoises off the Polish coast. For these purposes acoustic and visual detection rates were compared between several survey blocks in the Baltic Sea and adjacent waters.

METHODS

Survey design

Five survey blocks were defined and covered during 2001 and 2002 (Fig. 1; Table 1). In 2001, survey transects were laid out only in Polish coastal waters (block 5). In 2002, the survey was expanded to include waters north up to the Swedish coast (block 4). Additionally in 2002, adjacent German and Danish waters to the west (the Little Belt and the Kiel and Mecklenburg Bights, blocks 1-3) were surveyed, in order to obtain data on relative abundance in areas suspected to have higher population densities of porpoises (Hammond *et al.*, 2002). To aid future data comparisons, these blocks corresponded to those used by other researchers conducting aerial surveys for harbour porpoises in 2002 (Scheidat *et al.*, 2004; Berggren *et al.*, 2004), with the exception that the most westerly aerial survey block was split into two (Little Belt and Kiel Bight). The reason for this was that the original aerial survey tracks frequently crossed land and would have been inefficient for boat-based surveying.

Survey lines (transects) were laid out systematically with random starting points, to provide non-zero, approximately even coverage within each block. The boat followed the planned transects as closely as possible, given the constraints of navigational safety and the need for at least 10m of water for the deployment of the hydrophone. Data collection continued even when the vessel was off track, but unless explicitly stated, only data collected on track are presented here. In 2002, survey effort alternated between blocks over the period of the survey in order to reduce any effects from seasonal changes in distribution. Individual transects were not surveyed in any particular order but were selected based on the requirements of port visits for crew changes and the weather conditions on any particular date.

Table 1

Dates of survey by block. Dates represent only the first and last day of data collection within each block. Survey effort alternated between blocks to avoid the risk of biasing the result due to seasonal movement between blocks. The actual survey effort within each block is given in Table 2.

Survey block	Start date	End date
Polish coast	19 Aug. 2001	15 Sept. 2001
Little Belt	8 Jun. 2002	7 Aug. 2002
Kiel Bight	10 Jun. 2002	6 Aug. 2002
Mecklenberg Bight	15 Jun. 2002	4 Aug. 2002
Baltic Sea	26 Jun. 2002	3 Aug. 2002

Data collection

The surveys were conducted from the 14m auxiliary powered sailing vessel *Song of the Whale*. The vessel was operated under engine power in low wind conditions and when visual surveys were taking place (to maintain an approximately constant survey speed and so that sails did not obstruct the forward view of the observers). When not surveying visually, the vessel was sailed whenever the desired course could be maintained at a survey speed of approximately six knots. Global positioning system (GPS) data (position, speed, course over ground) were logged automatically to a database every 10 seconds. Environmental data (wind speed and direction, water temperature and depth) were logged automatically every minute. Other data, which could not be collected automatically, were entered manually into the database every 30 minutes (wave height, sea state, weather, visibility), or whenever they changed (engine on/off).

The vessel was equipped with an automatic porpoise detection system (Gillespie and Chappell, 2002), which was developed to detect the high frequency sounds produced by

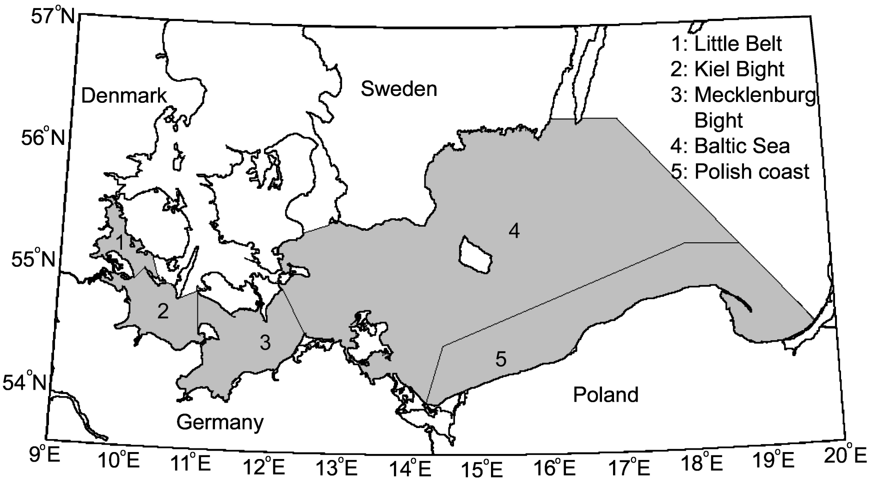

Fig. 1. Map showing the five survey blocks. Blocks 1-4 were surveyed in 2002, and block 5 in 2001. Note that block 4 extends right to the Polish coast and so block 5 is a sub-area of block 4.

harbour porpoises. It consisted of a two-element hydrophone towed 100m astern of the survey vessel. Analogue electronics modules split the signals from the hydrophones into different frequency bands and carried out envelope tracing to reduce the signal frequency. The signal envelopes were then digitised and analysed in real time for porpoise-like clicks using software running on a computer onboard the survey vessel.

During daylight hours (06:00 to 20:00) in clear weather with sea states of Beaufort two or less, two observers were stationed on an A-frame observation platform. This provided them with a clear view ahead, with an eye height of approximately 5.3m above sea level. The port side observer scanned from 270° to 15° and the starboard observer from 345° to 90° relative to the vessel's direction. Although observers only searched for porpoises ahead of the vessel, once spotted, they were tracked as far astern as possible to assist with linking sightings and acoustic detections in possible future dual visual-acoustic data analysis. Observers scanned with the naked eye and estimated ranges to sightings visually. Angle boards were used to measure bearings to sightings. Sightings were recorded on paper by a third person (so that the observers did not need to avert their eyes). Sightings data were transcribed into the database and automatically cross-referenced to the vessel's GPS co-ordinates.

Acoustic data analysis

A fully automatic algorithm to assign clicks to individual porpoises would be desirable but has not yet been developed. The acoustic data were therefore scanned by an analyst for trains of porpoise-like clicks, using the software described in Gillespie and Chappell (2002). The software has a screen display of the data showing amplitude, waveform envelopes and bearing information of the clicks over time. Sequences of clicks can also be played back through headphones. Individual clicks were classified as 'porpoise' if they had a minimum amplitude of 105dB re. 1μPa, and a signal strength in the 115-145kHz 'porpoise' band at least 25dB above the mean signal strength measured at two lower control frequencies. Click classification errors are discussed in Gillespie and Chappell (2002). The chosen analysis settings give a >50% correct classification for porpoise clicks and a low, but >0% false-positive rate.

For this analysis, click train selection was a two-stage process. In the first stage, the operator scanned files for sequences of porpoise clicks which were detected on both hydrophone elements and showed a clear change of bearing going from ahead to astern of the survey vessel. A subjective judgement was made based on the appearance of the track and the sound of the clicks (both porpoise and unclassified clicks) played back over headphones and the click trains labelled as 'likely' or 'possible'.

Porpoise clicks are highly directional (Au *et al.*, 1999) and it has been found that many porpoise events, as well as having a number of clearly identifiable porpoise clicks, also contain 'unclassified' clicks which have a lower amplitude and cannot be clearly identified as porpoise clicks purely from the signal amplitudes in the different frequency bands. These 'unclassified' clicks often lie on a bearing consistent with clicks, and from their regularity (apparent when they are played back through headphones) are clearly part of the porpoise click train.

In the second stage, a more objective classification was applied to the first-pass analysis. It was found that none of the click trains labelled as 'possible' contained more than six porpoise clicks, however, some click trains labelled as 'likely' had fewer than seven porpoise clicks. In order to keep the probability of false detections low, all click trains with fewer than seven porpoise clicks were discarded.

An example of a bearing-time plot for a porpoise detection is shown in Fig. 2. The track of a porpoise passing from ahead to astern of the survey vessel is clearly visible. Random non-porpoise clicks are also shown. The single 'porpoise' click off the main track is a typical false possible classification of a non-porpoise sound.

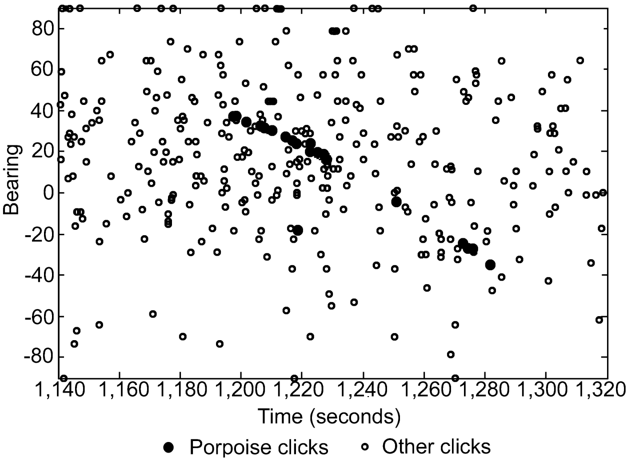

Fig. 2. An example bearing-time plot showing detections from an encounter with a porpoise. Porpoise clicks are characteristically narrowband in the 125-150kHz range while other clicks (open circles) are broadband. The detector found 28 clicks (filled circles) over a period of 80 seconds, as the porpoise passed from approximately 40° ahead to 40° astern. The 'porpoise' click below the main track in the figure is a false classification of another noise.

Statistical analysis

The variances in the number of detections, *n*, and the detection rate *n*/100km, were calculated using transects as sampling units (Buckland *et al.*, 2001, pp.78-80). The differences in the detection rates between the Baltic Sea block and the other three blocks to the west were compared using a randomisation test (10,000 re-samples). The standard error (SE) and a variance inflation factor $\hat{b}$ = var(n)/n (Buckland *et al.*, 2001) were also calculated for each block. The factor $\hat{b}$ measures the extent of clustering in the distribution of animals.

RESULTS

The total distance surveyed acoustically and visually in each survey block, the number of detections and detection rates for each block are shown in Table 2.

Fig. 3 shows the survey tracklines, the off-track survey route and the visual and acoustic porpoise detections along the Polish coast in 2001. Only one detection was made on-track in 2001, this was a single porpoise sighted northeast of the Polish port of Swinousjie. However, a single acoustic detection was also made while the vessel was off-track, less than 1km from the Polish coast, approximately 30km east of Swinousjie. These were the only detections in Polish waters during the 2001 and 2002 surveys.

Fig. 4 shows the acoustic survey tracklines and detections for 2002. The highest acoustic detection rate was in Danish waters in the Little Belt (16.8/100km). In broad terms, detection rates decreased from west to east dropping to 0.1/100km in the Baltic Sea. There were only three acoustic

Table 2

On-track survey effort and detections in 2001 and 2002. For acoustic surveys, the number of individuals detected (n), the detection rate (n/100 km) and the standard error se are shown. The symbol $\hat{b}$ denotes the ratio var(n)/n for the acoustic survey in each block. For visual surveys, the number of groups detected (n) and detection rate (n/100 km) is shown with number of individuals in parentheses.

			Acoustic survey					Visual survey		
Block	Year	Area (km^2)	km	n	n/100km	se (n/100 km)	$\hat{b}$	km	n	n/100km
Polish coast[1]	2001	17,000	1,692	0	0	0	-	292	1 (1)	0.34 (0.34)
Baltic Sea	2002	56,000	2,946	3	0.1	0.08	1.68	253	0	0
Mecklenburg Bight	2002	6,000	713	23	3.2	0.75	1.30	190	0	0
Kiel Bight	2002	3,200	494	52	10.5	1.96	1.96	97	1 (1)	1.03 (1.03)
Little Belt	2002	1,300	291	49	16.8	3.71	2.70	158	13 (18)	8.2 (11.4)

[1] In addition a single acoustic detection was made during 518km off-track survey route in 2001.

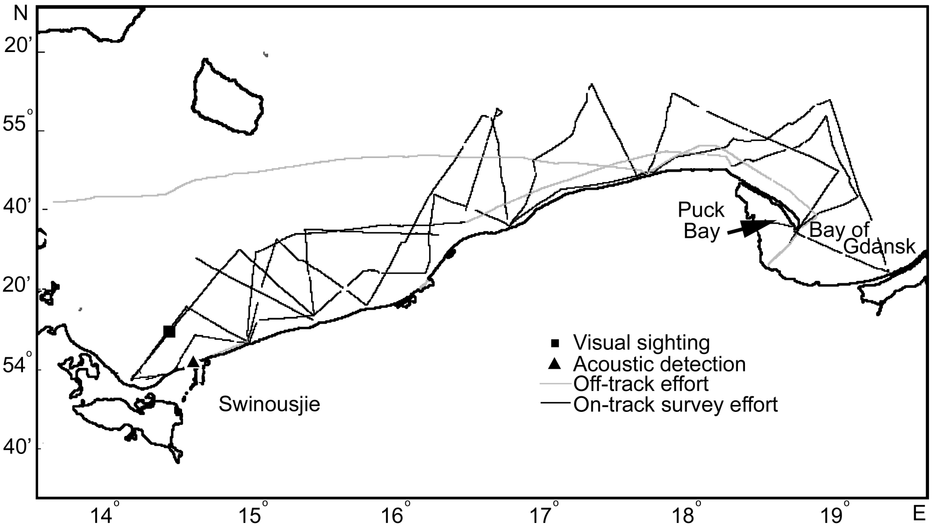

Fig. 3. Survey effort (and off-track effort) along the Polish coast in 2001. A single visual porpoise detection was made northeast of Swinousjie. An acoustic detection made off-effort approximately 1km from the coast is also indicated.

detections in the Baltic Sea block; two of these were close to the western edge of the block while the third was in the far northeast of the survey area in Swedish waters.

The visual on-track survey effort and sightings for 2002 are shown in Fig. 5. All sightings were made in the Little Belt (see Table 2). However, the visual on-track survey effort had limited coverage in all survey blocks limiting any useful comparisons between blocks. Eight single animals and five pairs were observed in the Little Belt, giving an average pod size of 1.4 animals. One of the pairs appeared to be a mother and calf.

The acoustic detection rate in the Baltic Sea block was one or two orders of magnitude lower than in the western blocks (Fig. 6). The vast majority of the visual and acoustic detections were made in the Little Belt and the Kiel and Mecklenburg Bights. Very few porpoises were detected acoustically in the southern part of the Mecklenburg Bight. The factor $\hat{b}$ was greater than one in all four blocks where detections were made while surveying (Table 2), indicating clustering in the distribution of porpoises. The randomisation test gave a probability $P(W>w)=0.0017$, where W is the random variable and w is the observed value. The difference in detection rate between the Baltic Sea and the other three blocks is therefore highly significant.

DISCUSSION

Distribution of porpoises

The pattern of acoustic detections indicates a gradient in the density of porpoises falling from west to east (Table 2, Fig. 4). Only one porpoise was detected while on-track (a sighting) in Polish waters during the survey conducted in 2001 (an additional acoustic detection was made off-track). We therefore conclude that Polish coastal waters do not contain a significant and uncounted part of the Baltic Sea population.

Apart from two porpoises detected at the extreme west of the Baltic Sea block, the only detection in the Baltic Sea in 2002 was in the extreme northeast of that block (see Fig. 4). The low porpoise detection rate in the survey of the entire Baltic Sea block agrees in a broad sense with the low density found in the 1995 aerial survey (599 porpoises in a 43,000km^2 study area; Hiby and Lovell, 1996) in international waters (this survey excluded the Polish coast). Furthermore, Berggren and Arrhenius (1995a) report only a single sighting in a five-year opportunistic Swedish sightings observer programme in the Baltic Sea.

Information from incidental sightings and bycatch (ICES, 1997; Kuklik and Skóra, 2003) has indicated that Puck Bay in the east of Poland (Fig. 3) may contain a relatively high

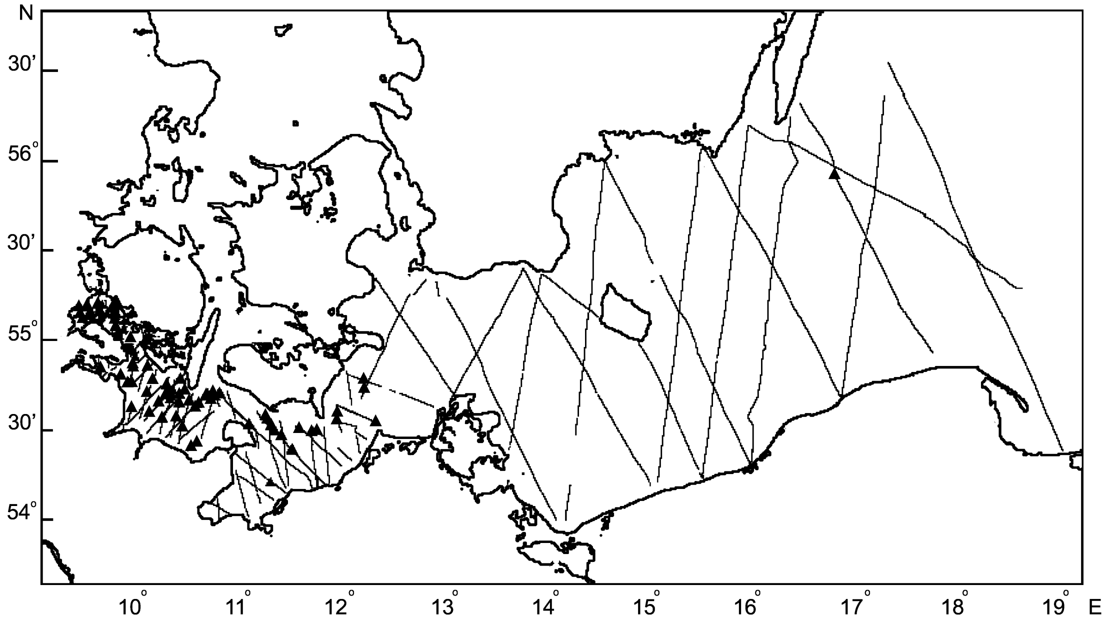

Fig. 4. Acoustic survey effort and detections (▲) in 2002. A total of 124 detections were made in the western blocks 1-3. There were only three detections in block 4 (Baltic Sea), two of which were in the extreme west and one in the northeast.

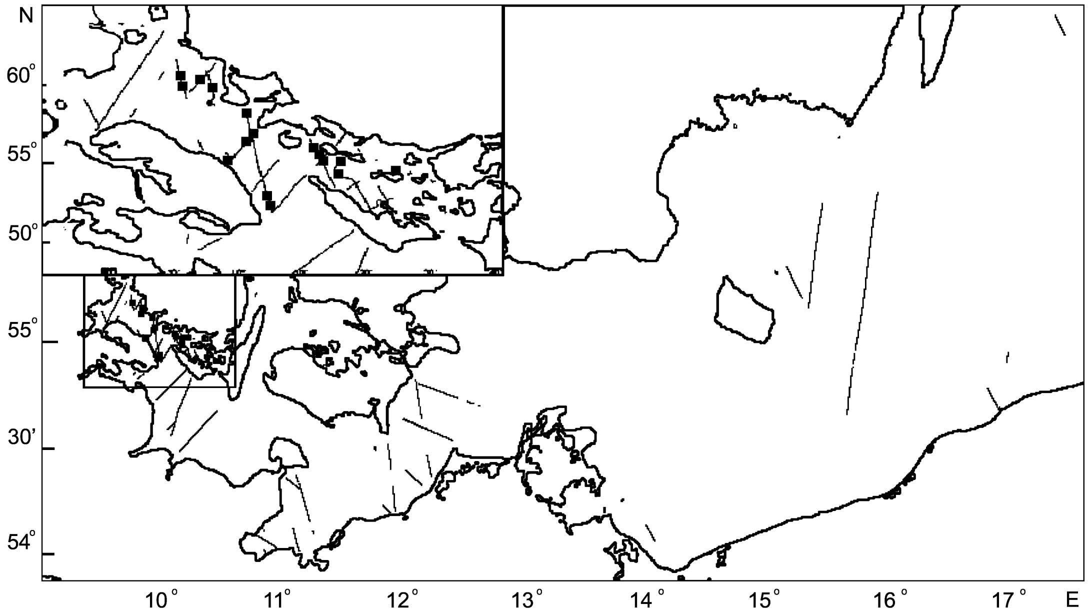

Fig. 5. Visual on-track survey effort and on-track sightings (|) in 2002. The inset is an enlarged view of the Little Belt region, where all sightings were made.

density of porpoises. However, it is possible that the relatively high occurrence of porpoise bycatch in Puck Bay is an effect of a very intense gillnet fishery in this area, rather than a higher density of porpoises. It is suggested that further research be carried out to clarify the cause of the high occurrence of bycatch in this area. Most bycatch is known to occur between December and April, although it has been reported in all other months except June (Kuklik and Skóra, 2003). Since this and other surveys (e.g. Hiby and Lovell, 1996) took place during the summer, we cannot rule out the possibility that there is seasonal movement in and out of the Baltic Sea.

Validity of survey results

An important assumption for a valid measure of relative abundance is that detectability is constant across the survey area (Pollock *et al.*, 2002). Acoustic detectability is a function of various measurable external variables or covariates (in particular ambient noise), and the vocal behaviour of porpoises. Noise level measurements recorded every second by the porpoise detection equipment showed that mean noise levels in the different blocks varied by less than 0.2dB. Acoustic cues from the survey vessel (sounds from the engines, propellers or depth sounder) could alert a porpoise to its presence, leading to changes in movement or vocal behaviour. The depth sounder was run continuously throughout the survey. The percentages of on-track survey effort with engine on were: Little Belt (83%), Kiel Bight (65%), Mecklenburg Bight (83%), Baltic Sea (50%). How engine noise affects porpoise behaviour is not known, but if detectability falls with engine use, due to directed motion of porpoises away from the survey vessel, then in this study the detection rate in the Baltic would have been positively biased. Conversely, if porpoises were attracted to the engine

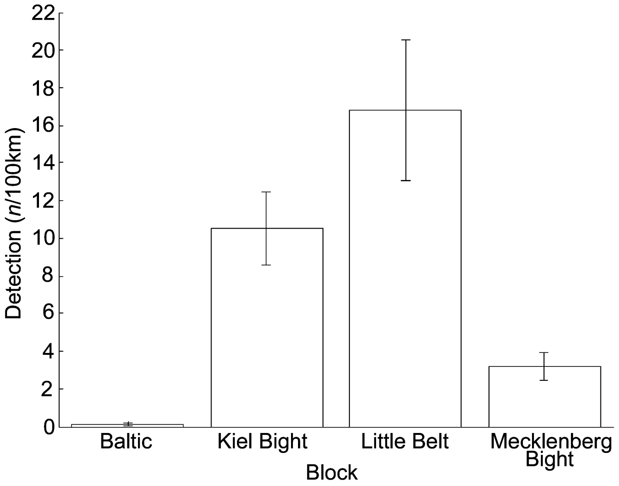

Fig. 6. Detection rates (*n*/100km) and their standard errors by survey block.

noise, the detection rate in the Baltic would have been negatively biased.

Another important underlying assumption for a measure of relative abundance is that the false detection rate is low, if not zero, compared to the detection rate. In this study, conditions were placed on click trains that would eliminate most false detections, but as a consequence some true porpoise detections may have been discarded. Ideally, both efficiency and false detection rates should be measured. Measuring efficiency is not possible with free ranging animals since even if it were known exactly how many animals were in the vicinity, their vocalisation rate would be unknown. Similarly, it is never possible to be sure that a detection is a false-positive, since it is impossible to be sure that no porpoise was there. This rate could potentially be measured by proxy using data collected in an area with similar levels of background noise, but known to be free of porpoises. No areas were visited during the study that met this requirement. However, the consistently low detection rate across the Baltic Sea block, and the similar measures of background noise between blocks indicates that the false detection rate was low.

Future work: estimation of absolute abundance

If acoustic detections are to be used for absolute abundance estimation, two major issues to consider are the estimation of $g(0)$ and the effects of responsive movement. Borchers (1999) described double-platform methods which, using the type of dual visual-acoustic data collected in this study, may allow the estimation of $g(0)$ when it is less than one. Furthermore, he outlined the use of the Buckland-Turnock approach (Buckland and Turnock, 1992), which is robust to responsive movement, using an observer team looking far enough ahead of the vessel to make visual detections before animals react.

Analysis of the dual visual-acoustic data from this study is underway and the effects of responsive movement are a major concern. Some data were collected using a second platform of observers searching ahead of the vessel from a crows nest at an eye height of approximately 10m above sea level. Unfortunately only eight of the crows nest sightings were at distances greater than 200m forward of the beam. The number of crows nest sightings is insufficient for a Buckland-Turnock analysis, and in any case the threshold distance, beyond which porpoises do not react to the survey vessel, is not known. Any comprehensive analysis of the data from this survey will therefore need to further consider the effects of responsive movement.

Conservation action

The results from this study confirm the limited occurrence and very low relative abundance of harbour porpoises in the Baltic Sea reported in the 1995 aerial survey. Further, the results do not support the existence of a porpoise 'reservoir' in Polish coastal waters. This further emphasises the endangered status of this population. Although it would be useful to conduct further surveys, priority should be given to reducing further anthropogenic mortalities, and hence to prevent extinction of the Baltic Sea population (e.g. see ASCOBANS, 2002). A number of factors may have contributed to the decline of the Baltic Sea population, including hunting, severe winters, pollutants and bycatch in fishing gear (ASCOBANS, 2002; Koschinski, 2002). Reducing bycatch in this region should be given high priority, because any is significant, relative to the low estimated abundance in the Baltic Sea, and bycatch is a form of anthropogenic mortality that can be mitigated immediately.

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Management of gillnet bycatch of cetaceans in New Zealand

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ABSTRACT

Bycatch of dolphins in inshore gillnets first attracted scientific and management attention in New Zealand in the 1980s. During 1984-88, 50-150 dusky dolphins were killed each year at Kaikoura in gillnets set at the surface to catch bait for rock lobster. At the same time, annual catches of 20-100 Hector's dolphins occurred in Canterbury waters in bottom-set commercial and recreational gillnets. These catches resulted in the banning of surface-set gillnetting at Kaikoura in 1989 and in the creation of the Banks Peninsula Marine Mammal Sanctuary in 1988 to protect Hector's dolphin. An additional gillnet closure was established to protect North Island Hector's dolphin in 2003. A key problem is that current information on catches in these and other areas is scant. One observer programme has been successfully implemented in a commercial gillnet fishery (Canterbury area, 1997/98 fishing season). Its estimate of Hector's dolphin bycatch (17) is clearly unsustainable by the local population. Pingers have been voluntarily used in these fisheries, but there are no data establishing their effectiveness, and it has not been possible to ensure consistency of pinger use. There are no reliable estimates of numbers taken in recreational fisheries. Area closures are used to mitigate gillnet bycatch of Hector's dolphin, however it appears that the Banks Peninsula Marine Mammal Sanctuary is not large enough to ensure the persistence of the Canterbury population. There is a bycatch limit in place for this population, although it is unenforced. We argue that management of this species via bycatch limits is not practical, however. Hector's dolphin's low abundance and separation into several distinct populations means that appropriate bycatch limits would be very small, and this necessitates very comprehensive observer coverage to be confident they are not exceeded. We propose that increasing the size of protected areas is the most reliable option for conservation.

KEYWORDS: CONSERVATION; FISHERIES; GILLNETS; INCIDENTAL CATCHES; REGULATIONS; SANCTUARIES; STATISTICS; SUSTAINABILITY; TRAWLS

INTRODUCTION

Incidental catch in gillnets is probably the most serious conservation issue facing small cetaceans (e.g. Bjørge *et al.*, 1994; Perrin *et al.*, 1994; Hall and Donovan, 2002). In some cases such bycatch endangers local populations (e.g. Martien *et al.*, 1999; Dawson *et al.*, 2001; Secchi and Wang, 2002) and species (Taylor and Rojas-Bracho, 1999). Few people outside the community of marine mammal scientists realise how effective gillnets are at catching small cetaceans. For example, in Peru a directed fishery has killed many thousands of dolphins annually for human consumption. No technique more effective has been found for this than gillnetting (Read *et al.*, 1988; Van Waerebeek *et al.*, 1999).

Three generic strategies are available to ensure that bycatch is sustainable. The most obvious strategy is to remove the relevant fishing gear (e.g. gillnets) from the area of interaction. Time/area closures, if properly designed and enforced, will eliminate bycatch in the closed area. Such measures are seldom popular with fishermen, however, and by displacing fishing effort from one area to another can act to move the entanglement problem rather than solve it (e.g. see Murray *et al.*, 2000). One beneficial side product of such a strategy is that gillnet-free areas established for marine mammals can have conservation benefits for other species, especially fish and seabirds (e.g. Darby and Dawson, 2000).

The second strategy involves modification of the fishing gear in order to reduce its likelihood of catching cetaceans. This may involve changes to the way the gear is rigged (e.g. Hembree and Harwood, 1987) or more recently the addition of acoustic pingers to nets to displace cetaceans from the area around the net, or warn them of its presence (e.g. Kraus *et al.*, 1997). Gear modification is appealing because fishing can continue, but requires detailed research to find effective measures, and long-term monitoring to ensure that gear modifications remain effective (Dawson *et al.*, 1998; IWC, 2000).

The third strategy involves setting a sustainable bycatch limit (e.g. 'Potential Biological Removal' or PBR; Wade, 1998). The fishery, if it exceeds that limit, is closed, or required to formulate a 'take reduction plan' to ensure that the limit is not exceeded in future. The PBR approach is used routinely by the National Marine Fisheries Service (NMFS) in the US. It is also used to manage trawl bycatch of New Zealand (NZ) sea lions (*Phocarctos hookeri*) in New Zealand, where its implementation has resulted in the early closure of the Auckland Islands trawl fishery for squid each year from 1996-2002[1], and has motivated the industry to explore ways of reducing bycatch. As in the gear modification strategy, high observer coverage is required to ensure that the number of incidental takes is known with reasonable precision.

The vast majority of gillnet vessels in New Zealand are small (96% were <15m in 1995, the most recent year for which data are published; Peacey, 1996). While gillnetting is used commercially to target a wide range of species, most gillnetting on open coasts is directed towards Chondrichthyan species, notably rig (*Mustelus lenticulatus*), school shark (*Galeorhinus galeus*) and elephant fish (*Callorhychus milii*). In 1995, these gillnet fisheries together contributed less than 1% of the primary value of all New

[1] The fishery was officially closed in each of the years. In several years the boats pulled out early, correctly anticipating closure (Childerhouse, pers. comm.).

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Zealand fisheries combined (Peacey, 1996). These species are targeted mostly in summer, in waters less than 100m deep.

In most countries, gillnets are used only by a licensed commercial fishing industry. In New Zealand, any member of the public may use a gillnet (gillnets used by recreational fishers are required to be <60m long). No licence is required and there are few detailed data on recreational gillnetting practices, or on the amount and distribution of effort.

Given that New Zealand is seen in some quarters as an example to follow in terms of fisheries management (e.g. Batkin, 1996), it seems appropriate to evaluate its management of the bycatch of cetaceans in gillnets.

INCIDENTAL CAPTURE OF CETACEANS IN NEW ZEALAND

Four small cetacean species are regularly seen off New Zealand coasts. They are Hector's dolphin (*Cephalorhynchus hectori*), dusky dolphin (*Lagenorhynchus obscurus*), common dolphin (*Delphinus delphis*) and bottlenose dolphin (*Tursiops truncatus*).

Common and bottlenose dolphins

The distribution of common and bottlenose dolphins has little overlap with intensive gillnetting, and therefore they may largely avoid gillnet entanglement. However, there are no observer programmes or other systematic attempts to estimate the level of bycatch for common and bottlenose dolphins.

Dusky dolphin

From specimens that were submitted for dissection, and from his discussions with fishermen, Cipriano (pers. comm.) estimated that 50-150 dusky dolphins were killed at Kaikoura each year between 1984-88. The highest mortality was from gillnets set at the surface to catch kahawai (*Arripis trutta*) to bait pots for rock lobster (*Jasus edwardsii*). Fishermen spoke of 'disaster sets' in which 20 or more dolphins were killed in one net.

Dusky dolphins are occasionally caught in commercial bottom-set gillnets set for tarakihi (*Nemadactylus macropterus*), rig (*Mustelus lenticulatus*), ling (*Genypterus blacodes*) and groper (*Polyprion oxygeneious*) at Kaikoura. This netting occurs inside and along the edge of the continental shelf, in waters 100-500m deep, typically within 500m to 18km of shore. Dusky dolphins feed mostly at night, beyond the shelf edge, on animals associated with the deep scattering layer. During the day they mainly rest and socialise inshore (Würsig *et al.*, 1997). Thus, during the day, when they are near the nets on the shelf edge they are probably not diving deep, and are hence not so vulnerable to nets set on the bottom. Local gillnetters believe that setting and hauling operations appear to pose the greatest risk (D. Burkhart, G. Melville, pers. comm.). With the removal of part-timers from the fishery when the Quota Management System was established in 1986 (Clark *et al.*, 1988), gillnetting practices appear to have improved, and reports of dusky dolphin catches have dropped substantially. However, in the absence of an observer programme or other systematic attempts to assess the level of bycatch, neither the magnitude of catches nor their impact on the population is known.

Hector's dolphin

Due to its close inshore distribution, Hector's dolphin has the greatest spatial overlap with gillnetting grounds and is known to be caught throughout its range (Dawson *et al.*, 2001). Despite this, serious attempts to quantify numbers caught have been made only in the Canterbury area. An interview programme established that commercial and recreational fishing killed some 20-100 per year in this area during 1984-1988 (Dawson, 1991). These catches, along with population viability analyses (Slooten and Lad, 1991) and studies of survival rate (Slooten *et al.*, 1992) contributed to the establishment of the Banks Peninsula Marine Mammal Sanctuary in 1988 (see Dawson and Slooten, 1993 for details).

Research since the creation of the sanctuary has confirmed its necessity. Pichler and Baker (2000) found a significant loss of genetic diversity in Canterbury from nine historical lineages, to five current. The timing of the loss matched closely with the introduction of mechanised gillnet fishing, and the resultant high dolphin bycatch (Dawson, 1991; Pichler and Baker, 2000). Additionally, Martien *et al.*'s (1999) modelling, based on gillnet effort and a catch rate determined from Dawson's (1991) interviews, indicate that the number of Hector's dolphins on the east coast of South Island has probably been in decline since the late 1970s when gillnetting became widespread. Using the most conservative estimates for maximum population growth, the model estimates that the population in 1984 was about half its size in 1970.

An observer programme (Starr and Langley, 2000) in 1997/98 off the Canterbury coast observed 6 mortalities in 214 gillnet sets, and 1 mortality in 434 trawl shots (Baird and Bradford, 2000). When stratified by area and season, observed gillnet bycatches extrapolate to an estimated total of 17 (ignoring setnet effort for spiny dogfish; Starr, 2000). Since commercial gillnetting is now illegal within the sanctuary, these catches are distributed to the north, south and offshore of it. Recent line-transect surveys indicate a total population of 1,198 in this region (Motunau to Timaru, CV= 27%; Dawson *et al.*, 2004). It can be argued that since the role of the sanctuary is to protect the dolphins within it, those animals should be omitted from any calculation of what might be an allowable bycatch. However, even if included, the US PBR model only yields an 'allowable' bycatch for this area of about two dolphins per year[2]. The estimated mortality in 1997/98 is more than eight times this figure. Due to paucity of data and low observer coverage, Baird and Bradford (2000) did not attempt to estimate numbers taken in trawls. The total trawling effort in Canterbury inshore waters was about 14,900 shots in 1997/98 (data from Catch Effort Landing Returns data only, areas 20 and 22; Baird and Bradford, 2000). It is clear that even low capture rates could result in a significant number of captures. Trawling is permitted within the sanctuary.

Frequent catches of Hector's dolphins occur in gillnets set off the west coast of South Island. The commercial gillnetting fleet here is small, but is supplemented by significant amateur fishing on certain parts of the coast. For example, off Ngakawau (25km north of Westport) local fishermen set nets attached to stakes permanently driven into the sand. The catch is cleared at low tide, and in some cases traded. It is illegal to set nets from stakes, or in such a way that fish can be stranded by the falling tide, and non-

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[2] Using the NMFS system, for a dolphin listed as endangered, results in an allowable bycatch of 0.2% of the lower 60% confidence interval of abundance.

commercial fishermen are prohibited from trading their catch. Ministry of Fisheries officials have met with fishermen to clarify these points, but there has been very little action to ensure compliance, despite this longstanding problem. Hector's dolphin densities in this area are high (Slooten *et al.*, 2004) and several net-marked dolphins are found beachcast each year (Neale, pers. comm.). There are no formal estimates of bycatch for this fishery, or any other west coast gillnet fishery.

The population of Hector's dolphin found off the west coast of North Island (Dawson *et al.*, 2001) is the most at risk of extinction. This population, now recognised as a separate subspecies (and renamed Maui's dolphin; Baker *et al.*, 2002) is very small (population estimate=111, CV=44%; Slooten *et al.*, Submitted), genetically distinct from South Island populations (Pichler *et al.*, 1998) and occurs within a range that, according to sightings by researchers and the public, appears to have shrunk since 1990 (Russell, 1999). Both population viability analyses (Martien *et al.*, 1999) and analyses of genetic diversity (Pichler and Baker, 2000) indicate that the population is declining. Beachcast, net-marked carcases provide direct evidence of gillnet bycatch, which has been confirmed in recent interviews of fishermen (Sylvester, pers. comm.). While other impacts (e.g. trawling, pollution) may contribute, gillnet bycatch alone is sufficient to explain the decline. Continued gillnetting at recent levels is likely to result in the extinction of North Island Hector's dolphins within decades (Martien *et al.*, 1999).

MANAGEMENT OPTIONS AND THEIR EFFECTIVENESS

Establishment of Banks Peninsula Marine Mammal Sanctuary for Hector's dolphin

As mentioned above, the Banks Peninsula Marine Mammal Sanctuary was established in 1988 in an attempt to reduce the impact of gillnetting on Hector's dolphins. The sanctuary is an 1,170km^2 area, in which commercial gillnetting is illegal and amateur gillnetting is restricted. Amateur fishermen may still use gillnets, but only outside the summer months (November-February). Unattended gillnetting is permitted only for flounder, in specially designated areas in the innermost parts of the Peninsula's four largest harbours. Elsewhere within the sanctuary amateur fishermen must stay with their nets.

While it attracted significant controversy when first established, the sanctuary is now socially well accepted and has resulted in net financial benefits to the region. There are no data to test whether recreational fishing has improved, although this is sometimes stated. Hector's dolphins are now the focus of several dolphin-watching and dolphin-swimming businesses. When the sanctuary was first created it was estimated that its annual economic impact on commercial gillnet fishers was $NZ550,000 (Department of Conservation and Ministry of Fisheries, 1994). The largest of the dolphin-watching companies now turns over more than twice this figure annually (Bingham, pers. comm.).

While clearly a step in the right direction, the sanctuary is not enough to ensure the sustainability of the Canterbury Hector's dolphin population. Mark-recapture analysis of photo-ID data gathered before and after the sanctuary's creation show no trend of increased survival rates (Cameron *et al.*, 1999; DuFresne, 2004). A stochastic, age-structured population model (Slooten *et al.*, 2000) indicates that there is a 94% chance that this population is still in decline. This is most likely due to bycatch of animals outside the sanctuary's protection.

Incidental capture still occurs immediately north, south and offshore of the sanctuary in recreational and commercial gillnets, and there are occasional catches inside the sanctuary. The 1997/98 observer programme provides the best estimate (17) of current catch in commercial gillnets (Baird and Bradford, 2000; Starr, 2000). In the 2000/2001 summer, five dead dolphins bearing gillnet (4), or rope marks (1) were found beachcast along the beach of Pegasus Bay (4) or on the north side of Banks Peninsula (1). These mortalities were thought to be caused by amateur gillnetters (Department of Conservation and Ministry of Fisheries, 2001). A further dolphin was recovered from an amateur gillnet set in Pegasus Bay. It is likely that combined commercial and recreational gillnet bycatch for Canterbury is at least 15-30 animals per year. The lower end of this estimate is seven times greater than would be allowed using the PBR approach (Wade, 1998).

In May 2002, the Minister of Fisheries responded by extending the ban on recreational gillnetting from 1 October to 31 March and has extended the geographic boundaries north to the Waiau River (42°46.8'S, 173°22.4'E) and south to the Waitaki River (44°56.5'S, 171°08.5'E). The Minister also established a bycatch limit of three Hector's dolphins for this area, warning that this area would be closed to all gillnet fishing for the remainder of the year if the limit was exceeded (Hodgson, 2002). However, there is no formal bycatch monitoring for either recreational or commercial fishing.

North Island Hector's dolphin

In August 2001, the Minister of Fisheries closed a substantial section of the North Island west coast to gillnet fishing, to protect the critically endangered population of North Island Hector's dolphin (Maui's dolphin). This decision was successfully appealed by the fishing industry, and the ban on commercial fishing was lifted. However, after extensive discussions with stakeholders, in January 2003, the Minister again decided that closing a large area to gillnetting was the only option likely to reduce takes to sustainable levels. The protected area extends 210 n.miles (390km) alongshore from Maunganui Bluff to Pariokariwa Point (Fig. 1), to 4 n.miles offshore, and includes a small part of one of the harbours in which Hector's dolphins have been sighted. All fishing methods other than gillnetting (both commercial and recreational) are permitted in the area. The Minister is also considering placing observers on trawling and Danish seining vessels working off this coastline, to assess the entanglement risk posed by these two fishing practices. Planned research will address the proportion of time Hector's dolphins spend in west coast harbours, where gillnet fishing effort is high.

Use of pingers to reduce Hector's dolphin bycatch in Canterbury

In light of studies showing that pingers reduce entanglement rates of harbour porpoises (*Phocoena phocoena*) in New Hampshire gillnets (Kraus *et al.*, 1997), and that the mechanism appears to be avoidance of the ensonifed area (e.g. Gearin *et al.*, 2000), Stone *et al.* (1997) investigated the responses of Hector's dolphins to pingers. Hilltop observers documented surfacing positions in the vicinity of a moored pinger which was activated remotely without observers knowing. The study reported a statistically significant difference in dolphin distribution, and provided the foundation for the introduction of pingers by Canterbury gillnetters. As in several similar studies, however, the data

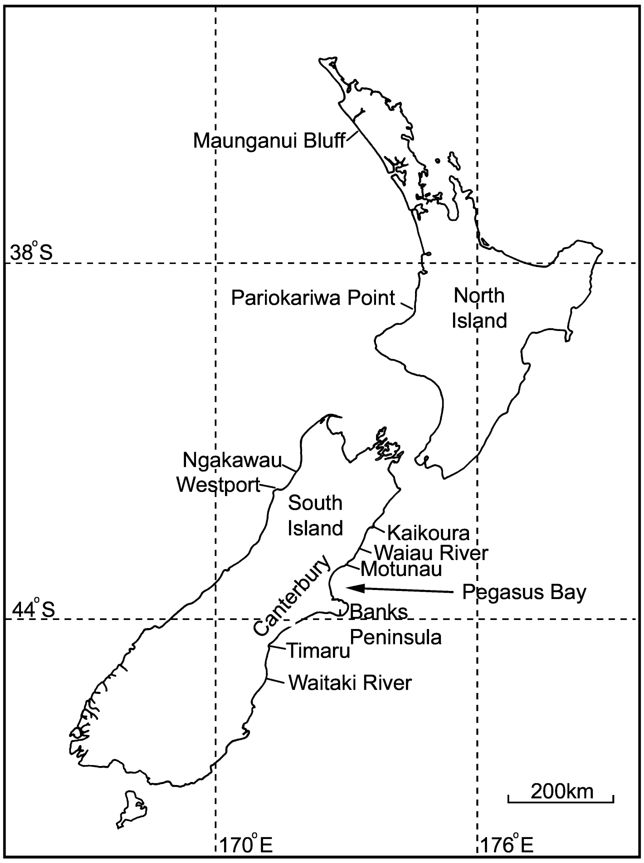

Fig. 1. Map showing place names mentioned in the text. The area closed to gillnetting from January 2003 is from Maunganui Bluff to Pariokariwa Point.

analysis suffered from pseudo-replication and the statistical result is unreliable (Dawson and Lusseau, 2005). Plots of surfacing positions (figs 3 and 4 in Stone *et al.*, 1997) show no clear avoidance of the pingers.

Nevertheless, Canterbury fishermen voluntarily use pingers under a 'Code of Practice' (Southeast Finfish Management Company, 2000) which, in addition to pinger use, encourages the setting of nets with the tide and the avoidance of setting nets in depths of less than 30m or when dolphins are around the vessel; it also advises on what might reasonably be considered best practice. In addition, some gillnetters have voluntarily shifted their fishing operations away from areas with high densities of Hector's dolphins.

It has been difficult, however, to ensure that pingers are used as required. While most of the skippers in the Canterbury gillnet fleet (Motunau to Timaru) have been co-operative, one refused to carry observers. Another insisted that it was dangerous for his crew member to attach pingers to the net as it is set. Since he believed that setting and hauling operations pose the greatest risk, he dangled pingers from his boat during these times. His nets, when set, were unalarmed. Of the 68 gillnet sets observed in Canterbury in 1999/2000, only 28% complied with the COP instructions for pinger deployment (Blezzard, pers. comm.). It is in the nature of fishermen to vary practices to find what seems the best solution, but this can mean that it is difficult to ensure effective use by everyone.

Without observers it is impossible to gain reliable data on entanglement and the effectiveness of pingers and other management measures. The organisers of the first observer programme in this fishery (1997/98) found it extremely difficult to gain a fleet coverage of 46% and 39% in the two statistical areas of primary interest (areas 20 and 22 respectively, Baird and Bradford, 2000; Starr, pers. comm.). Subsequently, coverage has been about 10% and 20% respectively in 1999/2000, approximately a third of that in 2000/2001 and there has been no coverage since this time.

It is impossible to say whether pingers are effective in reducing entanglements of Hector's dolphin, for two reasons. Firstly, because pingers are used in combination with several other measures intended to reduce entanglement rate, their effect (if any) is hidden. Secondly, there has been insufficient observer coverage to determine whether these measures, even in combination, are effective. In the 1997/98 fishing year, an observer programme detected 6 mortalities in gillnets from 214 sets (Baird and Bradford, 2000). Observer effort since then has been minimal (see above). One entanglement was observed in the 1999/2000 season, and the animal released alive. If we assume the simplest case of no area or season effects, bootstrapping can be used to judge the likelihood that the zero observed mortality in 1999-2001 is due to chance alone. Even when the two years with observer coverage are combined, there is a 14% chance that zero bycatches could be seen if the true bycatch rate is 6/214. Using power analysis the number of observed sets needed to detect any given reduction in this rate can be estimated (Dawson *et al.*, 1998). If a target of 80% reduction in bycatch is set and a nominal value of α=0.10 is accepted, it would take approximately 320 observed sets to detect a significant difference with 80% power.

As the power analysis shows, more observer coverage is needed to determine whether pingers significantly reduce incidental capture. Given that fishermen have implemented several changes simultaneously, it is difficult to determine the effectiveness of pingers. This unfortunately limits the guidance that can be offered to other fisheries which entangle Hector's dolphins.

Restrictions on surface set gillnetting at Kaikoura

With the aim of reducing bycatch of dusky dolphins, legislation now restricts the net height of gillnets set at the surface, and fishers must stay within 100m of the net. Possibly as a result of this rule, Kaikoura fishers appear to have stopped setting gillnets at the surface. Elsewhere in New Zealand, surface driftnets up to 1,000m are legal in the exclusive economic zone, but are rarely used.

FUTURE MANAGEMENT ACTIONS

Further closed areas

So far, two Hector's dolphin populations have been protected. One population is comparatively large (the Banks Peninsula population), while the other (North Island population) is very small and considered critically endangered. It makes sense to use a mixed conservation strategy, including large populations that provide good 'insurance' for the species' persistence, as well as preserving small, highly threatened populations. In this context it can be noted that gillnet fishing is used throughout the geographic range of Hector's dolphin, and the level of bycatch appears unsustainable for at least 10 of 16 local population subunits (Martien *et al.*, 1999).

To help mitigate mortality of New Zealand sea lions in the trawl fishery for arrow squid on the Auckland islands shelf, a Marine Mammal Sanctuary excludes all fishing within 12 n.miles of the shore (Slooten and Dawson, 1995). In 2003, this area was reclassified as a no-take marine reserve. No protected areas have yet been created for other marine mammal species in New Zealand waters.

Limits on bycatch

An annual bycatch limit of three Hector's dolphins has been established for the Canterbury gillnet fishery (Hodgson, 2002). The limit is loosely based on the PBR formula used by NMFS (Wade, 1998). However, there are two problems with the way it has been calculated. Firstly, all Hector's dolphins on the South Island east coast have been considered as one stock, which we believe is unrealistic. Both movement (Brager *et al.*, 2002) and distribution data (Dawson and Slooten, 1988; Dawson *et al.*, 2004; Slooten *et al.*, 2004) suggest that in some cases a local population might be effectively isolated from others as little as 100km away. Secondly, the NMFS system uses a minimum estimate of population, which is defined as the lower 60% confidence interval of abundance. The Ministry of Fisheries limit is based instead on the point estimate of abundance.

Wade and Angliss (1997) suggest that if a species has small sub-populations with rare, low or moderate dispersal, the sub-populations should be split for the purposes of the US Marine Mammal Protection Act. If this suggestion is followed for Hector's dolphins, the area over which bycatch limits should apply would be small. Corresponding bycatch limits may thus be as little as fewer than one every several years in some cases. Without extremely high levels of observer coverage, it will be difficult to determine when the limit has been reached, with the possibility that the limit could be exceeded, perhaps substantially. In addition, given that observed catches could close the fishery, the very small bycatch limits would place great pressure on observers. For these reasons we believe that management via bycatch limits is impractical for Hector's dolphin.

EVALUATION OF EFFECTIVENESS OF MANAGEMENT MEASURES

Effectiveness of the Banks Peninsula Marine Mammal Sanctuary

Detecting change in population size or in population parameters (e.g. survival rate) is inherently difficult (e.g. Taylor and Gerrodette, 1993). This is especially so for dolphin populations, which are difficult to study, and whose low reproductive rates mean that population growth happens very slowly. Intensive population biology studies have been in place at Banks Peninsula since 1985. Adult survival rate is the most influential parameter in population models (Slooten and Lad, 1991; Slooten *et al.*, 2000). We expect our estimates of this parameter to gain precision with time, as the study continues, however, at this stage there is no indication that survival rates are increasing (DuFresne, 2004).

A lack of increasing survival rates is consistent with the fact that bycatch continues immediately north, south and offshore of the sanctuary (see above). Recent aerial surveys conducted in summer and winter (Slooten, Dawson and Rayment, unpublished data) indicate that in summer some 80% of the Banks Peninsula dolphins are found inside the sanctuary and in winter this drops to around 35%. In addition, occasional entanglements have occurred in gillnets set illegally inside the sanctuary by amateur fishermen. In this case it appears that the protected area is not large enough, and compliance with its restrictions is incomplete (Slooten *et al.*, 2000).

The extension, in time and geographic area, of the restrictions for recreational fishermen will help reduce bycatch mortality. If these new regulations are extended to commercial fishing, they may lead to a sustainable Hector's dolphin population in the Canterbury area.

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A preliminary note on population structure in eastern South Pacific common bottlenose dolphins, *Tursiops truncatus*

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ABSTRACT

Previous studies of eastern South Pacific common bottlenose dolphins, *Tursiops truncatus*, defined offshore and inshore ecotypes in Peru based on cranial and tooth morphology, documented the presence of a single resident inshore community (‘pod-R’) in central-north Chile, and confirmed the presence of offshore bottlenose dolphins off Chile. Here, mtDNA control region (331bp) was examined to evaluate genetic relationships between four geographic areas: inshore pod-R (n=8), Chilean offshore population (n=8), Peruvian inshore (n=3) and offshore (n=12) ecotypes. This is the first genetic analysis of *T. truncatus* in this ocean basin. Phylogenetic analysis grouped the three Peruvian specimens morphologically identified as inshore ecotype in an independent cluster, supported by 100% bootstrap value. The net genetic distance between Peruvian inshore and Peruvian offshore ecotypes was estimated at 2.9%, and even higher when compared with Chilean bottlenose dolphins. Morphological and mtDNA evidence combined argues for considering inshore and offshore ecotypes as evolutionary significant units, to be managed accordingly. Despite its inshore behavioural ecology, pod-R presented a high divergence from the Peruvian inshore ecotype and a relatively closer affinity with the Chilean offshore stock (3.41% and 0.87% net interpopulational distance, respectively). However, homogeneity tests showed significant genetic differences of pod-R with all other groups, including Chilean offshore. This, combined with a low nucleotide diversity (0.0069) and behavioural observations, suggest that pod-R may be reproductively isolated and active protection measures are recommended. Only one haplotype from a total of 21 was shared by Peruvian and Chilean offshore animals. Their net genetic distance was estimated at 0.024 and no significant differences were found in haplotype frequencies, suggesting a single, wide-ranging ‘Peru-Chile offshore stock’.

KEYWORDS: GENETICS; STOCK IDENTITY; COMMON BOTTLENOSE DOLPHIN; PACIFIC OCEAN; SOUTH AMERICA; TAXONOMY; CONSERVATION; MANAGEMENT

INTRODUCTION

While the taxonomic status of the *Tursiops* genus (bottlenose dolphins) is still under discussion, most authors including ourselves presently recognise two species, one of cosmopolitan distribution, the common bottlenose dolphin, *T. truncatus*, Montagu, 1821 and the Indo-Pacific bottlenose dolphin, *T. aduncus*, Ehrenberg, 1833 (Ross, 1977; Curry, 1997; LeDuc, 1997; Hale *et al.*, 2000; Krützen *et al.*, 2001; Shirakihara *et al.*, 2003 and Wang *et al.*, 1999). However the existence of other species or subspecies cannot be ruled out.

Using general comparative morphology, cranial, diet and parasite load differences, as well as mtDNA analysis, two distinct common bottlenose dolphin ecotypes, offshore and inshore (syn. coastal), have been described for the eastern North Pacific (USA), western North Pacific (China and Japan), western North Atlantic (USA), eastern South Pacific (Peru), western South and Indo-Pacific (Australia and Solomon Islands), eastern South Atlantic (Namibia, South Africa), and the east coast of South Africa (Ross, 1977; 1984; Walker, 1981; Duffield *et al.*, 1983; Ross and Cockcroft, 1990; Van Waerebeek *et al.*, 1990; Findlay *et al.*, 1992; Mead and Potter, 1995; Hoelzel *et al.*, 1998 and Wang *et al.*, 1999).

In the Southeast Pacific Ocean, common bottlenose dolphins are known to occur from the Galápagos archipelago, continental Ecuador, the entire coast of Peru, northern and central Chile south to at least Quenu Island, off Calbuco, X Region (41°45’S, 73°10’W)[1] and the offshore Chilean archipelagos of San Ambrosio and San Félix, Salas and Gómez Islands and the Juan Fernández Islands (e.g. Lévèque, 1963; Aguayo, 1975; Donovan, 1984; Guerra *et al.*, 1987; González *et al.*, 1989; Van Waerebeek *et al.*, 1990; Félix, 1994; Félix and Samaniego, 1994; Findlay *et al.*, 1998; Sanino and Yáñez, 2001).

Van Waerebeek *et al.* (1990) found distinct cranial differences between offshore and inshore ecotypes of bottlenose dolphins in Peru, as well as clear differences in diets and helminth parasite loads, suggesting reproductively isolated populations. Santillán (2003), studying a larger sample, confirmed cranial differences. Recent management concerns, including direct takes (Sanino and Yáñez, 2000; 2001) led us to biopsy animals from an inshore dwelling community, named pod-R, at Choros Island (29°15’S, 71°26’W) in central-north Chile. These were originally thought to form part of a wider coastal population of bottlenose dolphins.

Video-identification studies revealed high site-fidelity of the 28 pod-R members, as well as morphological and behavioural differences in observed pods of offshore bottlenose dolphins (Sanino and Yáñez, 2001). The issue was raised as to whether pod-R bottlenose dolphins are

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[1] Live-stranding of two adult bottlenose dolphins, trapped by fast receding tide, on 2 August 2004 (CMMR files, positive identification from photographic evidence published in *El Llanquihue* No. 36.642, 4 August 2004; Sociedad Periodística Araucanía, Puerto Montt, Chile).

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largely reproductively isolated or are part of, and mix with, other communities to form a wide-ranging Chilean coastal population.

In this study[2] mtDNA control region sequences were used to examine genetic diversity and phylogenetic relationships among common bottlenose dolphins from different localities in the southeast Pacific Ocean, with particular emphasis given to pod-R dolphins in central-north Chile and the Peruvian inshore ecotype.

MATERIALS AND METHODS

Samples and localities

Sampling localities and sample sizes for the bottlenose dolphins are as follows (Fig. 1): inshore Choros Island (Chile) (CL-I, *n*=8), offshore Chile (CL-O, *n*=8), Peruvian inshore (PE-I, *n*=3) and samples from Peru that included individuals from both confirmed offshore ecotype and indeterminate but like-offshore specimens for which no skulls were collected (PE-O, *n*=12). Morphological characteristics used to distinguish ecotypes in Peru include tooth diameter, the morphology of pterygoids, palatine bones, antorbital process and the separation of occipital condyles (Van Waerebeek *et al.*, 1990). Accompanying fisheries data were also taken into account, especially if the landing process was monitored by one of the authors. All tissue samples from Peru (PE-I and PE-O) were taken from either freshly landed specimens captured in a variety of fisheries, or from body remains found on beaches near fishing towns. Most were stored in dimethylsulphoxide (DMSO) solution with the remainder being stored in 70% ethanol.

CL-I skin samples were collected using a 'Golden Bear' long-bow with modified darts, as described in IWC (1991), mounted with a 6mm (diameter) tip. The samples were soaked in 70% ethanol for three weeks, after which the hypodermis was eliminated and the epidermis/dermis was transferred to a DMSO saturated saline solution. Examination of photographs and videotapes for individual identification precluded the possibility of more than one sample coming from the same animal. CL-O samples were collected with a Barnett crossbow and tethered bolts (IWC, 1991) from the bow of the R/V *Shonan Maru 2* during the third blue whale cruise of the IWC/SOWER programme (Findlay *et al.*, 1998). Sampling error was considered negligible as biopsies of bowriding specimens were taken near-randomly over a wide area (8° of latitude).

Extraction of DNA

Total cell DNA was extracted from samples of skin or other tissue. DNA extractions followed phenol/chloroform/isoamyl/alcohol protocols as described by Sambrook *et al.* (1989). Extracted DNA was resuspended in 500μl 0.1M Tris-HCl (pH 8.0), 0.05 mM EDTA.

Amplification of mtDNA control region

The first 500 nucleotides at the 5' end of the mitochondrial control region were amplified by the polymerase chain reaction. The oligonucleotides employed in the PCR amplification were MT4 (Arnason *et al.*, 1993%) and P2R (5'-GAA GAG GGA TCC CTG CCA AGC GG-3'). Reactions were carried out in 50 μL volumes containing 100 mM KCl, 20 mM Tris-HCl, 0.1 mM EDTA, 1 mM DTT; 0.5% *Tween* R20, 0.5% *Nonidet* RP-40, 200 μM dNTPs, 2.5 pM each oligonucleotide and one unit of *Taq* DNA polymerase. After an initial denaturation step at 95° C for 5 minutes, a PCR amplification regime of 30 seconds at 94°C, followed by 30 seconds at 50°C and 30 seconds at 72°C was repeated 30 times. The amplification was completed with a final extension step of 10 minutes at 72°C. Subsequent cycle sequencing reactions were performed with 100ng of products generated in the above PCR amplifications using the *Prism*™ dRhodamine Terminator Cycle Sequencing Kit (Applied Biosystems, Inc.). The oligonucleotides used to prime the cycle sequencing reaction were the same as employed in the initial PCR amplification listed above. A total of 25 cycles with 10 seconds at 96°C, 20 seconds at 56°C and four minutes at 60°C were performed. The nucleotide sequence of each cycle sequencing reaction was determined by electrophoresis through a 5% *Long Ranger*™ (FMC, Inc.) denaturing polyacrylamide matrix on a DNA *Prism*™ 377 DNA Sequencer (Applied Biosystems, Inc.) under standard conditions. Both strands were sequenced in their entirety for all samples.

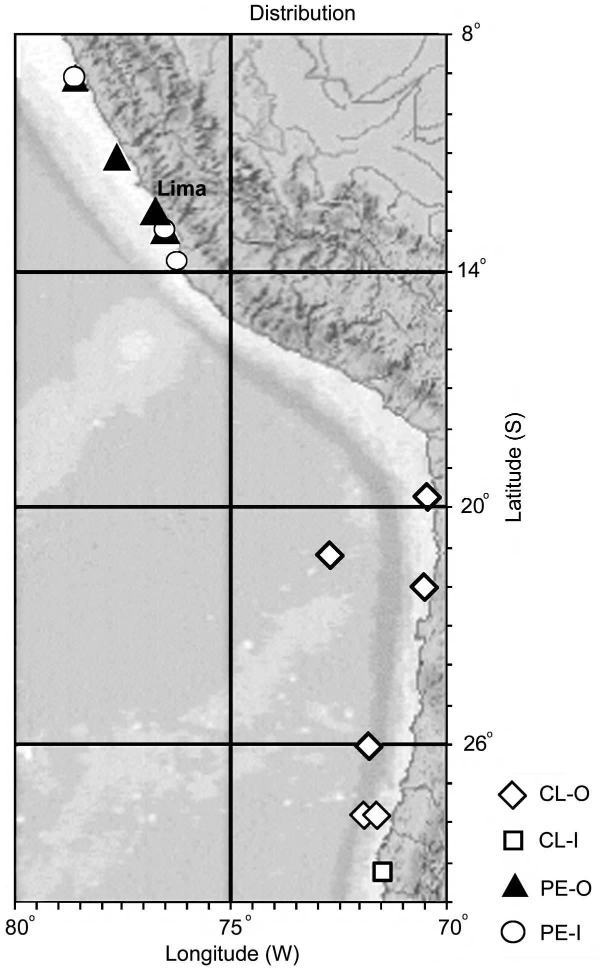

Fig. 1. Distributions of common bottlenose dolphins sampled in Peru and Chile.

Sequence analysis

Sequencing was performed with 100ng PCR products using the *PRISM™ Ready Reaction Dye Deoxy Terminator Kit* (Applied Biosystems) (ABI). Primers used for sequencing were the same as indicated above. The reaction was performed through 25 cycles of 96°C for 10sec, 56°C for 20sec and 60°C for 4min. The nucleotide sequence for each amplification was determined by electrophoresis through a 5% *Long Ranger™* polyacrylamide matrix on an ABI *DNA*

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[2] An earlier version of this paper (SC/55/SM22) was presented to the IWC Scientific Committee in 2003.

Prism™ 377, following the manufacturer protocols. For each sample both forward and reverse strands were sequenced. Sequences were aligned using the DNA sequence comparison software '*Sequence Navigator*' developed by ABI.

Levels of polymorphism

Genetic distances among different haplotypes were estimated using Kimura's two parameters method based on genetic distance among haplotypes (Kimura, 1980). Nucleotide diversity (π) was estimated following equation 10.5 of Nei (1987). The net genetic distance between populations (d_A) was estimated by subtracting the average level of variation within each population, following equation 10.21 of Nei (1987).

MtDNA genealogy

Phylogenetic reconstruction of haplotypes was made using the neighbour-joining method (Saitou and Nei, 1987). To evaluate the confidence limits of phylogenies, 1000 bootstrap simulations were conducted (Felsenstein, 1985). The phylogenies were rooted using the homologous sequence from a common dolphin (*Delphinus* sp.; GenBank accession number: U02652).

Homogeneity test

Homogeneity tests were conducted using the sequence (Kst*) and haplotype (Hst) statistics proposed by Hudson *et al.* (1992). The degree of divergence was inferred as being larger than zero, if an equal or more extreme value of the Kst* or Hst was observed in less than 5% of 10,000 Monte Carlo simulations.

RESULTS

Level of polymorphism

The first 331 nucleotides were determined in the mtDNA control region for each of the 31 samples. A total of 32 polymorphic sites were detected (30 transitions, one transversion and one deletion), which defined 21 unique haplotypes (Table 1).

Nucleotide diversity for the whole sample population was estimated to be 0.02193, and the nucleotide diversity within a single sample ranged from 0.00201 in the Peruvian inshore (PE-I), to 0.02007 in the Chilean offshore group (CL-O) (Table 2).

Geographic distribution of haplotypes

The frequency of haplotypes in the bottlenose dolphin samples is shown in Fig. 2. Apart from haplotype '9', which was shared by CL-O and PE-O, no shared haplotype occurred among CL-I, CL-O, PE-I and PE-O. All individuals in the PE-O group showed a different haplotype. Six haplotypes were defined in eight CL-O individuals, while only two haplotypes were defined in the eight CL-I animals. Two individuals of the Peruvian inshore ecotype shared the same haplotype ('14'), although they were landed in ports 556km apart (Chimbote and San Andrés).

MtDNA haplotype genealogy

A neighbour-joining-based phylogenetic tree of the haplotypes (Fig. 2) shows haplotypes '4' and '14' as highly divergent from all others. Haplotype '14' includes two individuals from the Peruvian inshore ecotype. Haplotype '4' was represented by a single inshore individual (MFB-465), as determined by cranial and capture data. This was supported by sequencing results since MFB-465 clustered very near to haplotype '14' (with 100% bootstrap support). Two other clusters with a branch supported by a high bootstrap value (70%) were identified, however these clusters included individuals from different localities, but no Peruvian inshore specimens.

Net inter-populational distances

Table 2 shows the net inter-populational distances between areas. Individuals with haplotypes '4' or '14' were identified as the Peruvian inshore type based on morphological features (Van Waerebeek *et al.*, 1990) and capture circumstances. All pairwise comparisons involving this ecotype (PE-I, *n*=3) showed large genetic distances ranging from 0.02900 to 0.03412. The other pairwise comparisons resulted in genetic distances between 0.00024 and 0.00870. The smallest genetic distance was found between CL-O and PE-O.

Homogeneity tests

Pairwise comparisons between areas predominately resulted in significant genetic differences. However, the comparison between CL-O and PE-O, showed no significant difference and the comparison between CL-I and PE-I was only near-to-significant for Hst. The latter is presumably due to the small PE-I sample size since the corresponding Kst* was highly significant (Table 3).

DISCUSSION

The three bottlenose dolphin specimens from Peru identified as the Peruvian Inshore ecotype through the evaluation of cranial characteristics, tooth diameters (Van Waerebeek *et al.*, 1990) and fisheries data (e.g. Van Waerebeek *et al.*, 1997; 2002), were found to be phylogenetically distinct from all other bottlenose dolphins studied in the eastern South Pacific. Furthermore we found a high net inter-populational distance of 2.9% between Peruvian inshore and offshore ecotypes, and even higher values when compared with another area (Chile). These results suggest that the Peruvian inshore ecotype should be considered an evolutionarily significant unit, and should be managed separately from the offshore ecotype, coinciding with morphological and ecological evidence (Van Waerebeek *et al.*, 1990; Santillán, 2003).

Inshore Chilean bottlenose dolphins (pod-R) were highly divergent from the Peruvian inshore ecotype based on phylogenetic analysis and the net inter-population distance. This is surprising since pod-R dolphins reside inshore with high site fidelity, which are characteristic of all inshore bottlenose dolphins. In fact, pod-R appeared more closely related to the Chilean offshore stock. However, homogeneity tests for pod-R in pairwise comparison with the Chilean offshore and the Peruvian offshore groups revealed significant genetic differences. Furthermore, the eight CL-I individuals showed only two haplotypes (Fig. 2) and a concomitant low nucleotide diversity. Strictly speaking, mtDNA divergence does not necessarily signify reproductive isolation since mtDNA is maternally inherited. However, these mtDNA analysis results are consistent with intensive field observations on pod-R, which suggest that the group may be reproductively isolated (Sanino and Yáñez, 2001), and a study to ascertain whether this is the case using nuclear DNA markers is planned. If reproductive isolation is indeed confirmed, the long-term survival of this community of some 30 individuals looks uncertain (Sanino and Yáñez, 2001). Pod-R might actually constitute the only remnant pod of a population. To date there is no evidence for

Table 1

Variable sites defining 21 haplotypes in bottlenose dolphins from the eastern South Pacific. The numbers above (reading vertically) list the nucleotide positions of the polymorphic sites starting from the 5' end of the mtDNA control region. Haplotypes '2' through '21' are listed with reference to haplotype '1'. A dot indicates an identical nucleotide at the position relative to haplotype '1'. A hyphen indicates a deletion.

		11	1111222222	2222222222	33
		233349901	1146034456	6777778899	00
		7013461270	1597545815	9237892334	35
1	MFB185/7	AGCTCTTTAT	GGCTCTCGCT	CATTTTTCCT	AT
2	MFB186	-.........	....T.....	........TC	..
3	MFB441	-.........	C	......C.T.	..
4	MFB465	-AT.T..C..	A..C..TATA	C	..
5	MFB701	-..C.....C	..T...T..C	....CCC.TC	G.
6	MFB702	......C...	.AT...T..C	....C...TC	G.
7	KVW2393	-.........	C	......C...	..
8	KVW2412	-.........	..T...T..C	....C.C.TC	G.
9	KVW2417	......C...	.AT...T..C	....C.C.TC	G.
10	KVW2439	-........C	C	T.....C...	..
11	KVW2440	-.........	....	C	..
12	RBC54	-.........	....	...C..C..C	.C
13	JAS12	....	C	....	..
14	JAS47	-AT.T..C..	A.....TATA	C	..
15	GPSV-BP001	-.........	A...T.....	..C...C.T.	..
16	GPSV-BP002	-.........	....	......C...	..
17	TS97121301	......C...	.AT......C	....C.C.TC	G.
18	TS97121303	-....C....	....	......C...	..
19	TS97121802	-.........	A.........	......C..C	.C
20	TS97122602	-.........	....	......C..C	..
21	TS97122604	........G.	..T..C...C	.G...C.TTC	..

Table 2

Nucleotide diversity (π, diagonal in bold) and net interpopulational distances (d_A, upper right), with respective sample sizes, for bottlenose dolphins in four geographic areas of the eastern South Pacific. Confidence intervals of the estimates are in parentheses.

Area	PE-I (*n*=3)	PE-O (*n*=12)	CL-I (*n*=8)	CL-O (*n*=8)
PE-I (*n*=3)	**0.00201** (±0.00190)	0.02900 (±0.00820)	0.03412 (±0.01122)	0.03349 (±0.01004)
PE-O (*n*=12)		**0.01794** (±0.00446)	0.00564 (±0.00160)	0.00024 (±0.00008)
CL-I (*n*=8)			**0.00691** (±0.00228)	0.00870 (±0.00290)
CL-O (*n*=8)				**0.02007** (±0.00766)

Table 3

Results of homogeneity tests for bottlenose dolphins in four geographic areas of the eastern South Pacific. Hst values are shown above the diagonal; Kst* values below diagonal. *=significant at the $P<0.05$ level; **=significant at the $P<0.01$ level.

Area	PE-I (*n*=3)	PE-O (*n*=12)	CL-I (*n*=8)	CL-O (*n*=8)
PE-I (*n*=3)	-	0.0577*	0.2177	0.1036*
PE-O (*n*=12)	0.2438**	-	0.1156**	0.0223
CL-I (*n*=8)	0.5100**	0.1284**	-	0.1633**
CL-O (*n*=8)	0.3271**	0.0207	0.1991**	-

the existence of a widely distributed inshore bottlenose dolphin population in Chile south of Punta Coloso (23°43'S), near Antofagasta (see Aguayo, 1975; Sielfeld, 1980; 1983; Guerra *et al.*, 1987; Van Waerebeek *et al.*, 1990). North of Punta Coloso, several undetermined ecotype specimens are curated at the University of Antofagasta (Guerra *et al.*, 1987), and unconfirmed reports of dolphins in the surfzone off beaches around Iquique require further investigation.

No significant differences were found between Peruvian and Chilean offshore bottlenose dolphins, but only one haplotype was shared between them (No. 9, Fig. 2). These dolphins often travel at great speeds with steady bearing, performing high, energetic jumps (personal observations) and are thought to cover great distances with ease. Chilean and Peruvian offshore bottlenose dolphins probably form a single wide-ranging population, which we have provisionally named the 'Peru-Chile offshore bottlenose dolphin stock'. Affinities with other nominal bottlenose dolphin species described from the eastern Pacific Ocean[3] should be established, including insular animals found around archipelagos.

The mtDNA results presented here, together with morphological data and parasite load differences (Van Waerebeek *et al.*, 1990) show that the Chilean and Peruvian inshore stocks of bottlenose dolphins should each be managed as distinct reproductive units.

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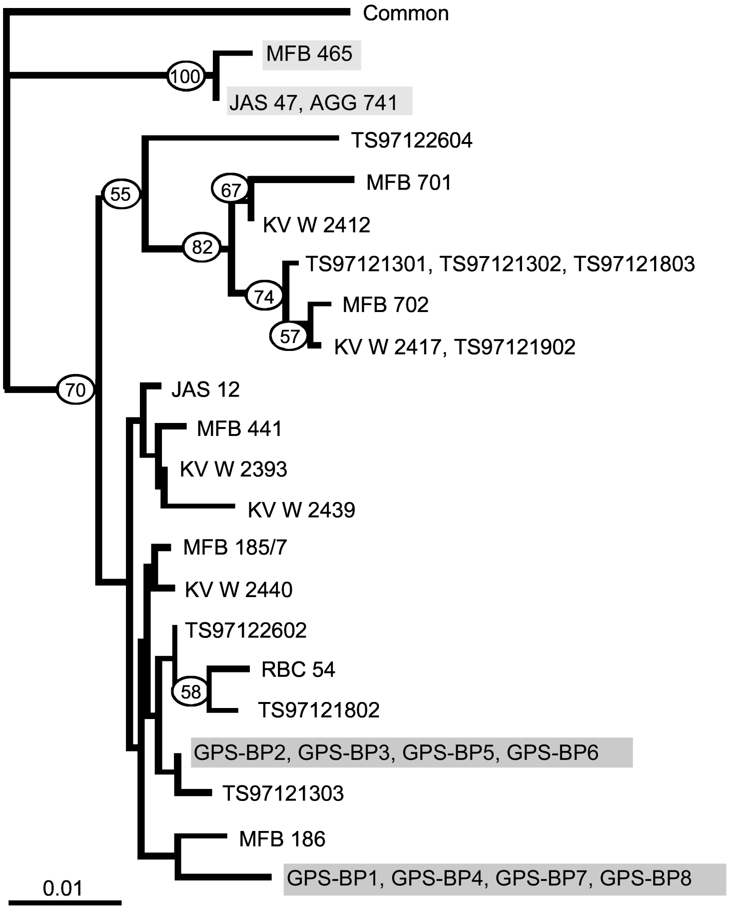
[3] In particular *Tursiops nuuanu* Andrews, 1911 of the eastern tropical Pacific and *T. truncatus gillii* Dall, 1873 of southern and Baja California (see comparative discussion by Andrews, 1911).



Sampling area

H	CL-I	CL-O	PE-I	PE-O
4			1	
14			2	
21		1		
5				1
8				1
17		3		
6				1
9		1		1
13				1
3				1
7				1
10				1
1				1
11				1
20		1		
12				1
19		1		
16	4			
18		1		
2				1
15	4			

Fig. 2. Phylogenetic tree and frequencies of haplotypes (H) for common bottlenose dolphins from the four study areas (CL-I: Inshore Chile, CL-O: offshore Chile, PE-I: inshore Peru, PE-O: offshore Peru). In circles are the bootstrap values over 50% —in 1,000 Monte Carlo simulations.

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A note on a pygmy right whale (*Caperea marginata*) sighting in the southwestern Pacific Ocean

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ABSTRACT

There are no previously published sightings of pygmy right whales (*Caperea marginata*) from the southwest Pacific Ocean. This note reports a sighting of 14 pygmy right whales approximately 445km southeast of Christchurch, New Zealand, at 17:30 on 5 January 2001 (46°26'S, 177°18'E). No cow-calf pairs were observed. Although feeding was not directly observed, several animals defecated during the observation period. The faeces were bright red and formed into small red clumps that bobbed at the water's surface which implies that the whales had recently been feeding in the immediate area. It appears that pygmy right whales may aggregate at the Sub-tropical Convergence during the austral summer and that this area may represent an important feeding ground.

KEYWORDS: PYGMY RIGHT WHALE; SURVEY-VESSEL; DISTRIBUTION; FEEDING GROUNDS; PACIFIC OCEAN; SOUTHERN HEMISPHERE

INTRODUCTION

The pygmy right whale (*Caperea marginata*) is the smallest (<7m) of the baleen whales. Little is known of its life history and it has only rarely been observed in the wild (e.g. Baker, 1985; Kemper, 2002a). It is found only in the Southern Hemisphere and is presumed to have a circumpolar, temperate distribution based on the available records. This note represents the first published sighting from the southwest Pacific Ocean.

MATERIALS AND METHOD

The 2000-2001 International Whaling Commission – Southern Ocean Whale and Ecosystem Research Programme (IWC-SOWER) Circumpolar Cruise was conducted from 5 January to 5 March 2001. The cruise was the 23rd in a consecutive series of Antarctic cruises conducted by the IWC primarily to estimate the abundance of Antarctic minke whales (*Balaenoptera bonaerensis*) (e.g. see Matsuoka *et al.*, 2003). The vessels departed Wellington on 5 January. The date change associated with crossing the International Date Line was made upon leaving Wellington. Permission was granted by the Ministry of Foreign Affairs and Trade, New Zealand, for a sightings survey within the 200 n.mile Exclusive Economic Zone (EEZ) of New Zealand, but only conducted in passing mode. Sighting effort was conducted by the bosun and topmen from the barrel (crow's nest) and the upper bridge where the helmsman, captain or officer-on-watch, international researchers, and the chief engineer or deputy were also present. Primary search effort was only conducted in acceptable weather conditions. These conditions are defined as being able to see a minke whale blow at a distance of at least 1.5 n.miles, with wind speed less than 20 knots. The sighting survey was conducted at an average of 11.5 knots.

RESULTS

The sighting was made at 46°26'S, 177°18'E (approximately 445km southeast of Christchurch, New Zealand) while *en route* to the Antarctic aboard the *Shonan-Maru No.2* at 17:30 on 5 January 2001. Weather conditions were optimal at the time of the sighting with visibility estimated at 13km and a wind speed of 11 knots. The temperature was 15.1°C, the sea surface temperature 13.1°C and the water depth was approximately 2,500m. The initial radial sighting distance was estimated to be 1.5km. Shortly after the initial sighting, vessel speed was reduced from 11.5 to 2.5 knots as the animals crossed the trackline within 35m of the bow.

The animals were identified as pygmy right whales by the presence of an arched jawline (Fig. 1), and a prominent dorsal fin positioned about two-thirds back on their bodies. Body lengths were estimated to be 5.5-6.5m except for one probable juvenile estimated to be 4m long. These length estimates are in agreement with Ross *et al.* (1975); the largest animal they measured was an adult female 6.4m long. There were 14 animals present and no cow-calf pairs, although the 4m animal did appear to keep company with another, larger animal.

As the whales swam past the bow it was possible to look almost straight down at them from above, the head shape of a pygmy right whale has a very narrow profile, similar to an Antarctic minke whale and quite unlike a right whale (*Balaena* spp.). The head was small, narrow and pointed, with a noticeably raised blowhole, visible when the animals exhaled. A mid-dorsal ridge extended from the leading edge of the blowhole to the tip of the rostrum. The arched lower lip was sometimes evident when the head was lifted during surfacing, but was usually not seen. The dorsal fin was medium-sized, pointed and falcate. The whales swam surprisingly quickly, suggesting that these whales are hydrodynamically efficient (during the observation a splinter group briefly increased their swim speed to an estimated 6-8 knots without apparent effort).

The body colour was dark greyish above and pale ventrally with some white sweeping up on the sides that was also similar to minke whales. The body colour showed some variation that may possibly be age-related. The apparent juvenile (4m long and very little visible body scarring – see below) was greyish above and pale pinkish ventrally (Fig. 2). Presumed adults (>5.5m long, heavy body scarring)

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appeared more brownish in photographs. Another colour pattern perhaps age-related was the presence of a thin white outline on the dorsal surface of the lower lip on what appeared to be older individuals (Fig. 3); some less-scarred individuals, including the smallest one, did not show this feature. All of the animals observed well had a fairly conspicuous, pale chevron varying in width and shape on their sides and back (Figs 1-3). The chevron extended from about the area of the pectoral flippers and angled forward, with the apex on the mid-body line of the back and pointing forwards.

Fig. 1. Pygmy right whale showing arched lower jaw and blowhole ridge. Photograph by Robert Pitman.

Fig. 2. An apparently juvenile pygmy right whale identified by the lack of oval scars and different colouration (see text for details). Photograph by Robert Pitman.

All of the animals had concave, oval scars scattered over most of the visible parts of their bodies (Figs 1-3). These scars were approximately 5-10cm at their widest, and in most cases the long axis of the scar was parallel to the long axis of the whale. These were presumably healed wounds from the bites of cookie-cutter sharks (*Isistius* spp.; Jones, 1971). The number of visible bites per animal ranged from <10 to dozens and it appeared that bigger animals were the most heavily pocked with scars. No fresh bites were seen on any of the animals, suggesting that they had been bitten in another part of their geographic/temporal range. Baker (1985) suggested that this species is a winter breeder based on observations of stranded animals. Matsuoka *et al.* (1996) also noted the presence of presumed cookie-cutter shark bites on animals observed in the southern Indian Ocean.

Fig. 3. Pygmy right whale showing chevrons, oval scars presumably from bites of cookie-cutter sharks (*Isistius* sp.) and a thin white outline on the lower lip. Photograph by Robert Pitman.

Initially, the animals formed a fairly tight group, milling and rolling at the surface. During most of the 10-minute observation period, the whales swam slowly near the surface, at an estimated 3 knots. No prolonged diving was observed; the longest dive did not exceed three minutes. When surfacing, the head and blowhole usually appeared before the dorsal fin was exposed, although sometimes the head and fin were exposed simultaneously. The blows seen initially were thin and columnar and very similar to Antarctic minke whales, but on closer approach the blows became smaller and puffier. The whales appeared to be responding to the vessel because after closing within approximately 300m, most of their blows were not visible anymore. This inconspicuous surfacing behaviour has also been reported by Ivashin *et al.* (1972) and Ross *et al.* (1975).

DISCUSSION

Although no feeding was directly observed, most animals defecated during the observation period. The faeces were bright red and formed into small red clumps that bobbed at the surface. This implies that the whales had recently been feeding in the immediate area. Stomach contents of pygmy right whales indicate that they eat copepods of the genus *Calanus* (Ivashin *et al.*, 1972). The stomach of one individual that stranded in False Bay, South Africa, in March 1990 contained 99.9% copepods, mainly *Centrophages brachiatus* and *Calanoides carinatus* (Sekiguchi *et al.*, 1992). The bright red faeces observed are consistent with a diet of copepods. Also, the milling, apparently resting, behaviour initially seen may also be related to the small scale vertical movement of copepods in the water column, with a considerable number of patches moving to the surface in the evening (Kawamura, 1974).

There are no previously published sightings of pygmy right whales from the southwest Pacific Ocean, although the sighting reported here is within the expected geographical range of this species (Kemper, 2002b). The sighting occurred just at the Sub-tropical Convergence (sea surface temperature was 13.1°C), where large concentrations of *Calanus tonsus* are known to occur (Kawamura, 1974). Kawamura (1974) indicated that the mean summer surface temperature at the Sub-tropical Convergence is 9°-13°C. He found large, dense patches of copepods and small euphausids on or near the Convergence south of Australia and New Zealand (Fig. 4). A single sighting of 80 pygmy right whales in late November 1992 in the southeast Indian

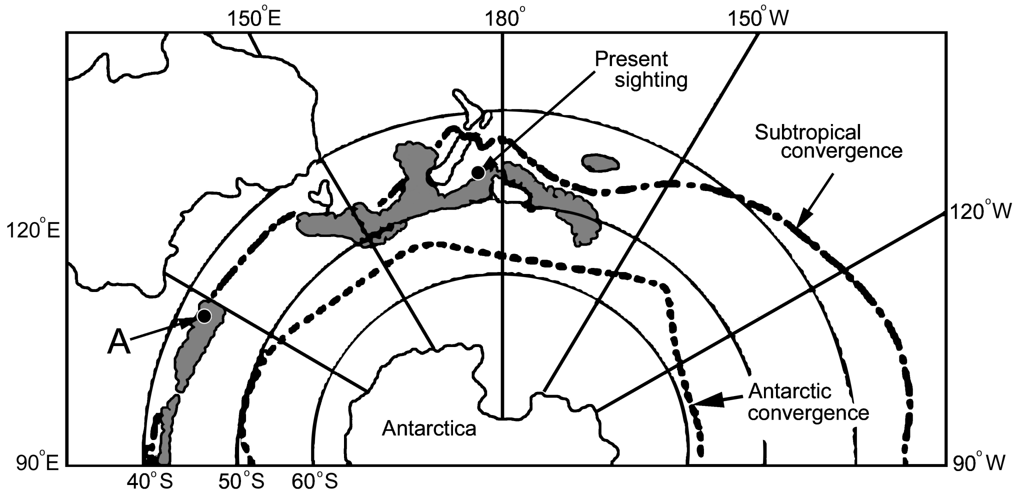

Fig. 4. Location of pygmy right whale sighting described in this note and distribution of *Calanus tonsus* in the Southern Ocean (shaded area; from Kawamura, 1974; see text for details). A: sighting by Matsuoka *et al.* (1996).

Ocean in the morning was also just south of the Sub-tropical Convergence in an area with a large concentration of *C. tonsus* (Matsuoka *et al.*, 1996), no cow-calf pairs were sighted. This limited information provides some evidence that pygmy right whales may aggregate at the Sub-tropical Convergence during the austral summer and that this may represent an important feeding ground.

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Aspects of the ecology and behaviour of bottlenose dolphins (*Tursiops truncatus*) in Santa Monica Bay, California

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ABSTRACT

The occurrence, distribution, site fidelity, group size and behaviour of common bottlenose dolphins (*Tursiops truncatus*) were assessed during a photo-identification study conducted between 1997-2001 in Santa Monica Bay, California. Bottlenose dolphins occurred year-round in the bay and were encountered on 56.8% of all surveys (*n*, total surveys=211). This species was found in waters within 0.5km of shore in 80.0% of the sightings (*n*=157), but sometimes found in deeper waters further offshore (>0.5km). No correlations between anomalies in sea surface temperatures during the 1997-98 El Niño event and sighting frequencies were observed. Group sizes varied significantly between schools observed inshore and offshore in the bay, with the largest groups sighted offshore. A total of 290 dolphins were individually photo-identified based on long-term natural marks on their dorsal fins. Forty-four individuals (15.2%) were encountered in both inshore and offshore waters, showing no exclusive fidelity to inshore waters. The low resighting rates of known individuals provided little evidence of long-term year-round site fidelity for Santa Monica Bay, revealing a range greater than the chosen study area. Several individuals, however, were resighted over one or two year periods, generally during more than one season. This suggested that these dolphins were highly mobile within the inshore waters of the Southern California Bight but they also spent time foraging and feeding in Santa Monica Bay, probably due to the presence of submarine canyons within this area. The behavioural budget for this species showed a predominance of activities characterised by travel and dive-travel (69.0%) and feeding (5.0%), indicating a fairly high proportion of time devoted to searching for prey and feeding in the study area.

KEYWORDS: SURVEY-VESSEL, SOCIAL, BEHAVIOUR, FEEDING, HABITAT, SCHOOL SIZE, DISTRIBUTION, MOVEMENTS, PHOTO-ID, EL NIÑO, CONSERVATION

INTRODUCTION

The genus *Tursiops* is found widely in temperate and tropical waters. Populations of bottlenose dolphins are known to inhabit pelagic waters as well as coastal areas, including bays and tidal creeks (Leatherwood *et al.*, 1983). These populations also show morphological, osteological and molecular differentiations (LeDuc and Curry, 1998; Rossbach and Herzing, 1999). The frequent presence of bottlenose dolphins along the coastline has made this one of the best studied cetacean species in the world.

Long-term studies on free-ranging bottlenose dolphins in the Southern California Bight have been focused mostly along the San Diego coastline (less than 1km from shore; Defran and Weller, 1999). In 1996, a preliminary series of cetacean surveys in the waters of Santa Monica Bay revealed that bottlenose dolphins could be found there throughout the year, making the area suitable for a long-term study of the social ecology and behaviour of this species. This is an area of conservation interest because of possible habitat degradation due to the adjacent metropolis of Los Angeles. It is also of interest due to the large effects of some El Niño events[1] on dolphin distribution reported elsewhere along the California coast (e.g. Wells *et al.*, 1990; Bonnell and Dailey, 1993). This five-year (1997-2001) longitudinal study represents the first attempt to describe the occurrence, distribution, site fidelity, group size and behaviour of bottlenose dolphins in Santa Monica Bay using an observational approach and systematic photo-identification.

MATERIALS AND METHODS

Study area

The Santa Monica Bay study area (approximately 460km^2, Fig. 1) is a shallow shelf, bounded by the Palos Verdes Peninsula to the south (33°45’N, 118°24’W), Point Dume to the north (33°59’N, 118°48’W) and the edge of the continental shelf to the west. The bay contains two shallow water submarine canyons (Dume and Redondo) and one deeper canyon, the Santa Monica Canyon. This begins at a depth of about 100m, at the edge of the continental shelf. The bay has a mean depth of about 55m and a maximum depth 450m. A shallow shelf between the Santa Monica and Redondo Canyons extends as a plateau from the 50m contour. The study area is characterised by mild temperatures, short rainy winters and long, dry summers. Normal water surface temperatures range from 11 to 22°C although during the 1997-98 El Niño, three peaks of sea surface temperature (SST) anomalies were evident: May-June 1997, September-October 1997 and August 1998, with an increase in temperature of +2°C above the norm (Nezlin *et al.*, 2003).

Data collection and analysis

Surveys were conducted from January 1997 to December 2001 (Table 1), with an average of 3.5 surveys per month (*n*=211). Inshore (distance from shore <500m) and offshore (>500m) surveys were carried out in the morning and early afternoon. Boat surveys were conducted from a 7m (1996-2000) and a 10m (2001) powerboat at an average speed of 18km h^{-1}. The number of kilometres spent at the different bathymetric locations in the bay (grid comprising of 82 3.7 × 3.7 units) was calculated to determine the evenness in the coverage of the study area. No significant difference was observed in surveying the different locations (t=1.92, DF=28, P >0.05).

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[1] El Niño is an irregular climatic phenomenon that results in *inter alia* abnormally warm sea temperatures in the eastern tropical Pacific (e.g. see *www.elnino.noaa.gov*).

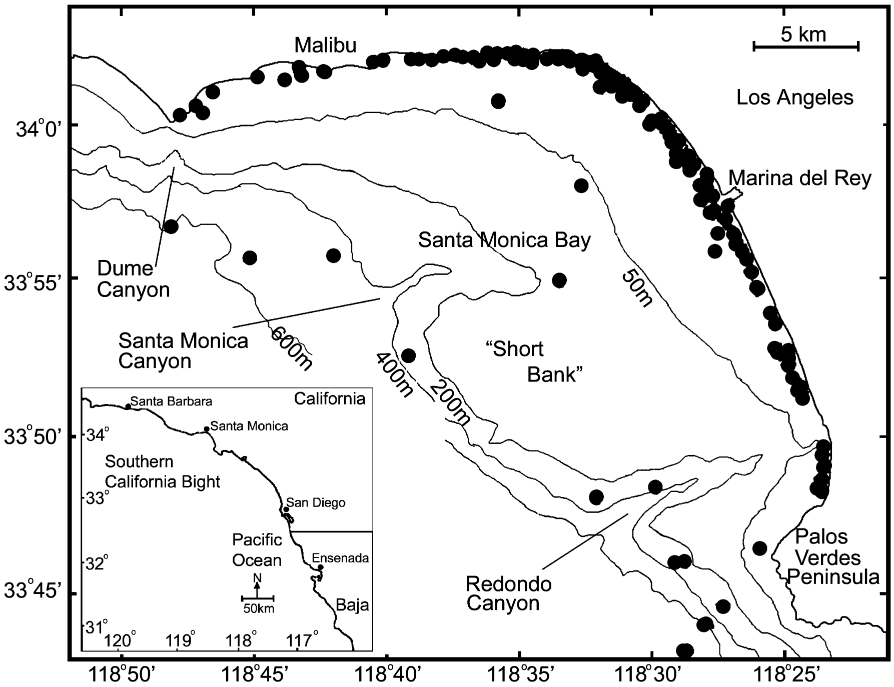


Fig. 1. The study area and the distribution of bottlenose dolphins in the bay. Each symbol (●) represents initial GPS coordinates of sightings.

Table 1

Number of surveys and summary of research effort in Santa Monica Bay for the years 1997-2001.

	1997	1998	1999	2000	2001	Total
Surveys						
Inshore surveys	5	17	12	6	7	47
Offshore surveys	23	3	5	4	6	41
Combined inshore/offshore surveys	11	38	27	27	20	123
Total number of surveys	39	58	44	37	33	211
Research effort						
Hours spent in the field	144	224	178	149	137	832
Hours spent searching for cetaceans	110	136	130	105	82	563
Hours spent with cetaceans	34	88	48	44	55	269
Hours spent with bottlenose dolphins	18	65	32	19	20	154
Total N of 5 min behavioural samples	295	1,065	698	525	675	3,258
N of 5 min behavioural samples for bottlenose dolphins	134	814	368	262	187	1,765

Data were collected with laptop computers and occasionally with tape recorders. When dolphins were seen, data on the number of animals, size classes and behaviour (Table 2) and aggregation with other species were recorded at five minute intervals throughout the sighting. The number of dolphins and size classes were additionally verified later through photo-identification analyses.

The majority of observations (91.7%) were conducted in good conditions (Beaufort scale ≤ 2, sea state 0 and visibility >300m). The dolphins' positions and speeds (±30m from the boat) were approximated to the boat's position using a GPS. Boat speed was reduced in the presence of dolphins and sudden speed or directional changes were avoided.

Photo-identification followed the methods of Würsig and Jefferson (1990) and Bearzi *et al.* (1997). For each sighting, an attempt was made to photograph all individuals present in the group. Colour photos were taken with 35mm cameras equipped with 75-300mm lenses using slide film (64-200 ISO). During the sightings, researchers also videotaped and recorded the animals' behaviour with Hi8 mm and Mini DV Video Camcorders.

Behavioural data collected opportunistically from July to December 1996 (58 hours of field observations) provided a framework of information to design the behavioural sampling procedures systematically adopted from January 1997 (Bearzi, M., 2003). Videos and photographs were reviewed in a laboratory to validate field observations.

Of over 21,500 cetacean pictures taken in the years 1997-2001, 11,909 were of bottlenose dolphins. Of these, 797 were scanned and matched using a computer-assisted identification system (*Finscan*; Kreho *et al.*, 1999). During matching, marks and scars likely to have been inflicted by sharks were also documented. Adult individuals consistently accompanied by a calf over a two-month period were assumed to be females.

Table 2

Definitions of size classes, group formation, physical contact, surfacing mode, directional behaviour, behavioural states and boat disturbance.

Size classes (from Bearzi *et al.*, 1997, modified)	
Adult	Dolphins approximately 2.7-3.2m long.
Calf	A dolphin estimated to be one half the length or less of an adult, that swam next to an adult so as to be touching it at most times and that is always accompanied by an adult during the course of the observation.
Group formation (from Shane, 1990a, modified)	
Tight	Dolphins generally less than one body length apart.
Loose	Dolphins generally between 1-5 body lengths apart.
Dispersed	Dolphins generally more than 5 body lengths apart.
Variable	Dolphins irregularly spread; or a group that meet more than one of the above criteria.
Convergent	The joining of one or more subgroups to the focal group.
Physical contact	
None	No physical contact among individuals is observed during the 5 min sample.
Occasional	Physical contact among two or more individuals is occasionally observed during a 5 min sample.
Regular	Consistent physical contact among some individuals (less than one half of the total group size) is observed during a 5 min sample, while the other group members are engaged in other activities.
Intense	Consistent physical contact among the majority of the group members is observed during a 5 min sample.
Unknown	Impossible to determine (e.g. the majority of the animals are diving or feeding at depth and their behaviour is not observable).
Surfacing mode	
Calm	Most surfacing is slow and does not produce foam or water turbulence.
Active	Most surfacing is fast and produces foam or water turbulence.
Variable	Both calm and active surfacing consistently occur during 5 min sample.
Directional behaviour	
None	No directional heading and geographical movement during a 5 min sample.
Poor	Changes of direction and short stops may occur during a 5 min sample, although consistent geographic movement does occur.
Straight	Steady directional heading and geographic movement during a 5 min sample.
Zig-zagging	Continuous zig-zagging movements during a 5 min sample.
Behavioural states	
Travel	Moving steadily in one direction (Shane, 1990a) during the 5 min sample.
Dive	No steady directional movement; dives longer than 30 sec occurring during the 5 min sample.
Feeding	Obvious feeding activities performed close to the water surface (Shane, 1990a); dolphins are often seen catching fishes by pursuing them parallel to the water surface or moving in circle and diving in the same small area; birds usually concentrated over the dolphins (Bearzi *et al.*, 1999) during the 5 min sample.
Socialise	Some or all group members in almost constant physical contact with one another, oriented towards one another and often displaying surface behaviours; no steady directional movement (Shane, 1990a) during the 5 min sample.
Play	Any activity that incorporates the use of a 'foreign object' during the 5 min sample.
Milling	Moving in varying directions in one location, but showing no surface behaviours and no apparent physical contact between individuals; usually staying close to the surface (Shane, 1990a) during the 5 min sample.
Others	Any other behaviour not included in the above list.
Boat disturbance (Würsig *et al.*, 1998, modified)	
Avoidance	When an individual or focal group moved away from the vessel or appeared to dive in response to the vessel.
No response	The animal/s showed no apparent response relative to the approach or pass-by of the vessel.
Approach	The animal/s moved toward the vessel during at least part of the observation period.
Bowride	Special case of an approach response.

Data analyses were performed using Statview 5.02 and Grapher 3.02; data on species distribution were plotted with Arcview GIS 3.2 and Surfer 6.02. For sighting frequency analysis, different sightings of the same individual during the same day were considered only once to avoid pseudo-replication.

Definitions

Aggregation refers to distances between one or more individuals of two different species being less than 100m, and *close aggregation* as instances when the distance between one or more individuals of two different species was about 1m. A *mixed group* is an interspecific group of cetaceans in continuous aggregation with each other for at least 10 minutes, displaying similar activities at least during part of the aggregation.

A *focal group* is any group of animals (of the same species) observed in association, moving in the same direction and usually engaged in the same activity (Shane, 1990a). Groups of animals not belonging to the observed focal group and spotted at distance were recorded but their numbers were excluded from group size calculations.

A *dolphin school* is one where all dolphins (of the same species) are in continuous association with each other and within visual range of the survey team (Weller, 1991).

A *behavioural state* is defined as a broad category of activities, such as feeding behaviour, that integrates a number of individual behaviour patterns into a recognisable pattern (Table 2; Weaver, 1987). *Mating* refers only to copulation occurring between individuals of different sexes.

RESULTS

Field effort

Data were collected during 47 inshore surveys, 41 offshore surveys and 123 combined inshore/offshore surveys in the bay for the years 1997-2001. A total of 516h was spent searching for cetaceans in good weather conditions (Beaufort scale ≤ 2), while 269h were spent observing 331 cetacean groups encountered during sightings lasting on average 40 minutes (range 2-266 minutes); 157 bottlenose dolphin schools were photographed during sightings lasting, on average, about one hour (mean=59.6 minutes, SD=41.97, SE=3.38, range 3-266 minutes, n=154; Table 1).

Occurrence, distribution and site fidelity

The bottlenose dolphin was the species most frequently sighted year-round (48.5%, *n*=157 schools) in Santa Monica Bay. Sighting of only a single school per day was most common (62.0% (*n*=131) of all survey days (*n*=211)). Multiple schools, which ranged between two and six schools, were sighted on 12.3% (*n*=26) of all survey days.

The sighting frequencies (sightings per hour) and the total sightings for this species are presented in Table 3 and Fig. 2a. The presence of other groups not included in the focal groups was recorded during 9.6% of the sightings (*n*=15). A significant difference in the number of sightings was observed (*t*=4.1, DF=4, *P*<0.05), with more sightings during 1998 than in all other years. A significant difference in the sighting numbers was also observed by season (Winter: Jan-Mar, Spring: Apr-Jun, Summer: Jun-Sep, and Autumn: Oct-Dec) for each of the five years (1997: *t*=3.6, DF=3, *P* <0.05; 1998: *t*=8.9, DF=3, *P* <0.001; 1999: *t*=6.7, DF=3, *P* <0.001; 2000: *t*=10.2, DF=3, *P* <0.001; 2001: *t*=15.6, DF=3, *P* <0.001). In 1997, 2000 and 2001, the sightings were more frequent during winter whereas in 1998 and 1999 they were more frequent in spring and summer.

Table 3

Sighting frequency (sightings/hour) of bottlenose dolphins in the bay for the years 1997-2001.

	1997	1998	1999	2000	2001	Total
Number of sightings	19	61	33	24	20	157
Sighting frequency	0.13	0.27	0.19	0.16	0.14	0.19

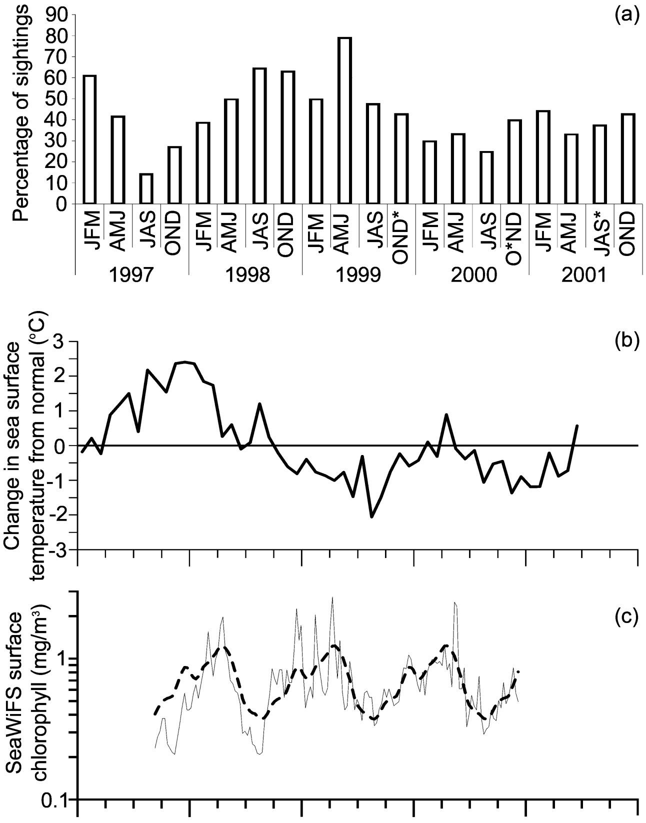
The presence of calves was recorded during 43.3% of the sightings (*n*=61) with a range of 1-6 calves per sighting (mean=1.0, SD=1.12, SE=0.09, *n*=141).

Bottlenose dolphins were observed with other cetacean species in 11.5% of the 157 sightings: common dolphins, *Delphinus delphis* or long-beaked common dolphins, *D. capensis*, 50.0% of mixed sightings, *n*=9; Risso's dolphins, *Grampus griseus*, 33.3%, *n*=6; Pacific white-sided dolphins, *Lagenorhynchus obliquidens*, 5.5%, *n*=1; common minke whales, *Balaenoptera acutorostrata*, 5.5%, *n*=1; and gray whales, *Eschrichtius robustus*, 5.5%, *n*=1. All mixed groups were observed in offshore waters except one aggregation with a single common dolphin and one with a single gray whale that occurred in inshore waters.

The potential effects of the 1997-98 El Niño were examined by comparing percentages of bottlenose dolphin sightings to variations in sea surface temperature and variations in surface chlorophyll (Fig. 2a,b,c). Visual inspection of the data suggested no apparent correlations between temperature changes brought about by the 1997-98 El Niño event and changes in bottlenose dolphin occurrence in the study area.

To identify distinct individual dolphins for the study area, matching procedures focused on 138 sightings (87.9% of total bottlenose dolphin sightings). A total of 290 distinct individuals (36.4% of total identified and resighted individuals, *n*=797) were recognised in the study area between 1997-2001.

The discovery curve (rate at which new individual dolphins were identified) is presented in Fig. 3. Most individuals were first identified during 1998. The number of new identifications gradually decreased after this year but has not yet levelled off.



*No data collection.

Fig. 2a,b,c. (a) Percentages of total bottlenose dolphin sightings recorded during 3-month periods (Jan-Mar, Apr-Jun, Jul-Sep, Oct-Dec) in Santa Monica Bay. (b) Variations of sea surface temperature anomalies during January 1997 – July 2001 for Santa Monica Bay based on Pathfinder AVHRR SST data (courtesy of N. Nezlin). (c) Variations of remote-sensed surface chlorophyll during 1997-2000 for Santa Monica Bay. The dashed line indicates averaged over the entire period of observations seasonal cycle (from Nezlin *et al.*, 2003).

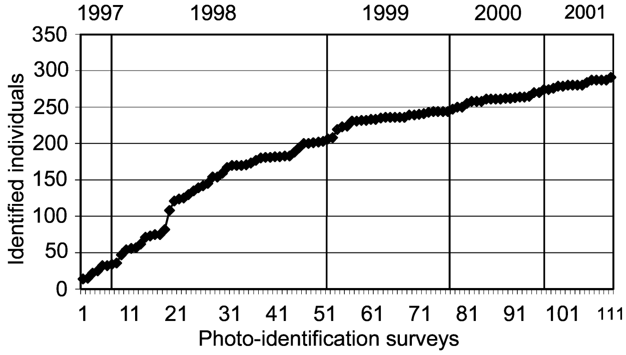


Fig. 3. Cumulative rate of identification of new individuals over time ('rate of discovery') in the years 1997-2001 for Santa Monica Bay.

The sighting frequencies for the identified dolphins ranged up to 15 days (mean=2.8, SD=2.18, SE=0.12, *n*=290; Fig. 4); 38.0% (*n*=111) were sighted only once while 22.0% were sighted five or more times. The identified individuals were observed over a one or two year period (mean=1.6, SD=0.81, SE=0.04, range 1-4, *n*=290), generally during more than one season, indicating a low degree of site fidelity as far as Santa Monica Bay is concerned. The majority of the individuals were sighted during spring in the years 1998-2001 and during winter in 1997 (see Bearzi, M., (2003) for a complete list of identified and resighted individuals during different seasons in the years 1997-2001).

Among the 290 catalogued individuals, 43 (14.8%) were classified as female. No significant differences between seasons in the number of calves sighted were observed (DF=3, *t*=3.83, *P* >0.05).

The distribution of bottlenose dolphins in relation to the bathymetry of the bay is presented in Fig. 1. The species was found regularly in inshore waters (<500m; 79.6%, *n*=157), most often within 50-100m of the shoreline, and less often in offshore waters (>500m) near submarine canyons and escarpments. To determine whether identified individuals exclusively frequented inshore waters as reported by other authors (Hansen, 1990; Hanson and Defran, 1993), the numbers of individuals observed both in inshore and offshore waters were determined. Most individuals (80.0%, *n*=157) were observed within 500m of shore; 36 individuals were seen only in offshore waters up to 15 miles from the coast (12.4%); but 44 individuals (15.2%) were sighted both in inshore and offshore waters.

No marks and scars that could clearly be attributed to sharks were observed, although a few individuals carried unusual wounds that might have been caused by either predatory attack or severe physical contact (e.g. rocks, boat propellers, etc.). No interactions between sharks and bottlenose dolphins were seen; killer whales, another potential predator of dolphins (Würsig and Würsig, 1979), were rarely observed in the bay and never in inshore waters.

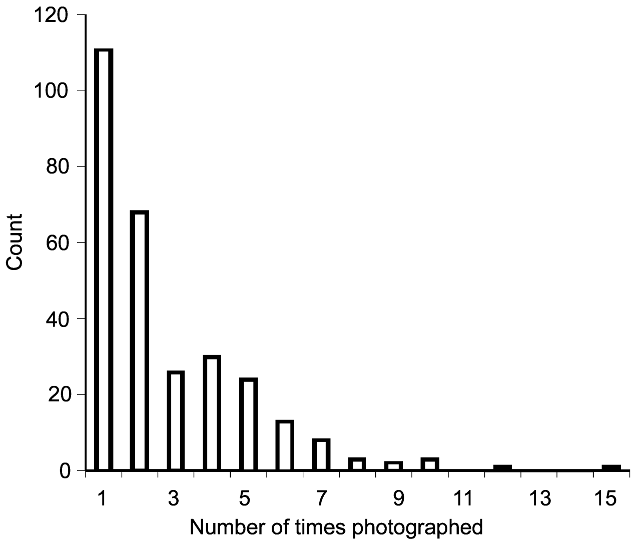


Fig. 4. Sighting frequencies for dolphins identified from 1997 to 2001.

Group sizes and group formations

Mean group sizes for inshore and offshore bottlenose dolphin schools inhabiting the study area are compared with group sizes reported by other authors in Table 4. The mean group size of inshore schools (8.8) was significantly different from offshore schools (15.0) with the largest groups observed offshore (mean difference=5.7, *t*=2.81, DF=29, *P* <0.001). Single animals constituted 3.4% of the samples (*n*=5).

The most frequent group formation observed was *variable* (inshore schools: 67.5% of the 5 minute samples; offshore schools: 79.0% of the 5 minute samples, *n*=253 calculated on a subset of data selected at random), followed by *tight* (inshore schools: 18.2% of the 5 minute samples; offshore schools: 10.3%), *dispersed* (inshore schools: 6.5%; offshore schools: 8.3%) and *loose* (inshore schools: 7.6%; offshore schools: 2.0% of the 5 minute samples). No significant difference was observed between inshore and offshore schools for the four group formation categories (*t* =0.01, DF=3, *P* >0.05).

Table 4

Group sizes for inshore and offshore bottlenose dolphin schools in Santa Monica Bay and mean group sizes reported by other authors.

	Mean	SD	SE	N	Min.	Max.
Bottlenose dolphins	10.1	7.60	0.63	145	1	57
Inshore (<500m from shore)	8.8	5.31	0.49	115	1	35
Offshore (>500m from shore)	15.0	12.05	2.20	30	1	57

	Mean	Sources
Bottlenose dolphins	19.8	Defran and Weller (1999), Southern California, inshore dolphins.
	18	Hansen (1990), Southern California, offshore dolphins.
	10	Scott and Chivers (1990), Eastern tropical Pacific Ocean, inshore dolphins.

Behavioural patterns

The behavioural budget recorded for bottlenose dolphins is presented in Table 5 and Fig. 5. The budget shows a predominance of *Travel* (49.0%; *n* 5 minute sample=1,757) and *Dive-Travel* (20.0%) activities. *Feeding* was observed in 5.0% of the sightings, also in association with other activities such as *Travel* (*Travel-Feeding*: 4.5%), *Socialise* (*Feeding-Socialise*: 4.5%) and *Dive* (*Dive-Feeding*: 1.5%).

Table 5

Overall behavioural state budget recorded for bottlenose dolphins in Santa Monica Bay. Behavioural state data at less than 99.5% level were not included in the table. A hyphen between two behavioural states refers to activities performed simultaneously by different focal group individuals during 5 min sample (e.g. dive-travel, socialise-travel, etc.). A comma between two behavioural states refers to activities performed by different focal group individuals to express consecutive behaviours occurring during 5 min sample (e.g. travel, feeding means travel followed by feeding).

Behavioural states	No. of 5 min samples	Frequency distribution of observed behaviour	Total 5 min samples states
Travel	861	49.01	1,757
Dive-travel [a]	351	19.98	
Feeding	87	4.95	
Feeding-travel	78	4.44	
Socialise-travel	78	4.44	
Dive	37	2.10	
Socialise	34	1.93	
Dive, travel [b]	31	1.76	
Travel, feeding	29	1.65	
Dive-feeding	27	1.54	
Travel, dive	24	1.36	
Feeding-socialise	19	1.08	
Travel, socialise	18	1.02	
Socialise, travel	15	0.85	
Dive-travel-feeding	11	0.62	
Dive-socialise	10	0.56	
Travel-milling	9	0.51	

[a] Simultaneous behaviour occurring during 5 min sample. [b] Consecutive behaviour occurring during 5 min sample.

A significant difference was observed in the behaviour patterns observed (Fig. 5) over the five years of study (χ^2= 154.78, DF=36, *P* <0.001), with most travel and dive-travel recorded in 1998-99 and most feeding observed in 1997 and 2000.

Bottlenose dolphins travelled at an average speed of 4.3km h^{-1} (mode=1.8, SD=2.56, SE=0.07, range 0.9–16.6, *n*=1,107 from subset of 5 minute samples data selected at random). Offshore schools travelled at a significantly higher speed than inshore schools (offshore: mean=5.8, SD=1.29, SE=0.04, *n*=118 from subset of data selected at random; inshore: mean=4.2, SD=1.80, SE=0.16, *n*=118; mean difference=2.2, DF=117, *t*=13.47, *P* <0.001). During travelling, the surfacing mode was usually *calm* for both inshore and offshore schools (73.9% of the total sightings) or *variable* (25.5%); *active* surfacing was observed only rarely (0.6%).

To determine potential differences between the behavioural states of bottlenose dolphins during the El Niño years *versus* a random period of normal temperatures, different behavioural states were compared over 11 months of increased and normal sea surface temperature (high SST: June 1997-April 1998; normal SST: June 1999-April 2000). No significant difference was observed between the most common behavioural states seen during years of high SST versus normal years (χ^2= 63, DF = 56, *P* >0.05).

Physical contact among two or more individuals in a school was recorded to be *occasional* 21.6% of the time during 5 minute samples (*n*=1,515), *regular* 3.4% of the time and *intense* 2.8% of the time. Mating behaviour was only recorded during 14 sightings (8.9%, *n*=157) in the study period.

DISCUSSION

Occurrence, distribution and site fidelity

The bottlenose dolphin was the species most often observed in Santa Monica Bay, followed by long-beaked common dolphins and common dolphins (Bearzi, M., 2003; see also Bonnell and Dailey, 1993; Forney and Barlow, 1998). Other cetaceans were occasional or rare inhabitants of the bay (Bearzi, M., 2003).

The distribution of bottlenose dolphins in the study area was largely within 500m of shore, in agreement with prior observations off the San Diego coastline (Defran and Weller, 1999; Defran *et al.*, 1999). However, they were also occasionally observed aggregating offshore near submarine canyons and escarpments (Fig. 1), where prey may be relatively more abundant (Hui, 1979). In Santa Monica Bay, bottlenose dolphins and the two common dolphin species generally differed both in distribution and prey preference, which has been suggested to be due to habitat partitioning, as a consequence of prey specialisation and competition for resources in inshore waters (Bearzi, M., 2003).

El Niño, is a pole-ward propagation of nutrient-poor warm water along the coast of Western America caused by a break-down of trade wind circulation (Tomczak and Godfrey, 1994). It usually reduces primary productivity throughout most of the coastal Eastern Pacific (Cane, 1983) and has been correlated with shifts in the distribution of

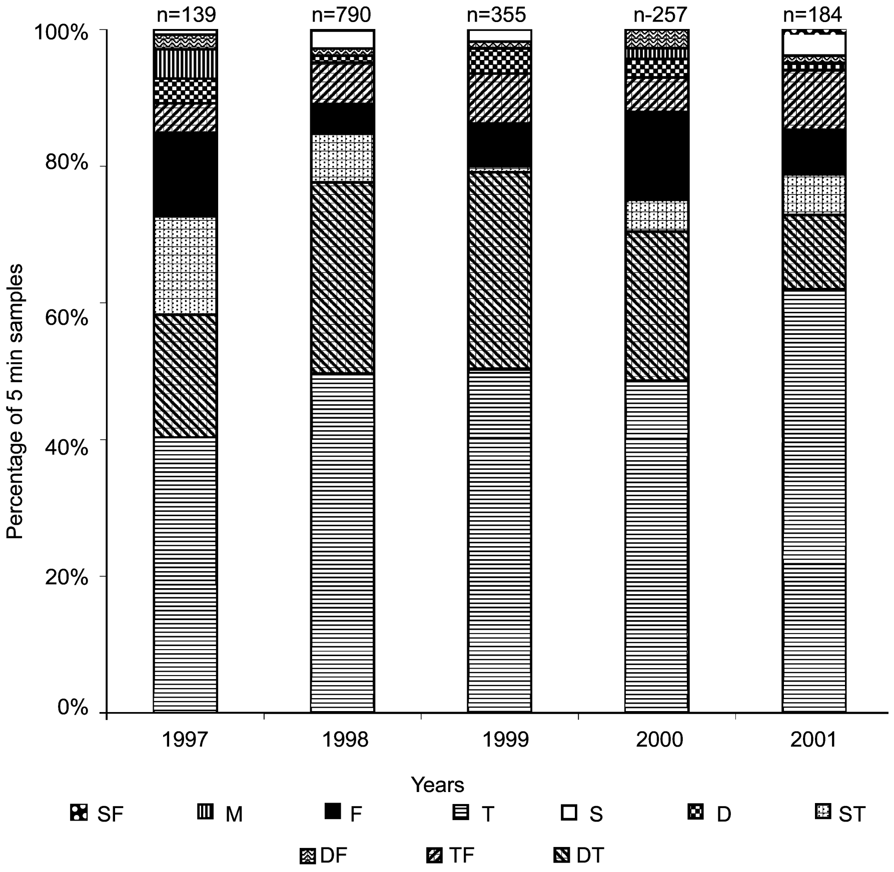


Fig. 5. Behavioural budget computed for the 10 most common patterns of activities during the study period. T=Travel, DT=Dive-Travel, ST=Socialise-Travel, F=Feeding, TF=Travel-Feeding, D=Dive, M=Milling, DF=Dive-Feeding, S=Socialise, SF=Socialise-Feeding. These behaviours include activities performed simultaneously and in sequence by different focal group individuals during 5 minute sample.

marine mammals and their prey (Shane, 1995a; b; Defran *et al.*, 1999; Hill, 1999). By contrast, La Niña is characterised by upwelling-favourable trade winds that restore levels of primary productivity (Chavez *et al.*, 1999). In the present study: (1) bottlenose dolphins occurred throughout the period, including during the strong 1997-1998 El Niño and the following La Niña events of 1998-2001; and (2) the number of bottlenose dolphin sightings peaked during the winter months in the years 1997, 2000 and 2001 and during spring and summer in 1998 and 1999, showing a high number of sightings during the El Niño event.

During the 1982-83 El Niño coastal bottlenose dolphins off San Diego were observed to extend their range to central and Northern California; this was believed to be a response to prey abundance fluctuations (Wells *et al.*, 1990; Dailey *et al.*, 1993). No systematic data were collected that would suggest a similar northward migration of coastal dolphins during the 1997-98 event, but K.J. Dudzik (pers. comm.) observed a slight decline in numbers of this species along the San Diego coastline at this time that may have contributed to the high numbers of dolphin sightings in Santa Monica Bay. The apparent lack of severe effects may be explained by prey abundance during these years for the Southern California Bight. Whilst the 1997-98 El Niño event had a dramatic negative affect on several important fisheries (e.g. decreases in squid and anchovy catches), some fisheries including the Pacific sardine (*Sardinops sagax*), white seabass (*Atractoscion nobilis*) and splitnose rockfish (*Sebastes diploproa*), showed improved catches in the Santa Monica Bay area (California Department of Fish and Game, 2000). Bottlenose dolphins are opportunistic species that feed on different prey, based on availability and abundance (Bearzi, G. *et al.*, 1999). During the El Niño years, bottlenose dolphins may have fed on species that were more abundant in Santa Monica Bay, eliminating the need to leave the study area.

The rate at which previously unidentified dolphins were discovered in the study area increased most rapidly in 1998 (when effort was highest). The discovery curve of new individuals for this study was comparable to the trend reported for Southern California (Weller, 1991; Defran and Weller, 1999), but it was in contrast with some other studies around the world where asymptotes were recorded over shorter periods of time (Wells, 1986; Ballance, 1990).

The number of identified dolphins in the study area was lower than the number reported for the San Diego area, approximately 140km to the south (n=290, this study, vs n=373, Defran and Weller, 1999), but given the discovery curve shown in Fig. 3, it is probable that not all dolphins had been identified by the end of this study in 2001.

The variability in bottlenose dolphin sightings and the generally low individual sighting frequencies were consistent with data from San Diego (Defran and Weller, 1999). The variations in time between many of the resightings of identified dolphins in Santa Monica Bay suggested that the area represents part of a larger 'home range'[2] within the California coast. Defran *et al.* (1999) reported that 58.0% (n=120) of a total population of 207 individuals exhibited back-and-forth movements over 470km of coastline between three discrete regions where photo-identification studies were conducted: Santa Barbara and Orange County, California; and Ensenada, in Baja California, Mexico, with no evidence of site fidelity to any particular area. These authors concluded that the high mobility of dolphins within a relatively narrow coastal zone reflected the extremely dynamic nature of this coastal ecosystem and the associated patchy distribution of food resources available (Dailey *et al.*, 1993). Previous work has also shown that movements by bottlenose dolphins were more evident where temperature and prey abundance fluctuate seasonally (Wells *et al.*, 1990; Bräger *et al.*, 1994).

In Santa Monica Bay, located between Santa Barbara and San Diego (see Fig. 1), most of the identified individuals were not observed year-round, showing the absence of a strong residency pattern. This suggests that they belonged to the same highly mobile and behaviourally flexible open coastal population observed by Defran *et al.* (1999), but has not yet been confirmed by comparing the two photo-id catalogues. However, in Santa Monica Bay: (1) at least some individuals appeared to utilise the area on a seasonal basis; and (2) the overall proportion of dolphins sighted only once in the bay (38.0%; n surveys in 1997-2001=211) was significantly lower than the proportion reported in other areas along the California coast (71.0% Orange County, 69.0% Ensenada, 53.0% Santa Barbara, n surveys in 1981-1989=241; Defran *et al.*, 1999), showing some degree of fidelity by bottlenose dolphins to Santa Monica Bay.

Although most identified individuals were observed within 500m of shore, some of the same individuals were also observed in offshore waters (>1km from shore). This contrasts with data reported for the Southern California waters where dolphins were almost always encountered within 1km of shore, showing a high fidelity to a 'coastal corridor' between 10 and 30m depths, with no matching between inshore and offshore individuals (Shane, 1994; Defran and Weller, 1999; Defran *et al.*, 1999). Although bottlenose dolphins in Santa Monica Bay also showed a preference for a coastal corridor, they did not adhere to a rigorous 1km boundary. This may be due to the different bathymetry and oceanography of the two study areas. Presumably, the presence of the same individuals in both inshore and offshore waters for Santa Monica Bay was related to the presence of submarine canyons and escarpments (Dartnell, 2000), which are optimal features for mixing of nutrients and consequently are rich in prey for dolphins. On the contrary, the San Diego area showed different oceanographic characteristics. There, the open coastal waters were relatively dynamic with a substantial variability in water temperature and, consequently, in abundance and composition of patches of prey over years or decades (Dailey *et al.*, 1993; Defran *et al.*, 1999). Weller (1991) reported that these variations were responsible for the behavioural flexibility demonstrated by bottlenose dolphins to changes in habitat ecology along the Southern California coastline.

In conclusion, the bottlenose dolphins of Santa Monica Bay appear to exhibit similar movements to the dolphin population of the open coastal waters of the San Diego area but they also showed some characteristics common to individuals living in more protected areas. They exhibited some degree of site fidelity, inshore-offshore movements (Connor and Smolker, 1985; Wells *et al.*, 1987; Bearzi, G. *et al.*, 1997) and an apparent indifference to El Niño shifts.

Group sizes and group formations

There was a substantial range in observed group sizes for bottlenose dolphins in Santa Monica Bay (range 1-57 individuals). The results of this study are similar to the findings of Hansen (1990) and Scott and Chivers (1990;

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[2] Home range was defined following Burt (1943) as that area traversed by the individual in its normal activities of food gathering, mating and caring for young.

Table 4). The higher difference in group size for inshore dolphins reported by Defran and Weller (1999) off San Diego in comparison to this study may be due to the use of dissimilar methodologies and definitions.

Bottlenose dolphins showed an increased group size from inshore to offshore waters, in agreement with the findings of Defran and Weller (1999) for Southern California. A similar trend for this species was observed by Scott *et al.* (1990) in Florida waters, where school sizes increased with water depth. As suggested by Wells *et al.* (1980), larger group sizes may benefit from cooperative feeding on patchy, rich food resources found in deeper habitats. In Santa Monica Bay, the increase in group size from inshore to offshore waters was perhaps a response to a patchy distribution and abundance of prey (Dailey *et al.*, 1993). Some authors have suggested that predation may be responsible for increased group sizes in dolphins (e.g. Norris and Dohl, 1980; Norris and Schilt, 1988; Scott and Cattanach, 1998). However, the low numbers of scars inflicted by sharks on identified individuals and the low number of predators observed at the surface in the study area suggested that predation pressure was not a major factor in determining group size.

Offshore bottlenose dolphins were also often found in aggregations with other cetaceans, which effectively increased school size. The presence of mixed species aggregations in offshore feeding grounds rich in prey can promote schooling behaviour of prey and facilitate the capture of food for one or both predators (Magurran, 1990; Similä and Ugarte, 1993; Norris and Johnson, 1994).

Although bottlenose dolphin schools showed a *variable* group formation, with no significant difference observed between inshore and offshore schools, inshore groups were sighted more often in a *tight* group formation than offshore groups. The *tight* formation is possibly related to a different feeding strategy used by dolphins when food is less patchily distributed (Wells *et al.*, 1980).

Behavioural patterns

Behavioural data collected for bottlenose dolphins in Santa Monica Bay were similar to those reported for the San Diego coastline (Hanson and Defran, 1993). In both areas, bottlenose dolphins spent a fairly high amount of time travelling (this study: 69.0% travel plus dive-travel; San Diego: 63.0% travel plus dive-travel). Feeding activities were observed 19.0% of the time near San Diego and 16.0% in Santa Monica Bay. Different methodologies and definitions probably account for the minor differences in behaviour recorded in the two areas. The relatively high proportion of behavioural states such as travel and feeding across seasons is likely to have been related to a year-round occurrence of prey, as also reported by Hanson and Defran (1993). Although feeding at the surface was observed occasionally, the rather large amount of time spent dive-travelling and diving may have been related to food searching or feeding activities not directly observed by the researchers (Bearzi, G. *et al.*, 1999). The significantly greater amount of travel and dive-travel recorded during the years 1998-99 for the study area may have been linked to an increased need to forage for prey, which had become more patchily distributed during those two years.

Bottlenose dolphins in Santa Monica Bay were regularly observed travelling in a *calm* surfacing mode and at speeds usually lower in comparison to those calculated for bottlenose dolphins in other areas worldwide (this study: 4.3km h^{-1}; Würsig and Würsig, 1979: 6.1km h^{-1}; Shane, 1990a: 5.5km h^{-1}; Bearzi, G. *et al.*, 1999: 7.1km h^{-1}). Considering that travelling has the primary function of locating food and conspecifics (Shane, 1990b), bottlenose dolphins in the study area may have travelled at lower speed to inspect shallow waters and forage for prey along the shore. In addition, a recurrent transition among behaviours indicative of foraging were often observed (Table 5), suggesting a consistent effort devoted to feeding-related activities, as also reported by Bearzi *et al.* (1999). Other behavioural activities unrelated to foraging and feeding were occasionally seen year-round during the study period, with irregular physical contact and mating observed mostly during socialising.

Bottlenose dolphins spent most of their time moving along favourite 'corridors' within 50 metres of shore, as also recorded by Defran and Weller (1999). Frequent direction changes along the coastline were often observed, which are likely to be related to prey movements. Dolphin travel activities were often followed by feeding activities in spots that may have reflected the presence of demersal prey on sand flats. On a larger scale than the study area, Defran *et al.* (1999) showed the high mobility of inshore bottlenose dolphins along the California coast is probably due to a shift in prey distribution. It is likely that the same dolphins frequented both the San Diego coastline and the study area following prey distribution.

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