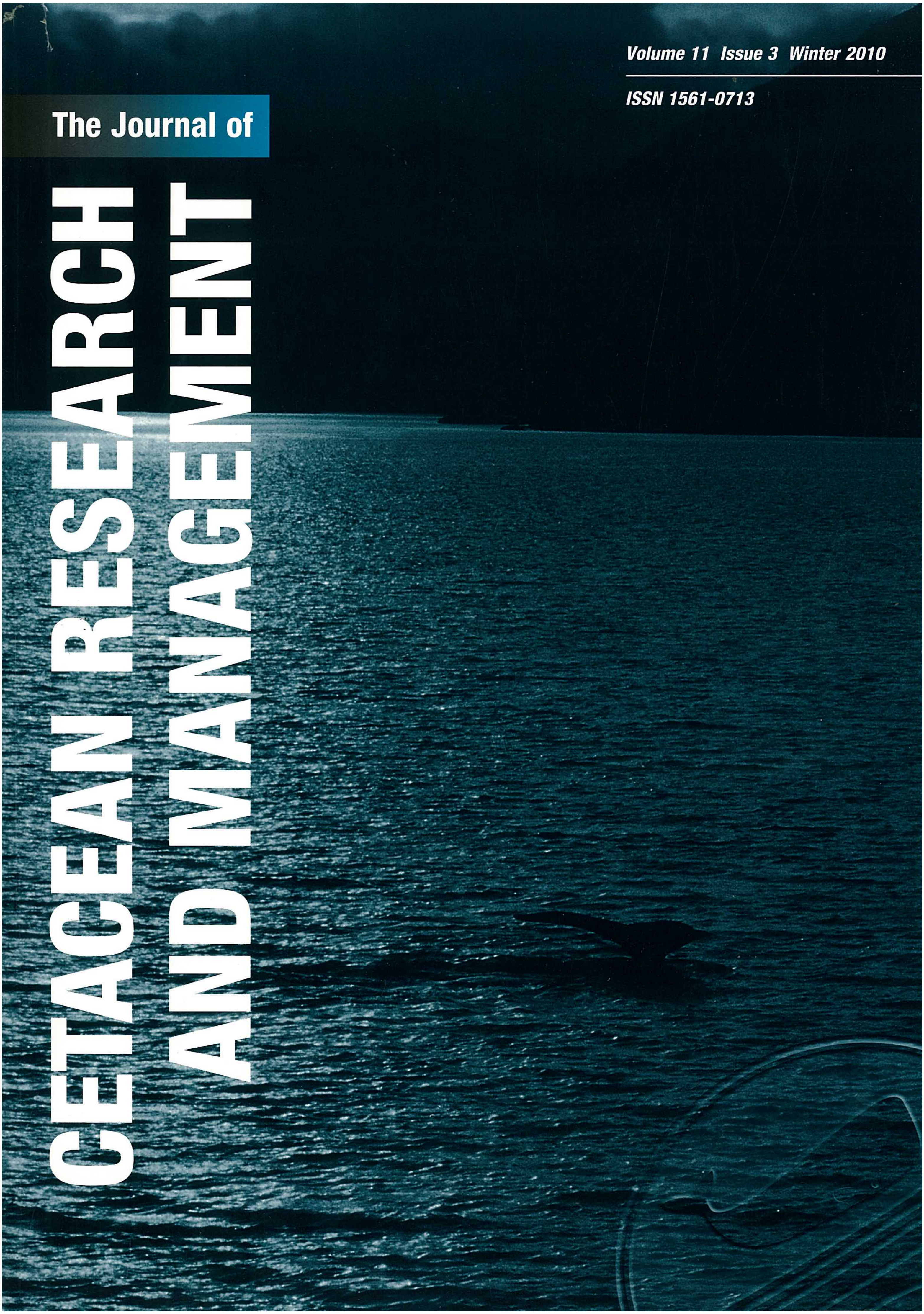

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The Journal of CETACEAN RESEARCH AND MANAGEMENT

Contents

Editorial

Welcome to this the final issue of Volume 11 of the *Journal of Cetacean Research and Management*.

2009/2010 have been challenging years for the Journal with respect to both printing companies and personnel (congratulations to the two mothers!). These have now been overcome and the Journal has also developed a completely online submission and review system that will begin to be used from June 2011[1]. A total of 31 papers have been published in Volume 11 covering a wide range of subjects related to the conservation and management of cetaceans. There have been a total of over 100 authors from Europe, Asia, Australia, Africa and North and South America. The first supplement to Volume 11 included the full report of the Scientific Committee held in Santiago, Chile in May 2008 and the results of six intersessional workshops. The second supplement to Volume 11 included the full report of the Scientific Committee held in Funchal, Madeira, Portugal in May/June 2009 and the results of eight important intersessional workshops. The author and subject index for Volume 11 can be found at the end of this issue.

The present issue contains 11 papers covering a wide range of subjects related to conservation.

Without knowledge of abundance and trends, it is difficult if not impossible to assess the status of populations and prioritise the anthropogenic activities that may impact upon them. Many of the papers in this volume contribute to developments in both practical and theoretical ways to the development of better estimates or produce estimates themselves. 'Distance'-based line transect methods are amongst the most commonly used in the cetacean community. Gillespie *et al*. and Leaper *et al.* provide companion papers examining the practicalities of integrating data collection on such surveys with an emphasis on better estimation of distance and angle – the key parameters needed for distance-based estimation methods. Schwarz *et al.* examine the implications of two alternative field approaches (closing mode and passing mode) for dolphin surveys in the eastern tropical Pacific whilst Kleppe *et al.* examine the implications of the choice of hazard probability function on effective strip width. Koski *et al.* review the possible use of unmanned aerial systems to provide quantitative information on cetacean abundance in offshore areas.

Two papers provide abundance estimates: Garrison *et al.* examine the habitat and abundance of cetaceans in the continental slope waters off the US southeastern Atlantic coast while Lauriano *et al.* provide estimates of abundance of striped dolphins in the Mediterranean Pelagos Sanctuary.

Anthropogenic activities that may affect population status range from direct harvesting to whalewatching. Frost and Suydam provide information on the subsistence harvest of white whales in Alaska while Montero-Cordero and Lobo provide information on the effect of tourist vessels on the behaviour of pantropical spotted dolphins off Costa Rica. Before designing appropriate methods to estimate abundance and trends it is important to understand the distribution of cetaceans in the area. Minton *et al*. provide the results of surveys undertaken in the coastal waters of the Sultanate of Oman that can provide baseline data for future conservation efforts in the region.

The final paper in the volume provides a major review of the 28 species of toothed whales found within the boundaries of the IWC's Southern Ocean Sanctuary, incorporating data and information from many different studies in the area including the IWC's research cruises carried out under the IDCR and SOWER programmes over a period of 30 years.

Finally, I would like to thank the 50 scientists that have acted as anonymous reviewers for the papers published in Volume 11 (M. Andre; R. Angliss; R. Baldwin; L.T. Ballance; J. Bannister; S. Berrow; P.B. Best; J. Brandon; M. Bravington; R.L. Brownell; D. Butterworth; J. Calambokidis; A. Canadas; C. Carlson; J.G. Cooke; P. Corkeron; P. Ensor; J. Forcada; C.M. Fortuna; T. Gerrodette; A. Gilles; G. Givens; P. Hammond; L. Harwood; S. Hedley; M.-P. Heide-Jorgensen; S. Ingram; R. Kenney; J. Laake; F. Larsen; J. Lawson; R. Leaper; W. McFee; G. Notarbartolo Di Sciara; N. Øien; D. Palka; S. Panigada; D. Pike; A. Punt; A. Read; R. Reeves; S. Reilly; J. Savelle; T. Schweder; L. Thomas; P. Thompson; K. Van Waerebeek; R. Waples; R.S. Wells; J. Zeh). Without their diligence and hard work, the papers in the *Journal*, and more importantly the contribution they make to the wise management and conservation of cetaceans, would be considerably poorer. A full list of the reviewers and their affiliations can be found at: *http://www.iwcoffice.org/publications/contents_reviewers.htm#review.*

G. P. DONOVAN
Editor

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[1] *http://www.iwcoffice.org/publications/JCRM.htm*

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An integrated data collection system for line transect surveys

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ABSTRACT

A computer based system for the collection of line transect survey data is described. The primary goals of the system were to measure (rather than estimate) distances and angles wherever possible, to provide accurate time-stamps for surfacing events as an aid to duplicate identification and to facilitate accurate data collection by using computers to automate data collection wherever possible. Distance and angle measurements were made using established photogrammetric techniques. Collection of photogrammetric data from video was automated and included a system of data buffering so that several seconds of data prior to each observer sighting could be captured. An additional goal of the system was to eliminate the need for post-cruise data entry and validation through the use of on-board data validation software. The system was successfully used during the 2005 SCANS-II and the 2007 CODA surveys.

KEY WORDS: SURVEY–VESSEL; PHOTOGRAMMETRY

INTRODUCTION

Visual surveys to estimate cetacean abundance rely on observers detecting cues from the target species during brief periods when animals are at the surface. A common analysis approach for line transect surveys is to use the observed locations at which animals are initially sighted to estimate the relative detection probability as a function of perpendicular distance from the track-line (Buckland *et al.*, 2001). Critical data for such analyses include species, group size and sighting location relative to the vessel (usually recorded as range and bearing). More complex analyses that attempt to estimate absolute detection probability frequently use more than one independent team of observers and require some method of assessing whether sightings reported by different observers are of the same animal or group and can thus be classified as duplicates. Detection probability can also be a function of several covariates including number of observers, sea state and weather conditions. These effort-related parameters need to be recorded along with the track of the survey vessel. On surveys, data are often collected in difficult conditions, for example in a cramped location exposed to strong winds and cold and observers are required to continuously scan the sea; all factors that make it difficult to make accurate written notes. A further complication is that for some methods (such as those relying on independent observers) it can be important that independent observers are unaware of the observations of others, while other personnel (e.g. data recorders or duplicate identifiers) need to be able to receive data from all observers.

The potential for computerised data entry systems has long been recognised. However there are considerable challenges to operating complex electronic systems at sea and an understandable reluctance to move away from simple reliable systems based on pencil and paper. The use of paper forms takes the attention of observers away from looking for whales and considerable subsequent effort is required to enter such data into a computer for analysis. An additional disadvantage is that it is not possible to make use of an automatic time-stamp. Simple dictaphones to record verbal commentaries have also often been employed. These have the advantage over paper forms that the observer does not have to take their attention away from the sighting and it is frequently possible to identify events with an automatic time-stamp. However, data still need to be transcribed offline, which can be a more onerous task with verbal recordings than with paper forms.

One of the first real-time computerised data entry systems was the Logger software developed by Lex Hiby and Phil Lovell of Conservation Research Ltd in the early 1990s. This was intended to facilitate data collection during commercial whale watch cruises where it was required that data could be entered in a standardised manner with a minimum of effort by people primarily involved in other activities (Leaper *et al.*, 1997). The original Logger software ran under the MS-DOS operating system. It automatically logged GPS and wind instrument data and also had a fixed number of forms for manual entry of effort, environmental and sightings data. Apart from the content of drop down lists, the forms could not be altered by the user. Following several years of successful use by a number of groups, a new version, Logger 2000[1], was developed with a much more flexible user interface, enabling users to create any number of data entry forms with user defined data entry fields and also interface to external hardware, such as sightings buttons, sound cards and video cameras. This has been used to collect survey and behavioural data on a number of studies in the past decade, (e.g. Gillespie *et al.*, 2005; Matthews *et al.*, 2001). This software forms the basis for the integrated data collection system developed for the SCANS-II survey in 2005 (SCANS-II, 2008) and subsequently used on the CODA survey in 2007 (CODA, 2009).

[1] The Logger 2000 software was developed by the International Fund for Animal Welfare to assist with benign research on cetaceans and is available to download from *http://www.ifaw.org/sotw*.

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Even with the addition of real-time computer data entry, the majority of surveys still rely entirely on human observers to estimate and collect key data items, with limited scope for identifying or rectifying errors. This contrasts with most other fields of science and engineering where the use of calibrated instruments to take and record measurements is considered the norm. The SCANS-II system attempted to measure data wherever possible, to record data in ways that allowed errors to be identified, and allowed backwards comparability with previous surveys.

The problems of using subjective human judgement to estimate the radial distances and angles that distance based methods rely on have been identified in a number of studies (Leaper *et al.*, 1997; Schweder, 1997) but rarely accurately quantified. This is because attempts to quantify errors in judgement have relied on experiments that were not fully representative of the process involved (Williams *et al.*, 2007). In addition, survey data may be subject to recording errors, but during most surveys there is generally little scope for validating data beyond excluding values that are beyond the possible range. Systematic bias in distance and angle data will result in biased estimates of total abundance. In addition, random errors can affect both the accuracy and precision of estimates. Surveys using independent observers also rely on matching data on time, location and a description of what was seen in order to identify duplicate sightings. Failure to correctly identify duplicates can also cause bias in abundance estimates (Hammond *et al.*, 2002).

When an observer detects a sighting cue the key data are species, group size, time, distance and bearing together with ancillary data related to behaviour. For observers using binoculars mounted on a stand, a protractor on the stand is generally used to read out angles relative to the vessels heading and distances are measured through use of binocular reticules. Reticule readings allow the angle of dip between the horizon and the whale to be measured and this allows the distance from an observation platform of known height to be calculated. These measurements are difficult to make when the object (a surfacing cetacean for example) is only fleetingly visible and the boat itself is pitching and rolling in a seaway. For observers searching with the naked eye, angles are generally measured through use of an angle board (a protractor fixed to the vessel, with a movable pointer which the observer lines up on the animals or in the direction of their last observed location). Estimation by eye is the primary method of distance measurement, with observers generally being trained in distance estimation at the start of the survey using objects at known distances (e.g. a navigation buoy tracked with the vessels radar). Observers may also use sightings sticks – a marked stick held at arms length or attached round the observers neck with a horizon mark and marks indicating various distances as an additional aid to distance estimation.

For observers using binoculars, Leaper and Gordon (2001) describe methods for measuring distances and angles using video and stills cameras mounted on the same stands as the binoculars. The distance measuring system measures the angle of dip from the horizon in the same way that reticules or distance sticks do, but provide a record of the data which can be measured carefully offline and at much higher resolution than the other methods. The angle measuring system simply takes photos of fixed marks (typically parallel lines) on the deck beneath the binocular stand. For observers using only the naked eye it is much more difficult to develop systems which will capture an image that allows the location of the sighting to be measured. Trials with low resolution helmet mounted video cameras were not successful due to the movement of the human eye within the head requiring the camera to have such a wide field of view that few sightings could be detected on the images.

This paper describes both the Logger 2000 system in general terms, giving an overview of its functionality and how it may be used in a wide variety of applications as well as more specific information about the configuration employed during the SCANS-II and CODA surveys. A separate piece of software was developed for off-line data entry, validation, processing of verbal commentaries, and photo-grammetric measurements from images. The practical problems encountered and recommendations for development of systems for future surveys are also discussed. Results are given for the overall performance of the system, but comparison of data collected by different methods during the SCANS-II survey and the potential effects of measurement error on abundance estimation are the subject of a companion paper (Leaper *et al.*, 2010).

LOGGER 2000 SOFTWARE

The Logger 2000 software has been used since 1999 by a number of research groups for the collection of both visual and acoustic data. The Logger 2000 software provides the user with a flexible interface which can be configured in a variety of ways and a number of modifications were made specifically for the SCANS-II survey in order to deal with high volumes of data coming in from five different observers. These included the option to have multiple instances of the same form open at once, the opening of forms by remote ‘action buttons’ and automatic video and stills image capture.

The Logger 2000 software stores data collected automatically (e.g. from a GPS or other NMEA compatible wind and navigation instruments or temperature probes and other sensors linked to an analog to digital data acquisition device) or entered by the operator, in a Microsoft Access database. In its most basic configuration, the software does nothing but store GPS data on a timer (default every 10 seconds) and has a form for comments entered by the operator. Under normal use, however, a number of forms will have been configured for entry of other types of data, such as sightings and environmental data. The software can also be configured to make sound recordings, capture webcam images and display forms or buttons which can initiate a variety of actions.

User defined forms

User Defined Forms (UDFs) are used to specify the characteristics of all the non-standard forms in a particular Logger configuration. Their creation is realised by adding tables to the Access database with table names beginning with the characters ‘UDF_’ and having a pre-determined column format. On start-up, Logger reads the content of the UDF tables and for each UDF table, a form for data entry and an output database table are created. For example, the existence of the table UDF_Sightings would cause the creation of a data entry form on the Logger display called ‘Sightings’ and a corresponding table in the database also called ‘Sightings’. Each row of data in the UDF table constitutes an instruction, which either sets a parameter governing the behaviour or appearance of the form, or creates a data entry field on the display panel and a corresponding column for data storage in the output data table. Any number of UDF tables may be created and the data entry forms are laid out on a tabbed panel, enabling the operator to navigate between them easily. If multiple forms of the same type are required to be open simultaneously (as is often the case with sightings forms) they are laid out on a secondary tab panel contained within the main Logger tab panel display. The Logger display also contains a map, showing the vessel’s track, coastal outline and depth contours. Summary information from each data entry form (such as the locations of sightings) can be overlaid on the logger map and there is scope for customising line colours and symbols used for plotting contours, sightings and detections.

The main data entry types allowable within the Logger 2000 UDF system are detailed in Table 1. For a complete set of commands, readers are referred to the Logger 2000 Help (available once the software is installed). Some of the data types are either preset (such as counters) or can be collected automatically (e.g. time-stamps, NMEA data). Drop down lists of selectable items, such as species or observer names can also be created. Operators may create forms that mix both automatic data and manually entered data. For instance an environmental data form may contain an NMEA data field which collects wind speed directly from the ships instruments and have a separate field where the operator enters sea state. If a form is created which only contains data fields which are filled automatically, then the software can be configured to save those data automatically either on a timer or every time GPS data are read. All data recorded by Logger are automatically cross referenced in the database to the most recent GPS data record.

Sound recording

The software can be configured to contain one or more sound recorders which acquire sound data from PC sound cards or other data acquisition devices and store the data in wave files on a hard drive. While in standby mode, the recorders can be configured to write acoustic data continuously to a circular buffer so that when recording starts, several seconds of data prior to the actual recording start time are saved. The sound recorders were originally developed to record cetacean sounds from underwater hydrophones. Recording can be continuous (in which case new files are started at regular intervals to stop individual files becoming too large), can be made at user defined timed intervals or can be started and stopped manually or triggered automatically when Logger is used in conjunction with other acoustic detection program, or the buttons described below.

Video image capture

The software can be configured to capture still images from DV camcorders, USB webcams, PCI capture cards, TV cards, USB capture devices and IEEE 1394 (Firewire) cameras. The moving video images and captured stills are displayed in a window on a Logger form for quality checking. Images are grabbed and stored in either jpeg or bitmap format in response to a user pressing a button or to a timer that can operate at fixed or randomised time intervals. A sequence of images can be stored in a buffer (similar to the buffered sound recorder) so that images recorded prior to a trigger event (button press or timer) can be stored.

Action buttons

Logger 2000 can be also configured to contain one or more forms of action buttons. As well as existing on the screen as standard software buttons which can be clicked on using the mouse, the buttons can be linked to the keyboard function keys (F1 … F12 at the top of most keyboards) and can also be linked to external physical buttons via a digital interface card (e.g. Measurement Computing USB-1208LS[2] or similar).

Button configuration is realised in a similar way to the user defined forms with UDB_ tables in the database defining each form of buttons; each row in the UDB_ table specifies a button on the form. When a button is pressed, the time-stamp and button reference code are immediately stored

Table 1

Logger data entry and command fields.

Data type	Description	Required configuration data
SHORT/INTEGER	16 or 32 bit signed integer data	
FLOAT/DOUBLE	32 or 64 bit double precision data	
CHAR	Character/text data	Maximum length
LOOKUP	Character data, selectable from a dropdown list such as a list of species or a list of observer names	A list of selectable items
COUNTER	Automatically incrementing integer number	
TIMESTAMP	Date and time data	
TIME	Time only	
NMEAINT/NMEACHAR/ NMEAFLOAT	Integer, character or floating point data from ships or instruments outputting data in National Marine Electronics Association (NMEA) format	NMEA sentence name and position within the sentence
ANALOG	An analogue voltage from either a Measurement Computing or a National Instruments data acquisition board	Channel number, channel gain, multiplicative and additive scaling factors to convert voltage (e.g. into a pressure measurement)
DIGITAL	A digital bit from either a Measurement Computing or a National Instruments data acquisition board	Channel number

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[2] Measurement Computing Corporation, 10 Commerce Way, Norton, MA 02766, USA. See: *http://www.mccdaq.com*.

in the associated output database table and a number of other actions can also be initiated:

(1) open single or multiple data entry forms;

(2) start a sound recording (the user sets which sound recorder to start and how long recording should continue for);

(3) start a series of video frame captures;

(4) send a command to a serial (RS-232) port;

(5) after a defined delay, automatically 'press' another button on the same or on a different button form. This allows sequences of commands to be created;

(6) broadcast software messages which can be picked up and acted on by other Windows programs.

Clock synchronisation

Data containing an accurate time-stamp are output by GPS systems at intervals varying between 0.5 and several seconds (more recent models tending to output data more frequently). Logger 2000 automatically updates the PC clock with a time-stamp from the GPS each time the software starts using one of the first data strings received from the GPS. All subsequent times used by the program are then read from the PC clock. This avoids either using a GPS time that is slightly out of date or having to wait for a new GPS time-stamp. During data collection, each GPS record is written to the database with both its own GPS time-stamp and the PC clock time so that any drift in the PC clock is detectable offline. In our experience, PC clocks are very accurate, it has never been necessary to correct PC clock times even when Logger has been running continuously and therefore not made further clock updates for many days. All times recorded in the database can therefore be considered accurate to better than 1s.

SCANS-II DATA COLLECTION

The SCANS-II survey was primarily designed to estimate the abundance of harbour porpoises and other small cetaceans in European shelf waters. The survey protocol closely followed that of the SCANS I survey of 1994 (Hammond *et al.*, 2002) using mark recapture distance sampling methods (Buckland and Turnock, 1992). Two teams of observers searched for animals. Two observers on the 'Tracker' platform used binoculars to search as far ahead of the survey vessel as possible, while two observers on the 'Primary' platform searched with the naked eye on either side of the track-line within 500m of the ship. The Tracker sightings were then used as trials for whether an animal first seen at a particular location was seen by the Primary observers. By maximising the detection range of the Trackers it was hoped that animals would be seen before they responded to the approaching vessel. In an attempt to maximise team efficiency over a range of different conditions, one Tracker used 7 × 50 and the other used 25 × 100 binoculars. Once an animal or group had been detected, Tracker observers attempted to record all subsequent surfacings in order to allow possible duplicates with the sightings from Primary observers to be identified. A fifth person acted as a Duplicate Identifier, receiving data in real time from all observers and making a judgement as to whether or not sightings were duplicates. A sixth person acted as a Data Recorder. Mark recapture distance sampling implemented in this way requires that the Primary observers have no knowledge of detections made by the Tracker platform, but the Trackers can be aware of Primary sightings. The Trackers, Duplicate Identifier and Data Recorder were therefore accommodated on the same platform on each vessel, whereas the Primary observers were stationed as far away as practically possible.

The primary goals of the SCANS-II data collection system were to:

Measure distances and angles wherever possible, rather than estimate them;

Automate data collection; and

Cross validate though multiple measures of critical data items.

Logger 2000 was used as the main data entry program. Data from the four observers and the Duplicate Identifier were audible to the Data Recorder sitting at the Logger computer. The Logger configuration contained the following data entry forms:

(1) primary Sightings (sightings from the primary platform);

(2) primary Resightings (resightings from the primary platform);

(3) tracker Sightings (sightings from the tracker platform);

(4) tracker Resightings (resightings from the tracker platform);

(5) effort (activity, observer and weather information);

(6) personnel Data (including eye heights for video range measurements);

(7) incidental Sightings (any other sightings made by non-observers).

In addition to the data entry forms, Logger was also configured to contain:

(1) two sound recorders (one for Primary and one for Tracker observers);

(2) a sightings button form;

(3) a webcam frame capture form for angle measurement (see below);

(4) a buttons form sending serial port commands to the video capture system used for distance measurement (see below).

The SCANS data collection system is shown schematically in Fig. 1. A data flow diagram, including actions and information from observers, the software and the data recorder is shown in Fig. 2.

Sightings buttons

In order to record sighting times accurately and to ensure that sightings were not missed by the Data Recorder, each observer was equipped with a microphone and two buttons, one for sightings and one for re-sightings. The buttons were mounted close to each observer, connected to the main data

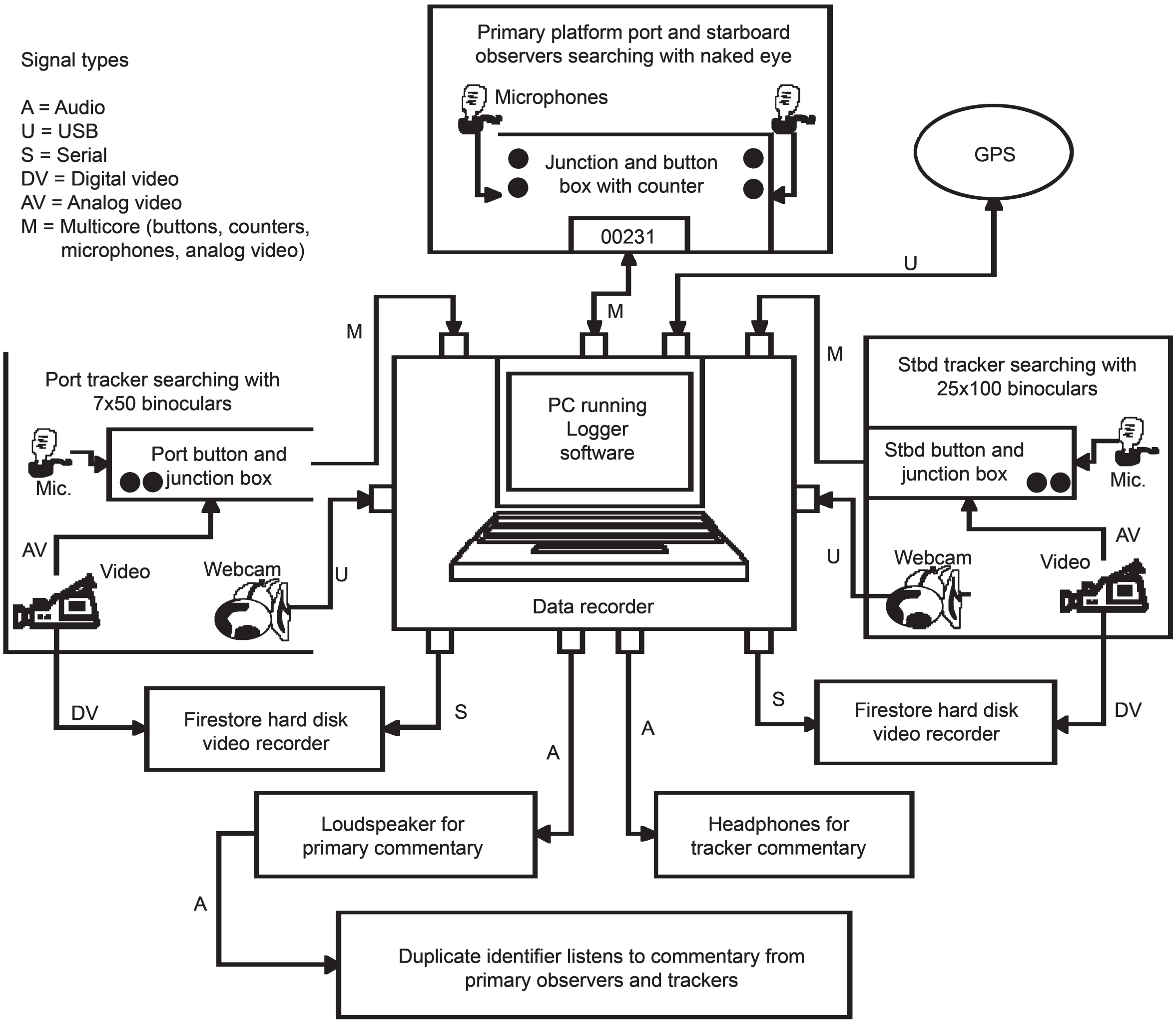

Fig. 1. Schematic diagram of the data collection system.

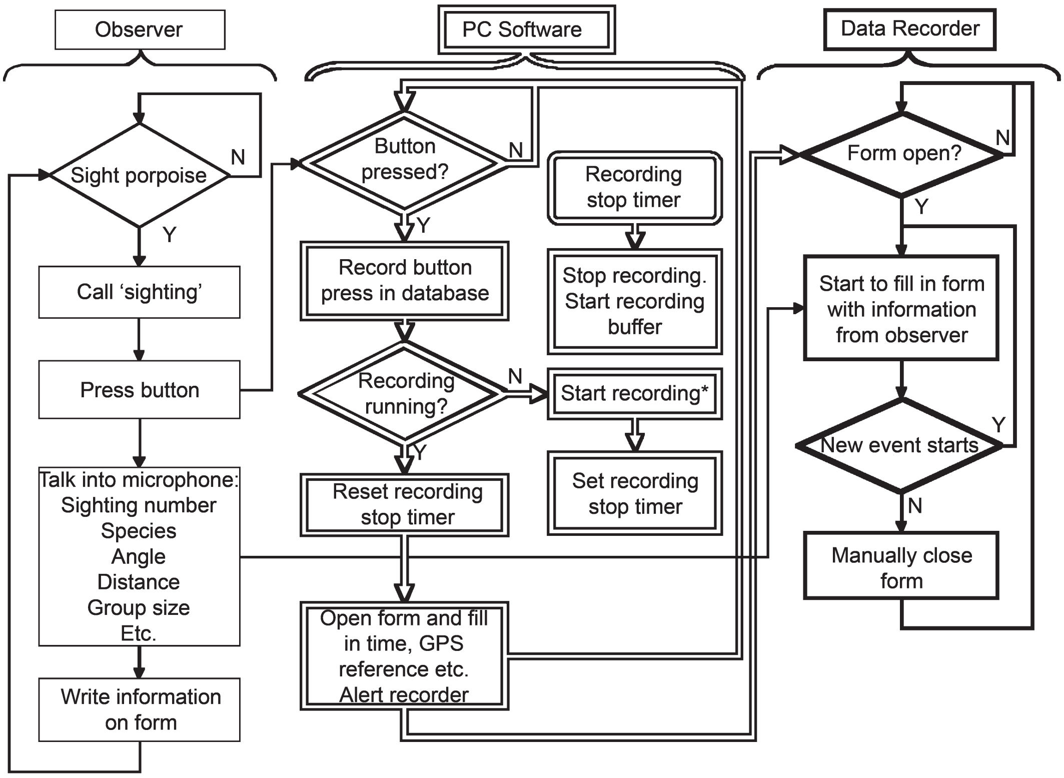

Fig. 2. Observer and data recorder actions and Software flow chart for the SCANS-II data collection system. *Tracker platform also takes a sequence of webcam photos for angle measurement and starts/stops video capture.

collection computer by cable and linked to Logger software buttons as described above. When a button was pressed, a number of actions took place within the Logger software:

(1) the date, time and button id was written to a table in the database;

(2) the appropriate sighting or resighting form opened on the Logger display, with fields filled in with the time, the button id and the incremental sighting/resighting number;

(3) the appropriate sound recorder started to record a voice track or, if it was already running, the timer controlling the recorder stop time would be reset.

If the button was a Tracker sighting or resighting button, the following actions also took place:

(1) video recording started (see below), or if it was already running, the stop timer reset so that recording would continue for a further 6s;

(2) a sequence of webcam images (see below) for angle measurement were stored.

Sighting numbers

The Primary platform button box also contained a counter which showed the same incremental sighting number as the Logger sighting form. The observers included this number in their commentary for cross-referencing during data validation. Tracker platform observers, who were stationed close to the data recorder, were given sighting numbers by the data recorder so that they did not have to take their eyes from the binoculars.

Sound recording

Two sound recorders were configured in Logger. Each had a 10 second data buffer allowing it to acquire voice commentary prior to each button press and would record for two minutes after each button press before stopping. Each sound recorder recorded data from a separate external USB sound card (Edirol UA20) and was configured to write stereo 16 bit wav files so that a separate channel could be used for each observer: the port and starboard Primary observers using the left and right channels of one recorder and the two Trackers the two channels of the other. Microphones were generally tucked inside the clothing of each observer to keep them out of the wind. The Data Recorder monitored the headphone output of the sound cards to listen to the Primary platform observers. The Data Recorder was generally close enough to the Tracker observers to hear them, but would use headphones or speakers on the output of the tracker sound card if required.

Observers were instructed to give information in the same order that data entry fields appeared in the Logger forms. The Data Recorder could talk back directly to the Tracker observers. A two-way radio was used to talk to the Primary observers.

Video capture and range measurement

When using the photo-grammetric video range methods described in Leaper and Gordon (2001) an animal on the captured video image may only be a few pixels in size. The video is therefore never used to detect an animal and identifying it on the video can usually only be reliably achieved by using information from the observer viewing the animal through binoculars either in the form of an audio commentary or some other accurate time-stamp to indicate an animal surfacing. Previous experiments had shown that it was not possible to rely on the video system to measure distances to 100% of sightings and so reticule readings were always recorded as well. The aim was to obtain sufficient measurements from the video to be able to measure any bias in the reticule readings made when video measurements were not available. In addition, comparison of measured distances and visual estimates from naked eye or reticules was informative in the context of other surveys.

The major factors in the choice of video camera for distance measurement are the quality of still images that can be achieved and the field of view of the lens. A narrow field of view maximises the size of distant cues in terms of pixels but at the risk of missing either the horizon or the cue. A wider field of view allows measurements from closer cues, but at lower resolution. Leaper and Gordon (2001) used a video camera with a field of view 2.7° vertically which was narrower than the 7° field of view of the 7 × 50 binoculars, but this was compensated for by a natural tendency for observers to place the object of interest in the centre of the binoculars field of view. For the SCANS-II survey, Canon XM1 cameras were used on the 25 × 100 binoculars and Sony HC90E camcorders on the 7 × 50 binoculars. These had a slightly wider field of view and were chosen in order to allow tracking closer to the vessel and to make the system easier for less experienced observers. The camera used on the CODA survey was the Canon HV20 high definition (HD) camera with a vertical field of view of 4.9°. On the 25 × 100 binoculars the same camera was used with a Canon TL-43 2× converter giving a vertical field of view of 2.9°. This HD video camera gave an effective image resolution of 1920 × 1080 pixels which was a considerable improvement over the 720 × 576 pixels of standard digital video used on SCANS-II. All cameras used had the progressive scan facility such that both interlaced fields in the video image could be captured simultaneously and shutter speeds were set to 1/1000s or faster. It was found that auto-focus systems were often not effective when scanning and so the camera was always set to manual focus at ∞ (a camera which has a control to set the focus to ∞ is much easier to use than one where this has to be done through manual adjustment).

In previous implementations, digital video tape was used to store data. The obvious disadvantages of using tape is that it must either be left running continuously, which generates vast quantities of data to review following the cruise, or recording must be started once animals are sighted. In this latter case the initial surfacing will be missed because the cameras generally enter a standby mode or turn off completely if not operated for a few minutes and it can take several seconds to re-load the tape and start recording when the record button is pressed.

Although the Logger software can capture sequences of still images from video and buffer sequences of images, it cannot perform video buffering in order to store video prior to a button press. Therefore a hard drive based video capture system was used (Firestore FS-4 from Focus Enhancements). These units are basically a computer hard drive configured to store data from the IEEE 1394 (Firewire) output of a digital video camera. Data are stored as video files which can be uploaded to a computer for editing or analysis. The great advantage of the Firestore units is that, like the sound recording system in Logger, they can be set to write data continuously to a 6s circular buffer so that when recording is started, data are recorded from 6s prior to the operator starting the system. This allows the first surfacing of any sighting to be captured on video which is particularly useful for harbour porpoises. The Firestore units have a serial (RS-232) interface connection which was used to start and stop recordings and set up the buffered operation. Sequences of Logger buttons were programmed to send the necessary sequences of commands to the Firestore units in order to make a 12s recording for each Tracker sighting, which would start 6s before the button press and end 6s after it. In the event of multiple resightings occurring within 6s of each other, recording would continue until 6s after the last button press.

Commentary from the camera microphone, or an external microphone mounted on the camera, was also recorded in the Firestore data. The commentary was the same as that recorded by Logger (but truncated to 6s either side of the button press). Generally, it was found that identifying surfacings within the short 12s video clips using the commentary, or the simple expectation of a surfacing about five seconds into each video clip, was considerably easier than searching for surfacing in the longer taped sequences captured in previous studies.

Accurate alignment of the video camera with the binoculars on which it is mounted is critical for successful video distance measurement as is the need for the horizon to be visible in every image. For the 7 × 50 binoculars a custom mount was designed that located the video camera above the binoculars and also held the webcam for bearing measurement (Fig. 3). This was supported on a monopod with a tripod ball head which allowed the observer to move freely to compensate for the motion of the vessel. The 25 × 100 binoculars were mounted on a fixed stand with the video camera mounted on top of the binoculars. There were no easy attachment points for mounting the camera and so a plate was fixed to the binoculars with steel bands which passed around the whole binocular body. On the SCANS-II survey, a custom mounting was built to allow adjustment of the alignment of the camera while on the CODA survey geared tripod heads (Manfrotto Junior Geared Head) were used. These proved easier to use, allowing small alignment corrections to be made when necessary.

It was often not practical to view the screens of the digital video cameras, so the analogue video output from each camera was fed back to the Data Recorder position and input to a small monitor. The Data Recorder could therefore make periodic checks of vertical and horizontal camera alignment. A video switch was used to monitor both cameras alternately with a single monitor.

Angle measurement (webcam capture)

Leaper and Gordon (2001) describe a method of angle measurement using downward pointing digital stills cameras mounted on Tracker binoculars which are used to photograph marks (generally parallel lines) on the deck below the observer. This basic methodology was updated for the SCANS-II survey by using the video capture utilities in Logger to acquire images from low cost webcams. When a Tracker sighting or resighting button was pressed, a series of eleven webcam images were stored, one per second from five seconds prior to the button press to five seconds after it. Images were also acquired at random time intervals in order to investigate observer scanning patterns.

Implementation

The data collection system described above is relatively complicated, requiring a computer with a number of external interfaces and cables to each observer position to carry button, audio, digital and analogue video signals and webcam images. Mains power was also required to run the computer, the Firestores and the video cameras.

The computer, sound cards, USB hub, USB to serial

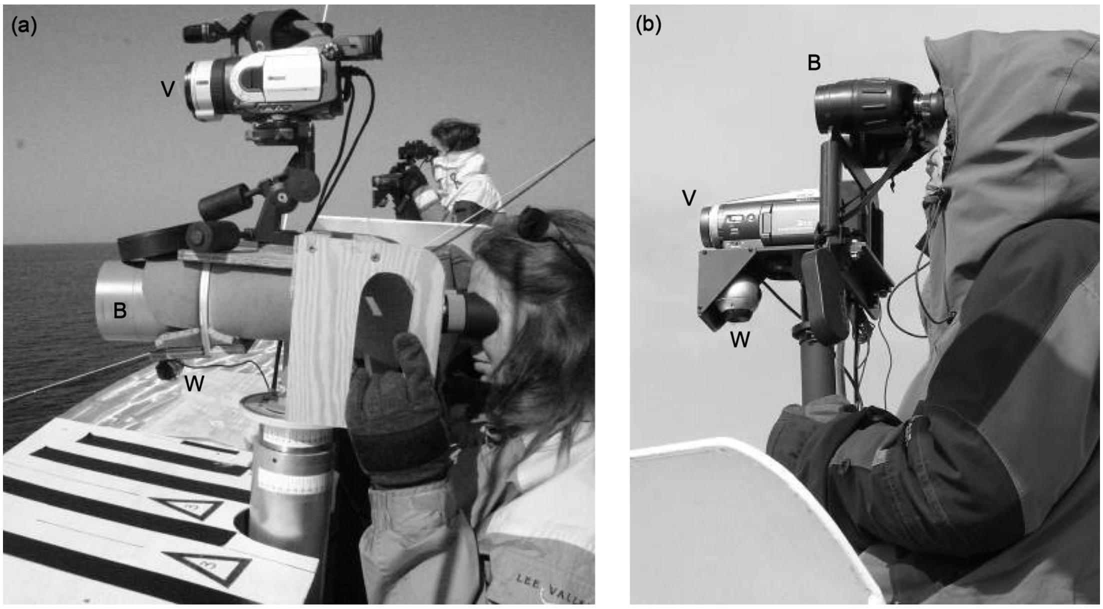

Fig. 3. (a) 25 × 100 and (b) 7 × 50 binocular stands on the tracker platform showing the webcams (W) the video cameras (V) and binoculars (B). Also visible are lines on the angle measurement board mounted below the webcam in front of the 25 × 100 stand.

adapters, button interface card and audio junction box were all built into an aluminium flight case which could be easily carried to the observation platform each day, provided a degree of environmental protection, and also provided a shield to aid viewing of the laptop screen in bright sunlight.

The system was built using standard components, requiring no specialist electronic expertise, the most complicated item being the junction boxes and cabling which combined audio and button signals into a single 10 or 15m long multi core cable to the two Primary observers and cables carrying button, audio and analog video to each of the Trackers. Digital video signals were carried down standard Firewire cables, which have a maximum length of 4.5m and the USB webcam signals were carried on 5m long USB extension cables which restricted the arrangement of the Trackers and data recorders on the platform.

In the event of inclement weather it was necessary to shut down the system quickly because the cameras and much of the other equipment used was not in any way water resistant. Most could be put into the aluminium case with the computer.

Two important aspects of the data collection system were: (a) that it would be backwards compatible with the data collection system employed in the first SCANS survey of 1994 (Hammond *et al.*, 2002), so that a direct comparison of the two surveys could be conducted; and (b) that it would contain sufficient redundancy that data collection could always continue. Thus, in the event of damage to the cable from the Primary platform, two way radios could be used for communication with the data recorder and an audio output from the radio at the data recorder station input to the sound card to record the Primary platform voices, with the software buttons in Logger being used in place of the wired buttons. The Trackers were within talking distance of the Data Recorder and the software buttons could again be used should the wired buttons fail. Video range tracking was very much an add-on to the survey protocol, and had no backup beyond the binocular reticule measurements. Compass roses and pointers on the binocular stands could be used to read angles manually. All vessels were also provided with paper forms for data recording. The laptop used was a standard, non specialised model, so in principle the software could have been reinstalled on a new machine, without specialist help, should the need have arisen.

Testing and training

The system was tested, reviewed and where necessary modified during a dedicated two week pilot survey in late April, 2005. This left a two month period prior to the main SCANS-II survey for construction of hardware interfaces and software development. Cruise leaders from each survey vessel were also trained in using the system during the pilot survey. Additional tests and training of one additional person per vessel were conducted during a one week passage on a platform of opportunity in early June. Cruise leaders were responsible for training other observers on each vessel.

DATA VALIDATION

At sea data validation

One of the potential benefits of entering as much data as possible into a computer in real-time is that it allows for data validation algorithms to flag potential problems that may be possible to correct while the details of the sighting are still fresh in the observers' minds. This type of processing needs to be balanced against the need for efficient and reliable data entry. For SCANS-II, with so many other new components to the data collection system, it was decided that data validation should be done entirely off-line in order not to compromise or further complicate the data collection task.

An additional requirement of the system was that at the end of the survey there should be a database with complete data on each sighting including measurements of distances and angles and that no further processing should be required (such as watching video or listening to commentaries) to extract the basic data. The data recording system was designed such that the Data Recorder would enter as much data for each sighting as possible. However, if there were several events happening at the same time then data items could not all be entered in real time and would need to be entered from recordings at the end of the day. In these circumstances the Data Recorder's main task was to monitor the commentaries and check that they were clear (e.g. all microphones were working and well positioned) and to remind observers if they had missed key data items. At the end of each observation session there were thus a full set of sightings and resightings forms, but often with gaps in some data fields. Hence an off-line data entry and validation system was developed which allowed observers to listen to commentaries, analyse video and measure bearings from the webcams at the end of the survey period each day. For each record the validation software listed possible problems as 'errors' or 'warnings'. 'Errors' were problems with the data that would preclude standard distance analysis (e.g. distance, angle, species) and 'warnings' were problems with ancillary data (e.g. swim direction, cue type).

As well as identifying blank fields, errors and warnings were triggered by the validation algorithms if identified values were outside of predetermined ranges for all parameters, inconsistencies between the observer reporting the sighting and the effort status, or inconsistencies in duplicate status.

For records that had errors or warnings, the observer listened to the verbal commentary to fill in blanks and try to resolve any discrepancies. The record was then saved with a code indicating whether there were no warnings or errors, whether further processing was planned, or if the data could not be resolved.

The validation software could be run simultaneously on multiple PC's, all networked to the central Logger laptop containing the database. Some vessels ran a network cable from the Logger laptop during data collection so that observers could process and validate their recent sightings during rest periods, immediately after coming off watch as other observers continued data collection.

The validation software also allowed cruise leaders to extract summary statistics of the type of errors that were occurring and plots of distances and bearings (diagnostics similar to those recommended by the International Whaling Commission Scientific Committee for cruise reports). These could be selected by species or observer and were particularly intended to identify problems such as rounding in estimated values that might be corrected during the survey.

Post-survey data validation

The database from each vessel contained records with two values for some parameters for many of the sightings (e.g. range from reticule, range from video, angle from angle board, angle from webcam). These included estimated and measured values of angles and distances. These were not directly compared by the data validation software at sea in order to allow for a more informative post-cruise analysis of errors. Comparisons of distances and bearings indicated a variance associated with estimation error but also occurrence of major errors which were assumed to be due to mistakes in data entry. Visual inspection of plots suggested that checking the 10% of sightings with the largest discrepancies should capture the majority of these gross errors. Although the photo-grammetric systems allowed for careful measurement, this was still done by an operator and so there was scope for error in these measurements too. All video and still images used for distance and angle measurement were linked to the database and so could be retrieved and re-measured. In cases where the image was clear and a discrepancy remained, it was assumed that the error was in the estimated distance or angle.

OVERALL PERFORMANCE OF THE SYSTEM

The full system was used on seven vessels on the SCANS-II survey in 2005, and five vessels on the CODA survey in 2007. In addition, systems were supplied to three vessels on the 2007 T-NASS survey of the North Atlantic. Components of the system have also been used on the IWC SOWER cruises.

Video distance measurement

The proportions of sightings that were successfully captured on video such that distance measurements could be made are given in Table 2. Success rates for the 7 × 50 and 25 × 100 binoculars were similar but varied considerably among vessels as a result of different conditions experienced and some technical problems. One vessel on the CODA survey experienced a total technical failure of the video equipment. If these data are discounted then the overall success rate for the CODA survey (66%) was higher than that for SCANS-II (37%). This was probably due to the use of high definition video cameras that resulted in much better image quality meaning that fewer surfacings were missed due to camera resolution and the fact that harbour porpoises, which made up the vast majority of sightings during SCANS-II but were absent on CODA, were particularly challenging subjects.

The most common problems encountered were with control of the Firestore hard-disc recording units; it was later found that these function more reliably on mains power if their internal rechargeable battery is removed. On one vessel on the CODA survey these failed completely which seemed to have been mainly a result of a failure of communication between the PC and the Firestore.

Bearing measurement

On the SCANS-II survey, the bearing cameras generally worked well, with an overall 94% success rate. On CODA there were more problems due to hardware conflicts related to the number of USB devices connected to the computer resulting in a lower success rate of 85%. Achieving a high success rate of bearing measurement using webcams should be possible, however recent developments in other angle measurement devices (e.g. magnetic sensors) may ultimately give better results.

The light levels during surveys varied from very dull conditions to bright sunlight. The video cameras used for distance measurement were designed for such a range of conditions and the automatic exposure compensation worked well. The webcams were generally designed for indoor use under artificial light and needed additional filters over the

Table 2

Number of Tracker sightings for all species and the percentage of sightings recorded for each measurement from SCANS-II and CODA surveys.

Survey	Observer	Vessel code	Number of sightings	Angle Estimated %	Angle Measured %	Distance Estimated %	Distance Measured %
SCANS-II	25 × 100	GO	128	98.4	86.7	98.4	17.2
		IN	91	95.6	95.6	89.0	51.6
		MC	20	45.0	100.0	100.0	0.0
		SK	77	97.4	90.9	100.0	36.4
		VH	108	99.1	100.0	100.0	73.1
		WF	57	93.0	93.0	86.0	22.8
		ZI	86	95.3	94.2	98.8	8.1
SCANS-II	7 × 50	GO	144	95.1	93.8	100.0	14.6
		IN	124	97.6	95.2	94.4	67.7
		MC	66	51.5	86.4	100.0	10.6
		SK	61	98.4	98.4	100.0	21.3
		VH	97	97.9	97.9	97.9	62.9
		WF	52	98.1	98.1	82.7	51.9
		ZI	100	100.0	92.0	99.0	41.0
	Total	**All**	**1,211**	**93.9**	**94.0**	**96.7**	**37.2**
CODA	25 × 100	IN	147	99.3	83.7	100.0	68.7
		MC	38	92.1	84.2	86.8	63.2
		RA/GE	100	100.0	88.0	100.0	0.0
CODA	7 × 50	IN	345	98.8	80.6	98.8	67.8
		MC	71	97.2	95.8	95.8	69.0
		RA/GE	142	97.9	90.1	98.6	0.0
	Total	**All**	**843**	**98.5**	**85.1**	**98.3**	**48.4**

lens to prevent overexposure in bright sunlight. Dark lenses from cheap sunglasses were found to be an effective form of filter.

An additional advantage of the bearing measurement system was that bearings were collected at random intervals (with a mean interval of 30s) in order to examine the scanning patterns of observers. Detailed analyses of scanning patterns and relative sighting rates are the subject of further analyses but simple plots made during surveys could be used to identify whether observers were scanning the appropriate angle sectors.

Data validation and workload for observers

Unlike the Logger software which had evolved over more than 10 years with considerable feedback from users, the validation software was written specifically for the SCANS-II survey. The software suffered from a lack of flexibility which meant that changes implemented during the pilot survey involved writing new code. Thus the first real test of the system was the survey itself. The system performed adequately in allowing the playback of audio recordings and measurements from images. However, this did prove a very time consuming process, especially for observers who had not seen the system before. Although several improvements that would speed up data entry were identified, the main problems that caused most lost time were small bugs and glitches. Vessels varied in their ratio of survey time to bad weather time and while some teams were able to keep up with data entry and validation, for others it proved a rather onerous task.

DISCUSSION

The Logger 2000 software has been used for several years on a variety of projects and has provided a reliable and flexible method for semi-automated data collection during line transect surveys, mitigation monitoring, photo-identification and behavioural studies of marine mammals. The system of user defined forms allows different users to configure the software in different ways without having to modify the program code itself. However, in order to deal better with high volumes of concurrent data coming in from five different people and to improve the way in which sightings data are recorded more generally, a number of modifications were made to the Logger code specifically for the SCANS-II survey. All of these modifications have been included in software releases since the SCANS-II survey and are freely available. Information on circuits for external interfaces are available from the authors on request.

Although accurate input data are clearly critical for line-transect surveys and serious biases can be caused by measurement error, surprisingly little attention has been given to data collection compared to that devoted to data analysis methodology (Williams *et al.*, 2007). In addition, experiments that have attempted to quantify range and bearing measurement error have used static targets which are unlikely to provide realistic error data. The data collection methods used on the SCANS-II and CODA surveys generated range and bearing measurements for a proportion of sightings allowing both better abundance estimation from these surveys and also comparison of estimated and measured distances and angles under real survey conditions. Such comparisons are of more general value in revealing the likely extent of measurement error on other surveys that have used conventional reticule binoculars and angle-boards. Full analyses of these results are described in Leaper *et al*. (2010).

The data collection system worked effectively on all seven vessels taking part in the SCANS-II survey, although the complexity of the system and the large number of interconnected components working in a harsh environment required a certain level of enthusiastic vigilance on the part of the operators to keep it running. The most commonly encountered problems were with the video capture system. This appears to be due to either the Firestore units overheating or a failure of communication between the video cameras and the Firestores or the Firestores and the computer. At the time of the survey, the Firestore units were a new and innovative product. It is likely that this technology will become more standard, and therefore cheaper and more reliable in the future. During the CODA survey, problems were encountered with the webcams used for angle measurement. This appears to be due to the use of newer webcams sending higher quality data and also an audio signal to the PC, which overloaded the USB system. We also note that the Edirol sound cards used in 2005 are no longer manufactured and we have yet to identify a replacement with the same specification. Recreating the system for future surveys would therefore require a certain amount of re-development in order to recreate a stable system.

As computer hardware capabilities develop it is likely that the optimum means of implementing a system like this may change more fundamentally. For example, some of the rather cumbersome cabled connections used here might be replaced by wireless links.

The Logger 2000 software is written in C++ and runs on a PC under Windows. The software itself is not open source and cannot therefore be modified by the operator. We hope, in the future, to incorporate Logger features into the PAMGUARD software (*http://www.pamguard.org*; Gillespie *et al.*, 2008) which is both free and open source and is also more likely to be supported in the future. The validation software is coded in Visual Basic and was written specifically for the SCANS-II survey. Unlike Logger, the database structure used by the validation software is fixed, so the validation system can currently only be used with the Logger SCANS-II configuration. For future systems, validation should have the same flexibility as Logger in terms of user defined data base structure based on the user's choice of forms.

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Comparisons of measured and estimated distances and angles from sightings surveys

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ABSTRACT

Photogrammetric systems using video cameras were used to measure radial distances to sightings during the SCANS-II, CODA and SOWER surveys. These surveys included sightings of a variety of species from harbour porpoise, at distances of a few hundred metres, to large baleen whales at distances greater than 10km. A total of 910 initial sightings with estimated distances from reticles and measured distances from video, using 7 × 50 (636) or 25× 'Big Eye' (274) binoculars, were compared. Bearings to sightings were also measured from still images. The CV_{RMSE} in distances varied between 0.19 and 0.33 for reticle binoculars. Comparisons of measured distances to simultaneous sightings by other observers using naked eye gave a CV_{RMSE} of 0.39 for naked eye estimates. There was a consistent, non-linear pattern in all data sets, of over-estimating close distances to sightings of surfacing cetaceans and under-estimating those further away. However, this pattern was not evident from the distance experiments on SOWER to fixed targets which also had a much lower variance ($CV_{RMSE} = 0.13$). Bearing data from SCANS-II and CODA showed around 5% of estimates had gross errors greater than 20º that were attributed to mistakes. For the remaining values, RMS errors were in the range 5.7º–7.2º for SCANS-II and CODA and 4.9º for SOWER. Both distance and angle errors will make a substantial contribution to the variance of abundance estimates and simulated data showed that the observed non-linear nature of distance errors may cause considerable bias even when linear regressions might suggest little bias. There still remain technological challenges in operating complex electronic systems at sea to measure distances and bearings, but investment in these methods should be a cost effective way of reducing bias and improving precision of cetacean abundance estimates.

KEY WORDS: SURVEY–VESSEL; PHOTOGRAMMETRY

INTRODUCTION

Distances and angles to sightings during line-transect surveys are critical data items but often rely on estimates from observers that may be subject to considerable error. These errors are a widely acknowledged problem for cetacean abundance estimation (Williams *et al.*, 2007). Photogrammetric methods have been used for some time to measure distances and angles to cetacean sightings and have been incorporated into the data collection system on recent surveys (Gillespie *et al.*, 2010). On the SCANS-II[1] (Small Cetaceans in the European Atlantic and North Sea) and CODA[2] (Cetacean Offshore Distribution and Abundance) surveys in the Northeast Atlantic in 2005 and 2007, photogrammetric systems were part of a fully integrated, computer-based data collection system. On the IWC SOWER (Southern Ocean Whale and Ecosystem Research) surveys in 2006/07 and 2007/08, the use of video cameras to measure distances and digital still cameras to measure angles was limited to experimental periods.

The implications of measurement error for bias and precision in abundance estimates have been examined for theoretical models, showing the potential for severe bias in the case of both large unbiased measurement error and biased errors (Marques, 2007). Distance and angle experiments to artificial visual targets such as buoys are also conducted during many surveys to assess the variance and, sometimes, to try to correct for distance errors. However, the extent to which such experiments are representative of the real situation for cetacean sightings is difficult to assess. Most methods to correct for distance errors have also relied on either additive models (e.g. Chen, 1998; Chen and Cowling, 2001) or linear multipliers (e.g. Marques, 2004). Such models may not always be appropriate for correcting distance errors. For example, Alldredge *et al.* (2007) reported non-linearities in distance errors to calling birds and suggested the need for more complex error correction methods. The aim of this paper is to compare measured and estimated values to the sightings made during surveys and examine the implications of measurement error for abundance estimates.

SURVEY METHODS

The integrated data collection system used on the SCANS-II and CODA surveys, described in Gillespie *et al.* (2010) included photogrammetric measurement of distances and angles to sightings using the methods of Leaper and Gordon (2001). Observers on the surveys consisted of two 'Primary' observers searching with naked eye and two 'Tracker' observers, one searching with 7 × 50 binoculars and one with 25× 'Big Eyes' (Monk Leviathan) to implement Mark Recapture Distance Sampling methods (Buckland and Turnock, 1992). Measurements from digital video sequences

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[1] *http://biology.st-andrews.ac.uk/scans2/*

[2] *http://biology.st-andrews.ac.uk/coda/*

of cetacean surfacings were used to calculate distances and digital still images were used to calculate angles for the observers using binoculars. The general principle behind the use of video cameras to measure distances at sea is the same as with using reticle binoculars and involves measuring the angle of dip from the horizon to the whale from a platform of known height. Eye heights on SCANS-II and CODA vessels varied between 10 and 14m. One of the main challenges to the system is capturing an image of the first surfacing reported by the observer of sufficient quality to allow measurements to be made. Photogrammetric measurement of bearings used a downward pointing camera taking a still image of reference marks on the deck of the vessel. These methods can only be used for observers searching with binoculars but some estimates of errors from naked eye observations were available from the SCANS-II survey where photogrammetric measurements from the Tracker could be compared with naked eye estimates from the Primary for the same surfacing event. Estimated angles were obtained using angle boards for the Primary observers and using angle pointers attached to the binocular mounts for the Trackers. The 7 × 50 binoculars were supported on a monopod with the angle pointer at the bottom, the Big Eyes were on a fixed pedestal with an angle scale just below the binoculars.

A subset of the full system described in Gillespie *et al.* (2010) was used during experimental periods of the IWC SOWER surveys in the Southern Ocean. The video system was used for observers in the top observation barrel on SOWER cruises in 2006/07 and 2007/08 from an eye height of 20.5m. Observers on SOWER use 7 × 50 binoculars with a non-linear reticle scale which is marked in nautical miles: angles are measured using angle boards (see Fig. A1 in Appendix). On the SOWER 2007/08 cruise, the video system was also used during a distance estimation experiment where distances were also measured to a buoy in the water using radar. This experiment served as an additional calibration check. Observer distance estimation errors during standard distance experiments were also compared to the errors to whale sightings.

ANALYSIS METHODS

Calibration tests of the photogrammetric systems are described in Leaper and Gordon (2001) and indicated sufficiently small errors (root mean square error in distance of 3.5% and in angle of 1.5°) that measured values were treated as 'true' values when compared with visual estimates for the analyses in this paper. We only used initial sightings, except for analysis of simultaneous sightings by naked eye observers during SCANS-II, to avoid autocorrelation, because distance estimation errors to re-sightings may be strongly influenced by the initial estimates.

Unlike the computer controlled system for capturing angle images on SCANS-II and CODA, which used webcams, the still camera system on the SOWER surveys needed to be completely self-contained. This system is described in Appendix I and followed similar experiments on a previous cruise in the same series of surveys in 1983/84 (Thompson and Hiby, 1985). In addition to using the still camera to measure angles to sightings with images captured when the observer pressed a button, images were also captured at intervals to examine observer scanning patterns. On SCANS-II and CODA, images were captured at random intervals with a mean interval of 30s. On SOWER, the interval was fixed at 30s because the camera did not support random intervals but the variation in the observer's scanning patterns would effectively generate a random sample.

Following the surveys, pairs of simultaneous distances and angles were compared. These are referred to as 'measured' for values derived from the photogrammetric system and 'estimated' for naked eye estimates, reticle and angleboard readings by the observers. For distances, the errors are likely to scale with the distance and a convenient measure is the CV of the root mean squared error (CV_{RMSE}) defined as the root mean squared error divided by the mean of the observed values. For angles, the root mean squared (RMS) error is more appropriate.

For analysis of naked eye estimates, distances and angles to reported surfacings from the Primary and Tracker from SCANS-II that occurred close together in time and location but were not necessarily classed as duplicate sightings, were compared. Sightings had to occur within 10 seconds (the Tracker sighting did not necessarily need to occur first as with usual duplicate sightings) and on a similar bearing (±10°).

It was anticipated that patterns of errors in distance estimates would be complex and non-linear (Williams *et al.*, 2007). In addition to simple linear regression of estimated distance against measured, non-linear effects were investigated by plotting log(estimated)–log(measured) against log(measured). Investigative analyses were also carried out using Generalized Additive Models (GAMs).

Simulation study of the effects of measurement error on estimated strip widths

For non-linear errors, simple simulations of the detection process were used to investigate some of the effect of measurement error on estimated strip width. Errors will affect both the accuracy and precision of estimates with the effect on precision being strongly influenced by the number of data points. To investigate bias, 10,000 simulated sightings were generated with and without distance error. Software DISTANCE (Thomas *et al.*, 2010; Thomas *et al.*, 2006) was used to fit detection functions to these two data sets so that estimated strip widths could be compared.

To simulate sightings, a fixed vessel speed of 5ms^{-1} was assumed with whales distributed randomly within a box ahead of the vessel. The probability P, that a whale surfacing at a particular location was detected, was modelled by the hazard probability function:

$$P(r,\theta) = \frac{e^z}{1 + e^z} \tag{1}$$

Where r is the radial distance, θ is the angle from the trackline and

$$z = a_0 + a_1 r + a_2 r^3 \tag{2}$$

Where a_0, a_1 and a_2 are parameters of the detection function

The choice of functional form for the hazard probability function was based on sightings from surveys of minke whales (Cooke and Leaper, 1998). The aim of the simulations was to investigate general implications of distance error rather than specific results for any particular survey. Thus parameters were not species specific and a pattern of a dive time of 120s followed by 3 surfacings was assumed in all cases with whales travelling in a straight line with a speed of 0.5ms^{-1}. Whales were introduced into the box according to the method of Hiby (1982) in order to ensure the correct distribution of whale headings. Values of a_1 and a_2 were adjusted to create detection functions with different effective strip widths. The inclusion of whale movement and multiple surfacings was designed to ensure that simulated data without error did not fit perfectly to a simple parametric detection function but were a more realistic representation of real data, even though the parameters themselves were not conditioned to any actual data. These simulations did not include a term in θ or the implications of angle error, but the data on search patterns using binoculars gathered by the photogrammetric systems on these surveys did allow the detection probability by angle to be estimated.

For each simulated sighting that occurred with position (r,θ), the position with distance error (r_e, θ) was generated by calculating r_e from r using the regressions derived from the data for the survey and observation method being investigated.

RESULTS

Performance of video systems

Gillespie *et al.* (2010) describe the performance of the video systems on the SCANS-II and CODA surveys. On SCANS-II the majority of sightings were of harbour porpoise (*Phocoena phocoena*) and distances were successfully measured on video from 448 (37%) of 1,211 sightings. The CODA survey had a greater variety of species, including large whales, and 405 (48%) of 843 sightings were measured. The combined success rate for two of the three CODA vessels analysed was 67% whereas the third vessel suffered a total failure of the video recording system. The higher success rate on CODA was likely due to a combination of larger, more visible species and the use of high definition video.

On the 2006/07 SOWER cruise, seven minke whale (*Balaenoptera bonaerensis*) surfacings were measured on video out of a total of 21 sequences that were recorded (33%). The main reason for sightings not being detectable on video appeared to be related to image quality and the characteristics of minke whale blows. The maximum distance that a minke body was detected on video was 3.6km and the maximum distance that a blow was detected was 1.9km. Even at this distance, this sighting was only detected for certain due to being a combined blow/body cue. On the 2007/08 cruise, experiments were mainly conducted in the presence of large baleen whales using a high definition video camera. In this case, 34 video measurements were obtained out of a possible 64 sequences (53%). Large baleen whale blows were detected out to measured ranges of 10km.

The experiment on the SOWER cruise that compared distances to a buoy between radar and the video gave the linear regression *Video = 1.03 × Radar* with CV$_{RMSE}$ = 0.05 assuming radar measurements had no error. However, it is not known which of radar or video is more accurate and the small bias of 3% apparent in the video could be explained by refraction effects (which would affect reticle binoculars in the same way). These results were consistent with the calibration tests reported in Leaper and Gordon (2001).

Angle measurements were obtained for 94% of sightings on SCANS-II and 85% on CODA. The lower success rate on CODA was due to conflicts between USB devices connected to the computer which caused the webcams to stop working periodically.

Comparison of estimated and measured distances

After a first comparison of the estimated and measured data, the 90th percentile of largest distance errors were re-examined for errors due to data recording, transcribing or measurement mistakes.

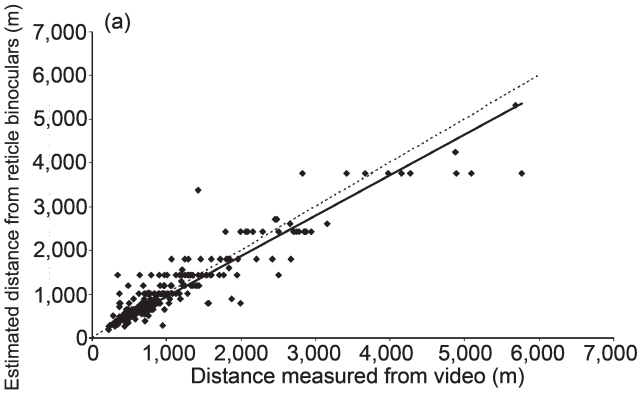
Plots of estimated against measured radial distances are shown by survey and binocular type in Figs 1a–e together with linear regressions (regression coefficients are given in Table 1). Rounding to certain reticle values, indicated by points in a horizontal line, is particularly apparent for the

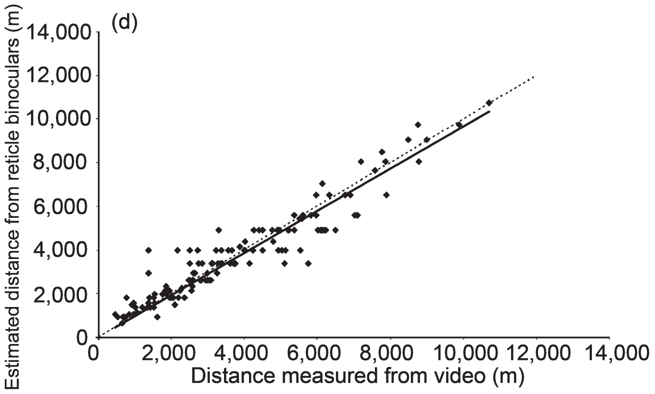
Table 1

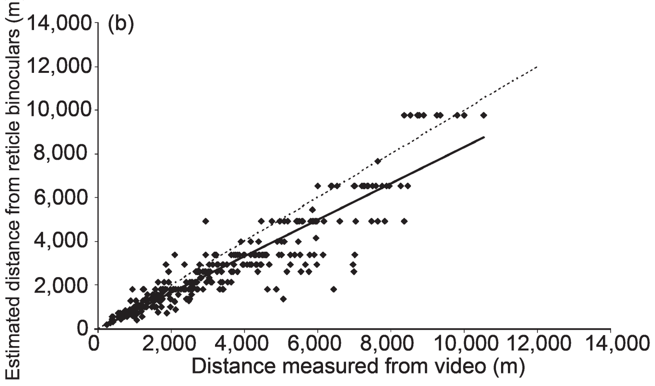
Comparison of estimated and measured radial distances.

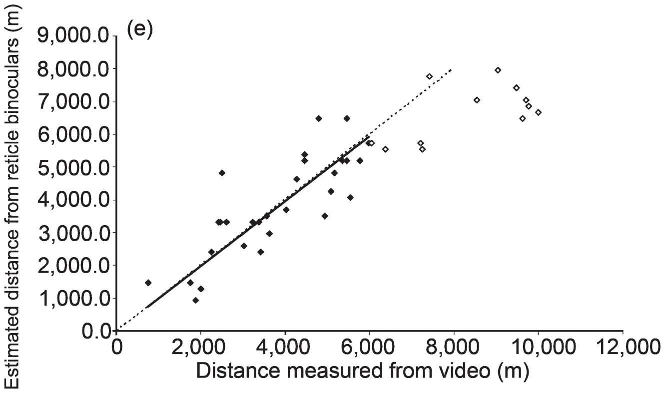
Survey	Searching method	*n*	Linear regression slope *m* (with intercept forced to 0)	CV$_{RMSE}$	Regression on log of distance: *a*, standard error in ()	Regression on log of distance: *b*, standard error in ()
SCANS-II	7 × 50	245	0.93	0.31	–0.13 (0.03)***	0.96 (0.18)***
SCANS-II, 5% of furthest estimated distances (>3.4km) truncated	7 × 50	233	0.97	0.36	–0.15 (0.03)***	1.05 (0.21)***
CODA	7 × 50	321	0.83	0.32	–0.08 (0.02)**	0.44 (0.14)**
CODA, 5% of furthest estimated distances (>6.5km) truncated	7 × 50	305	0.78	0.35	–0.12 (0.02)***	0.70 (0.15)***
SCANS-II	Big Eye	136	1.07	0.33	–0.20 (0.04)***	1.61 (0.31)***
SCANS-II, 5% of furthest estimated distances (>4km) truncated	Big Eye	129	1.06	0.36	–0.25 (0.05)***	1.94 (0.35)***
CODA	Big Eye	138	0.97	0.19	–0.18 (0.03)***	1.45 (0.21)***
CODA, 5% of furthest estimated distances (>8km) truncated	Big Eye	131	0.94	0.21	–0.20 (0.03)***	1.62 (0.22)***
Combined SCANS-II and CODA	Big Eye	274	0.98	0.23	–0.18 (0.02)***	1.42 (0.16)***
SOWER	7 × 50	41	0.86	0.26	–0.22 (0.07)**	1.74 (0.59)**
SOWER truncated at estimated distances >6km (10 values removed)	7 × 50	31	0.92	0.23	–0.21 (0.10)*	1.65 (0.82)
SCANS-II	Naked eye	28	0.81	0.39	–0.36 (0.17)*	2.00 (1.0)
SCANS-II truncated at estimated distances >600m (3 values removed)	Naked eye	25	0.74	0.44	–0.53 (0.18)**	2.95 (1.1)*

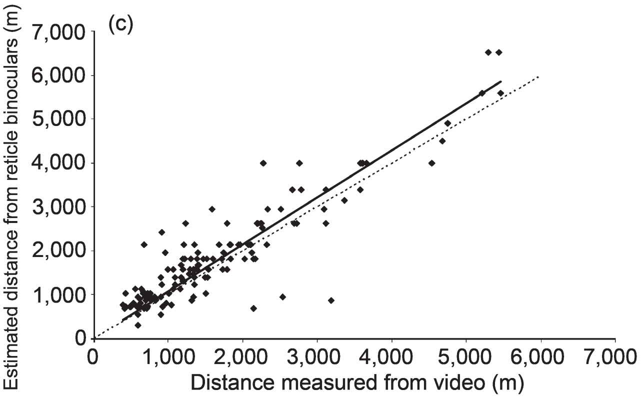
***Regression significant at p<0.001. **Regression significant at p<0.01 *Regression significant at p<0.05. *n* is the number of observations.
m is the slope term *estimated* = *m* × *measured*. *a* and *b* are the slope and intercept in ln (*estimated*)-ln(*measured*) = *a* × ln(*measured*) + *b*.











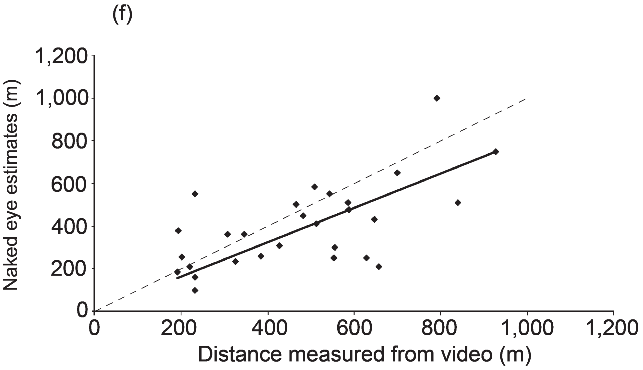


Fig. 1a. Estimated against measured distances from SCANS-II for 7 × 50 binoculars. Dotted line indicates no error, solid line indicates fitted linear regression.

Fig. 1b. Estimated against measured distances from CODA for 7 × 50 binoculars. Dotted line indicates no error, solid line indicates fitted linear regression.

Fig. 1c. Estimated against measured distances from SCANS-II for Big Eye binoculars. Dotted line indicates no error, solid line indicates fitted linear regression.

Fig. 1d. Estimated against measured distances from CODA for Big Eye binoculars. Dotted line indicates no error, solid line indicates fitted linear regression.

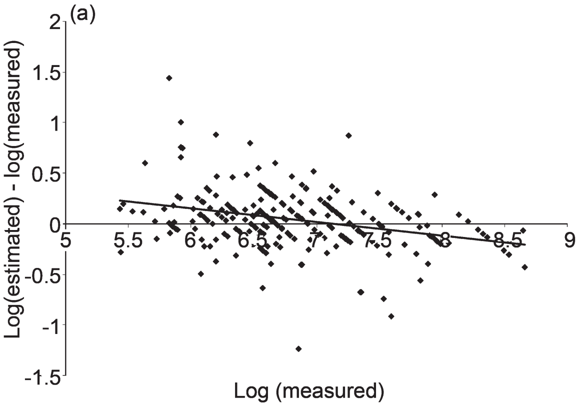
Fig. 1e. Estimated against measured distances from SOWER cruises in 2006/07 and 2007/08. Solid circles represent measured distances <6km, open circles>6km. Dotted line indicates no error, solid line indicates fitted linear regression up to a truncation of measured distances of 6km.

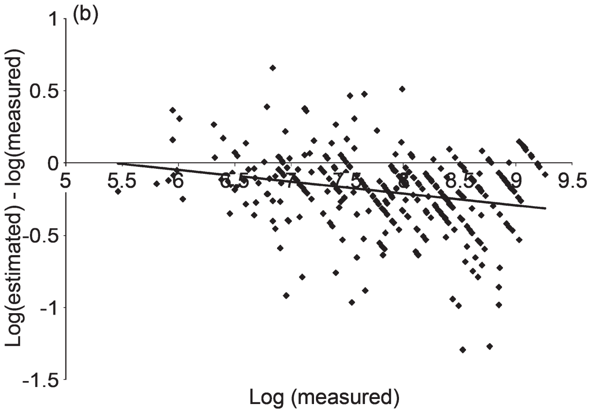
Fig. 1f. Estimated distances from naked eye against measured distances from Tracker from SCANS-II. Dotted line indicates no error, solid line indicates fitted linear regression.

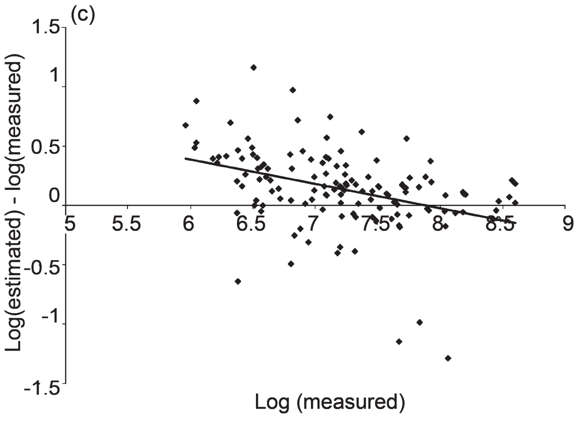
SCANS-II and CODA 7 × 50 binoculars, particularly at larger distances (Figs 1a and b). Less rounding is apparent with the Big Eyes which have a finer reticle scale. The 7 × 50 estimates on CODA and the full data set from SOWER (including distances out to 10km) were the only ones that showed overall bias of greater than 10%. For SOWER, this can be explained by the difficulties of using reticles to estimate very small angles of dip for whales close to the horizon. When distances were truncated at measured values >6km, the bias was negligible (Fig. 1e). However, truncating on the basis of estimated distances >6km did less to reduce the bias (Table 1). The reason for bias in the CODA 7 × 50 binoculars is unclear but these were a different model to those used on SCANS-II whereas the same Big Eyes were used in both surveys.

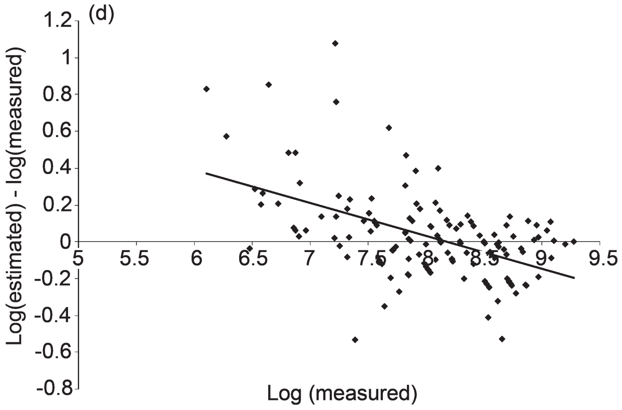
The magnitude of the errors is indicated by the CV_{RMSE} given in Table 1. These varied between 0.19 for the CODA Big Eyes to 0.33 for the SCANS-II Big Eyes. The CV_{RMSE} of the CODA Big Eye sightings was strongly influenced by a single observer who accounted for 36% of all the sightings and had an individual CV_{RMSE} of 0.09. For the SOWER data, the CV_{RMSE} for the data truncated at 6km (approximately the maximum distance used in the buoy experiments) was 0.23, considerably greater than the CV_{RMSE} of 0.13 from all observers in the buoy experiments on the 2007/08 cruise.

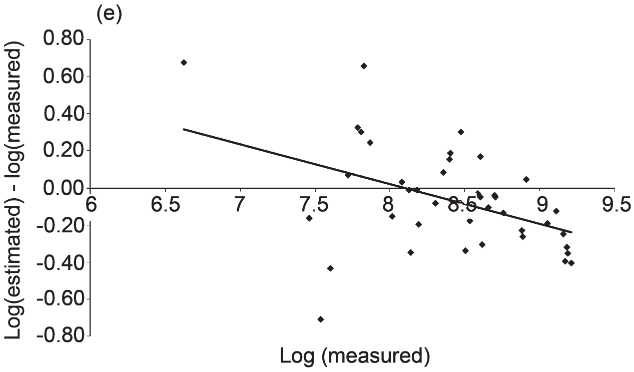
In all cases for sightings of surfacing cetaceans, there was

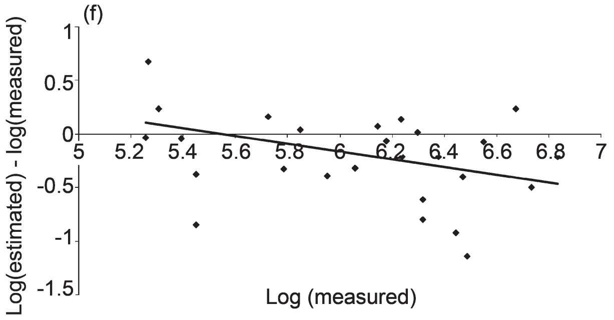












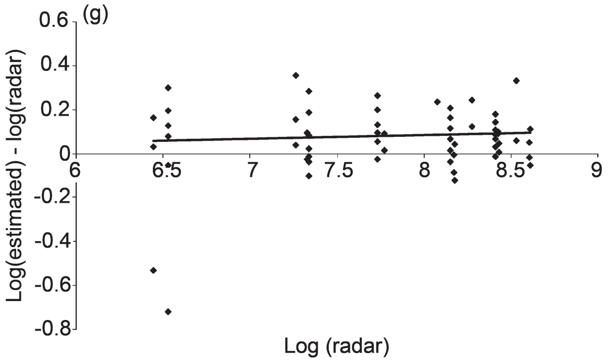


Fig. 2a. Log(estimated)–log(measured) against log(measured) distances from SCANS-II. Measured values are distances from video in metres, estimates are from 7 × 50 reticle binoculars.

Fig. 2b. Log(estimated)–log(measured) against log(measured) distances from CODA. Measured values are distances from video in metres, estimates are from 7 × 50 reticle binoculars.

Fig. 2c. Log(estimated)–log(measured) against log(measured) distances from SCANS-II. Measured values are distances from video in metres, estimates are from Big Eye reticle binoculars.

Fig. 2d. Log(estimated)–log(measured) against log(measured) distances from CODA. Measured values are distances from video in metres, estimates are from Big Eye reticle binoculars.

Fig. 2e. Log(estimated)–log(measured) against log(measured) distances from SOWER cruises in 2006/07 and 2007/08. Measured values are distances from video in metres, estimates are from reticle.

Fig. 2f. Log(naked eye estimate)–log(measured) against log(measured) from simultaneous sightings during SCANS-II. Measured distances are from Tracker platform.

Fig. 2g. Log(estimated)–log(measured) against log(measured) distances from the SOWER distance experiments in 2007/08. Measured values are from radar and estimates from reticle binoculars. Solid line shows linear regression which was not significant.

evidence of a non-linear relationship between error in distance and distance, with over-estimation of close distances and under-estimation of far distances. Figs 2(a–f) shows plots of log(estimated)–log(measured) against log(measured) with regression coefficients in Table 1. The slope of these regressions was significantly different from 0 at $p<0.05$ in all cases, indicating a change in distance bias with distance. By contrast, there was no evidence of a similar pattern in the errors to the fixed buoy in the distance experiments (Fig. 2g). Visual examination of the residuals from each of the regressions in Fig. 2 indicated a uniform spread, suggesting an adequate model. Exploratory investigations with GAMs suggested complex models with 5–8 degrees of freedom. The log based models were chosen for consistency between surveys and simplicity.

For 7 × 50 binoculars, the angle of dip from the horizon to the whale at which distances changed from over to under-estimation was approximately 0.26° for SOWER and 0.37° for SCANS-II (when binocular magnification is taken into account these would result in angles of 1.82 and 2.59° subtended at the eye). For the 25× Big Eyes, these angles were 0.08° and 0.13° (2.00 and 3.25° subtended at the eye). For naked eye on SCANS-II, this angle of dip was 2.6°. These indicate a fairly consistent angle of dip between the horizon and the whale, perceived at the eye, at which distance bias changes from positive to negative.

The effect of truncation at larger radial distances was also investigated for the 7 × 50 and Big Eye data from SCANS-II and CODA. Following the 'rule of thumb' suggested by Buckland *et al*. (1993), the largest 5% of estimated distances were truncated. Unlike for SOWER where there was clear evidence of increasing bias for large distances, truncation of the SCANS-II and CODA data did not generally reduce bias. In addition, the slope of the regression of log(estimated)–log(measured) against log(measured) and the CV_{RMSE} increased in all cases (Table 1). There is also some selectivity in the dataset resulting in both the closest and furthest distances being less likely to be measured from video. For SCANS-II and CODA the closest distance measured on video was 230m and 390m for 7 × 50 and Big Eyes respectively. The effects of truncation were most apparent in the naked eye data from SCANS-II. If 30% of the furthest estimated radial distances (>500m) were truncated then there was no longer a significant correlation between estimated and measured distances ($r = 0.42$, df = 18, $p>0.05$).

Comparison of estimated and measured angles

Where large discrepancies between estimated and measured angles were observed, these were resolved wherever possible by listening to the commentaries and re-analysing the bearing images. Bearing images were taken in sequences, one second apart, and so it was possible to measure whether the observer was looking steadily at a target, or still scanning when the sighting button was pressed. For the 7 × 50 binoculars this resulted in 651 initial sightings where both estimated and measured bearings were available from SCANS-II. Of these, 5% (34 sightings) showed gross errors of more than 20° which could not be resolved and were assumed to be either observer error or related to angle pointers becoming mis-aligned. For the remaining sightings, the RMS error was 7.1° for SCANS-II and 7.2° for CODA. For the Big Eyes there were 355 sightings with both estimated and measured bearings of which 6% of sightings showing errors of more than 20°. Excluding these sightings with large errors gave a RMS error of 6.0° for SCANS-II and 5.7° for CODA. For the simultaneous sightings from naked eye observers during SCANS-II where there was also a measured angle from the Tracker platform, the RMS error was 5.9°. However, this value may be influenced by the selection criteria used for simultaneous sightings; angles needed to be within +/– 10° and hence, sightings with larger angle errors were eliminated.

On the SOWER 2008/09 cruise there were a total of 62 sightings where bearings were both estimated from angle boards and measured photographically. There was evidence of a small systematic bias of around 2° and an overall RMS error of 4.9°. Of the 62 sightings, 45 (73%) were humpback whales (*Megaptera novaeangliae*), nine (15%) were sperm whales (*Physeter macrocephalus*) and five (8%) were southern bottlenose whales (*Hyperoodon planifrons*). There were no significant differences in mean squared error between these species (Anova, df = 2, $p = 0.88$). There were only four sightings where the cue was not recorded as a blow or blow/body and so it was not possible to investigate the accuracy of bearings with respect to cue type.

Perpendicular distance is proportional to the sine of the angle, so this was used to investigate potential bias in perpendicular distance due to angle error. Fig. 3 shows sin(estimated) against sin(measured) for the SCANS-II Tracker angles. The linear regression is given by $y = 1.01x$ showing no evidence of overall bias.

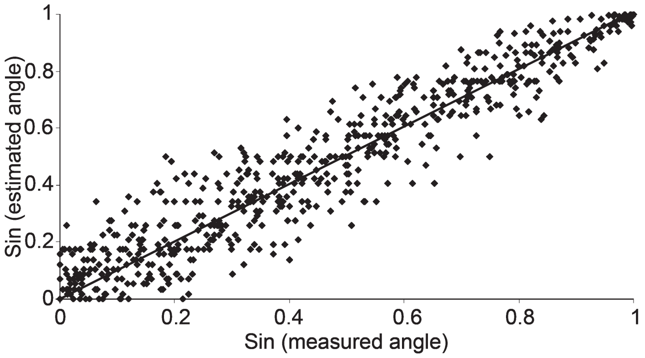


Fig. 3.Angle component of perpendicular distance, sin(measured angle) against sin(estimated angle). Data from SCANS-II survey.

Effect of measurement error on estimated strip widths

The estimated strip widths from simulated data with and without error are given in Table 2. The intercept and slope from the regressions of log(estimated)–log(measured) against log(measured) in Table 1 were used to generate a distance with error (r_e) from the distance a simulated whale was detected (r). It can be seen that there is scope for substantial bias, although the extent of the bias depends on the distribution of observed radial distances in relation to the distance at which distance errors tend from over-estimation to under-estimation. These results should be treated as illustrative of the level of bias that may occur based on the distance error relationships estimated for each survey rather than actual estimates of potential bias for these surveys. The parameters of the detection function were adjusted to

Table 2

Estimated strip widths from simulations with and without measurement error. Parameters for the simulations were adjusted to generate two different estimated strip widths (ESW) for each model to investigate the effects of measurement error under different sighting conditions.

	No measurement error		With measurement error		$\frac{ESW_{error}}{ESW_{noerror}}$
Source of measurement error model	ESW (m)	CV	ESW (m)	CV	
SOWER (all data)	850	0.016	1,229	0.020	1.45
SOWER (all data)	1,386	0.017	1,795	0.021	1.30
SCANS-II 7x50	585	0.016	680	0.018	1.16
SCANS-II 7x50	391	0.013	493	0.015	1.26
Combined SCANS-II/CODA; Big Eye	857	0.016	1,174	0.020	1.37
Combined SCANS-II/CODA; Big Eye	1,956	0.019	2,276	0.026	1.16
SCANS-II naked eye	377	0.016	388	0.021	1.03
SCANS-II naked eye	628	0.016	506	0.024	0.81

Strip widths calculated from simulated data (10,000 sightings) in Distance selecting half-normal key with cosine adjustments based on AIC.

generate different effective strip widths rather than fitted to the data themselves.

The simulations assumed an equal probability of detection for all angles between 0° and 90° (and zero for greater angles). No data were available for angular search effort from naked eye observers, but the angular search effort using binoculars is shown in Fig. 4 for combined 7 × 50 and Big Eyes (CODA survey). The function to describe searching effort by angle fitted to these data by least squares is given in equation 3. If detection probabilities in the simulations were multiplied by the fitted effort function in equation 3, this would reduce estimated strip widths to approximately 50% of what they would be assuming uniform search effort

$$y = \frac{\pi}{2\int^{\pi/2} \cos(x)^{6.9}\,dx}\cos(\theta)^{6.9} \qquad (3)$$

DISCUSSION

All the datasets of distances to sightings of surfacing cetaceans showed a consistent pattern of over-estimation of small radial distances and under-estimation of larger ones. This could be a result of rounding effects at small reticle readings if observers tend to round up the reticle reading, and difficulties in counting reticles at larger reticle readings. The same pattern was also apparent in the naked eye estimates but for naked eye this could also be explained by the high variance of the estimates. Williams *et al.* (2007) reported a similar error pattern from an observer using 7 × 50 binoculars from a platform height of 18.3m. In that case, the angle of dip at which errors changed from over-estimates to under-estimates was 0.25º, or 1.8º subtended at the eye. The consistency of the angle subtended at the eye (1.8º–3.25º) may provide some insight into the visual processes involved in distance estimation using reticles and corresponds roughly to the angle of foveal (high acuity) vision. This could be investigated further by specific experiments involving different magnification binoculars and different observation heights. However, such experiments would need to involve real sightings targets because the results indicate that distance experiments to fixed targets do not show the same patterns of distance errors.

The implications of the compression of the range of true

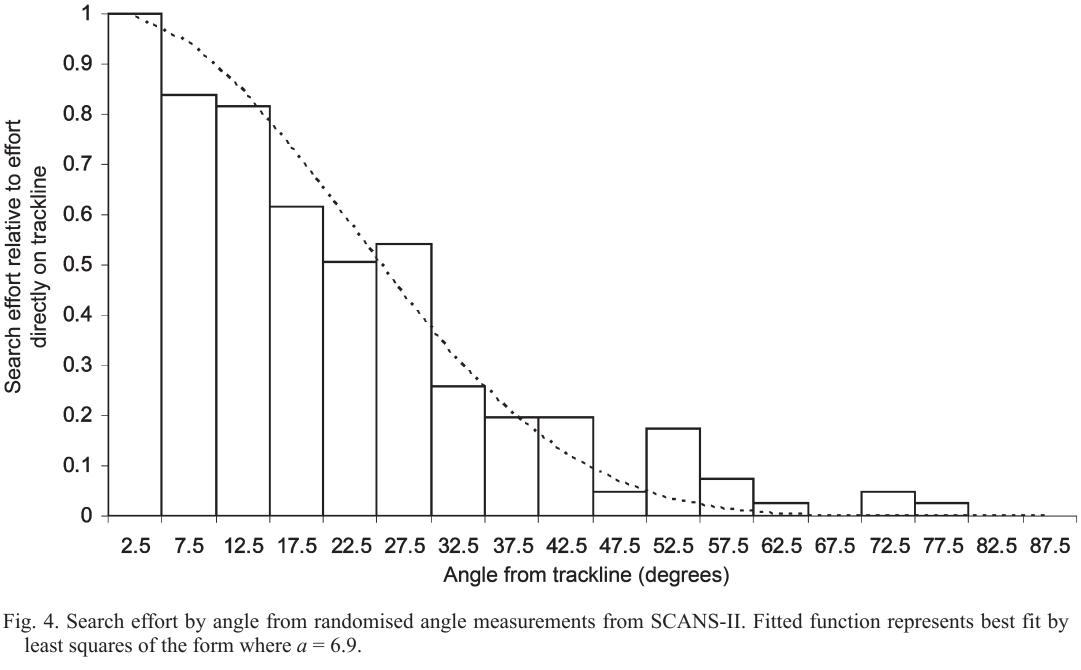


Fig. 4. Search effort by angle from randomised angle measurements from SCANS-II. Fitted function represents best fit by least squares of the form where *a* = 6.9.

distances for abundance estimation are not easy to predict. The overall distribution of radial distances to sightings will affect the direction and extent of any overall bias. We have only investigated the effects on methods using perpendicular distances but cue counting methods may be especially sensitive to non-linearity in errors in distance estimation (e.g. Borchers *et al.*, 2003) because these are based on area (i.e. square of distance).

One result apparent from the SOWER 2007/08 data was the comparison between the distance estimation errors during buoy experiments ($CV_{RMSE} = 0.13$ for observations from the barrel) and to whales during survey conditions ($CV_{RMSE} = 0.24$). It would be expected that estimated distances to a stationary object that remains at the surface are more accurate than those to whales and this is apparent from these results. There was also no evidence of the non-linear pattern in distance errors to buoy experiments that is common to the other datasets to actual sightings (Fig. 2g). These results suggest that distance experiments using fixed buoys may not yield much information about the errors that occur under real conditions. Williams *et al*. (2007) reached a similar conclusion, finding that errors in distances to transient cues were larger than those to cues that were visible for a longer period of time. There are also dangers in correcting for estimation error based on simple linear regressions. For example, in the case of the SCANS-II 7 × 50 estimates, a simple linear regression would suggest that distances were underestimated by around 7% (Table 1). Nevertheless, the simulation results in Table 2 would suggest that in this case strip width is likely to be overestimated (by 26% for a strip width without error of 391m). Thus a simple linear multiplier applied to distances would actually exacerbate the error.

The results presented here all involved data that have been through a careful validation process, both at sea and also prior to analysis. Recording distances and bearings by two separate methods allowed an initial screening for gross errors which could then be checked against the complete verbal commentary for each sighting. This validation process involved double checking around 10% of sightings which showed the greatest discrepancies. Although the majority of these cases involved errors with the estimated values, there were also errors in measured values. Errors in measured values could be corrected because all the raw images were stored. Overall, the rate of large discrepancies was higher than might have been expected, but was only apparent because of having two independent sets of data and there was no reason to assume that this was not typical of most surveys.

The patterns of non-linear measurement error observed in this study would appear difficult to correct without at least a substantial number of measurements to real sightings for comparison during a survey. The photogrammetric methods used provide such measurements and as techniques improve, measuring distances should be successful for an increasing proportion of sightings. The use of high definition video has resulted in a marked improvement in image quality on the most recent surveys (CODA and SOWER 2007/08). Detecting minke whale blows in the Southern Ocean on the standard resolution video images was identified as a problem in the 2006/07 SOWER data (Leaper, 2007). However, there were insufficient sightings of minke whales during the video experiments on SOWER 2007/08 to establish whether the high definition video was capable of detecting minke whale blows across the range of distances that blows are detected by visual observers.

Errors in angle measurements appear less likely to cause bias than errors in distances, but will affect the variance of estimates. There was no evidence of changes in angle errors with angle and thus an additive model should be appropriate for angle error. Additive errors for bearings will cause a small bias in perpendicular distances because for a true angle θ and angle error α

$$\frac{\sin(\theta + \alpha) - \sin(\theta)}{\sin(\theta) - \sin(\theta - \alpha)} < 1 \tag{4}$$

i.e. the increase in perpendicular distance due to a positive angle error will be less than the decrease due to a negative angle error. For a RMSE of α of 7° or less, this bias will be less than 1% for any θ and so is not a major concern. The effect on the variance of the perpendicular distances may need more consideration. Although there was little evidence of angle error causing overall bias, the contribution to the variance will be dependent on the distribution of angles to sightings (Fig. 3). For sightings at 10°, 20° and 30° from the trackline, an RMSE in angles of 7° would contribute to a CV of perpendicular distances of 0.69, 0.34 and 0.21 respectively. Measurements of the proportion of time spent searching by angle sector do show differences between surveys, with 80% of search effort within 26°, 37°, 34° for SCANS-II Big Eye, SCANS-II 7 × 50 and SOWER, respectively. Thompson and Hiby (1985) found that over 80% of sighting effort was within 22.5° of the trackline on the 1983/84 IDCR cruise.

In conclusion, the contribution to the CV of the final abundance estimate from distance and angle estimation errors may be considerably greater than typical CVs for cetacean surveys that do not take these factors into account. In addition, estimation errors may also cause biases of similar or greater magnitude. Although simple linear regressions indicated that none of the surveys showed substantial overall bias, bias can nevertheless occur due to the non-linear relationships between errors and distance. In the case of the simultaneous sightings from SCANS-II, the bias would have been 29% if the survey had been reliant on naked eye estimates. The lack of a significant correlation between the truncated naked eye estimates (over the distance range of 200–500m) and measured distances, highlights the difficulties of estimating distances by naked eye. Distance errors are difficult to predict or correct from typical distance experiments using fixed targets and ultimately there appears no substitute for measuring these at sea. Video systems are still not at the stage where close to 100% success in obtaining images to sightings can be expected, but high definition cameras have allowed considerable improvements. Operating and maintaining complex electronics in harsh marine environments also remains a challenge. For example, one vessel on the CODA survey had major technical problems resulting in no measured distance data. Nevertheless, compared to increased ship time, investment in measurement technologies would appear likely to be a more cost effective way of reducing the CV of the resultant estimates, in addition to reducing the possibility of bias.

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Appendix

Leaper and Gordon (2001) described a system for photogrammetric measurement of bearings based on a digital camera attached to the binoculars used by the observer. Mounting the camera on the binoculars has the advantage of moving with the observer and ensuring alignment in a vertical plane because the observer will be holding the binoculars horizontal. The disadvantage is the additional weight for the observer. Observers on the SCANS-II and CODA surveys used a monopod with the 7 × 50 binoculars which took the full weight of the system. On SOWER, observers use a shorter binocular support and are sensitive to additional weight. Thus the system used for SOWER involved downward pointing cameras mounted above the observer. Two cameras were used, one with a remote shutter release (infra-red) which was pressed to obtain a bearing to a sighting and a time-lapse camera taking images every 30s to investigate scanning patterns. This system was very similar to that used on the 1983/84 IDCR cruise (Thompson and Hiby, 1985) except that the cameras were only used to monitor the starboard observer rather than the whole barrel. Two digital cameras, Pentax Optio S10 (for bearings to sightings) and GEC A835 (for time lapse) were mounted in a small, waterproof Lexan case as close to vertically above the observer as possible (Fig. A1). A white stripe was attached along the line of the binoculars to allow measurements. The infra-red remote control for the Pentax Optio was also mounted in a small waterproof box with a large waterproof push button.

It was not possible to position the camera box directly above the observers and so there was some error in bearing measurement due to parallax. This was measured using images of the angle board and found to be less than 1° for all angles within the search area of the starboard observer (the error to the binoculars will be slightly less than this because these were closer to directly beneath the cameras).

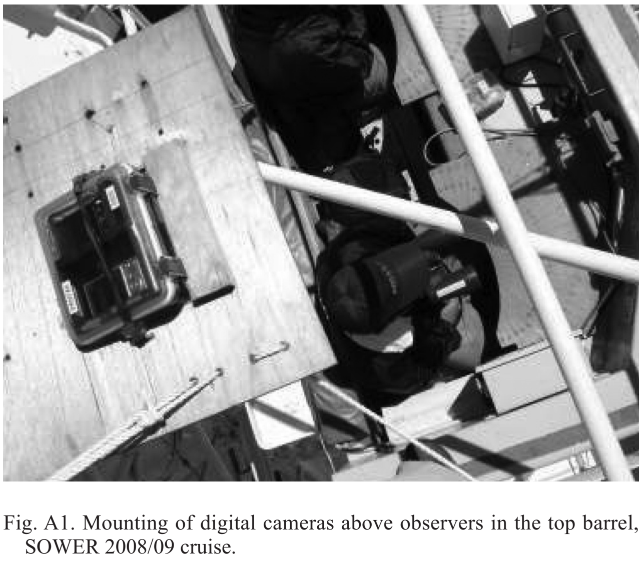

Fig. A1. Mounting of digital cameras above observers in the top barrel, SOWER 2008/09 cruise.

An inventory and evaluation of unmanned aerial systems for offshore surveys of marine mammals

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ABSTRACT

A literature review, internet searches and communications with personnel working with unmanned aerial systems (UAS) were used to identify the capabilities of UAS throughout the world. We assessed their ability to replace manned aerial surveys for marine mammals, sea turtles and seabirds and to monitor, in real time, sea ice and other physical features that might influence marine mammal distribution. The vast majority of the systems identified were either too expensive or their capabilities did not meet minimum standards necessary to perform the tasks required of them in real time. Eight systems were identified that might be able to perform some of the desired tasks. Several other systems had similar capabilities but had not been tested or would require upgrades. Installation of high-definition (HD) video and better stabilisation systems would improve UAS performance. It is recommended that development of HD video with real-time data transmission and improved stabilisation systems for UAS be pursued and that side-by-side comparisons of a few of the best systems be conducted.

KEY WORDS: INDEX OF ABUNDANCE; MONITORING; NOISE; SHORT-TERM CHANGE; SURVEY-AERIAL; VIDEO; UNMANNED AERIAL SYSTEM

INTRODUCTION

Dwindling oil and gas supplies and increased demand for existing reserves have prompted exploration and production (E&P) activities to expand into offshore areas that were considered inaccessible in the past. In many jurisdictions, concern about the potential impacts of these activities on marine resources, particularly marine mammals, sea turtles and seabirds, has created a requirement for E&P companies to assess and monitor marine resources to help minimise impacts of their activities on these resources. Because some species of marine mammals appear to react to the presence of E&P activities at distances that cannot be monitored from the platforms conducting the activities (Miller *et al.*, 1999; Richardson *et al.*, 1995), observations from other vessels or aircraft are sometimes required to document such behaviour. In these cases, accepted monitoring and mitigation methods cannot be used when vessels are too far offshore to safely conduct manned aerial flights, and some E&P activities face temporal and spatial restrictions. Thus, new tools and methods are urgently needed to effectively monitor marine resources in offshore areas so that activities can be conducted there without having adverse impacts on species of concern.

Marine mammals have been the main marine resource of concern because they tend to be more sensitive to sounds produced by offshore activities than sea turtles or seabirds. Currently, visual vessel-based marine mammal monitoring programs are conducted from most seismic vessels (and some other E&P platforms) used for offshore oil and gas exploration (Johnson *et al.*, 2007; Moulton *et al.*, 2006; Patterson *et al.*, 2007; Stone, 2003) and, more recently, academic geophysical research (Holst *et al.*, 2005). Observations have also been conducted from artificial islands where production facilities are present (Richardson, 2006). The focus of these monitoring programs has been to detect marine mammals that are close to the activity so that mitigation measures can be implemented to avoid adverse effects on them by such measures as reducing or ceasing activities when marine mammals are observed within project-specific safety distances. When the zone of responsiveness has been too large to monitor from a vessel, aerial survey programs have been conducted at sufficient distances ahead of the vessel to allow surveyors to modify the timing and locations of activities so that the activities do not impact those species, particularly sensitive components of the population such as mother-calf pairs (Yazvenko *et al.*, 2007a; 2007b). An alternative method of real-time monitoring marine mammal presence has been by the use of towed passive acoustic monitoring systems (PAM) to record or detect animal vocalisations. PAM can be used at night and during periods of bad weather. However, detection rates are often lower than with visual methods, locations of calling animals are often not precise enough to use for estimating density or to determine if animals are within defined safety radii of the activity and call detection range often is not sufficiently large to monitor safety radii around intense energy sources such as large airgun arrays. In addition, towed PAM arrays are not effective for species with low vocalization rates or near noisy activities that cause animals to cease or reduce calling. If the technology were verified, unmanned aerial systems (UAS) launched and recovered from a vessel may be able to provide unique platforms to monitor marine mammal distribution and abundance in areas where aircraft cannot safely operate. They may be able to survey a large enough area to monitor sound-based safety radii such as those required to be monitored by the US National Marine Fisheries Service (NMFS) for marine

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mammals around intense energy sources and, unlike manned aircraft, would not be restricted as to how far from land they could operate.

Selection of UAS that might be suitable for use in offshore areas is challenging because the technology is new and rapidly evolving, a large number of systems are available and few systems have been tested specifically in offshore areas. Today, about 45 countries fly more than 600 different UAS models; in the USA alone, there are approximately 280 companies, academic institutions, and government groups developing more than 200 different UAS designs ranging in price from $1,000 to $26 million (www.thirtythousandfeet.com/uav.htm).

Currently, surveys with manned aircraft are conducted in nearshore and offshore areas within ~200km of shore to obtain unbiased real-time estimates of animals present because the aerial survey platform does not influence the distribution or behaviour of the animals that are being counted. In far offshore areas, where aerial surveys are not conducted due to safety concerns, ship-based surveys are used to survey animals. It is known that many species of marine mammals and seabirds are either attracted to or avoid vessels (e.g. Barlow *et al.*, 2006; Würsig *et al.*, 1998), resulting in biased estimates of distribution and abundance. If UAS are found to be a suitable platform for conducting marine wildlife surveys, then unbiased estimates of their distribution and abundance in offshore areas could be obtained. These data can be used to assess and manage potential impacts of various types of activities on marine mammals.

UAS can also be used to collect environmental data that might influence marine mammal distributions. Sea ice affects marine mammal distribution and UAS can provide real-time information on ice and ice movements and other physical features of the offshore environment. In many situations these data could not be collected using methods such as satellite imagery because of cloud cover over the survey area or because resolution of the imagery does not provide sufficient detail. Even low-resolution imagery from UAS equipped with infrared sensors may be more effective than manned surveys to detect some marine mammals such as polar bears and walruses.

With all of the above uses in mind, the objectives of this study were to:

- compile UAS characteristics deemed important for monitoring marine animals and physical features such as ice, and compile research on UAS that might be applicable for their use in offshore areas and harsh environments;
- review and assess each UAS with respect to its cost, availability and technical characteristics;
- evaluate the ability of existing UAS and sensors to meet requirements for use in offshore areas and review studies that have tested this technology;
- identify areas of further technological development that would improve the ability of UAS to accurately detect, classify and track marine mammals, turtles and seabirds; and
- identify political or regulatory barriers (including patents) to advancing the state of knowledge and acceptance of the technology.

METHODS

Initially, a list of the range of capabilities of UAS and sensors was developed. Capabilities of UAS vary from model airplanes that are controlled by a joystick within a range of a few kilometres to high-altitude UAS used for military applications that have ranges of 1,000s of km and can fly at 15,000m above sea level. The information on the low-tech UAS, in particular, is voluminous, and setting boundaries on

Table 1

Criteria used to evaluate whether UAS are suitable as real-time data collection platforms for wildlife surveys.

Vehicle characteristic	Requirements
Size	UAS of all sizes were considered, but if range (<200km) or flight duration (<4h) would not permit launch and recovery from land, then vehicles needed to be small enough to be handled by 1–2 people aboard a vessel.
Cost	Aircraft needed to be <$250,000 because of risk of loss and the need for multiple aircraft for back-up or to house different sensors for different applications.
Payload capacity	A payload capacity of 2kg or more was deemed necessary to carry sensors and fuel.
Vehicle control	Both real-time flight control and pre-programmed flight control were considered, but real-time flight control to 50km is necessary.
Distance of operation from base	UAS needed to be able to fly >20km from launch location if launched and recovered from a vessel and >200km if launched and recovered from land. See also Vehicle control requirements.
Flight duration	Minimum flight duration was 1h if operated from a vessel or 4h if operated from land. For most applications, flight duration in the survey area needed to be >4–6h.
Operating capabilities	UAS need to be able to operate in remote areas, such as the Arctic, with minimum logistics support and during most conditions when manned aircraft could fly.
Speed	A minimum airspeed of 46km h^{-1} is needed to permit flying during moderate winds.
Fuel	Fuel or power for the UAS had to be readily available and non-hazardous. Gasoline was considered acceptable.
Launch/recovery requirements	The aircraft could be launched and recovered either from land or from a vessel, depending on flight duration (see Flight duration).
Sensor capabilities	A wide variety of sensors was considered to meet a wide variety of needs. These included, but were not limited to, sensors to detect marine mammals (visual, infrared, UV, night vision); map ice conditions; measure water temperature, ocean currents, chlorophyll, weather variables including wind speed and direction, air temperature, humidity and cloud cover.
Sensor size	Sensors as large as 20kg were evaluated, but to be useable on current UAS sensors needed to be no heavier than 2–5kg.
Video resolution	Video resolution needed to be 640 × 480 pixels or better.
Image stabilisation	Imagery needed to be stabilised to reduce motion/vibration and to allow clear imagery when scanning a large area.

the information that would be integrated into the evaluation was necessary. Based on prior experience with using UAS in marine mammal monitoring (Koski *et al.*, 2009; 2007; Lyons *et al.*, 2009), a set of criteria for evaluation of UAS was developed (Table 1). The most important criteria included the ability to launch and recover the aircraft from a mid-size vessel; flight endurance of at least 4 hours; payload capacity of 1.5–2kg to accommodate high-quality sensors; a broadband datalink which allows National Television System Committee (NTSC), Phase Alternating Line (PAL) or Advanced Television Systems (ATSC or HD) video to be streamed back to a control station; and reasonable cost.

Based on the criteria in Table 1, a list of UAS and sensors was prepared using various data sources, i.e. technical reports, internet searches, UAS newsletters and contacts with UAS suppliers or people who have conducted research on UAS and various types of sensors. Personal contacts with companies' representatives provided much useful information. In some cases, a system that was best suited for offshore surveys was in development or only recently available, and therefore would otherwise have been missed. Alternatively, some systems that seemed highly suitable were rejected based on the additional information obtained from these individuals or because they were no longer in production.

Studies were identified that have evaluated UAS and potentially useful sensors for use in marine wildlife surveys. Because of the relative scarcity of the published and grey literature, internet and personal communications turned out to be the main sources of information on the present status in this area. A variety of websites were browsed, including manufacturer's sites, the sites of various UAS associations, meetings and exhibitions; various blogs were included in the subsequent analysis and forums related to UAS.

Technical parameters for each UAS and sensor that met the criteria in Table 1 were tabulated. The requirements in Table 1 were intentionally set low so that marginal systems would be included with the hope that future upgrades would improve performance. The tabulated data also included an assessment of availability for civilian use, and contact names and numbers of suppliers. When tables were completed, each system was evaluated as being good, fair-good, fair or poor based on the criteria in Table 1. During the ranking, emphasis was placed on cost, control (remote or autonomous), flight duration, operating range, the requirements for real-time vs. delayed data collection and analysis, and the potential to train biologists to operate such a system. In the evaluation and ranking, we considered two markets separately because of political and military boundaries: North America, Europe, Israel and Asia vs Eastern block countries, which included Russia and the countries of the former Soviet Union.

Research and testing that have been done on UAS and sensors were also reviewed and considered during evaluation (see Results and Discussion).

Finally, areas were identified where further technological development would improve the ability of UAS to provide the data required including the ability to accurately detect, identify and track marine animals. The political, regulatory and patent barriers to advancing UAS technology were also identified.

RESULTS AND DISCUSSION

Of the 600 or so UAS that are advertised, in production or in development, about 400 were briefly evaluated. Of these, 162 UAS (aircraft or aircraft plus sensors) and 15 sensors were entered into an evaluation matrix and information on their capabilities was summarised from the various sources mentioned above. Only 12 UAS (7.4% of those evaluated in detail) were considered 'good' prospects for use as a real-time survey platform for marine mammals in offshore areas. Eight additional systems (4.9%) were considered 'fair to good'. The majority of the systems were considered fair or poor and would require significant improvements before they could be used (Table 2).

The eight most promising systems are discussed here, and a general discussion of capabilities and deficiencies in other systems is included in the next section.

Table 2

Summary of numbers of UAS and payloads evaluated in detail.

	Aircraft	Aircraft plus payloads	Payloads	Total
Good	7	5	3	15
Fair to good	4	4	–	8
Fair	30	12	7	49
Poor	36	21	1	58
Could not be evaluated	12	1	4	17
Not available	10	4	–	14
Too expensive*	13	3	–	16
Total	112	50	15	177

*These systems would be classified as good if they were affordable.

Top-rated UAS

Eight UAS were considered to be potentially appropriate for use as real-time survey platforms for marine mammals in offshore areas, two from eastern block countries and six from other regions of the world. None of these systems have been fully tested to establish their efficacy for detection of marine mammals or other tasks for which UAS might be used. Because most of these systems have not been tested, it is likely that some of these UAS would need improvements before they could be used for many applications. Some have not been tested in the Arctic, where cold and icing pose problems not encountered in other regions. The strengths and limitations of each of these systems are discussed below.

The **Insight A-20** (also called the ScanEagle; Insitu Group, Bingen, WA and Evergreen Helicopters, McMinnville, OR) is one of the top-rated UAS in the size and cost range considered practical and is one of only three UAS that have undergone or are undergoing systematic testing of their capabilities as a platform for surveying and observing marine mammals in real time. The other systems tested for use with marine mammals, the Warrigal 2 and the systems tested by the University of Rostock, did not make the list of top-rated UAS (see below). The Insight A-20 was included among the top-rated systems because of the testing that has been done during 2006–2009 and because it appears to meet or exceed the capabilities of the other top-rated systems. In particular, the Insight A-20 can be manually controlled and sensor data can be obtained in real time out to 150km from the control station (depending on flight altitude and antenna height at the base station). Pre-programmed routes can be flown beyond 150km. The long endurance of the Insight A-20 (>20h) facilitates efficient surveying of large areas and minimises the number of launches and recoveries. It is small enough to be easily handled on a vessel (3.1m wingspan) and has an efficient launch and recovery system that can be deployed from an offshore platform or a vessel. It has a sophisticated ground control station (GCS) that provides real-time display and processing of imagery and storage of all data collected. The current video system (NTSC) appears to cover an area approximately the same as a single observer in a manned aircraft and with similar detection probabilities (Koski *et al.*, 2009). If a high definition (HD) video system were installed, it would allow coverage of a larger survey area than was possible during the tests conducted by Koski *et al.* (2009). It is likely that a HD video system would make the Insight, and other systems listed below, suitable for surveying birds and most species of marine mammals (see Discussion on HD video below).

The **Manta B**, which is a larger version of the Silver Fox (Advanced Ceramics Research, Inc., Tucson, AZ), is slightly smaller (2.7m wingspan) and less expensive than the Insight A-20, but has fewer capabilities. Its ability to operate in the Arctic has been proven during research in Greenland. However, currently it cannot meet the 'distance under control' requirements for many offshore marine mammal surveys (control to only 37km), and its endurance of >6h is marginal for large scale aerial surveys since it can be launched, but not recovered, from a vessel. A marine recovery system (in a net) is currently being developed and tested, which would improve its usefulness. The Manta B or Silver Fox could be used to conduct marine mammal surveys in nearshore areas or in offshore areas once the marine recovery system is verified.

The **Arcturus T-16 XL** (Arcturus UAC, Rohnert Park, CA) meets most of the performance criteria for use in offshore areas. It has a 24h flight duration and it can be launched and recovered (in a net) from a vessel. It is slightly larger (3.9m wingspan) than the Insight and Manta B, which would make it slightly more difficult to handle on a vessel. It is less expensive than either the Insight or the Manta B. The major flaws of the Arcturus T-16 XL are the small range under control (16–24km) and the fact that it has not been tested in Arctic conditions. In particular, extending the range under control would markedly increase the value of this system for offshore marine mammal surveys. In its current condition, it could potentially be used to conduct surveys which do not require acquisition of real-time data.

The **CryoWing** (Norut Northern Research Institute, Tromsø, Norway) is one of the UAS that could be used for collection of real-time data on marine mammals in offshore areas. It is relatively inexpensive (€30,000 for the aircraft) but among the larger UAS (3.8m wingspan) that could be deployed from a vessel. CryoWing has been specifically designed by a Norwegian team of scientists to operate in the Arctic and has been tested there. It has flight endurance of up to 20h at speeds of up to 160km h^{-1} and it can be manually controlled out to >70km from the control station. Pre-programmed routes can be flown beyond 300km. The current video system is PAL, which has slightly higher resolution than NTSC but is of similar clarity because of a slower refresh rate. Datalink options include 3G GSM (up to 1Mbit), and up to 7Mbit dedicated radiolink, which might permit use of HD video, but HD video has not been investigated or tested. The main weakness of the CryoWing is that it is not recoverable on a vessel (it is launched by a catapult that could be used on a vessel but it lands on its belly), so it would need to be recovered from land. Its long flight duration and the ability to pass control from one control station to another or pre-program the landing at the end of the flight makes this feasible. In this situation the UAS would become separated from the vessel after the first flight. As an alternative, it could be launched and recovered from land, but this is not practical if operations are far from shore. It is a light system (30kg) but has a relatively large wingspan (3.8m), which would make it slightly more difficult to handle than some of the smaller aircraft if vessel launch and recovery were implemented.

The **Elbit Skylark II LE** (Elbit Systems Ltd, Haifa, Israel) is a system recently developed by one of the world leaders in the UAS industry. The cost of the system was not given by the supplier who did not respond to our request for information. It appears to be one of the more advanced systems but has not been tested in the Arctic. It can carry 9kg of payload, has flight endurance of up to 17h at speeds of up to 74km h^{-1}, and can be manually controlled out to 50km from the GCS. The payloads of Skylark II are among the most sophisticated in its class; a gimballed and stabilised triple-sensor payload (Micro-CoMPASS) includes a colour CCD daylight camera, 3rd generation thermal-imaging night camera and a laser illuminator. Skylark II LE is not currently recoverable on a vessel, but a vessel-based launch and recovery system is undergoing sea trials. Considering the pace of its evolution, Skylark II LE is one of the systems to watch in the next 1–2 years. Elbit has been successful in obtaining recent military contracts, suggesting that it is one of the best UAS that are available. Their failure to respond to our requests, however, suggests that they may be too busy to be responsive to requests from non-military users.

The **Fulmar** (Aerovision Vehículos Aeros, S.L. San Sebastian, Spain) is one of the top rated UAS in the size and cost range (€20,000 for one fully equipped aircraft) considered for use as a platform to conduct real-time surveys of marine mammals. Fulmar has been specifically designed by a Spanish team of scientists to operate at sea, and its capabilities appear to meet most requirements for offshore use. In particular, it can be launched and recovered from a vessel into a net or by descending and sea-landing on a pneumatic skate. It is waterproof, and a satellite radio beacon is incorporated into the aircraft for recovery. Fulmar has flight endurance of up to 8h at speeds of up to 150km h^{-1}, it can be manually controlled out to 100km from the GCS and pre-programmed routes can be flown farther. The data link with the control station at 900 Mhz is out to 100km at 128kbps but the real-time video link at 2.4 Ghz has a maximum range of 50km. It is a light system (19kg) with a medium wingspan (3.1m) and can carry 8kg of payload including fuel.

The **ZALA 421-16** (A-Level Aerosystems, Izhevsk, Russia) is the top rated UAS for the Russia/FSU market. It has a 1.6m wingspan and the cost is €200,000 for two aircraft and a GCS. It is a newly released system (2009) and so is untested. Projects involving ZALA 421-16 on behalf of Rosneft, a Russian oil company, were conducted in offshore Arctic waters during summer 2010. Gazprom, a Russian natural gas company, has contracted the ZALA 421-16 and other A-Level aircraft to monitor its network of onshore pipelines in the Arctic and elsewhere; and the State fisheries committee is considering the ZALA for missions to search for illegal fishing boats offshore of the Kamchatka Peninsula. The ZALA 421-16 appears to meet most baseline requirements for use in offshore cold-water environments. It has flight endurance of 5–7h with speeds of 80–120km h^{-1} (marginal for some needs), can be deployed and retrieved from a vessel, can transmit real-time video to a GCS at distances up to 50km and can be manually controlled out to >70km from the GCS. Pre-programmed routes can be flown beyond 200km. As with the CryoWing, the communications bandwidth can be increased to 7 Mbits (possibly to 20 Mbits), which might make real-time transmission of HD video or medium-resolution (12 megapixel) still images possible.

The **R-100 Marine** (UAVia Pte Ltd, Kiev, Ukraine) can be launched and recovered from a vessel, is small (1.8m wingspan) and can be controlled up to 100km from the GCS. The current version has only 4h endurance (battery powered) but a 10h version (gasoline powered) is being developed. As with most eastern block systems, the R-100 Marine appears to be costly ($1.0M for 3 aircraft and GCS) and it has not been tested for surveys of marine mammals.

Other UAS

There are several other systems that are available or under development or that might become suitable for use for offshore surveys of marine mammals as systems are upgraded. These include the Aerosonde MK-4 and Shadow (Aerosonde Pty Ltd, Notting Hill, VIC, Australia and AAI Corp, Hunt Valley, MD), V-Bat (MLB Co., Mountain View, CA), Warrigal 2 (V-TOL Aerospace Pty Ltd, Brisbane, Queensland, Australia), Resolution (Airborne Technologies, Inc, Wasilla, AK), Skyblade IV (Singapore Technologies Aerospace, Paya Lebar, Singapore), Aerostar and Orbiter 3 (Aeronautics Defence Systems Ltd, Yavne, Isreal) and the S4 Ehécatl (Hydra Technologies, Zapopan, Mexico).

There are several large and sophisticated UAS used for military applications that exceeded the requirements of a system for use during offshore wildlife surveys. However, the cost of operating these systems would be prohibitive, which eliminated them from consideration. In addition, many of these systems are classified and are available only for military use. As the technology advances, and more research and development are done, some of the features in these large, sophisticated systems may become available to the smaller, more practical systems.

A review such as this relies on information provided by vendors and manufacturers. Thus, no actual tests or side-by-side comparisons of systems were made. Based on our experience working with several different UAS, the most common deficiencies have been poor image quality (primarily due to lack of image stabilisation), low or marginal flight duration and the lack of the ability to launch and recover the UAS from a vessel or offshore platform (Table 3). Because these deficiencies have been overcome in some systems, future generations of many of the UAS examined may address these deficiencies. In many cases, systems have not addressed these deficiencies because the

Table 3

Improvements needed for UAS systems to be useful to researchers working in offshore waters.

Limitation	Description of problem	Can this be improved?
Video resolution	The resolution of current systems does not permit monitoring of large areas because the pixel size or resolution is not high enough.	Yes, higher resolution video cameras are available and being tested by some providers. A study with HD video showed it to be as good as manned surveys for estimating densities and identification of birds (Mellor and Maher, 2008). Digital SLR cameras can be used if real-time data collection is not required.
Image quality	Movement and vibration degrade image quality.	Yes, in three ways. The more sophisticated UAS have built in image stabilisation systems and some high end cameras have image stabilisation built into the lens or camera body. In addition, post processing of the imagery can produce a clearer image. That is available in real time for some systems.
Real-time data transition rates	Real-time data transmission rates are limited which prevents use of higher resolution sensors in real time.	Yes, the technology exists for the military.
Limited range with real-time control of UAS	Some applications require real-time acquisition of data.	Better and higher antennas will increase range of control. Satellite linked data transmission is possible at increased cost.
Simultaneous use of multiple sensors	Smaller UAS can only support one sensor at a time because of payload limitations.	Sensors continuously get smaller and some of the larger models can hold multiple sensors. This can also be solved by flying two aircraft, each with a different sensor, at the same time.
Weather-proofing of systems	The ditching of a UAS into sea-water would damage the electronics and, in some cases, possibly the aircraft itself.	Yes, a few systems are designed for offshore operations. Waterproof casings can be designed for almost any system (or system components) and make them operational in offshore environments.
Icing	Systems can be prone to icing in certain arctic conditions.	Systems can be designed to better monitor this risk and reduce the likelihood of icing. Heat can be provided to key locations on the aircraft to reduce or prevent icing.
Launch and recovery limitations	Some systems that are otherwise suitable cannot be launched and recovered at sea	Yes, the smaller aircraft could be captured in nets or on a wire like the Insight™.
Cost	Many systems are too expensive.	Costs will come down substantially when these systems are used for commercial purposes. Current use is by the military and few units have been sold in comparison to the potential civilian market.

market for such systems had not been identified before we contacted the system marketers.

Studies on UAS

To date, few studies have been conducted with UAS either in offshore Arctic regions or for surveys of marine mammals. Six studies were identified that focused on marine mammals. The first was conducted in 2002 by the Office of Naval Research using the Silver Fox and the technology has advanced substantially since that test, so the findings are outdated. Even at that time, the researchers were able to detect and identify humpback whales (NOAA, 2006).

A 2006 study by Shell was the first systematic test of the ability of a UAS to detect objects of interest in a marine environment (Buck *et al.*, 2007; Ireland *et al.*, 2007; Koski *et al.*, 2009; 2007). The surveys were flown in winter conditions in Washington State (they included freezing rain, fog and high winds), which are similar to conditions that would be encountered in the Arctic during the late summer and autumn. Kayaks were used to simulate the dorsal surfaces of whales at the surface that would be available to be seen by marine mammal observers (MMOs) during manned aerial surveys. The kayaks were placed randomly in the search area and the MMOs, who were blind to kayak locations, used a systematic grid to search for them using an Insight A-20. Detection rates varied with sea conditions (greatest influence), kayak colour and kayak inflation, but detection rates with search swaths up to 600m were similar to those reported in the published literature for manned aerial or vessel-based surveys (Koski *et al.*, 2009). The authors concluded that the system tested (Insight A-20) was suitable for surveys of large cetaceans or large groups of small cetaceans, but noted that the search swath was narrower than that covered by a manned aircraft. The narrower search area could be compensated for by the longer flight duration of the UAS and by flying during periods with ceilings <300m when manned aircraft are not permitted to fly because they could disturb marine mammals.

A follow-up 2008 study by Shell and ConocoPhillips (Lyons *et al.*, 2009) showed that the Insight A-20 could be operated successfully in offshore Arctic waters. It was flown for 32h over a 10-day period, and several cetaceans and pinnipeds were sighted and captured on video. The 2008 study was constrained by US Federal Aviation Administration (FAA) requirements to remain within one nautical mile of the vessel and requirements for a cloud ceiling of at least 300m before the UAS could be flown. This prevented a useful evaluation of the efficiency of the UAS in comparison to surveys by manned aircraft.

From mid-May to mid-June 2009, the National Marine Fisheries Service (NMFS) conducted a series of tests in the Bering Sea with a ScanEagle launched from the NOAA research vessel *McArthur II.* The ScanEagle was fitted with a downward-facing digital SLR camera to identify and estimate densities of seals occupying pack-ice habitat in the Bering Sea (Cameron *et al.*, 2009). Although the tests were constrained by FAA operating requirements, the study confirmed the abilities of a UAS to operate in a variety of sub-arctic weather conditions and to obtain imagery of sufficient quality to distinguish the different species, ages, and occasionally, even the gender of ice-associated seals.

Two additional studies are underway to investigate the use of UAS for surveying marine mammals. One is at the University of Queensland, Australia, (Monaghan, 2008) and the other is at the University of Rostock, Germany (Grenzdörffer, 2008). Both studies are ongoing and results are pending.

Memorial University, Canada, and Provincial Aerospace Limited are testing an Aerosonde MK-4 for the potential monitoring of illegal fishing and pollution in the North Atlantic off Newfoundland and Labrador. This study is ongoing and results are not available yet. Of more importance to the present review, this group is also working on the development of an autonomous collision avoidance system for small UAS. As noted in the next section, development of such a device is important to permitting considerations for use of UAS in many areas.

NOAA and Airborne Technologies are testing the Resolution (one of the UAS listed in the ‘Other UAS’ section) for detection of abandoned fishing gear. An interesting finding by Churnside *et al.* (2009) during these tests was that an infrared sensor could detect whale tracks in temperate areas by thermal disturbance at the water surface. During earlier tests in the Arctic, however, biologists were unable to locate bowhead whales or their tracks with infrared sensors even though the whales could be seen in imagery collected using low resolution colour video (W. Koski, unpubl. data).

University of Colorado scientists used the Aerosonde MK-3 to study ice roughness and surface temperatures and they identified and implemented modifications to the UAS to permit flying in the Arctic (Curry *et al.*, 2005). The modifications suggested during these early UAS studies have resulted in increased safety and efficiency of UAS operations in the Arctic.

A study of the test of a Cineplex gyroscopically stabilised high-definition (HD) (1080 × 1920) colour video has been included in the review because HD video is being modified for use in some of the UAS reviewed. Mellor and Maher (2008) tested this system in a small fixed-wing aircraft flying at 600m above sea level with a 30–40m surface coverage. The objective of the test was to determine if the HD video in a fixed-wing aircraft was suitable for obtaining information on species, distribution and abundance of seabirds near offshore wind farms. The target species included alcids (Alcidae), common scoters (*Melanitta nigra*) and cormorants (*Phalacrocorax* spp.), which are dark-coloured birds that are difficult to detect and identify during manned aerial surveys. The smaller of these species are approximately 35cm long when swimming on the water. The study concluded that the target species could be detected and identified easily in the imagery that was obtained, and that birds were less likely to be disturbed than during lower-level manned surveys.

Problems with UAS use

There are many problems involved in using UAS to replace manned aerial surveys. These include acceptance of the technology by regulatory bodies that issue permits, responsiveness by UAS providers, export restrictions on UAS and aviation-related restrictions on flying UAS in many jurisdictions. However, the main problem with their use for conducting marine mammal surveys is that they have not been systematically tested and data collected from UAS have not been compared with those from manned aerial and ship-based surveys.

Benefits of using UAS

UAS with video streams or digital still cameras would have many advantages over manned aerial surveys. When still photography is used, there is constant detectability across the search swath, or if there is reduced detectability near the edges, it can be quantified through analysis of the imagery. The lateral distances from the trackline can be measured rather than estimated and would be more precise. Group sizes can be counted more accurately and the relative sizes of animals (adults vs subadults vs calves) can be determined from the film, resulting in collection of more information than is collected during most manned aerial surveys. Also, all data can be reviewed by more than one observer permitting estimation of counting and detection errors, which tend to be much lower during analysis of photographs than during manned aerial surveys (Heide-Jørgensen, 2004). The ability to review imagery eliminates the need to conduct double platform or independent observer experiments to quantify detection bias.

Another benefit of still and video imagery is that an estimation of the surfacing time of whales can be obtained by analysing sequential images as was done by Heide-Jørgensen *et al.* (2009). This alleviates the need for other studies to obtain estimates of availability bias. As a result of the ability to review imagery, Ferguson and Angliss (2010) note that a more precise estimate of group density might be obtained from UAS surveys than manned aerial surveys (using conventional distance sampling methodology and assuming the UAS platform allows for accurate species identification) because UAS data may be able to better account for or eliminate detection biases.

A potentially large benefit to users of UAS over manned aerial surveys or observers on vessels is that data streams from UAS can be transmitted in real time from the GCS, where data are received from the UAS, to all parts of the world through the internet. Some systems like the Insight A-20 and CryoWing have used this capability for some studies, and although not demonstrated for many systems, it is a relatively simple process to implement, provided that high speed internet access is available at the GCS. By using this capability, groups conducting offshore surveys could minimise the numbers of people on vessels in offshore areas and do some data processing in the office in real time. Because bunk space is usually limited during offshore activities, and it is safer and more cost-effective to have personnel working in the office rather than the field, this would provide significant cost and safety benefits to the users.

CONCLUSIONS

Many of the UAS investigated during this study would be suitable for collecting data on marine mammals and their habitats (i.e. ice cover and oceanic fronts), but only a small fraction of them may be useful for replacing manned aerial surveys. Those UAS that might be suitable have sensors with sufficient resolution to conduct surveys of large cetaceans or of large groups of small cetaceans, but the search area is smaller than that covered by a manned aircraft and the survey speed is slower; thus, a ~3–4h survey by a single UAS would be needed to obtain similar coverage as a 1h survey by two MMOs in a single manned aircraft. However, estimates from UAS may be more precise because they eliminate or can better correct for perception bias (Ferguson and Angliss, 2010). Given that some UAS can survey for up to 24h without refuelling, whereas manned aircraft cannot survey more than ~3–6h, and that UAS can fly at lower altitudes without disturbing animals, a UAS may be able to obtain the coverage needed to replace manned aerial surveys. In some situations, UAS might obtain coverage when a manned aircraft could not survey because of low cloud in the survey area or at the aircraft base.

The highest HD video currently available provides 6.75 times the number of pixels in a frame than does NTSC video; as a result, it could cover an area three times wider than NTSC video with the same resolution. Introduction of stabilised HD video into a UAS probably would provide imagery that would be as good, or better, than data collected during manned aerial surveys. As demonstrated during the Mellor and Maher (2008) study, in some cases HD video could provide better data than manned surveys because species identification from the video may be better than that possible during manned aerial surveys. In part, this is because of the ability to review characteristics of a sighting from the digital record, which cannot be done during real-time manned aerial surveys. Thus we would recommend that development of HD-video capture and transmission be encouraged. HD video may be the break through that would permit use of UAS for surveys of birds and small marine mammals in offshore areas.

Image stabilisation is another limiting factor in the use of UAS for wildlife surveys. UAS are small and unstable platforms for capturing visual data. Development of better stabilisation systems for sensors would increase the quality of imagery and permit more efficient surveying.

See-and-avoid systems should be developed for UAS. One of the major road blocks to using UAS in most jurisdictions is the lack of a see-and-avoid system that would prevent a UAS from colliding with an aircraft.

This study evaluated the ability of UAS to collect real-time data and that requirement eliminated from consideration many platforms that can collect data on board the UAS for later analysis. Digital still cameras provide higher resolution than video cameras, allowing coverage of a larger area and/or identification and counting of smaller species of marine mammals than can be conducted using video. UAS are therefore ideal platforms to use for high-resolution photographic surveys, particularly in nearshore areas such as the fjords in Greenland (Heide-Jørgensen, 2004) or nearshore areas in Canada (Richard *et al.*, 1994; Stenson *et al.*, 2002) where photographic surveys have been used to estimate numbers of narwhals (*Monodon monoceros*) and harp seals (*Pagophilus groenlandicus*).

In summary, several UAS are available that would be suitable for monitoring offshore ice conditions, oceanographic fronts, wave height and some other physical features of the offshore environment, but more testing is needed before UAS can be used as replacements for manned aerial surveys of marine mammals and birds. Side-by-side testing should be conducted using the most promising systems, and high-resolution digital still photography should be tested for counting marine mammals at haul-out sites and for estimating densities of marine mammals in offshore areas. Development of better image stabilisation systems and implementation of higher-resolution video is recommended to improve the capabilities of current UAS.

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Asymptotic bias of the hazard probability model under model mis-specification

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ABSTRACT

We compare the sensitivity of the estimated effective strip half-width with respect to choice of hazard probability function (*Q*). This is done by fitting the model under an erroneous assumption about the parametric form of *Q*, and comparing the estimated and true effective strip half-width. An 'infinite sample size' setting is employed, where fitting the model by maximum likelihood amounts to minimising the Kullback Leibler distance between the assumed and true models. The experiment is carried out in a situation that is relevant to minke whale sighting surveys both in the Antarctic and in the northeastern Atlantic. It is found that the hazard probability model is fairly robust with respect to the choice of parametric class for *Q*. The largest observed bias in the resulting effective strip half-width is less than 10%, while for most situations there is almost no bias.

KEY WORDS: ABUNDANCE ESTIMATE; g(0); SURVEY-VESSEL

INTRODUCTION

The hazard probability model has been used within the International Whaling Commission's Scientific Committee to model independent observer line transect data for minke whales, because it directly takes into account the discrete availability of the animals (Okamura *et al.*, 2003; Skaug *et al.*, 2004). The hazard probability function $Q(x, y)$ is defined as the probability of observing a cue that occurs at relative position (x, y), given that the observer is not previously aware of the whale. Here, x and y are perpendicular and forward distances (km), respectively. The purpose of the present paper is to study how sensitive quantities such as the effective strip half-width and the perpendicular distance density are to the choice of . For this purpose, we perform a pairwise comparison of four alternative parametric families for Q. For each comparison we take one Q as being the truth, with the other being treated as an approximation (Q^*). We then tune the parameters of Q^* such that the Kullback-Leibler (KL) distance between the models is minimised and finally we compare the corresponding effective strip half-widths, w and w^*.

MATERIAL AND METHODS

Hazard probability model for independent observers

Consider first a single observer with hazard probability function $Q(x, y)$, and assume that the whales are stationary (do not move) and surface according to a Poisson process. The detection function, i.e. the probability of detecting a whale that is present at perpendicular distance x, is given as

$$g(x) = 1 - \exp\left(-\frac{\alpha}{v}\int_0^\infty Q(x, y)\, dy\right),$$

where α is the surfacing rate of the whale, and v is the speed of the observer. The probability density of the relative position of the initial observations is given as

$$f(x, y) = \frac{\alpha}{vw} Q(x, y) \exp\left(-\frac{\alpha}{v}\int_y^\infty Q(x, u)\, du\right), \; 0 \leq x \leq W, y \geq 0,$$

where w is the effective strip half-width given by

$$w = \int_0^W g(x)dx = \int_0^W \left[1 - \exp\left\{-\frac{\alpha}{v}\int_0^\infty Q(x, y)\, dy\right\}\right]dx.$$

Observations falling outside the observation strip $(0, W)$ are discarded.

There are typically two or more independent observers (or observer platforms). In the common minke whale (*Balaenoptera acutorostrata*) surveys in the northeastern Atlantic a symmetric two-platforms design is used (Skaug *et al.*, 2004), while in the surveys for Antarctic minke whales (*B. bonaerensis*), three platforms, with a partly asymmetrical configuration, have been used (Okamura *et al.*, 2003). For simplicity, we shall adhere to the setting of Skaug *et al.* (2004) and assume that there are two independent observers, which we denote by A and B, having the same Q function. The combined observer $A \cup B$, i.e. viewing A and B as being a team, has hazard probability function

$$Q_{A \cup B} = Q_A + A_B - Q_A Q_B = 2Q - Q^2.$$

To get expressions for $g(x)$, $f(x, y)$ and w for the combined observer $A \cup B$ we can directly insert in the above formulae. Further, each animal detected by sets up an experiment with trinomial outcome $u \in \{A, B, AB\}$. Conditionally on the position (x, y) the probability distribution of u is

$$q(u \mid x, y) = \{Q_{A \cup B}(x, y)\}^{-1} \begin{cases} Q_A(x, y)\{1 - Q_B(x, y)\}, & u = A; \\ Q_B(x, y)\{1 - Q_A(x, y)\}, & u = B; \\ Q_A(x, y)Q_B(x, y), & u = AB. \end{cases}$$

Via the above formulae, the true hazard probability function Q^T and its approximation Q^* induce two different probability distributions for the datum (x, y, u). The KL distance between these distributions is given as

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$$KL = \sum_u \int_0^\infty \int_0^W \log \frac{f_T(x,y,u)}{f_*(x,y,u)} f_T(x,y,u)dxdy \qquad (1)$$

$$= \sum_u \int_0^\infty \int_0^W \left[\log \frac{q_T(u\,|\,x,y)}{q_*(u\,|\,x,y)} + \log \frac{f_T(x,y)}{f_*(x,y)}\right] f_T(x,y,u)dxdy$$

$$= \int_0^\infty \int_0^W KL(u\,|\,x,y) f_T(x,y)dxdy + \int_0^\infty \int_0^W \log \frac{f_T(x,y)}{f_*(x,y)} f_T(x,y)dxdy,$$

where

$$KL\,(u\,|\,x,y) = \sum_{u=A,B,AB} \log \frac{q_T(u\,|\,x,y)}{q_*(u\,|\,x,y)}\, q_T(u\,|\,x,y).$$

Here, we have exploited that $f(x, y, u) = q(u\,|\,x, y)\, g(x, y)$. In the expression for KL above, g denotes the density based on $Q_{A\cup B}$.

Experimental setup

The four parametric forms $Q_1 - Q_4$ considered are shown in Table 1. Each Q was in turn taken to be the true model (Q^T), while treating the three others as approximating models (Q^*). For a given Q^T the parameters of Q^* were chosen so that the KL distance (1) was minimised. The practical interpretation of this is to use maximum likelihood estimation under an erroneous model assumption, in a situation where an infinite amount of data (from the correct model) is available. The data being fit to consisted of two parts: (i) the initial position for the combined observer $A \cup B$, i.e. the position (x, y) where the whale was first detected (regardless of whether it was A, B, or both that actually made the detection); and (ii) the outcome $u \in \{A, B, AB\}$ of the trinomial trial. Observations falling outside a strip $(-W, W)$ were discarded.

The parameter values used as the 'true values' for each of the four functions are given in the first column of Table 2. For $Q_1 - Q_3$ – these values were based on Antarctic minke whale data (CP 3, Area 5, Okamura and Kitakado, 2009a) and for Q_4 the parameter values were based on northeastern Atlantic minke whale data (Skaug *et al.*, 2004). In the Antarctic setting (three upper panels of Table 2) we truncated at $W = 2$ km, while $W = 1$ km was used in the bottom panel of Table 2 due to the much shorter effective strip half-width in the Northeastern Atlantic. The vessel speed was taken to be 11.5 knots, and mean surfacing rate was 48 surfacings per hour.

Table 2

Parameter estimates of approximating models (columns 2–4) that minimise the KL distance to the true model (column 1).

True model		Approximation					
Model 1		Model 2		Model 3		Model 4	
σ_x	1.1779	σ_r	0.1995	σ_r	0.1704	λ_r	0.8029
σ_y	0.0354	σ_θ	3.3828	σ_θ	2.5958	λ_θ	1.3408
γ_x	1.0000	γ_r	1.6901	γ_r	1.7262	ρ_r	0.3086
γ_y	2.5100	γ_θ	0.2452	γ_θ	0.3048	ρ_θ	–81.8074
τ	1.1840	τ	–0.9196	τ	0.0000	μ	0.5531
w	0.9960		0.9252		0.9212		1.0903
$g(0)$	0.8463		0.9953		0.9962		0.8950
KL			0.0109		0.0128		0.0379
Model 2		Model 1		Model 3		Model 4	
σ_r	0.7856	σ_x	2.5982	σ_r	0.5722	λ_r	0.8001
σ_θ	1.0811	σ_y	0.1216	σ_θ	0.9661	λ_θ	2.0854
γ_r	1.0000	γ_x	0.0687	γ_r	1.1204	ρ_r	0.0201
γ_θ	1.5360	γ_y	1.8768	γ_θ	1.5574	ρ_θ	0.5905
τ	0.2940	τ	–0.6814	τ	0.7782	μ	0.3954
w	1.0156		1.1459		1.0113		0.9972
$g(0)$	0.7940		0.9956		0.8016		0.7840
KL			0.0305		0.0001		0.0001
Model 3		Model 1		Model 2		Model 4	
σ_r	0.5362	σ_x	0.1549	σ_r	0.7632	λ_r	0.8388
σ_θ	0.9158	σ_y	0.1436	σ_θ	1.0127	λ_θ	2.3433
$_r$	1.1800	γ_x	2.5193	γ_r	1.0428	ρ_r	–0.0756
γ_θ	1.6930	γ_y	1.8313	γ_θ	1.7125	ρ_θ	0.6965
τ	0.6460	τ	1.3817	τ	0.1259	μ	0.5195
w	1.1265		1.2564		1.1291		1.1237
$g(0)$	0.8472		0.7091		0.8322		0.8493
KL			0.0227		0.0002		0.0001
Model 4		Model 1		Model 2		Model 3	
λ_r	5.0000	σ_x	2.2041	σ_r	2.2290	σ_r	1.7967
λ_θ	5.7296	σ_y	1.9247	σ_θ	0.0832	σ_θ	0.0247
ρ_r	0.6923	γ_x	1.8436	γ_r	1.7725	γ_r	2.0347
ρ_θ	1.6183	γ_y	1.9934	γ_θ	-0.0073	γ_θ	6.7873
μ	0.3700	τ	0.5042	τ	0.2769	τ	0.8671
w	0.3151		0.3226		0.3346		0.3183
$g(0)$	0.4519		0.4616		0.4551		0.4672
KL			0.0043		0.0060		0.0025

The numerical minimisation of the KL distance, with respect to the parameters of Q^*, was done in Matlab. All integrals occurring above were evaluated using numerical integration in Matlab (precision 10^{-6}) as well. The integration range in the forward direction (y) was 0–6 km, except for the bottom panel of Table 2, where the range was 0–3 km.

The parameter of main interest for animal abundance estimation was, because the abundance estimate is inversely proportional to the estimate of w. Often, it is the single observer version of w, as opposed to $w_{A\cup B}$, that is being used

Table 1

Different hazard probability functions used in the study: Q_2 and Q_3 are from Okamura and Kitakado (2009) while Q_4 is from Skaug *et al.* (2004). Here, (r, θ) denotes polar coordinates, with $r = \sqrt{x^2 + y^2}$ is radial distance and $\theta \in [0, \pi]$ is the angle relative to the forward direction. Parameter values are given in Table 2.

	Parametric form	Parameter constraints
Model 1	$Q_1(x, y) = \left(1 + \exp(\sigma_x x^{\gamma_x} + \sigma_y y^{\gamma_y} + \tau)\right)^{-1}$	$\sigma_x, \sigma_y, \gamma_x, \gamma_y > 0$
Model 2	$Q_2(r, \theta) = \left(1 + \exp(\sigma_r r^{\gamma_r} + \sigma_\theta \theta^{\gamma_\theta} + \tau)\right)^{-1}$	$\sigma_r, \sigma_\theta, \gamma_r, \gamma_\theta > 0$
Model 3	$Q_3(r, \theta) = \exp(-\sigma_r r^{\gamma_r} - \sigma_\theta \theta^{\gamma_\theta} - \tau)$	$\sigma_r, \sigma_\theta, \gamma_r, \gamma_\theta\, \tau > 0$
Model 4	$Q_4(r, \theta) = \mu \dfrac{l\,[-\lambda_r\,(r - \rho_r)]\; l\,[-\lambda_\theta\,(\theta - \rho_\theta)]}{l\,[\lambda_r \rho_r]\; l\,[\lambda_\theta \rho_\theta]},\ l\,[x] = \dfrac{\exp(x)}{1 + \exp(x)}$	$\lambda_r, \lambda_\theta > 0,\ 0 < \mu \leq 1$

in the abundance calculation (e.g. Skaug *et al.*, 2004). So, although the parameters were estimated from double platform data, we measured the goodness of fit using single-observer versions of w, $g(0)$, and perpendicular distance density $f(x) = \int_0^\infty f(x, y)dy$. As a diagnostic for the fit to the trinomial trials we used, $q(AB \mid x, y)$, i.e. the probability that both observers detect the whale simultaneously.

RESULTS AND DISCUSSION

Table 2 shows parameter estimates, i.e. the values that minimises the KL distance, for all pairwise comparisons of the four hazard probability functions. The corresponding comparisons of the perpendicular distance densities $f(x)$ are given in Fig. 1. This figure also gives the ratios w_T / w_*, which are the key quantity of interest in the present study. When interpreting the density plots it is useful to recall that

$$\frac{w_T}{w_*} = \frac{g_T(0)}{g_*(0)} \cdot \frac{f_*(0)}{f_T(0)}.$$

A misfit in $f(x)$ at $x = 0$ can partly be compensated for by a counteracting misfit in $g(0)$. An example of this is Truth = Q_1 and Approx. = Q_2 for which $g_1(0)/g_2(0) = 0.85$ (Table 2) and $f_2(0)/f_1(0) = 1.27$, yielding $w_1/w_2 = 1.08$. Hence, the perpendicular distance density is not fully diagnostic, and the ratio $f_*(0)/f_T(0)$ does not play the same critical role as it does when $g(0) = 1$ is assumed. Another example of this occurs when Truth = Q_4 and Approx. = Q_1, for which the two density curves are almost identical (Fig. 1; lower left corner). The proportion of ($u = AB$), on the other hand, indicate that there is a misfit (Fig. 2; lower left corner). In a 45 degree sector from the transect line the true model predicts a higher proportion of duplicates than the approximating model (light colored area in the plot), and correspondingly there are too few duplicates in the remaining 45 degree sector.

It is clear from both Figs 1 and 2 that Q_1 differs from Q_1 – Q_4, while Q_2 – Q_4 between themselves yield models with very similar properties. The reason for this is that Q_1 is formulated in Cartesian coordinates (x, y), while Q_2 – Q_4 are formulated in terms of polar coordinates (r, θ). In particular, Q_2 – Q_4 can all be written in the separable form h_1 $\{h_2(r)h_3(\theta)\}$, where h_1 is a decreasing function, and h_2 and h_3 are increasing functions.

Generally speaking Q_1, predicts more observations close to the vessel than do Q_2 and Q_3. This holds both when Q_1 is taken to be the truth (first row of Fig. 1) and when Q_1 is being fitted (first column of Fig. 1). Further, Q_2 and Q_1 behave very similarly in the comparison with Q_1, also when it comes to the ratios w_T/w_* (Fig. 1) and $q_T(AB \mid x, y)/q_*(AB \mid x, y)$ (Fig. 2). The picture is less clear for the comparison of Q_1 versus Q_4. It is worth noting that the effective strip half-width is over estimated, both when Q_1 is taken as the truth (w_1/w_4 = w_T/w_* = 0.91) and when Q_4 is taken as the truth (w_4/w_1 = w_T/w_* = 0.98) .

From a conservation perspective a negative bias in is more

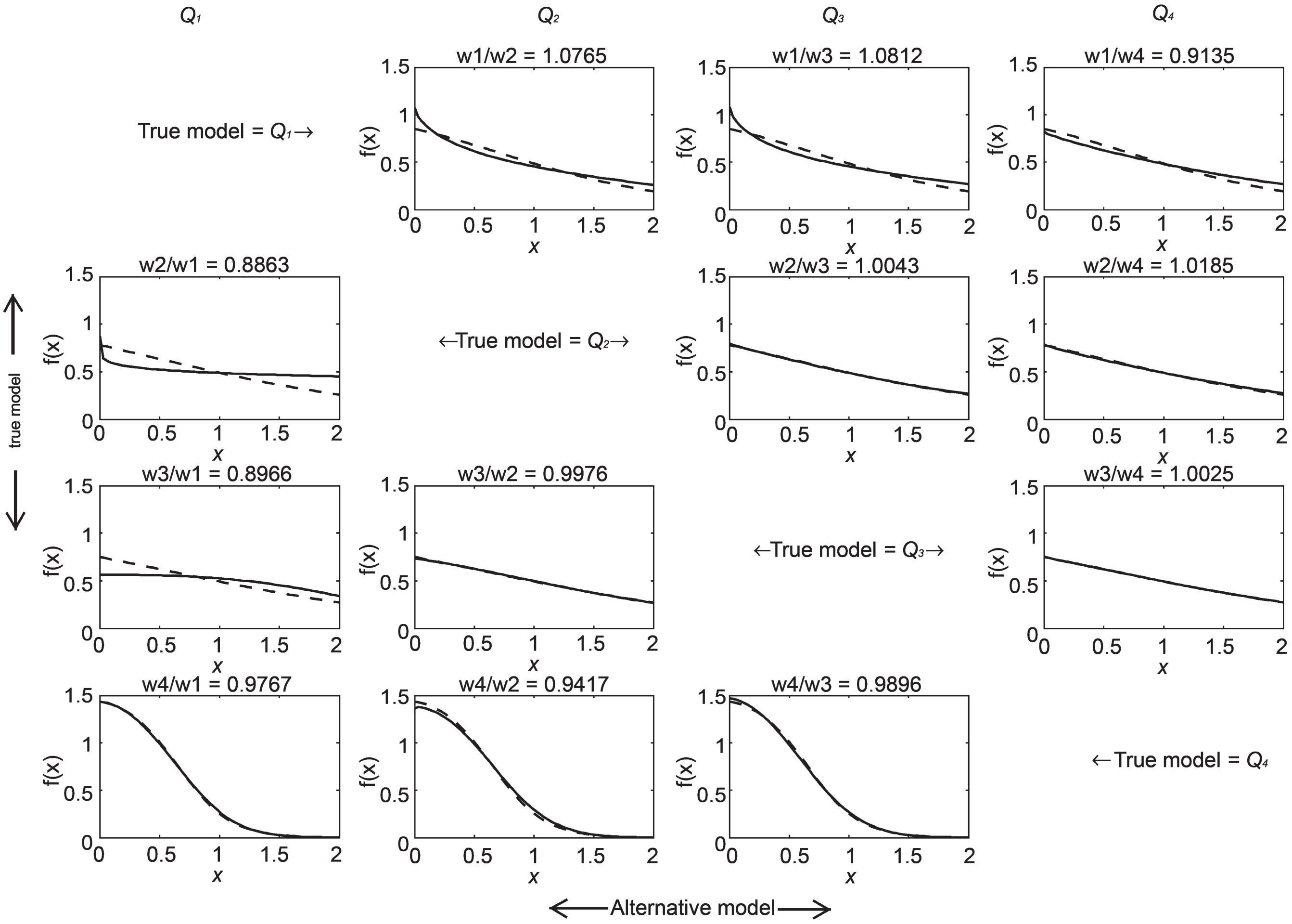

Fig. 1. Comparison of perpendicular distance (km) densities for true (dashed line) and approximating density (solid line), where x is the perpendicular distance. The corresponding ratios of effective strip half widths (w) are also given.

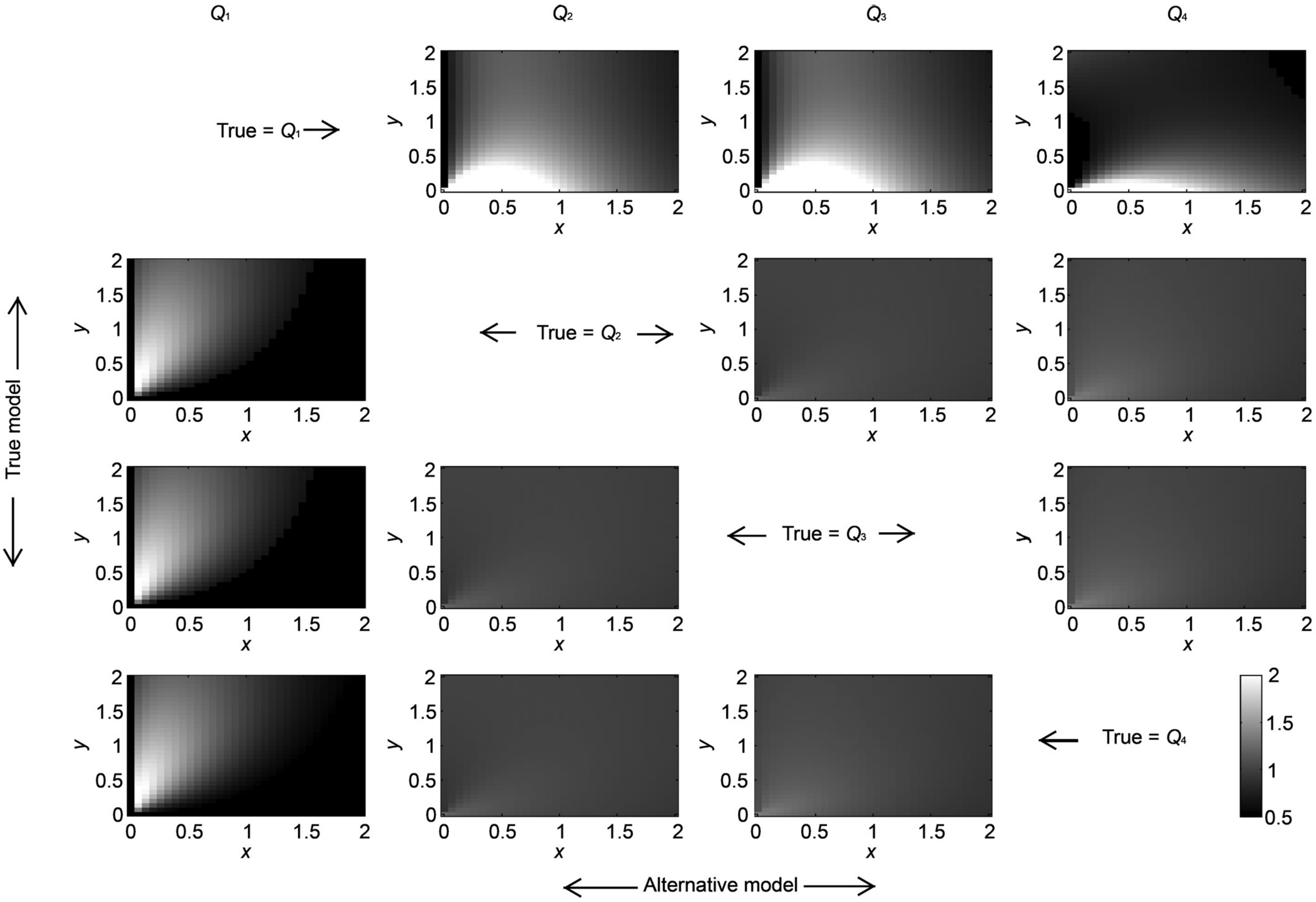

Fig. 2. Ratio $q^{(T)}(AB \mid x, y)/q^{(*)}(AB \mid x, y)$, where $q^{(T)}(AB \mid x, y)$ is the probability (under the true model) that a whale which is initially is observed at location x, y is detected by both A and B (duplicate sighting). Similarly, $q^{(*)}(AB \mid x, y)$ is the probability under the approximating model. The layout of the plot corresponds to that in Fig. 1. The darker the cell, the smaller the ratio.

critical than a positive bias, because the former will lead to a positive bias in the abundance estimate. The most severe underestimation of w found in the present experiment is $w_1/w_3 = w_T/w_* = 1.08$ (Fig. 1; upper row, second column from right). This occurs when Q_1 is the truth and Q_3 is the approximation. For this case Q_3 predicts a too large proportion of duplicate sightings (seen by both A and B) at short radial distance (Fig. 2 upper row, second column from right). The case with the largest over-estimation of w is $w_1/w_4 = 0.91$.

Within each row in Table 2 the smaller the KL distance is, the closer w_T/w_* is to unity. This means that model selection based on a likelihood ratio test, or the AIC criterion, will perform reasonable well for the purpose of picking a model that yields an unbiased estimate of w. The value of the KL statistic does not say anything about the direction of the bias of the estimated w, however.

CONCLUSION

For the purpose of estimating the effective strip half-width (w) the hazard probability model is fairly robust with respect to choice of Q. For all 12 pairwise comparisons considered in this study the fitted w falls within 10% of the true value in all cases. Strictly speaking these conclusions apply only to the version of the hazard probability model used in Skaug *et al.* (2004), and it has not been investigated they hold in the setting of Okamura *et al.* (2003).

We have chosen to use an infinite sample-size setting, which allowed bias arising from mis-specification of Q to be separated from the finite-sample properties of the maximum likelihood estimator. The latter can be studied by simulating data from the hazard probability model, and then applying the estimator on each simulated dataset. This has recently been done for the method of Okamura *et al.* (2003) which did not show any severe biases as a result of finite sample size alone (Okamura and Kitakado, 2009b).

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Comparison of closing and passing mode from a line-transect survey of delphinids in the eastern Tropical Pacific Ocean

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ABSTRACT

Line-transect ship surveys are the primary method used to estimate abundance of pelagic cetaceans. However, survey methods are often modified from traditional methods because observers must approach cetacean groups to identify species and estimate group size. Returning to the trackline after approaching a school dramatically reduces the amount of effective survey time, so ships often resume survey effort at the sighting but parallel to the original trackline (closing mode). Survey effort is no longer independent of group locations, and it is unclear how such methodological modifications affect overall abundance estimates. This research presents the results of a study designed to determine the effects of closing mode methods on abundance estimation for cetacean species in the eastern tropical Pacific. Species identification and group size estimation in closing mode are compared with results using survey techniques where the ship does not approach or slow down to investigate a sighting (passing mode). Both empirical data and simulations were used to compare group encounter rates in the two modes and to better understand the mechanisms that might lead to an encounter rate bias in closing mode. As seen in similar studies, observers are able to identify to the species level less frequently in passing mode (81% vs 57% of sightings), and point estimates of delphinid group size were 58% lower in passing mode than closing mode at distances between 1.0 and 5.5km from the trackline. In addition, uncertainty in group size both within and between observers was higher in passing mode. Closing mode delphinid group encounter rates were generally 20–25% lower than passing mode delphinid group encounter rates. Simulations showed the empirically lower encounter rates in closing mode are due to a loss in detection probability caused by the stop-start nature of the survey method. The closing mode encounter rate bias is greater when groups are in fewer and/or tighter clusters and when overall group density is higher. Methodological adjustments and analytical solutions to improve group size estimation and species identification in passing mode and reduce closing mode encounter rate bias are analytically complex and would also result in the loss of important additional life history data. Nevertheless, such avenues should be explored further.

KEY WORDS: SURVEY-VESSEL; SAMPLING STRATEGY; MODELLING; PACIFIC OCEAN; ABUNDANCE ESTIMATE; *g*(0); SCHOOL SIZE

INTRODUCTION

Line-transect analysis is a well-developed and widely used method of estimating density and abundance (Buckland *et al.*, 2001). Data for a line-transect analysis are collected by moving along a transect line and recording the perpendicular distances at which the objects of interest are detected. Detection probabilities are then estimated as a function of perpendicular distance from the trackline; density estimates are calculated as a function of detection probability, amount of trackline covered and overall cluster size (if objects are clustered).

Line-transect surveys are the primary method used to estimate abundance of pelagic cetaceans (e.g. Barlow, 2006; Branch, 2007; Branch and Butterworth, 2001a; 2001b; Branch *et al.*, 2004; Calambokidis and Barlow, 2004; Dolar *et al.*, 2006; Gerrodette and Forcada, 2005; Goodwin and Speedie, 2008; Hammond *et al.*, 2002; Miyashita, 1993; Mullin and Fulling, 2004). Cetacean line-transect surveys are typically conducted using one of two methods, called passing mode and closing mode. In a passing mode survey, the platform (ship or plane) moves continuously along a transect line. When a group of cetaceans is seen, visual observers determine species composition and estimate group size without leaving the transect line. In a closing mode survey, the platform stops normal searching procedures once a group of cetaceans is sighted (goes ‘off-effort’), leaves the transect line and approaches the sighted group to identify the species and stock composition of the group and to make reliable estimates of group size. Once observers have recorded such information, search effort begins again (going ‘on-effort’). The platform can either return to the trackline and resume effort at the point it was terminated or continue from the point of the sighting and travel along a new trajectory parallel to the original trackline. The latter is often employed during ship-based surveys since the amount of time needed to return to the original trackline would dramatically reduce overall sampling effort. Such a method also ensures the previous sighting remains ‘behind’ the ship and is not double counted.

When the ship does not return to the transect line but continues in a parallel direction from the point of the last sighting, the discontinuous searching effort of closing mode could bias data collection and abundance estimation in several ways. First, on-effort searching is usually carried out in the 180° arc ahead of the ship. When a sighting is made and the observers go off-effort, the area surveyed is calculated as a function of the distance travelled by the vessel from the start of effort to the point at which the observers go off effort, which does not include the area surveyed ahead of the ship (Barlow, 1997). The underestimate in area surveyed would lead to an overestimate of abundance. A second, related issue arises when observers resume scanning effort and there are schools already within visible range. Because these schools are within the search area for a shorter period

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of time, they have a lower detection probability (for the same perpendicular distance) compared to schools further down the trackline. Such an effect would lead to an underestimate of abundance. A third issue arises because the placement of the transect line is no longer random in relation to group distribution after the first sighting. If the ship is travelling through areas of non-uniform group distribution, the ship might get pulled in to high density areas on successive sightings (Haw, 1991). This non-random sampling would lead to an overestimate of abundance. Passing mode has been considered less biased when estimating school density, but closing mode is used because school size estimation and species identification are considered unreliable in passing mode.

By alternating passing and closing days during line-transect surveys, several studies have investigated differences in cetacean abundance estimates using the two survey methods. During the 1984/85 field season, the Antarctic minke whale survey alternated between passing and closing mode on successive days. Passing mode for minke whale surveys is also called independent observer (IO) mode because of the presence of an additional observer (Branch and Butterworth, 2001b). Haw (1991) found minke whale group density estimates to be 20–25% lower in closing mode than in passing mode. In addition, the overall perpendicular distance at which schools were detected appeared shorter in closing mode, leading the author to believe detection probabilities are lower in closing mode. Thus, Branch and Butterworth (2001b) applied a 17% correction factor for negative bias for closing mode surveys for minke whales. Results from the 1984/85 study also led to the incorporation of some passing mode effort during subsequent surveys (Matsuoka *et al.*, 2003). During cetacean line-transect surveys off the west coast of the USA, Barlow (1997) alternated two days in closing mode with one day in passing mode. Results showed species identification was particularly poor in passing mode, and passing mode school size estimates were often lower than closing mode estimates. He found common dolphin group encounter rates were significantly higher in closing mode than in passing mode. However, the preferred habitat for this species may have been undersampled in passing mode. Encounter rates were not significantly different for other species.

This paper reports the results of an experiment designed to compare passing and closing line-transect surveys in the eastern Tropical Pacific Ocean (ETP). Southwest Fisheries Science Center (SWFSC) has used closing mode line-transect surveys in the ETP since the 1970s to estimate abundance of several dolphin species affected by the purse-seine tuna fishing industry (Gerrodette and Forcada, 2005). Over a three month period, passing and closing mode surveys were conducted on alternate days on predetermined tracklines, repeated every 21 days. We compare the data collected in the two modes and use a simulation study to examine reasons for some of the differences.

METHODS

Field methods

To better understand the overall differences between passing and closing mode, we chose a study area thought to be rather homogeneous with respect to environment. The NOAA Ship *MacArthur II* surveyed in a region approximately 200–250 n.miles southwest of Manzanillo, Mexico from late August through late November 2007 (Fig. 1). The survey area is located in the middle of the eastern Pacific warm pool and is also far enough away from the coast to avoid coastal eddies and upwelling that lead to temporal and spatial shifts in the thermocline (Tomczak and Godfrey, 1994). The study area is also a region of relatively high cetacean abundance and is located close enough to land to allow for quick and easy resupply and exchange of personnel. The area was surveyed during four separate legs, each leg lasting three weeks. The duration of each leg was limited by ship supply needs. Timing of the survey coincided with the traditional timing of cetacean surveys conducted in the ETP. During each leg, the objective was to survey along the tracklines of two different diamond patterns twice. Each side of the diamond was to be surveyed on consecutive days, one day in passing mode and the other day in closing mode. The daily trackline distance (90 n.miles) could easily be completed in one day in passing mode but was also short enough that the ship could reach the end point of each line in time for nightly oceanographic stations even when closing mode prevented completion of the trackline. For more information on the methods and ecosystem data from the survey, see Archer *et al.* (2008).

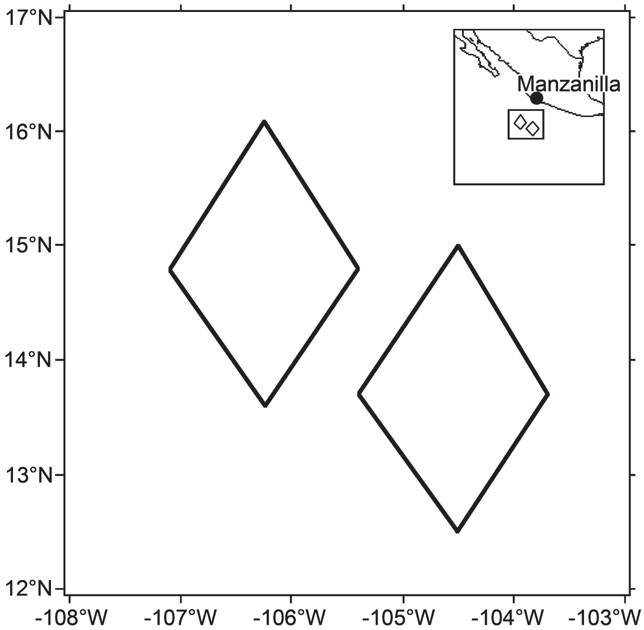

Fig. 1. Study area and tracklines.

Methods for cetacean line-transect surveys by the SWFSC have been consistent since 1986 (Kinzey *et al.*, 2000). Three observers are stationed on the flying bridge of the ship which is travelling at a constant speed of 18.5km/hr (10 knots). Two observers scan for schools using 25 × 150 binoculars mounted on each side of the ship. Each observer sweeps the area from the trackline (0º) to 90º to the left or right of the trackline. In this manner, the area 180º in front of the ship is scanned with high-powered binoculars. Prior to 2003, observers scanned an additional 10º over the trackline, so the 20º area directly in front of the ship was scanned by both observers. Such scanning overlap was not conducted in this study. A third observer in the middle, also acting as data recorder, scans for schools near the ship with the naked eye and hand-held 7× binoculars, covering the entire 180º area ahead of the ship. The searching method used in this study is exactly the same as methods used in previous studies. The difference between passing and closing mode occurs when a cetacean group is detected.

Passing mode is a form of line-transect sampling used in most non-cetacean surveys. Within the ETP, passing mode consists of the following steps. The ship travels at a constant speed (18.52km/h) along the predetermined trackline. When a cetacean school is sighted, only the observer who detected the school attempts to identify the school to species and estimate school size. Angle and binocular reticle readings are also taken for each group which are later converted to perpendicular and radial distances (Kinzey and Gerrodette, 2003). Observers do not terminate effort to investigate the school or to give an observer more time for species identification or school size estimation. The ship does not deviate from the trackline or slow down when a group is detected. Scanning effort is not continuous, since the observer must briefly stop scanning to determine school size and species. The methods are somewhat different from passing (IO) mode methods during Antarctic minke whale surveys where one group of observers continuously scans while another group is responsible for species ID and school size estimation (Haw, 1991).

Cetacean species identification and school size estimation can be difficult from a distance, so closing mode is often used in cetacean line-transect surveys. During closing mode, as in passing mode in the ETP, the ship begins on a predetermined trackline with a given bearing and constant speed. When a cetacean school is detected, the observers terminate search effort, and the ship manoeuvres to approach the school. All three observers work together to identify species within the school, and each observer makes an independent estimate of school size, recording best, high and low estimates. Additional data may be collected, such as photographs and biopsies (Jackson *et al.*, 2008). Depending on the school's distance from the ship, size and species composition, observers may spend anywhere from five minutes to several hours collecting data on a single school. When observers are finished investigating a school, the ship returns to its original bearing and speed, and scanning effort resumes. In almost all cases, the ship does not return to the original trackline before resuming effort. However, to prevent surveying too far from the original trackline, the ship bearing is altered to return to the line at a 20º angle if the ship has moved more than 18.5km (10 n.miles) from the line.

Analysis

Similar to other analyses of ETP data, we limited the data within certain sighting conditions (visibility > 5.5km and Beaufort sea state ≤ 5) and only included sightings detected within 5.5km perpendicular distance from the trackline.

Species identification

Species identification was based on a tiered system and the level of identification depends on the expert judgment of the observers. The least descriptive identification is 'Unidentified Cetacean.' Observers are often able to identify schools to a species or even sub-species level, and schools may have more than one species present. To compare the ability of observers to identify schools to the species or subspecies level, schools were tallied based on level of species identification (identified to species or subspecies level vs not identified to species or subspecies level). Using a binomial likelihood with a uniform conjugate prior distribution of $\beta eta(1,1)$, the posterior distribution of the probability a school is identified to the species or subspecies level becomes $\beta eta(n+1, N-n+1)$, where n is the number of schools identified to the species or subspecies level, and N is the total number of detected schools (Gelman *et al.*, 1995). The number of schools with more than one species identified were also tallied for both passing and closing mode and the same Bayesian binomial method was used to determine the probability schools were composed of more than one species (n = number of mixed-species schools, N = total number of detected schools). Comparison of passing and closing modes was done via comparison of 100,000 random samples from the respective beta distributions.

Identification during passing mode was poor, so comparison of school sizes and encounter rates in passing and closing modes on a species-by-species basis was not feasible (see Results). To create a reasonable sample size for further analyses of school size and encounter rate, several identification categories were combined in to a 'delphinid' category: unidentified dolphin, unidentified medium delphinid, unidentified small delphinid, spinner dolphin (*Stenella longirostris*; unidentified subspecies), pygmy killer whale (*Feresa attenuata*), Risso's dolphin (*Grampus griseus*), rough-toothed dolphin (*Steno bredanensis*), common bottlenose dolphin (*Tursiops truncatus*), pantropical spotted dolphin (*Stenella attenuata*; offshore), eastern spinner dolphin (*Stenella longirostris orientalis*), striped dolphin (*Stenella coeruleoalba*) and common dolphin (*Delphinus delphis*).

Delphinid school size

School size estimates were calculated using the methods described in Gerrodette and Forcada (2005). Prior to this study, each observer's school size estimates were 'calibrated' by comparing their estimates to school size counts from aerial photographs of the same school (Gilpatrick, 1993). The observers' school size raw estimates are adjusted by observer-specific regressions which include effects of year and school size (Barlow *et al.*, 1998; Gerrodette *et al.*, 2002). On occasion, observers will only record a low count if they feel they did not get a thorough look at the school. In such cases, the low count is used as an observer's best estimate. The logarithm of the point estimate on school size was:

$$\ell n\, \hat{s} = \sum_{i=1}^{n} w_i\, \ell n\, C_i \tag{1}$$

with variance

$$\mathrm{var}\,(\ell n\, \hat{s}) = \sum_{i=1}^{n} w_i^2\, \mathrm{var}\,(\ell n\, C_i) \tag{2}$$

where

n = number of calibrated estimates (C) for the school

$w_i = v_i^{-1} \big/ \sum v_i^{-1}$

v_i = var ($\ell n\ C_i$), the residual variance of the log-log regression of school size estimates vs photo counts for the observer

The above calculations and calibrations are based on school size estimates in closing mode only, when each observer has a prolonged opportunity to estimate school size at a close distance. School size calibration has never been done for passing mode estimates; so for this analysis, the same calibration coefficients and calculation methods were used for both passing and closing modes. Variance estimates account for differences among observers and uncertainty in the calibration relationship.

To analytically compare school size estimates for the two survey methods, a Bayesian analysis was conducted using a lognormal likelihood with unknown mean and variance. Data were school size point estimates (Equation 1), grouped by survey method and perpendicular distance from the trackline (0 to 1km vs 1 to 5.5km). Comparison of passing and closing modes was done via comparison of 100,000 random samples from the respective posterior lognormal distributions. When only one observer reports a low estimate for school size, variance calculations are not possible, so uncertainty in school size point estimates was not incorporated into the Bayesian model. However, to compare additional measurement uncertainty for each school and observer, the range of high to low estimates standardised by the best estimates was calculated. In addition, standard deviations of ln $\hat{s}$ as a function of distance from the trackline and survey method are reported when school sizes were based on more than one low estimate.

Delphinid school encounter rate

Since previous studies have shown that species identification and school size estimation were compromised in passing mode, our analytical methods focused on comparing school encounter rates (number of detected schools/km effort) between passing and closing mode. We first develop a model that assumes school placement is temporally and spatially random. However, daily encounter rates are probably not temporally random due to potential large scale movements of groups in to and out of the area, most likely due to environmental changes. Therefore, we also developed a hierarchical Bayesian model, treating each day as an independent estimate of encounter rate. Since encounter rates decline with sea state and our estimates did not adjust for such changes, data were partitioned and analysed separately by sea state.

To estimate school encounter rate, we started by assuming school location was random in relation to the trackline and each other. In that case, the number of detected schools follows a Poisson process:

$$p\ (x \mid \lambda t) = \frac{(\lambda t)^x}{x!}\ e^{-\lambda t} \tag{3}$$

where

x = count of detected schools

t = km of effort (known and constant)

λ = encounter rate

The above equation becomes the likelihood when estimating daily encounter rates. The prior on λt can be expressed differently and the data partitioned in various ways depending on how confident we are in the assumption of temporally and/or spatially random placement of schools. If we assume school placement is completely random, all survey data can be pooled and a conjugate gamma prior for λt can be used (Robert, 2001):

$$\pi\ (\lambda t) = \frac{\beta^{\alpha}}{\Gamma(\alpha)}\ (\lambda t)^{\alpha-1}\ e^{-\beta\lambda t} \tag{4}$$

where

α = gamma distribution shape parameter

β = gamma distribution inverse scale parameter.

With a conjugate prior, the posterior on λt is another gamma distribution (Robert, 2001):

$$\lambda t \mid x \sim \text{Gamma}(\alpha + x, \beta + 1). \tag{5}$$

By transformation, the posterior on encounter rate is also a gamma distribution:

$$\lambda \mid x,t \sim \text{Gamma}(\alpha + x, (\beta + 1)t). \tag{6}$$

The prior distribution becomes vague or non-informative as α and β approach zero (Robert, 2001). In this case, both parameters were set constant at 10^{-6}. Setting the constants smaller or larger by a magnitude of three did not affect the outcome of the analysis, indicating 10^{-6} was essentially zero.

A second estimate of school encounter rate relaxed the assumption that school density was the same for all days. In this region, school density may potentially change on a daily scale due to various ecosystem dynamics. So, another form of analysis used each day as an exchangeable and independent sample of encounter rate in a hierarchical Bayesian model. Daily encounter rates may not be independent because of changes in temporally correlated environmental variables. However, the experimental design randomised the environmental variables since passing and closing methods were performed in pairs of days.

We still assumed school distribution was random within any given day, so the likelihood was a product of Poisson distributions on a daily scale.

$$p(\underline{x} \mid \underline{\lambda t}) = \prod_{i=1}^{m} \frac{(\lambda_i t_i)^{x_i}}{x_i!}\ e^{-\lambda_i t_i} \tag{7}$$

where m is the total number of days of sampling. However, now we assume that each daily encounter rate is drawn from a gamma-distributed overall encounter rate, so the prior distribution for all daily encounter rates (λ) becomes a multiple of a gamma distribution.

$$p(\underline{\lambda} \mid \gamma,\phi) = \prod_{i=1}^{m} \frac{\phi^{\gamma}}{\Gamma(\gamma)}\ (\lambda_i)^{\gamma-1}\ e^{-\phi\lambda_i} \tag{8}$$

where

γ = gamma distribution shape parameter for overall encounter rate

ϕ = gamma distribution inverse scale parameter for overall encounter rate

Note the gamma prior distribution in the hierarchical analysis describes the distribution of overall encounter rates and is not the same as the conjugate gamma prior described for λt when pooling the data. In the hierarchical model, the gamma distribution was chosen for its flexibility and because values are limited above zero. We used Jeffery's hyperpriors on γ and ϕ: independent uniforms on the log of γ and ϕ (Miller, 1980).

$$\pi(\gamma,\phi) \propto \frac{1}{\gamma\phi} \tag{9}$$

Linking all probabilities together gives us the final joint posterior distribution of γ and . We also have posterior distributions for λ, but they are not relevant here since we are interested in the derived posterior distribution of overall encounter rate based on posterior distributions of γ and .

$$p(\underline{\lambda}, \gamma, \phi \mid \underline{x}, t) \propto p(\underline{\lambda} \mid \gamma, \phi) \cdot p(\underline{x} \mid \underline{\lambda t}) \cdot \pi(\gamma, \phi) \tag{10}$$

The hierarchical Bayesian analysis was done using program MTG (Metropolis within Gibbs) developed by Daniel Goodman of Montana State University (Schwarz, 2008). To maximise computer efficiency, simulations were performed on orthogonally transformed parameters when correlation between parameters was high (γ and ϕ). Simulations were set for a rejection rate near 0.7, a sub sampling (thinning) of 1 in 150 and a burn-in period of 150, continuing for a subsample size of 10,000 for each inference (see Cowles and Carlin, 1995). The resulting lag-1 autocorrelations were <0.1, and independent chains with different parameter starting values gave indistinguishable results. To verify convergence and stationarity within the final chains, we used the Heidelberger and Welch convergence diagnostic available from the CODA package in *R* using standard 10% increments and $p \leq 0.05$ (Heidelberger and Welch, 1983; Plummer *et al.*, 2006). Results are reported as the ratio of the derived posterior distributions of closing mode encounter rate over passing mode encounter rate.

In general, school detection probabilities decline with higher Beaufort sea state level (Barlow *et al.*, 2001; Gerrodette and Forcada, 2005; Teilmann, 2003), and the proportion of time spent surveying in each sea state was not equal for passing and closing modes simply due to random factors. Since encounter rates are not adjusted for changes in detection probability by sea state, data (observations and km of effort) were partitioned and analysed separately by sea state. However, Beaufort sea states 0 and 1 were combined since sample size was small for these two categories. Sighting conditions are nearly identical in such excellent weather conditions. Although data collection procedures were designed to produce pairs of days in passing and closing modes along the same trackline, data were not analysed in a pair-wise fashion for two reasons. First, we found fluctuations in encounter rates between paired days to be just as high as fluctuations between all days. Second, fluctuations in sea state were highly variable on a daily scale, making paired day comparisons inappropriate. Final 'delphinid' abundance estimates were not calculated because simulation results (see below) led us to believe we need to re-measure several variables and rethink some assumptions that go in to such a calculation.

Simulation

To understand the mechanisms that lead to differences in passing and closing mode encounter rates, a line-transect simulation was developed. The simulation exercise accounts for all known potential sources of bias in closing mode. All schools within detection range along the trackline have the potential to be detected based on the schools' radial distance from the ship, which accounts for scanning in front of the ship. With spatially clustered schools, ships in closing mode have the potential to spend more survey effort in high density areas. Lastly, the detection probability equations account for lower detection probabilities for schools closer to the ship when survey effort resumes. The simulation does not account for passing mode biases, namely school size estimates and species identification.

The simulation is based on the instantaneous probability of detecting a school as a function of radial distance from the ship (D_x).

$$p(Detected \mid D_x) = -\frac{1}{72} D_x + \frac{1}{6} \tag{11}$$

Distances are assumed to represent the distance from the bow of the ship to the centre of the schools. The exact shape of the above function for ETP dolphin schools is not known, although it most definitely varies by sea state and school size. For ease in transformation and integration, we chose a linear function. However, Equation 11 is based on some characteristics we would expect to find in the true function. The maximum possible detection distance is 12.0km, and the function assumes objects closer to the ship are easier to detect than objects further away. The overall resulting patterns in relation to school density and school clustering between the two modes hold if the true function exhibits the same characteristics but has a different shape. The simulation assumes all schools have the same number of individuals and does not allow for different Beaufort sea states. Such differences merely change the shape of the detection function.

Integrating over all possible radial distances, the overall probability of detecting the school becomes

$$p(Detected) = -\frac{1}{144} (D_{rs}^2 - D_p^2) + \frac{1}{6} (D_{rs} - D_p) \tag{12}$$

D_{rs} is the maximum possible radial distance between the ship and school at which the school can be detected. If the school is 12km or more away from the ship at the beginning of scanning effort, D_{rs} is 12.0km (distance to the horizon). Otherwise, D_{rs} is simply the distance between the ship and school at the start (or usually resumption in closing mode) of searching effort. The minimum distance at which a school can be detected is the perpendicular distance from the school to the trackline (D_p). Equation 12 would represent the 'detection function' in a standard line-transect analysis, assuming all schools are at least 12km away at the start of survey effort, and the detection probability for a school 12 or more kilometres away on the trackline ($D_{rs} = 12.0$, $D_p = 0$) is 1.0.

Because the ship travels at a constant speed, the probability of any specific distance from the ship to the perpendicular intercept is a uniform distribution from 0 to $\sqrt{D_{rs}^2 - D_p^2}$, assuming our simulated world is flat. By transformation we get:

$$p(D_x) \propto \frac{D_x}{\sqrt{D_x^2 - D_p^2}} \tag{13}$$

Combining Equations 11 and 13, Bayes' formula allows us to calculate the probability of D_x given a school was detected:

$$p(D_x \mid Detected) \propto \frac{-\frac{1}{72}D_x^2 + \frac{1}{6}D_x}{\sqrt{D_x^2 - D_p^2}} \tag{14}$$

The simulation first placed a predetermined number of schools in a survey region. Coordinates x and y were assigned to each school where $0 \leq x \leq 222$km and $-24.52 \leq y \leq 24.52$km. This region represented the area surveyed in one day (12h of effort at 18.5km/hr) in passing mode with the ship travelling through the area starting at the origin (0,0) and ending at coordinates (222,0). In closing mode, the ship starts at the origin (0,0) and can veer away from the trackline up to 10 n.miles (18.52km) in either direction. Similar to closing mode in the field, the ship did not stop to investigate schools detected further than $D_p = 6.0$ from the trackline. Nor were such schools used in encounter rate estimates.

School coordinates were generated to allow a controlled degree of spatial clustering. Since schools are the object of detection, the spatial Poisson process referred to clusters of schools, not clusters of individuals as traditionally defined in line-transect analysis. First, the coordinates for the centres of a predetermined number of clusters (1, 10 or 20) were generated randomly in the rectangle. Then each school, from a predetermined number of schools (50, 150 or 150), was randomly assigned to a cluster. The school's coordinates were generated based on a bivariate normal circular placement around the cluster centre with a known standard deviation (10, 20 or 30km). If the coordinates of the school were outside the survey region, were within 0.5km of a previous school, or were more than two standard deviations away from the cluster centre, the school coordinates were re-sampled. The process was repeated until a predetermined school density for the survey region (50, 150 or 250) was reached. Assuming school density, cluster number and cluster size are independent, there were a total of 27 different simulations. Although school density and cluster characteristics are highly variable within the ETP, the suite of simulations represents probable scenarios one would encounter in this region.

The overall spatial distribution of the schools is relatively simplistic compared to potential real spatial processes. The simulation produces circular school clusters; each cluster has the same radius, and densities of schools are highest in cluster centres. However, the spatial point process is able to illustrate the overall differences in passing and closing mode encounter rates when schools are clustered. Total number of schools in the area controls overall density, and cluster standard deviation controls how concentrated the schools are within the clusters.

In passing mode, the trackline was defined as the x-axis, and we assumed the ship travelled at a constant speed (18.52km/hr). The probability of detection was calculated for each school in relation to its perpendicular distance from the trackline and its distance from the ship when surveying began (Equation 12). Schools more than 12km from the ship at the beginning of surveying had the highest detection probability possible given their perpendicular distance from the trackline. Each school was randomly detected or not detected based on that probability. The simulation tallied the number of detected schools and calculated the encounter rate (number of detected schools/222.0km).

The encounter rate was then reassessed for the same spatial distribution of schools while the ship moved through the area in closing mode. In closing mode, the ship started at the origin (0,0) but then moved off the original trackline toward the first detected school (i) with $D_{pi} \leq 6$km. To determine which school was detected first, the radial distance at which each school was detected (D_{ri}) was randomly drawn from its probability distribution (Equation 14), limiting values between D_{pi} and D_{rsi}. Then the distance along the trackline (D_{ti}) at which the school was detected became

$$D_{ti} = D_{start} \sqrt{D_{ti}^2 - D_{pi}^2} \tag{15}$$

where D_{start} is the total distance between the starting point and the school's perpendicular intercept. The school with the shortest D_t was the first school detected (school A). All other schools were put back in the pool to potentially be detected later.

The total amount of trackline surveyed was D_{tA}, and the amount of survey time was calculated as the sum of the time on the trackline, the time it took to approach the school and the time it took to investigate the school.

$$Time = \frac{D_{tA} + D_{rA}}{18.52} + \frac{1}{6} \tag{16}$$

We assumed the observers terminate scanning effort as soon as a school is detected. The ship then travels directly to the school, covering the radial distance between the ship and school A (D_{tA}) at the same speed as searching speed (18.52km/h). We assume the school does not move in response to the ship. Once the ship arrives at the school, the observers spend 10min determining species composition and school size. Only the first assumption affects comparisons between passing and closing modes. If observers continued to search after finding a school in closing mode, the actual amount of survey effort would be underestimated, leading to an overestimate of encounter rate. Changing the other assumptions only decreases sample effort in closing mode.

When scanning effort resumes, the ship begins at the coordinates of the last detected school (x_A, y_A) continuing along a trackline parallel to the original trackline. The process of determining the position of the next detected school and calculating the amount of accrued survey time is repeated, adjusting for the new starting point of the ship. All schools 'behind' the ship ($x_i < x_A$) are no longer available for detection, and any schools within 12km of the ship ($D_r < 12.0$) have a lower detection probability than schools further away. The tally of number of schools detected (n_c) and sum of total surveyed trackline (D_c) continues until the ship reaches the edge of the survey area ($x = 222.0$), or the accrued survey time is 12.0h. If the ship moves more than 18.52km from the original trackline, the scanning trackline is angled at 20° toward the original trackline instead of running parallel to it. All distances and the definition of 'behind' the ship are adjusted for the angle change. Encounter rate for the survey day in closing mode is then calculated as n_c/D_c. Sample size was 1,000 sample 'days' for every combination of number of schools, number of clusters and cluster size.

Results are shown as the ratio of raw closing mode encounter rate over raw passing mode encounter rate. To understand potential biases in estimation of the detection function in closing mode, we also present the distribution of perpendicular distances of detected schools in closing mode for nine of the 27 simulations. They are graphically compared with the perpendicular distance distribution one would expect in passing mode given the same amount of covered trackline. Such comparisons indicate differences in detection probabilities one would estimate in the two modes if one was estimating abundance (Buckland *et al.*, 2001). A general description of perpendicular distances of sightings from the field data are then given by Beaufort sea state and survey method to compare with simulated results.

RESULTS

Survey effort

Overall, the ship spent 67 days surveying on the trackline, with a total of 9550km of effort and 765 cetacean sightings. Weather, including a hurricane, prevented completion of both diamonds on Legs 1 and 3. After limiting effort by sighting conditions (Beaufort sea state ≤5, visibility >5.5km), more days were spent in closing mode than passing mode, but distance on effort was longer in passing mode when combining days together (Table 1). Average daily distance on effort was 26% lower in closing than passing mode (113 vs 153km/day). At greater than 100km/d of effort in closing mode, the total number of daily sightings is negatively correlated with distance on effort, indicating longer distances on effort in closing mode are only achieved on days with fewer sightings (Fig. 2).

Table 1

	Survey method	
Resulting effort	Closing	Passing
Days	34	31
Km effort	3,832.5	4,741.8
Delphinid sightings	186	288

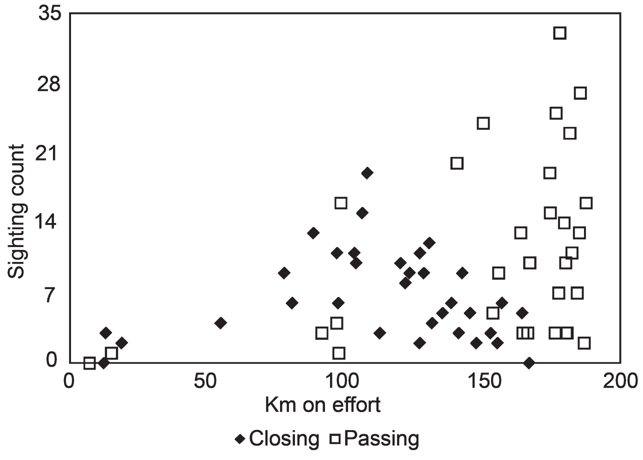

Fig. 2. Daily sighting counts and kilometers of effort for closing and passing modes (all cetacean species). Sighting conditions limited to Beaufort ≤5 and visibility >5.5km. Perpendicular distances of all sightings are limited to ≤5.5km.

Species identification

After filtering the data for visibility, Beaufort and distance, results indicated observers were able to identify cetaceans to the species or subspecies level 81% (±2% SD) of the time in closing mode and 57% (±3% SD) of the time in passing mode. The probability that the proportion of groups identified to the species or subspecies level in passing mode is equal to or greater than the proportion of groups identified to the species or subspecies level in closing mode is zero. In addition, the observers' abilities to identify to the species or subspecies level declined with distance in passing mode (Fig. 3). Mixed-species schools were identified 22% (±3% SD) of the time in closing mode and 7% (±1% SD) of the time in passing mode. The probability that the proportion of mixed species schools identified in passing mode is equal to or greater than the proportion of mixed-species schools identified in closing mode is zero. In closing mode, observers determined four schools that consisted of three different species. No schools with three or more species were identified in passing mode. The observers' ability to identify schools with more than one species was generally poor in passing mode regardless of distance (Fig. 4). The combined 'delphinid' category made up 79% of all sightings in closing mode and 84% of all sightings in passing mode. Only one mixed-species school contained both delphinids and non-delphinids. In general, species identification was poor in passing mode compared to closing mode. Only schools identified as delphinids were used in further analyses.

Delphinid school size

When using closing-mode calibration techniques for both passing and closing survey methods, passing mode delphinid school size estimates tended to be lower for schools further than 1.0km from the trackline (Fig. 5). Estimates of delphinid school size were 58% lower in passing mode than closing mode at distances between 1.0 and 5.5km from the trackline. The probability that passing mode school sizes were less than

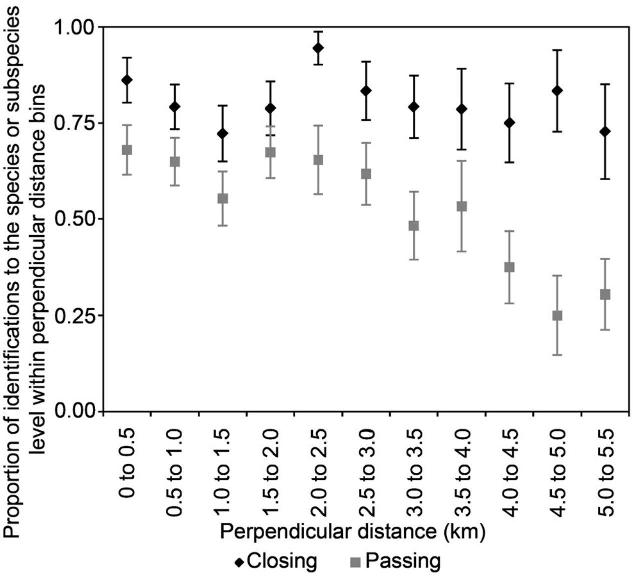

Fig. 3. The proportion of identifications to the species or subspecies level for all sightings as a function of perpendicular distance from the trackline and survey method. Bars are ± one standard deviation. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km.

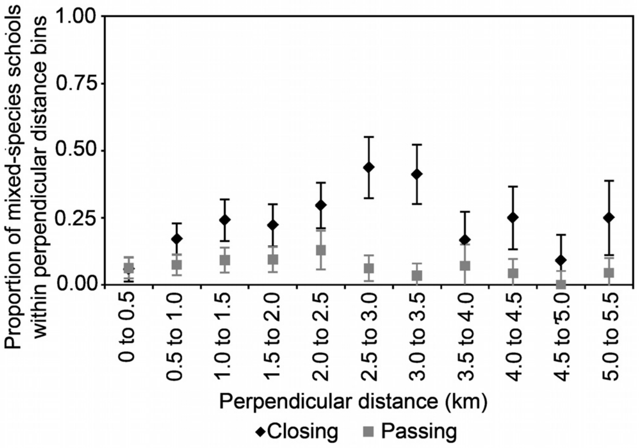

Fig. 4. The proportion of schools with two or more species for all sightings as a function of 0.5km perpendicular distance bins from the trackline and survey method. Bars are ± one standard deviation. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km.

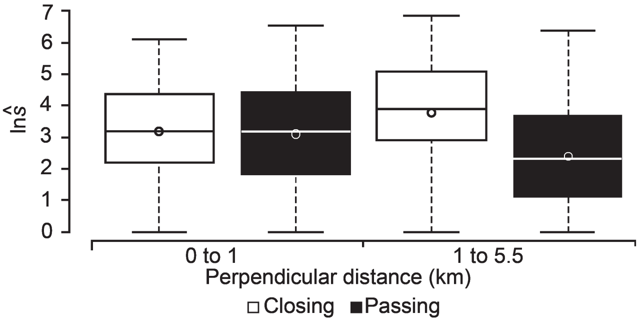

Fig. 5. Distributions of log of school size estimates in relation to perpendicular distance from the trackline and survey mode for delphinid sightings. Open circles are means. Boxes are 1st and 3rd quartiles, and lines within the boxes are medians. Whiskers are minimum and maximum values. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km.

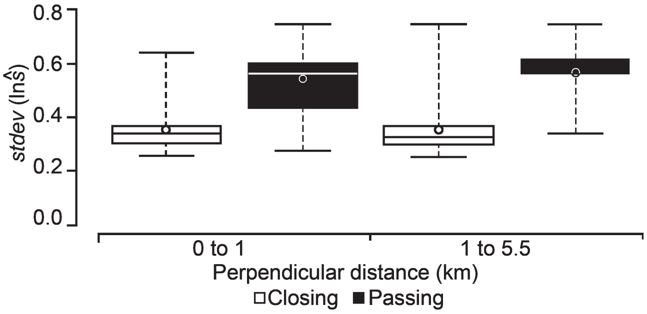

Fig. 6. Distributions of standard deviation of the log of school size given perpendicular distance from the trackline and survey mode for delphinid sightings. Each data point represents multiple observer estimates of the same group (Equation 2). Open circles are means. Boxes are 1st and 3rd quartiles, and lines within the boxes are medians. Whiskers are minimum and maximum values. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km.

closing mode school sizes was 0.51 between 0 and 1.0km from the trackline and 0.73 between 1.0 and 5.5km from the trackline. The standard deviation of school size estimates was higher in passing mode than in closing mode (Fig. 6). The better precision in closing mode school size estimates is due to lower within-observer variability as well as a larger sample size since more than one observer estimates school size in closing mode. Delphinid school size estimates were based on one observer's low estimate 6.5% of the time in closing mode and 36.1% of the time in passing mode. Differences between school size estimates in passing vs closing mode at greater distances are not as pronounced when estimates based on one low count are removed. Without low-count school size estimates, the probability that passing mode school sizes were less than closing mode school sizes was 0.47 between 0 and 1.0km from the trackline and 0.61 between 1.0 and 5.5km from the trackline. The standardised range between low and high school size estimates implies that within-observer uncertainty was generally higher in passing mode than closing mode, particularly with delphinid schools further from the trackline (Fig. 7). No changes in the difference between passing and

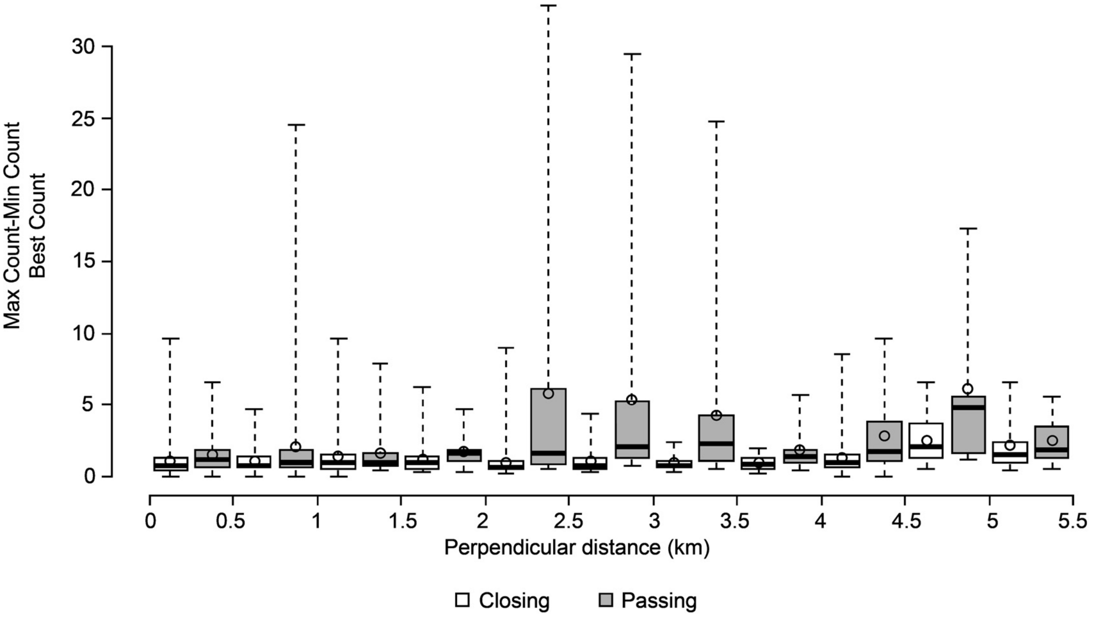

Fig. 7. Range of observer delphinid counts (maximum count-minimum count) standardised by the best count in relation to survey method and distance from the trackline (0.5km bins). Open circles are means. Boxes are 1st and 3rd quartiles, and lines within the boxes are medians. Whiskers are minimum and maximum values. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km.

closing mode school size estimates were seen in relation to sea state (not shown). Overall, delphinid school size estimates were lower in passing mode with a higher level of uncertainty compared to closing mode. However, if estimates are biased low, they could be recalibrated using new aerial survey data.

Delphinid encounter rates

Without adjusting for different levels of effort in different Beaufort states, the simple aggregate delphinid encounter rate was 20% lower in closing than passing mode (0.049 vs 0.061schools/km). When pooling data within each Beaufort state, encounter rate in closing mode is more likely to be lower than encounter rate in passing mode, except in Beaufort 5 conditions (Fig. 8). However, the posterior distribution of the ratio of closing mode encounter rate over passing mode encounter rate is much more variable in Beaufort 5 conditions due to small sample size. With the assumption that schools are randomly located in our survey area, the results indicate a 95% probability that the encounter rate bias is roughly between 40% and 120% of passing mode encounter rate, which would result in an underestimate of abundance in closing mode. Variability in the posterior of the encounter rate ratio is higher when treating days as exchangeable samples of an overall encounter rate (Fig. 9). When we incorporate temporal variability in to our bias estimates, there is a 95% probability the bias is around 25% to 400% of passing mode encounter rates.

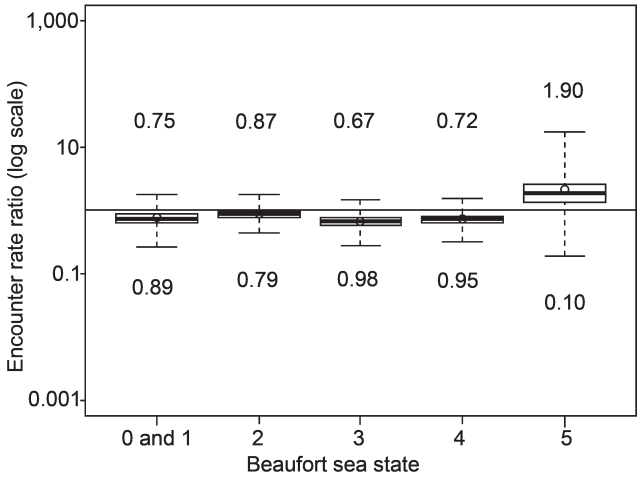

Fig. 8. Derived posterior distribution of the ratio of closing mode encounter rate over passing mode encounter rate by Beaufort sea state when days are pooled. Boxes are 1st and 3rd quartiles. Bars within the boxes are medians, and points are means. Whiskers are minimum and maximum values. Top numbers are medians. Bottom numbers are proportions of distribution <1.0. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km. Perpendicular distances of sightings are limited to ≤5.5km.

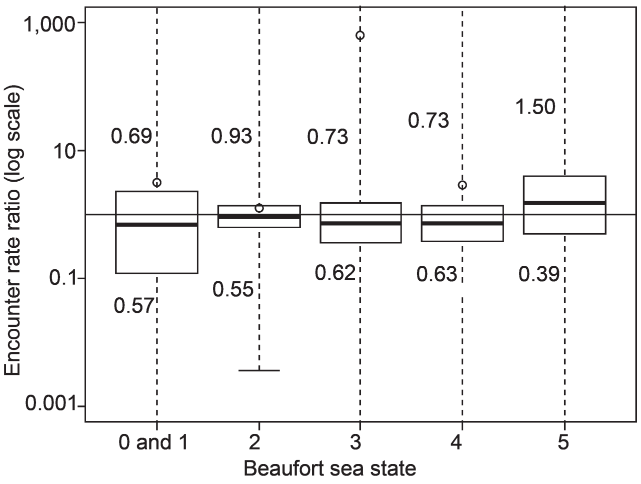

Fig. 9. Derived posterior distribution of the ratio of closing mode encounter rate over passing mode encounter rate by Beaufort sea state using a hierarchical model with days as exchangeable estimates of encounter rate. Boxes are 1st and 3rd quartiles. Bars within the boxes are medians, and points are means. Whiskers are minimum and maximum values (mostly off scale). Top numbers are medians. Bottom numbers are proportions of distribution <1.0. Sighting conditions limited to Beaufort ≤5 and visibility >5.5km. Perpendicular distances of sightings are limited to ≤5.5km.

Simulation

Overall, daily closing mode school encounter rate is lower than passing mode encounter rate when schools are in fewer clusters and when clusters are more concentrated (Fig. 10). The difference is more pronounced when there are more schools in the area, i.e. overall school density is larger (Fig. 10). There is a threshold at which school clustering becomes random, and, on average, the closing mode encounter rate is no longer biassed. Again, the threshold is dependent on the number and size of clusters as well as the true school density in the area.

Even when the overall difference between passing and closing mode is minimal, variability in the encounter rate ratio can be quite high simply due to random movement patterns of the ship in closing mode as well as random detection of each school. Variability is highest when schools

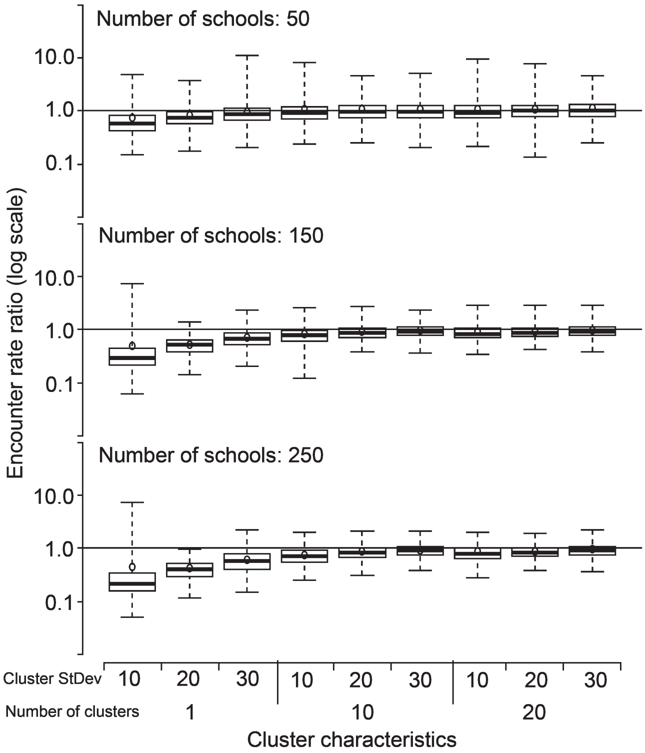

Fig. 10. Distribution of the ratio of closing mode encounter rate over passing mode encounter rate from simulations with differing overall school density and different clustering characteristics (most patchy to least patchy along the x-axis). Boxes are 1st and 3rd quartiles. Bars within the boxes are medians, and points are means. Whiskers are minimum and maximum values.

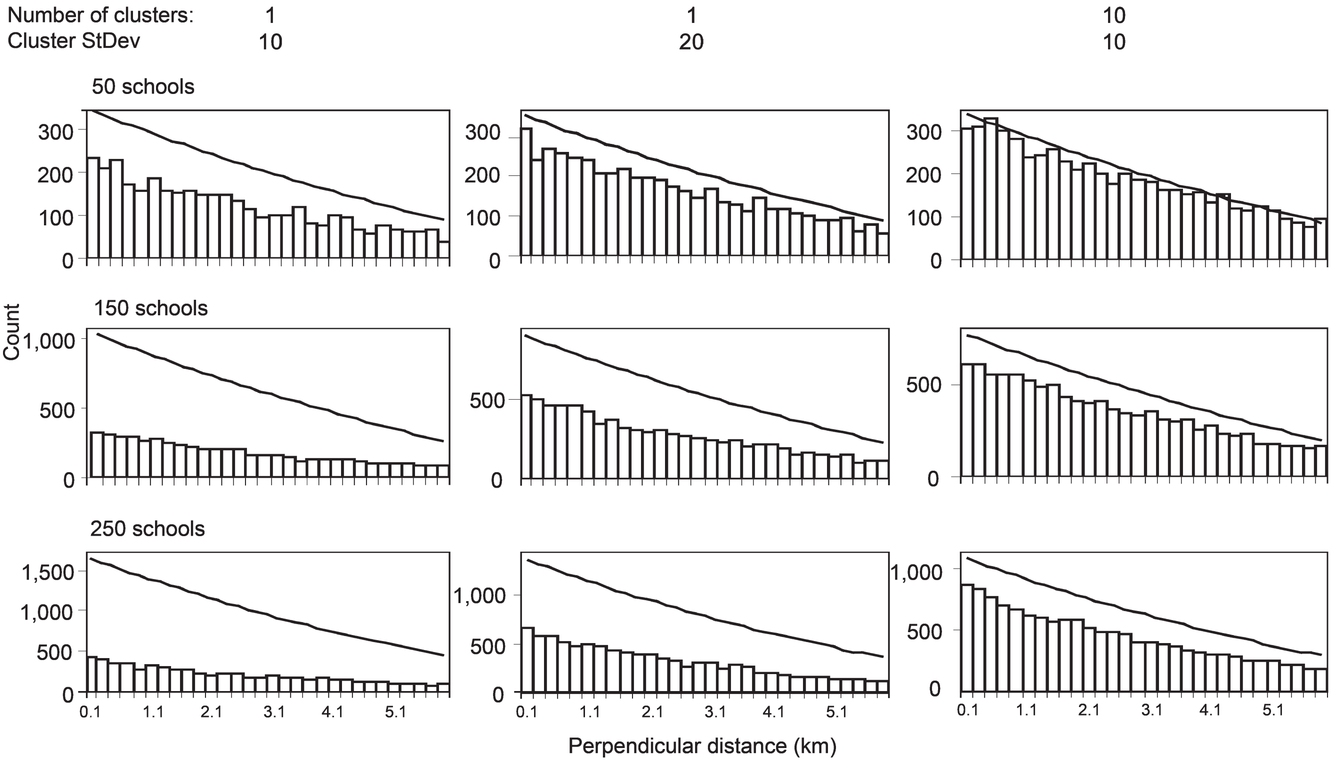

Fig. 11. Count of perpendicular distance of detected objects in closing mode simulation with differing overall school density and different clustering characteristics. Histogram bars are counts by 0.2km distances, so numbers on x-axis represent perpendicular distance midpoints for each bin. Curve represents the counts one would expect from a passing mode line-transect survey. Maximum counts are different for each scenario based on the amount of trackline covered in closing mode. One thousand simulations conducted for each density and clustering combination.

are located in one relatively tight cluster. Variability becomes more stable with more and larger clusters.

Distribution of detected sightings by perpendicular distance indicates detection probabilities by perpendicular distance are lower in closing mode than in passing mode when schools are in fewer clusters and when clusters are more concentrated (Fig. 11). The difference is more pronounced when there are more schools in the area (overall school density is larger) and for schools closer to the trackline (shorter perpendicular distances) (Fig. 11). One would expect from the simulation results that the overall distribution of perpendicular distance for detected sightings would be higher in closing mode than in passing mode, and the disparity would be larger when the encounter rate ratio is smaller. Overall, the differences in perpendicular distances of delphinid sightings are not very different by survey method (Fig. 12).

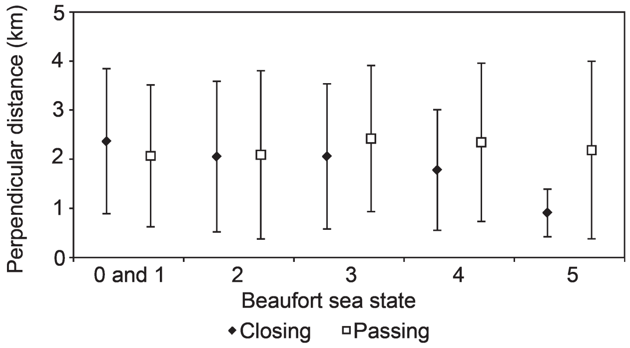

Fig. 12. Perpendicular distance as a function of sea state and survey method for delphinid sightings. Points are means and whiskers are ± 1 std. Sighting conditions limited to Beaufort ≤ 5 and visibility >5.5km. Perpendicular distances of sightings are limited to ≤ 5.5km.

DISCUSSION

The results of this study confirm observers' abilities to determine species composition and estimate school size are poor in passing mode. Results are similar to those seen in Barlow (1997) and Haw (1991). In addition, uncertainty in school size estimates both within and between observers was higher in passing mode. In general, we expect a positive correlation between detection distance and group size, since larger schools are easier to detect at distance than smaller schools. Such a correlation could explain higher school size estimates with distance in closing mode. However, the ships may also influence group behaviour, and multiple smaller schools may coalesce in to larger schools as the ship approaches an area. In passing mode, school size estimates actually declined with distance, implying poorer school size estimation at distance. Once a school is detected in passing mode, observers could miss individual animals when schools are further away. Even when data were filtered to only include sightings where the observer was able to give best, high and low estimates of school size, school sizes were lower in passing mode. The school size differences occur well within the 6.0km perpendicular distance commonly used in detection functions for delphinid species in the ETP. Another possible explanation for lower school sizes in passing mode is that observers may see different parts of a large school and may record it as multiple small schools rather than a single large one. Such an effect would contribute to smaller estimated school sizes, as well as to higher school encounter rates in passing mode. To eliminate any potential bias, passing mode school size estimates would need to be recalibrated (Barlow *et al.*, 1998). However, higher uncertainty in school size estimates in passing mode would still carry over as higher uncertainty in abundance estimates compared to closing mode.

As Haw (1991) found with Antarctic minke whales, overall our delphinid school encounter rates were 20–25% lower in closing mode than in passing mode. Results show the variability in encounter rates due to random movements of schools appears to be much larger than the bias one would see in closing mode on a daily scale. One result was not consistent with general findings. Beaufort 5 encounter rate ratios had a high probability of falling above one, meaning encounter rates were actually higher in closing mode than passing mode. However, the result may be due to a small sample size, with only four days and 126km of effort in closing mode and five days and 392km of effort in passing mode.

Empirical comparisons of overall abundance estimates would not be very informative at this point because of potential biases using both methods, particularly at longer distances. If groups simply coalesce when the ship approaches in closing mode or if observers detect large groups as several smaller groups in passing mode, neither survey method produces an inherent bias in abundance estimates. Otherwise, passing mode may produce lower abundance estimates due to smaller school size estimates while closing mode also leads to an underestimate in abundance because of lower encounter rates. In any case, passing mode abundance estimates would have higher variability due to higher uncertainty in school size estimates, and passing mode methods reduce the ability to identify groups to the species level.

However, simulations confirm the suspicions of Haw (1991), showing that the reduction in detection probabilities at the beginning of each segment of search effort contributes to a bias towards lower encounter rates in closing mode. Although closing mode vessels may get ‘trapped’ in high density areas, the loss in detection probability from stopping and re-starting effort in those areas overall makes up for a positive bias one might get in such a situation. The degree to which detection probability is affected in closing mode is a function of school clustering and overall school density in the sampling area.

The simulations also provide insight in to differences in detection probability on the trackline, $g(0)$, in the two modes. Closing mode $g(0)$ is always less than or equal to $g(0)$ in passing mode. If one assumes $g(0)$ is the same in both modes (which is often the case), detection probability in closing mode will be overestimated further from the trackline, which in the long run leads to an even larger underestimation bias in abundance estimates. The magnitude of the bias is also dependent on school clustering characteristics, the number of schools in the area and presumably the shape of the underlying radial detection function (Equation 11 is just one theoretical example). Therefore, the magnitude of the bias needs to be calculated on a case-by-case basis and is beyond the scope of this paper.

Variability in the ratio of daily closing mode encounter rate and passing mode encounter rate can be quite high even when school density in the area does not change and without complex hierarchical modeling. The random placement of clusters, random detection and random movement of the ship in relation to detected schools creates such variability. For example, if school density ranges from high to low along the planned daily trackline, a closing mode ship will undersample areas of low density since it will spend a higher proportion of time sampling in the high density area, leading to a higher encounter rate ratio. If school density ranges from low to high, the opposite occurs (undersampling of high density areas and a lower encounter rate ratio). If planned daily tracklines are randomly placed in relation to school density, such over- and under-sampling will be reflected in the distribution of the encounter rate ratio.

Direct comparisons of results from the empirical data and the simulations should be done with caution for several reasons. First, we do not know the detection probability in relation to radial distance (Equation 11) for the empirical data. Although the general patterns seen in the simulated results would be similar to the empirical results, the absolute quantities in the encounter rate ratio could be quite different depending on the shape of Equation 11. Second, the clustering characteristics of delphinid schools are unknown, which again play a role in the absolute quantities of the encounter rate ratios. Although the simulations were created to mimic as closely as possible potential clustering scenarios and closing and passing mode data collection techniques, the simulations only function as a method to understand the underlying mechanisms that would lead to different encounter rate ratios. They are not meant to be used, at this point, to quantify any sort of correction factor or determine the type of school clustering.

There are several analytical and methodological techniques that could adjust for the overall lower encounter rate in closing mode. If we have an accurate measurement of $g(0)$ and the correct form of the perpendicular distance detection function in closing mode, we have the appropriate correction factor. In addition, we could potentially derive an analytical correction factor for closing mode using the simulation techniques presented in this paper if we knew the correct form of Equation 11 and the level of school clustering in the data. As with Antarctic minke whale surveys, one could also adjust data collection protocols to include some passing mode effort. Then we could empirically estimate a correction factor, similar to Branch and Butterworth (2001b).

However, several aspects of the current data collection methods make such analytical and methodological adjustments difficult in the ETP. First, $g(0)$ in closing mode is not easy to measure without some sort of comparison in passing mode. Second, the cluster characteristics of data collected in closing mode are difficult to determine since the measured distance between sightings is a function of (1) the non-random movements of the ship potentially in response to clustering; and (2) the changes in detection probabilities that occur with such stop-start searching effort. In addition, the ship moves in response to almost all cetacean sightings, not just the species of interest. So, different species may have differing cluster patterns that could affect the movement of the ship and thus encounter rate estimates of all species. Determination of school patchiness with such analytical complexities is a topic in need of further study. In addition, school clustering is likely to be highly variable in space and time considering the very large area and variable ecosystem traits of the ETP. Lastly, detection probability as a function of radial distance from the ship is most certainly different for different school sizes.

The benefits of incorporating passing mode in to the protocol to remove closing mode bias need to be compared with any disadvantages. In addition to a loss in precision for species ID and school size estimates, passing mode days would lead to the loss of some additional, highly-valuable data from photographs and tissue samples that can only be collected in closing mode. Such data are important for population, genetics, contaminant, diet and life history studies (André *et al.*, 1990; Archer, 1996; Borrell *et al.*, 2004; Escorza-Treviño *et al.*, 2005; Kane *et al.*, 2008; Kellar, 2008; Kellar *et al.*, 2009; Kellar *et al.*, 2006; Olson and Gerrodette, 2008; Smith and Worthy, 2006).

In summary, the study confirms observers' abilities to identify species and estimate school size are improved using closing mode methods. However, both empirical data and simulations indicate closing mode methods produce an underestimate in encounter rates, potentially leading to an underestimate in abundance. The cause of the underestimation is due to the stop-start nature of the method, which results in lower detection probabilities of schools already within the 12km visual range when effort begins or resumes. On a daily scale, variability in abundance and changing spatial distributions of schools in the area produce a high level of uncertainty in encounter rate ratios. Adjustments to field methods in the ETP could remove the bias but result in higher levels of uncertainty in the overall abundance estimate and the loss of other important data. While difficult to implement, analytical and methodological methods to remove or reduce the bias should be explored further.

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Habitat and abundance of cetaceans in Atlantic Ocean continental slope waters off the eastern USA

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ABSTRACT

This study quantifies the abundance and spatial distribution of the cetacean community occupying continental shelf edge and inner continental slope waters along the US southeast Atlantic coast. A shipboard visual line-transect survey was conducted between June and August of 2004 that included effort in waters >50m deep encompassing the shelf break and inner continental slope off the US east coast between 28°N and 38°N latitude. The abundance of nine cetacean taxa was estimated using line-transect distance analysis and an independent observer approach to correct for visibility bias. Canonical correspondence analysis was used to examine the spatial distribution of the cetaceans encountered during the survey as a function of surface temperature, surface salinity, surface fluorescence, bottom depth, and bottom slope. The abundance estimates for most species were much higher than those from a study of the area conducted in 1998. This is primarily due to increased coverage of the shelf-break region and correction for visibility bias. The multivariate analysis indicated four distinct groups of cetaceans that partitioned habitat as a function of salinity, depth, and a latitudinal gradient. These groups were associated with specific water masses and hydrographic features including mid-Atlantic shelf waters (Group I), the shelf break (Group II), mid-Atlantic slope waters (Group III), and south Atlantic slope water (Group IV). Areas where water masses converge such as the continental shelf break along the mid-Atlantic and near Cape Hatteras, North Carolina are therefore areas of both high diversity and density of cetaceans.

KEY WORDS: CETACEAN HABITAT; LINE-TRANSECT SURVEY; MULTIVARIATE ANALYSIS

INTRODUCTION

Cetaceans are highly mobile predators that occupy a diverse range of habitats throughout the world's oceans. Habitat selection and spatial distribution are thought to be largely determined by prey density, particularly for the odontocetes (e.g. Baumgartner *et al.*, 2001; Kenny *et al.*, 1995). Cephalopods and pelagic fish are the primary prey of odontocetes occurring in deep continental shelf and slope waters (e.g. Cañadas *et al.*, 2002; Gannon *et al.*, 1997; Mintzer *et al.*, 2008). The density of such prey varies both seasonally and spatially. Oceanographic features such as water mass boundaries, mesoscale eddies, upwelling or downwelling regions and convergence zones have the potential to locally increase prey densities in response to increases in secondary production. These features may also increase the availability of prey to shallow-diving cetaceans by increasing the abundance of prey near the surface (Baumgartner *et al.*, 2001).

Given the strong correlation between bathymetry and underlying circulation patterns in shelf and slope systems, many studies have demonstrated differentiation of cetacean habitats within bathymetric zones (e.g. Azellino *et al.*, 2008; Cañadas *et al.*, 2002). In the northern Gulf of Mexico, for example, several species of delphinids showed preferences for distinct bathymetric zones separating species that occurred near the shelf break with steeper bathymetry from those with a more broad distribution over deeper waters. However, finer scale partitioning of habitat within these groups was driven by hydrographic features such as thermocline depth (Baumgartner *et al.*, 2001). Similarly, beaked whales (family Ziphiidae) and sperm whales (*Physeter macrocephalus*) along the northeast coast of the US occupied waters near the shelf-break and inner continental slope but partitioned habitat at smaller scales based upon water temperature (Waring *et al.*, 2001). A broader study of the northeast US pelagic cetacean community likewise demonstrated groupings of species by bathymetry and latitudinal range corresponding to water temperature (Hamazaki, 2002). The spatial distribution of these species groups shifted between years as a result of variations in water temperature. These studies demonstrate that cetaceans, like their pelagic prey, respond to environmental variation by moving to track preferred habitats (Redfern *et al.*, 2006).

The outer continental shelf and inner slope of the Atlantic Ocean along the US east coast between 28°N and 38°N (Fig. 1) encompasses a diverse suite of cetacean habitats. Over the southern portion of the survey area (south of Cape Hatteras, North Carolina), the shelf break (roughly the 200m isobath) is dominated by warm, high salinity waters of the Gulf Stream. On the western side of the Gulf Stream, South Atlantic Shelf waters are present, while the eastern side is dominated by the low productivity waters of the Sargasso Sea (Schmitz *et al.*, 1987). The Blake Plateau, with bottom depths of approximately 1,000m, extends approximately 300km east of the continental shelf break. North of Cape Hatteras, the Gulf Stream diverges from the continental shelf break. Between the shelf break and the Gulf Stream are slope waters with surface temperatures of 20–24°C and salinity ranging between 34–35psu. The waters over the shelf (Mid-Atlantic Shelf Water) in this region are both cooler and of lower salinity than the adjacent slope water. The shelf water

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bulges out over the shelf break in a wedge extending from the 100m isobath to the surface 30–50km seaward. Associated with this bulge is a pool of cold, low salinity (~33psu) water occurring at depths of 50–80m (Schmitz *et al.*, 1987). At the surface, the boundary between the shelf and slope waters is evinced as the shelf-break front with a strong cross-shelf salinity gradient (Gawarkiewicz *et al.*, 1996). Near Cape Hatteras, the hydrography is complex due to the interaction between the Gulf Stream and both the Mid-Atlantic and South Atlantic Shelf water masses.

Against the backdrop of this complex oceanographic structure, the cetacean community includes both sub-tropical and temperate species responding to a range of bathymetric and oceanographic regimes. In this study, we assess the abundance and spatial distribution of this diverse cetacean community and use constrained ordination analysis to examine the relationships between species groups and large-scale oceanographic and bathymetric features.

METHODS

Survey methods

A visual line-transect survey was conducted aboard the NOAA Ship *Gordon Gunter* from 22 June to 19 August 2004. The survey was conducted in water depths greater than 50m and covered waters including the outer continental shelf, the shelf break and the inner continental slope to the US Exclusive Economic Zone. Survey effort was conducted in three strata: South Atlantic slope (Area = 146,933km^2); the Mid-Atlantic shelf break (Area = 74,114km^2); and the Mid-Atlantic slope (Area = 194,326km^2, Fig. 1). Tracklines were arranged in a 'double saw-tooth' pattern perpendicular to the bathymetry with a randomised starting point to provide uniform coverage probabilities within each stratum. However, not all planned tracklines were covered due to weather conditions. Survey speed was typically 18km hr^{-1} (~10 knots). Survey effort was suspended during heavy seas (swell height >2m), rain, or other poor visibility conditions (sea state >5 on the Beaufort scale).

The survey was conducted using a two-team independent observer approach to estimate abundance and account for visibility bias (Laake and Borchers, 2004). The first observer team was stationed on the ship's flying bridge (average eye height above water = 13.7m), and the second team was stationed at a lower platform on the bridge wings (average eye height above water = 11.0m). The two teams were isolated from one another to avoid cueing each other to the presence of marine mammals. The flying bridge team included two observers searching with 25 × 150 'bigeye' binoculars and a centre observer searching with handheld binoculars and the naked eye. The bridge wing team consisted of two observers searching with bigeye binoculars. The bigeye observers searched the arc from the ship's bow (the trackline) to the vessel beam on each side, while the third observer on the flying bridge primarily concentrated on the trackline and near the ship.

A data recorder maintained independent communication with both teams and recorded data on sightings by each team. This coordinating observer was also responsible for identifying sightings that were seen by both teams. Upon a marine mammal sighting made by one of the teams, the position of the group was plotted, and the sighting team went off effort to continue to track the group. The second team was not informed of the sighting and remained on effort. If the mammal group went past the vessel's beam (relative bearing 90°) without being seen by the second team, then it was considered missed. Once the group was 'missed' or seen by the second team, then both teams went off effort, and the vessel was turned to approach the group for species identification and group size estimation. If the two teams saw two separate sightings of marine mammals at the same time, then the vessel typically turned to identify the closest group first and then attempted to relocate the farther group.

For each cetacean group sighted, time, position, bearing to the sighting, radial distance to the sighting, species, group size, behaviour, bottom depth, sea surface temperature, and associated animals (e.g. seabirds and fish) were recorded. The radial distance to sightings made from the bigeye binoculars was measured with reticles, while distances were visually estimated for groups sighted by naked eye or handheld binoculars. Survey effort data were recorded every two minutes and included the ship's position and heading, effort status, observer positions, and environmental conditions which could affect the observers' ability to sight animals (e.g. Beaufort sea state, trackline glare, cloud cover, etc.). Typically, if a sighting was within 3 n.miles on either side of the ship, the ship was diverted from the trackline to approach the group to identify species and estimate group size. Cetaceans were identified to the lowest taxonomic level possible. Unidentified animals were typically those that were not re-sighted after the initial cue.

There are three abundant taxonomic groups used in this study that combine species or genetically distinct populations that cannot be differentiated at sea and may have different habitat associations. The first is the pilot whales (*Globicephala* sp.) which may include both the short-finned (*G. macrorhynchus*) and long-finned (*G. melas*) species. The spatial range of the two species overlaps in the region between 35°N and 41°N, with the short-finned species having a generally more southern tropical and sub-tropical distribution (Leatherwood *et al.*, 1983). Given the spatial range of our survey, it is likely that our results are more indicative of the distribution of the short-finned pilot whale. The second group is the Atlantic spotted dolphin (*Stenella frontalis*) which occurs in a more near-shore coastal form and a more offshore form (Adams and Rosel, 2006). These two groups are genetically distinct from one another, and there is evidence for differentiation of habitat near Cape Hatteras. It is unclear if both groups are included in the current analysis. Finally, there are two distinct forms of bottlenose dolphins (*Tursiops truncatus*), a larger more robust type with a more offshore distribution and a smaller type with a more coastal distribution (Hersh and Duffield, 1990; Torres *et al.*, 2003). It is most likely that the current survey includes predominantly the offshore morphotype.

Abundance estimation

Abundance estimates for observed cetacean species were derived using the independent observer approach assuming point independence (Laake and Borchers, 2004) as implemented in the Distance computer program (version 5.0 release 2; Thomas *et al.*, 2006). Briefly, this approach is an extension of standard line-transect distance analysis that includes direct estimation of sighting probability on the trackline. The probability of sighting a particular group is the product of two probability components. The first probability corresponds to the 'standard' sighting function such that the probability of detection declines with increasing distance from the trackline following a known functional form (typically the half-normal or hazard function). The second component is the likelihood of detection on the trackline which is modelled using a logistic regression approach and the 'capture histories' of each sighting (i.e. seen by one or both teams). The logistic model can include factors that may affect the probability of detection such as viewing or weather conditions. Details on the derivation, assumptions, and implementation of the estimation approach are provided in Laake and Borchers (2004).

Sighting probability analyses were conducted separately for three groups of cetaceans, dolphins, pilot whales, and large whales, to account for differences in body size and surface behaviour and associated differences in sighting probability (Table 1; Barlow, 1995; Mullin and Fulling, 2003). While 'cryptic' species including beaked whales (family Ziphiidae) and pygmy/dwarf sperm whales (*Kogia* spp.) were observed, there were insufficient sightings of these species for reliable abundance estimation. For each species group, sighting probability was estimated globally across strata. The perpendicular sighting distances were right-truncated to remove roughly 10% of the sightings with the farthest distances (Buckland *et al.*, 2001) which corresponded to 5,000m for the dolphins and large whale groups and 4,000m for the pilot whales. The form of the sighting function (hazard vs. half-normal) and the inclusion of covariates (including group size, sea state, glare, swell height, wind speed) in the mark-recapture model were evaluated through model selection based upon the Akaike Information Criterion (AIC; Laake and Borchers, 2004). There was no evidence of reactive movements to the survey vessel, and environmental covariates had little effect on either the sighting distance or mark-recapture components of the model with the exception of group size which was important in the mark-recapture portion of the model for pilot whales and large whales. Stratified abundance estimates for each individual taxon were calculated using stratum and species level encounter rates (groups per km of trackline) and mean group size.

Habitat associations

Surface layer (measurements taken at <5m depth) salinity, temperature and fluorescence (recorded in micrograms per litre and used as a proxy for chlorophyll concentration) were recorded continuously throughout the survey using sensors deployed aboard the vessel. These hydrographic data were used to assess habitat associations and groupings amongst the species encountered during the survey. Bottom depth along the trackline was derived from the ETOPO2 global bathymetry dataset[1]. The bathymetric slope was derived from the bathymetry grid using tools in ArcGIS Spatial Analyst (ESRI, Inc.).

Habitat variables (temperature, salinity, fluorescence, bottom depth, and bathymetric slope) were summarised into 10 × 10km grid cells. For each cell, the mean of the environmental variables, total survey effort and number of marine mammal groups and individuals (by species) were calculated. The spatial cells were treated as the sampling unit in a multivariate analysis of marine mammal habitat associations.

Canonical correspondence analysis (CCA; Ter Braak, 1986) was used to examine the habitat associations of the marine mammal species encountered during the survey. CCA is a constrained ordination approach that quantifies the amount of variation in a multivariate response (i.e. species abundance) that can be explained by a selected suite of environmental/habitat characteristics (Ter Braak, 1986). The response matrix was the species composition within each cell expressed as the total number of animals of each species sighted in the cell. Several metrics of species occurrence were explored including dividing the number of animals observed by the amount of trackline in the cell as a proxy for animal density. In addition, log- and square-root transformations were applied to reduce the influence of rare species. The results of the analysis were insensitive to these transformations, and therefore simple counts were used. The explanatory matrix included the suite of environmental variables described above for each cell in addition to the X ('Easting') and Y ('Northing') location of the cell based on the Universal Transverse Mercator (WGS 1984, Zone 18N) projection of the grid. These spatial variables were included as main effects in the models after exploratory analyses indicated no significant spatial confounding of habitat relationships (Borcard *et al.*, 1992). All CCA analyses were conducted using the package 'vegan' implemented in the R statistics package (Oksanen *et al.*, 2008).

A stepwise selection approach was used to select explanatory variables to constrain the ordination. Each variable was first submitted as a single term, and its significance was tested using permutation tests (1,000 permutations) of the F-statistic. Those variables that were statistically significant ($p<0.05$) were then included in progressive two and three term models until all significant variables were included. Single terms were then sequentially dropped from this full model to verify their explanatory significance. The multiple term significance tests in CCA may be sensitive to the order of entry into the model (Oksanen *et al.*, 2008); therefore, the stepwise approach included the entry and removal of terms in differing orders to avoid this artifact. The model including all significant effects was used to examine correlations between species distribution and habitat variables, associations between species, and spatial patterning in marine mammal habitats.

RESULTS

Survey results

A total of 5,139km of effort was completed during the survey including 1,601km in the South Atlantic slope stratum, 1,798km in the Mid-Atlantic shelf break stratum, and 1,739km in the Mid-Atlantic slope stratum (Fig. 1). Several survey days were lost due to poor weather conditions including Hurricanes Alex and Charley. The vast majority (>90%) of the survey effort was conducted in Beaufort sea states of three or less. There were a total of 364 marine mammal groups sighted by one or more survey teams including 17 taxa (Table 1). The most common species sighted were bottlenose dolphins, sperm whales and pilot

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[1] *http://www.ngdc.noaa.gov/mgg/global/global.html.*

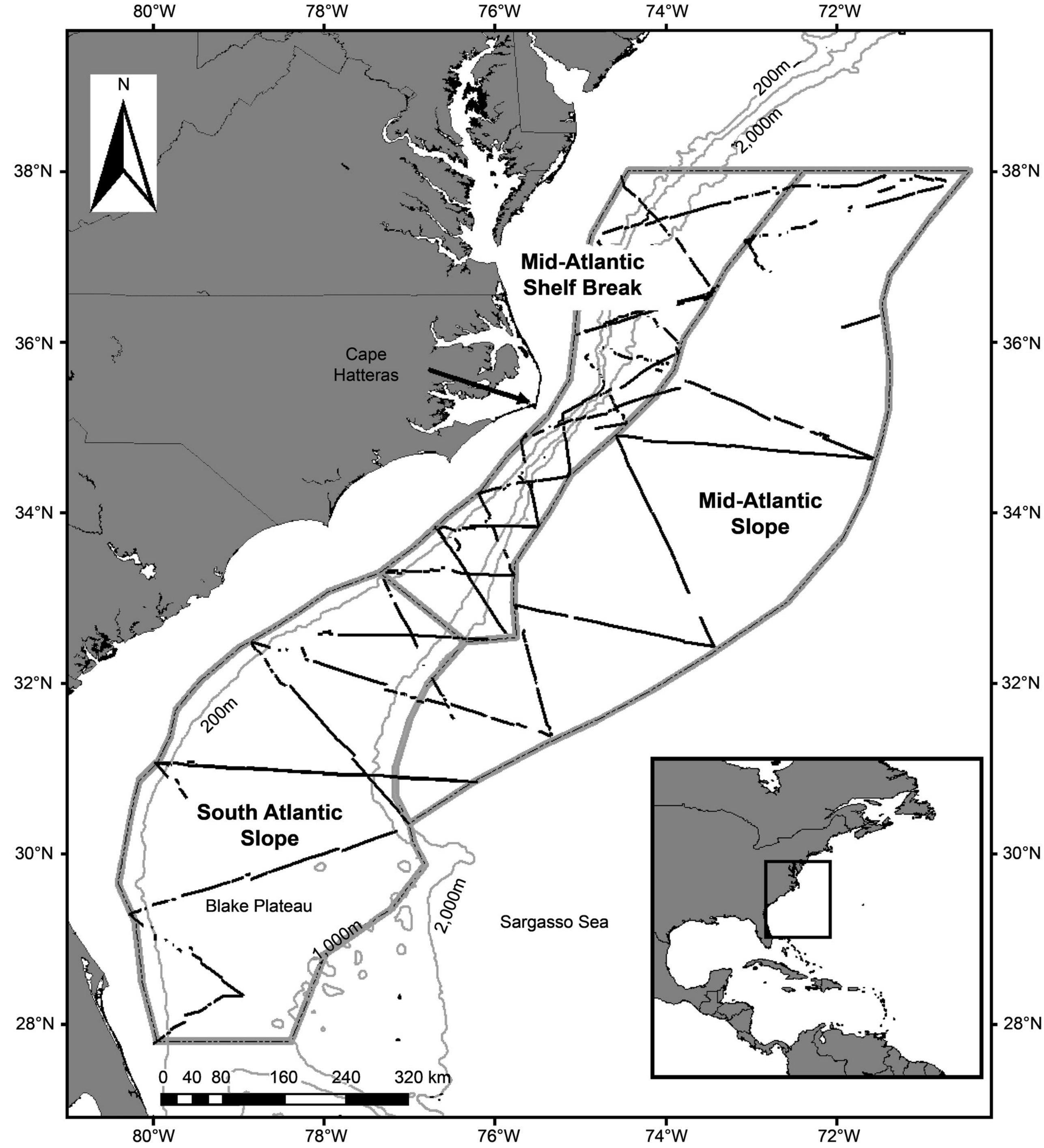

Fig. 1. Survey effort (black lines) and strata (within thick grey lines) during the summer 2004 Atlantic Cetacean Survey. Contours derived from the ETOPO-2 bathymetry grid are shown.

whales. Both the number of sightings and the number of species observed in the Mid-Atlantic shelf break stratum were much higher than in either of the other two strata. Mean group sizes for most taxa were consistent across strata (Table 1).

Abundance estimates

For each group, the best-fitting models were selected based upon the model with the lowest value of AIC. There was little evidence that covariates, other than group size for large whales and pilot whales, improved the overall model fit. In most cases, the addition of covariates to the models resulted in small increases in AIC (generally less than 2), indicating that the parsimonious model excluding covariates was at least as appropriate as the more complex models. Models including all covariates resulted in ΔAIC values greater than 7, indicating that there was little support for including this additional complexity. The choice of the form of the sighting function (hazard vs. half-normal) had strong support for each group with ΔAIC values of 4 or greater in each case. The selected models were effective at fitting the sighting probabilities of the different capture histories (seen by team 1 only, seen by team 2 only, or duplicates) as demonstrated by Chi-sq goodness of fit tests for all cases.

For the dolphin sub-group, the best fitting sighting function was a hazard model including no additional covariates in the mark-recapture component of the model (Fig. 2a). The average estimated sighting probability within the surveyed strip was 0.27 (CV = 0.157). The sighting

Table 1

Number of groups sighted (N) and mean group size by stratum for each marine mammal taxon encountered during the summer 2004 Atlantic cetacean survey. The coefficient of variation (CV) of the mean is indicated in parentheses.

	South Atlantic Slope		Mid-Atlantic Shelf Break		Mid-Atlantic Slope	
Species	N	Mean group size (CV)	N	Mean group size (CV)	N	Mean group size (CV)
Subgroup – dolphins						
Atlantic spotted dolphin (*Stenella frontalis*)	1	73.0 (–)	21	83.0 (0.19)	9	73.6 (0.44)
Bottlenose dolphin (*Tursiops truncatus*)	28	25.6 (0.19)	43	33.1 (0.17)	4	34.3 (0.14)
Bottlenose/Atlantic spotted dolphin	4	10.0 (0.44)	0	–	0	–
Short-beaked common dolphin (*Delphinus delphis*)	0	–	10	267.2 (0.36)	0	–
Pantropical spotted dolphin (*Stenella attenuata*)	2	52.5 (0.71)	0	–	1	70.0 (–)
Risso's dolphin (*Grampus griseus*)	6	23.3 (0.52)	8	24.3 (0.39)	1	15.0 (–)
Stenella sp.	0	–	0	–	2	6.5 (0.23)
Striped dolphin (*Stenella coeruleoalba*)	0	–	4	80.0 (0.21)	6	136.5 (0.21)
Unidentified dolphins	11	3.2 (0.27)	21	20.6 (0.35)	7	6.0 (0.54)
Unidentified odontocetes	3	2.0 (0.29)	2	1.0 (0.0)	3	2.0 (0.50)
Subgroup – pilot whales						
Pilot whales (*Globicephala sp.*)	11	24.2 (0.22)	37	19.8 (0.12)	5	12.6 (0.17)
Subgroup – large whales						
Fin whale (*Balaenoptera physalus*)	0	–	1	2.0 (–)	0	–
Unidentified baleen whales	0	–	1	12.0 (–)	1	1.0 (–)
Sperm whale (*Physeter macrocephalus*)	2	2.0 (0.50)	69	2.3 (0.08)	14	1.9 (0.15)
Unidentified large whales		–	3	1.0 (0.0)	4	1.0 (0.0)
Subgroup – cryptic species						
Beaked whales (*Ziphiidae*)	5	1.4 (0.17)	8	1.4 (0.27)	4	2.0 (0.20)
Pygmy/dwarf sperm whale (*Kogia* sp.)	0	–	1	1.0 (–)	1	1.0 (–)

function had a good overall fit to the observed sighting distances (Chi-sq Goodness of Fit [GOF], $p = 0.228$). The estimated sighting probability on the trackline for each team independently was 0.596 (CV = 0.068) and for the two teams jointly was 0.837 (CV = 0.039).

A hazard rate model including cluster size and sighting distance was the best fitting mark-recapture model for pilot whales and had a good fit to the observed data (Chi-sq GOF, $p = 0.281$; Fig. 2b). The average estimated sighting probability for pilot whales in the survey strip was 0.35 (CV = 0.31). The estimated sighting probability on the trackline for each team independently was 0.47 (CV = 0.18) and was 0.69 (CV = 0.13) for the two teams jointly.

For the large whales, the best mark-recapture model was a half-normal function including cluster size as a covariate (Fig. 2c). The sighting function model was a good fit to the observed data (Chi-sq GOF, $p = 0.640$), and the resulting estimated sighting probability in the survey strip was 0.483 (CV = 0.08). The sighting probability on the trackline for each team independently was 0.52 (CV = 0.13) and was 0.76 (CV = 0.08) for the two teams jointly.

The abundance and density estimates for each stratum, and overall, are shown in Table 2 for species where estimation was possible. The 'baleen whale' group in Table 2 includes both identified fin whales and unidentified baleen whales. The precision of the abundance estimates varied widely as a result of variable encounter rates and group sizes. The most precise estimates approach CV values of 0.30 for bottlenose dolphins and Risso's dolphins, while the least precise approach CVs of 1.0 for Atlantic spotted dolphins and the baleen whales (Table 2).

Among the abundant dolphin species, there are clear differences in density across strata. For example, Atlantic spotted dolphins occurred in high densities in the Mid-Atlantic shelf break and Mid-Atlantic slope strata while bottlenose dolphins were more abundant in the Mid-Atlantic shelf break and the South Atlantic slope strata (Table 2). In

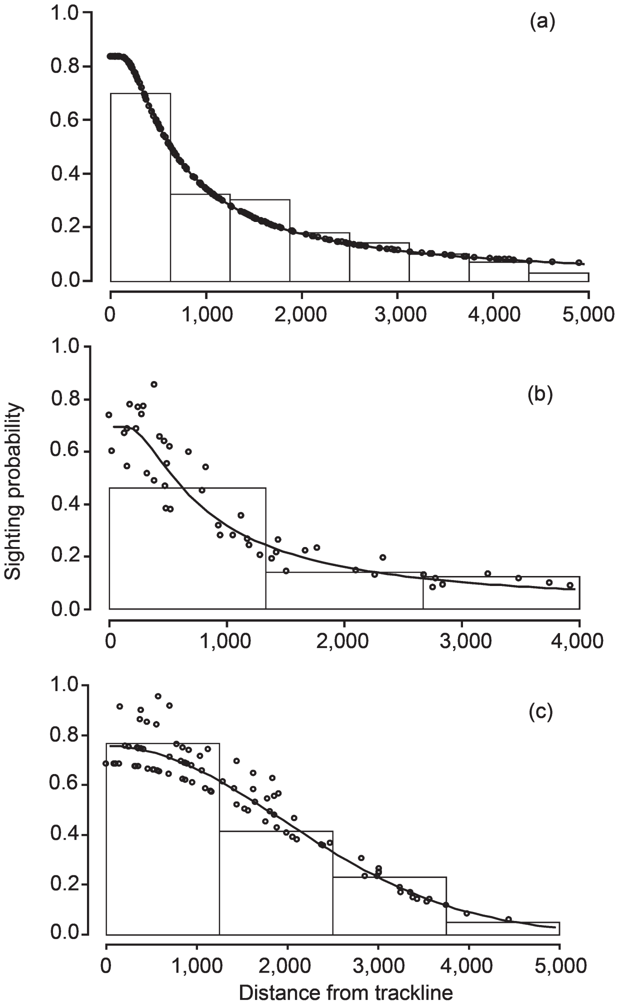

Fig. 2. Sighting detection functions pooled across observer teams for (a) dolphins, (b) pilot whales, and (c) large whales. The line indicates the fitted sighting function, and points are estimated sighting probabilities for individual groups. Bars are grouped distance intervals used for chi-square goodness of fit tests, though sighting function fits are based on ungrouped data. Plots were generated using Distance 5.0 (release 2, Thomas *et al.*, 2006).

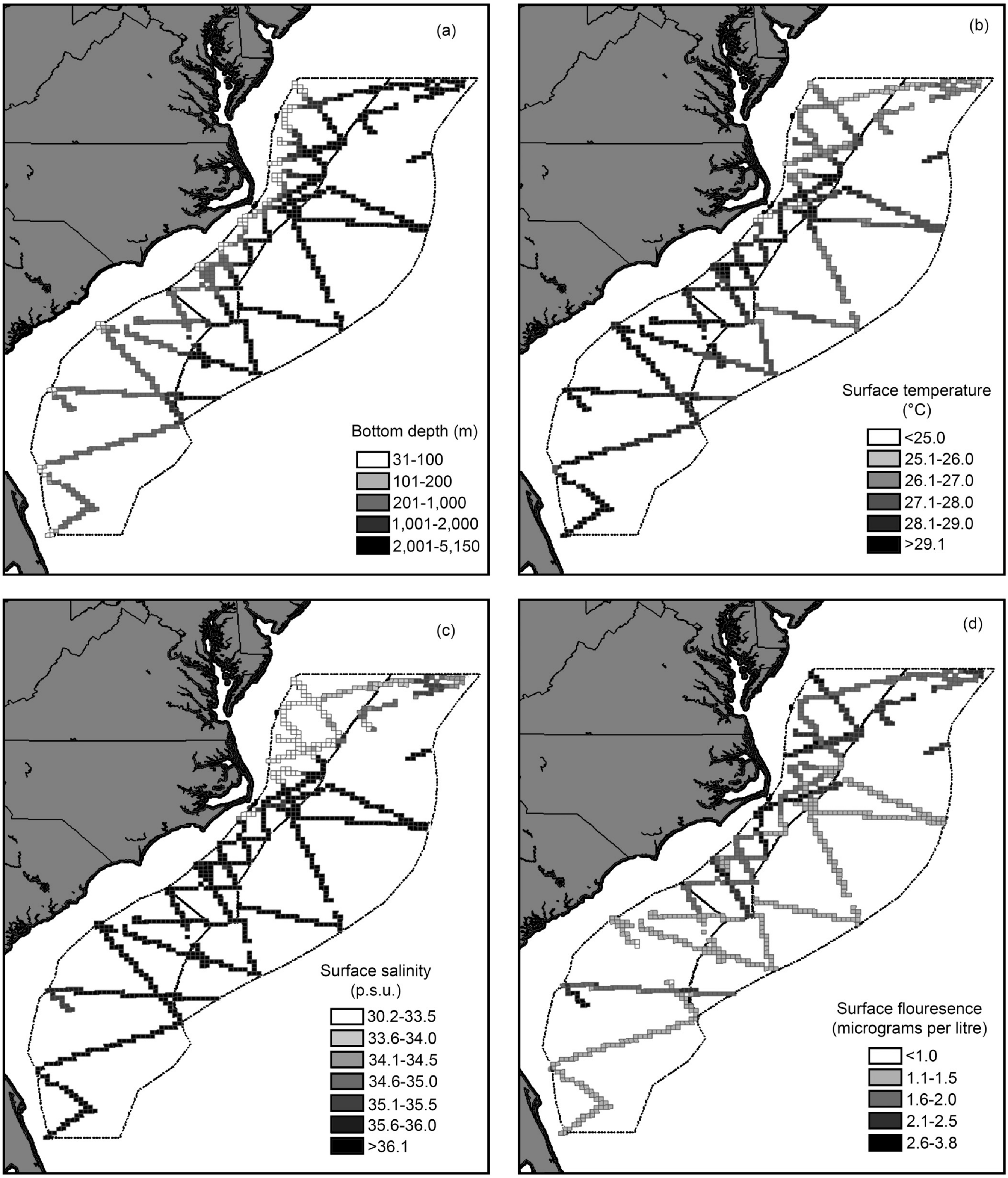

Fig. 3. Mean of environmental variables in sampled 10 × 10km spatial cells: (a) bottom depth, (b) surface temperature, (c) surface salinity, and (d) surface fluorescence.

contrast, striped dolphins occurred almost exclusively in the Mid-Atlantic slope stratum while short-beaked common dolphins were confined to the Mid-Atlantic shelf break stratum (Table 2).

Habitat associations

The sampled area included broad regions with distinct environmental conditions. In the northern portion of the survey range in waters generally < 2,000m depth corresponding to the outer continental shelf and shelf break, the water temperature was cool (< 26°C) and salinity was low (< 33.5psu, Fig. 3). Further offshore in the northern part of the region, water temperature was also cool, but salinity was greater than 35psu. The Gulf Stream (high water temperature, high salinity) was apparent near Cape Hatteras, North Carolina (~35.2°N Latitude) in close proximity to

Table 2

The estimated density (number/km^2) and abundance for each species by stratum from the summer 2004 Atlantic cetacean survey. The coefficient of variation (CV) for density estimates is indicated in parentheses.

	South Atlantic Slope		Mid-Atlantic Shelf Break		Mid-Atlantic Slope		Total	
Species	Density (CV)	Abundance	Density (CV)	Abundance	Density (CV)	Abundance	Density (CV)	Abundance
Atlantic spotted dolphin	0.020 (0.74)	2,891	0.418 (0.52)	30,997	0.164 (1.81)	31,923	0.158 (0.92)	65,812
Bottlenose dolphin	0.183 (0.37)	26,892	0.312 (0.46)	23,172	0.034 (0.69)	6,606	0.136 (0.30)	56,671
Short-beaked common dolphin	0 (–)	0	0.512 (0.85)	37,951	0 (–)	0	0.512 (0.85)	37,951
Pantropical spotted dolphin	0.028 (1.00)	4,158	0 (–)	0	0.017 (0.76)	3,375	0.018 (0.66)	7,534
Risso's dolphin	0.038 (0.53)	5,545	0.026 (0.56)	1,938	0.004 (0.96)	723	0.020 (0.41)	8,207
Striped dolphin	0 (–)	0	0.077 (0.48)	5,690	0.203 (0.694)	39,494	0.109 (0.61)	45,185
Pilot whales	0.029 (0.87)	4,262	0.176 (0.66)	13,055	0.019 (0.62)	3,737	0.0507 (0.52)	21,056
Baleen whales	0 (–)	0	0.002 (1.06)	125	0.0002 (1.35)	34	0.0004 (0.88)	159
Sperm whale	0.001 (0.53)	97	0.021 (0.42)	1,615	0.004 (0.98)	743	0.006 (0.42)	2,455

Mid-Atlantic Shelf and Slope waters. While South of Cape Hatteras, over the Blake Plateau, both water temperature (>28°C) and salinity (>35.5psu) were high (Fig. 3).

In the CCA analysis, salinity ($F = 0.332$, $p = 0.022$), depth ($F = 0.956$, $p = 0.002$), and the Y-coordinate ($F = 0.716$, $p = 0.002$) were retained as significant explanatory factors. The ordination including these variables explained 16.7% of the total inertia in the data. The first two canonical axes accounted for 89.4% of this explained variance. The CCA biplot arrows (Fig. 4) indicate the correlation between the canonical axes (CA) and the explanatory variables. The first axis (CA I) scores were negatively correlated with depth and salinity. The second axis (CA II) was primarily correlated with the Y-coordinate with northern sites having more negative scores and southern sites having more positive scores (Fig. 4). The large scale gradient indicated by the Y-coordinate is partially correlated with the latitudinal temperature gradient; however, accounting directly for this large scale spatial effect by including the Y-coordinate as a conditional variable did not improve the explanatory power of the CCA nor did it result in temperature being included in the selected model.

The CCA indicated four clearly differentiated groups of species based upon the species-environment relationships (Fig. 4). First, the baleen whales (including both unidentified baleen whales and fin whales) and short-beaked common dolphins were grouped and occurred in habitats of shallow water depth and low salinity (Table 3). These two taxa occurred in waters north of Cape Hatteras at depths <1,000m (Fig. 5a). The second group included bottlenose dolphins, Risso's dolphins and pilot whales. These species were distributed more broadly throughout the latitudinal range of the survey in waters between the 1,000–2,000m isobaths. In the northern portion of the survey range, this corresponded to waters near the shelf break, while south of Cape Hatteras, this corresponded to the shallow portion of the Blake Plateau (Fig. 5b). These species had mean temperature values of

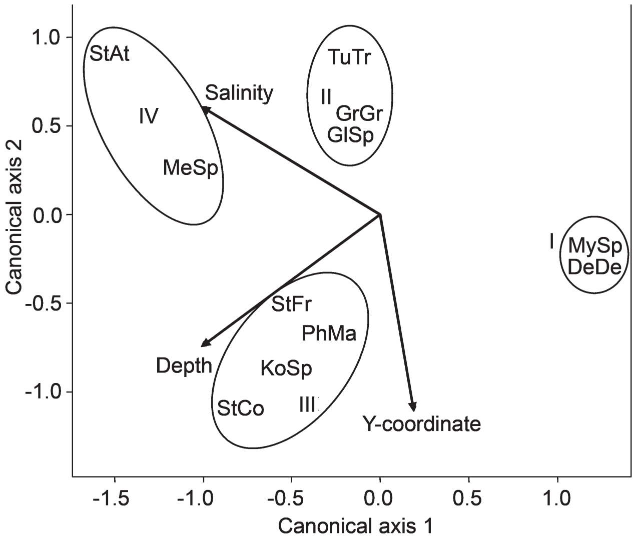

Fig. 4. Canonical correspondence analysis biplot. Arrows indicate the relative importance (length) and correlation (angle with axis) between each variable retained in the model and the canonical axes. Species scores on the canonical axes are indicated by abbreviations (see Table 3). Species groupings are indicated.

Table 3

Mean (weighted by abundance) environmental variable values for each species. The CCA group assignment for each species is shown in Figure 4. The standard deviation of the mean is indicated in parentheses. 'Northing' refers to Latitude projected into the Universal Transverse Mercator (Zone 18N) coordinate system.

CCA Group	Species (abbreviation)	Depth (m)	Temperature (°C)	Salinity (ppt)	Northing (km)
I	Common dolphin (DeDe)	305.6 (148.5)	26.6 (0.12)	31.4 (0.25)	4,020 (23)
	Baleen whales (MySp)	272.7 (546.5)	26.4 (0.33)	31.8 (0.94)	4,062 (122)
II	Pilot whales (GlSp)	1,341.8 (164.8)	28.3 (0.24)	34.9 (0.33)	3,829 (35)
	Risso's dolphin (GrGr)	1,161.4 (293.4)	28.5 (0.51)	35.0 (0.55)	3,839 (76)
	Bottlenose dolphin (TuTr)	1,002.1 (112.8)	28.2 (0.19)	35.5 (0.19)	3,765 (36)
III	Dwarf/pygmy sperm whale (KoSp)	2,605.6 (432.8)	26.7 (0.31)	33.9 (0.84)	4,060 (45)
	Sperm whale (PhMa)	2,186.9 (92.9)	27.0 (0.14)	33.6 (0.22)	4,031 (21)
	Striped dolphin (StCo)	3,050.3 (135.6)	26.5 (0.29)	34.5 (0.31)	4,150 (18)
	Atl. spotted dolphin (StFr)	2,377.4 (188.5)	27.5 (0.34)	34.4 (0.33)	3,998 (24)
IV	Beaked whales (MeSp)	2,280.7 (208.9)	27.8 (0.34)	35.8 (0.25)	3,790 (77)
	Pantropical spotted dolphin (StAt)	2,369.4 (1,230.9)	28.7 (0.82)	36.4 (0.06)	3,473 (229)

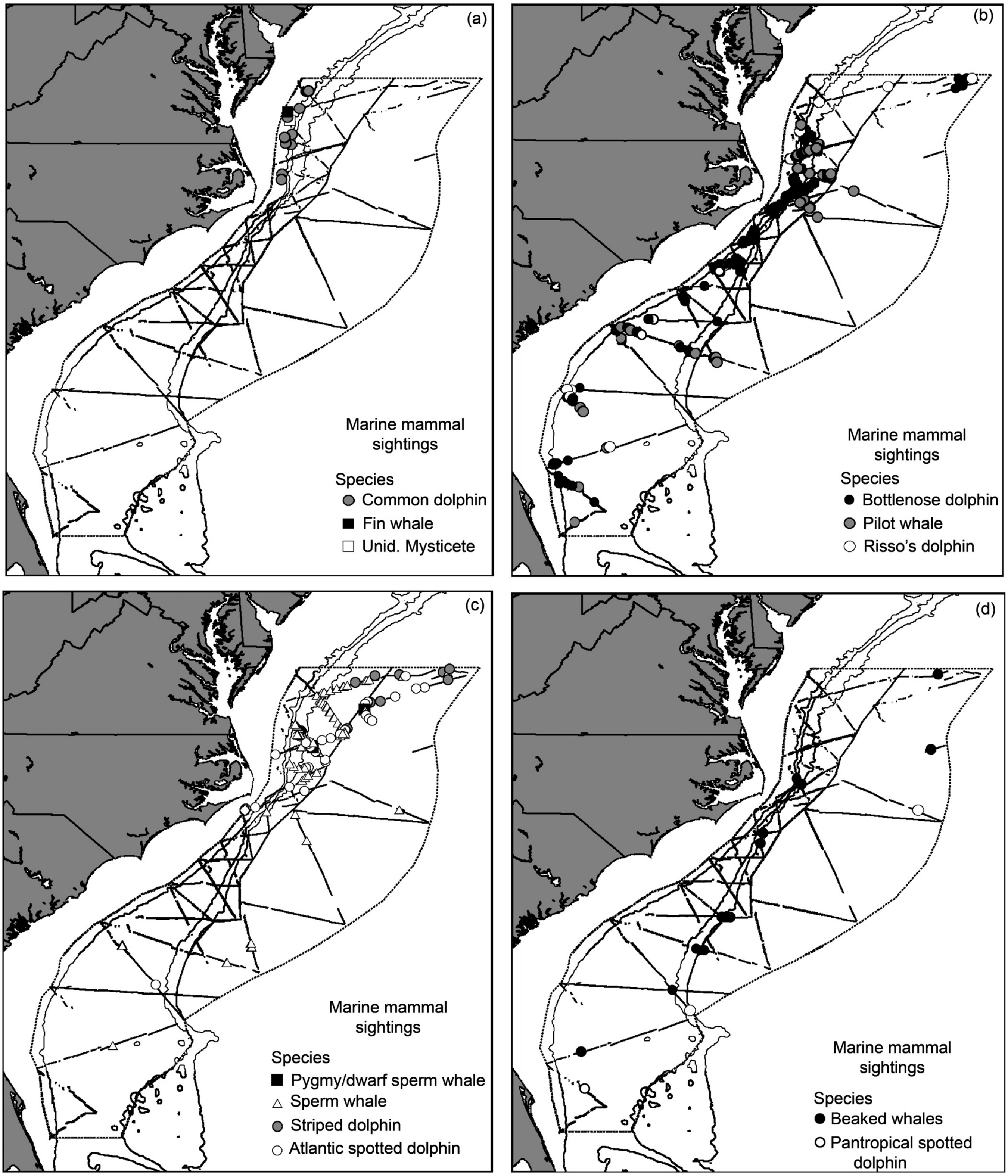

Fig. 5. Sightings of each species in groups identified by CCA including (a) Group I, (b) Group II, (c) Group III, and (d) Group IV.

28–28.5°C, mean salinity of 34.9–35.5psu, and mean depths of 1,000–1,300m (Table 3). The third group included sperm whales, Atlantic spotted dolphins, striped dolphins and *Kogia* sp. (Fig. 4). These species occupied the northern offshore waters deeper than 2,000m (Fig. 5c), and their habitats were characterised by lower water temperatures and intermediate salinities (Table 3). Finally, the beaked whales and pantropical spotted dolphins formed a group of more southern offshore species occurring in waters deeper than the 2,000m isobath (Fig. 5d) with high water temperatures (>28°C) and high salinities (>35.5psu, Table 3).

DISCUSSION

This study demonstrates that there are distinct communities of cetaceans inhabiting oceanic waters along the southeast US Atlantic coast, and that these communities correspond to distinct oceanographic regimes. Our study also provides improved and updated abundance estimates for the cetacean species encountered, which is a critical part of the management of these protected species.

This region was also surveyed during the summer of 1998 (Mullin and Fulling, 2003). The previous survey also conducted line transect sampling; however, there was no correction for visibility bias in the associated abundance estimates. In addition, the previous survey used a uniform sampling design that also included areas over the continental shelf. As a result, there was relatively little effort expended over the shelf break in the 1998 survey (Mullin and Fulling, 2003). Given these differences in survey design and analysis, it is not surprising that there are significant differences in estimated abundance. The 1998 survey used the same observer configuration as the flying bridge team in the 2004 survey. Based upon estimated sighting probabilities for individual teams (ranging from 0.470 to 0.596), it is expected that the estimates from the 2004 survey would be approximately 2× higher than those from 1998. Accordingly, the abundance estimates from the 2004 survey are much higher than those from 1998 for Atlantic spotted dolphins (2004: 65,812 vs. 1998: 14,438), bottlenose dolphins (2004: 56,671 vs. 1998: 24,671), pilot whales (2004: 21,056 vs. 1998: 5,109), striped dolphins (2004: 45,185 vs. 1998: 10,225), and sperm whales (2004: 2,455 vs. 1998: 1,181; Mullin and Fulling, 2003). The increased level of survey effort along the shelf break during 2004 is also an important component of these differences as the majority of pilot whales, Atlantic spotted dolphins, and Atlantic bottlenose dolphins were seen in this stratum, which was undersampled in the 1998 survey (Table 1).

The majority of species observed in 2004 were also observed in 1998. However, the 2004 survey did not include sightings of minke whales (*Balaenoptera acutorostrata*), Clymene dolphins (*Stenella clymene*) or rough-toothed dolphins (*Steno bredanensis*). These species were rare in the 1998 survey with a total of four sightings. Interestingly, the 1998 survey did not observe short-beaked common dolphins, which was one of the more abundant species in the 2004 survey and included encounters of large groups. During summer months, common dolphins occur primarily to the north of the surveyed area along the shelf break near Georges Bank (Selzer and Payne, 1988), and it is therefore expected that the abundance of this species may vary strongly between years as a function of environmental variation.

Risso's dolphin abundance was similar between the two surveys with an estimated abundance of 8,207 (CV = 0.41) during the 2004 survey and 9,533 (CV = 0.50) during 1998 (Mullin and Fulling, 2003). The spatial distribution of Risso's dolphins was similar between the two surveys with sightings occurring throughout the survey range but primarily in the South Atlantic slope and Mid-Atlantic shelf break strata (Fig. 5). As with common dolphins, Risso's dolphins occur primarily along Georges Bank during summer months (Waring *et al.*, 2007), and hence the abundance in the Mid-Atlantic strata may vary. However, it does appear that there is a relatively constant, but lower density, occurrence of Risso's dolphins in the south Atlantic slope stratum.

A study of marine mammal habitats just to the north of the current survey area identified four species groups as a function of water temperature, bottom depth, bottom slope and surface front probability (Hamazaki, 2002). This study included most of the species evaluated in the current analysis. Interestingly, Hamazaki's (2002) grouping of 'Mid-Atlantic Shelf Species' included bottlenose dolphins, Risso's dolphins, pilot whales, and common dolphins. The first three species correspond to our Group II and include species that are associated with shelf break waters and a broad latitudinal distribution. Common dolphins, however, have a markedly different distribution compared to these species as they are associated with shallow, low salinity waters on the eastern side of the shelf break front along with the fin whales (included Hamazaki's Northern Atlantic shelf species group). This difference in groupings is primarily a function of the more southerly spatial range of our study that better covers the range of the Group II species. In addition, our study included salinity as an explanatory factor. The boundary between the Mid-Atlantic shelf water and the Mid-Atlantic slope water is better defined by a salinity gradient in surface waters as opposed to a temperature gradient (Gawarkiewicz *et al.*, 1996). Our Group I species were associated with this cool, lower salinity shelf water, and this reflects the fact that our survey covered the southward extension of their more northerly habitats during summer months. Spatially, this results in dramatic changes in cetacean habitats over relatively small spatial scales. On the shoreward side of the shelf-break front we observed concentrations of these more northerly species. However, on the seaward side of the front, over distances <30km, both the underlying hydrography and the cetacean community are markedly different.

The convergence of water masses, and associated increases in both the density and diversity of the cetacean communities is most apparent near Cape Hatteras. In this region, there is a convergence of Mid-Atlantic shelf water, South Atlantic shelf water, the Gulf Stream, and Mid-Atlantic slope waters (Gawarkiewicz *et al.*, 1996). As a result, there is an associated convergence of all four of our identified species groups within a small and extremely dynamic region. Within the area between Cape Hatteras and the mouth of the Chesapeake Bay (~37ºN latitude) near the shelf-break we observed 9 of our 11 identified species (only fin whales and *Kogia* sp. were not observed) in 138 groups of marine mammals totalling 5,648 individuals. In addition, this area included numerous large groups of dolphins including 9 groups of more than 100 Atlantic spotted dolphins and 5 groups of common dolphins ranging between 350 and more than 1,000 individuals. This dynamic hydrographic region and the associated high productivity clearly supports a very dense and rich cetacean fauna.

The habitat associations of species in this study are consistent with findings in other areas of the world's oceans. For example, Risso's dolphins were found to be associated with areas of high bathymetric slope in the northern Gulf of Mexico (Baumgartner, 1997; Baumgartner *et al.*, 2001) and in two areas of the Mediterranean Sea (Azellino *et al.*, 2008; Cañadas *et al.*, 2002). These authors note that Risso's dolphin and other species associated with strong bathymetric slope primarily feed upon squids. Similarly, pilot whales (both long-finned and short-finned) are typically grouped with Risso's dolphins and other shelf-break associated species, as they are in our study, again presumably associated with a preference for squid prey.

The inclusion of bottlenose dolphins in this group is interesting as it suggests that perhaps they are also primarily squid predators in this habitat. The bottlenose dolphins encountered during this survey are certainly of the more pelagic 'offshore' morphotype as opposed to the 'coastal' morphotype that occurs in near shore continental shelf and estuarine waters (Torres *et al.*, 2003). These offshore animals appear to be more adapted to longer, deeper duration dives than coastal animals based upon hemoglobin profiles (Hersh and Duffield, 1990). This is consistent with the characterisation of the deep diving shelf-break associated species that consume primarily squids in pelagic habitats. Most diet studies characterise bottlenose dolphins as primarily piscivorous (e.g. Barros and Odell, 1990; Kenny *et al.*, 1995), but many of these previous studies are from animals inhabiting coastal habitats. The remaining abundant delphinids including common dolphins, Atlantic spotted dolphins and striped dolphins, are thought to be opportunistic piscivores with a diverse diet including small mesopelagic and pelagic fishes (Kenny *et al.*, 1995; Pauly *et al.*, 1998; Young and Cockcroft, 1994).

Interestingly, in some studies (e.g. Baumgartner *et al.*, 2001; Cañadas *et al.*, 2002; Davis *et al.*, 1998) sperm whales are also grouped with the shelf-break species, presumably associated with the similar dependence on squid prey. However, in our study, and similarly in Hamazaki (2002), sperm whale habitats were not as strongly associated with the shelf break areas and were spread more broadly across the inner continental slope. This suggests segregation of habitat, and perhaps prey resources, among the teuthophagic species in this region. Alternatively, sperm whales may be exploiting less persistent areas of high productivity associated with Gulf Stream eddies and rings. Similar associations of sperm whales with mesoscale physical features have been observed with Loop current eddies in the northern Gulf of Mexico (Biggs *et al.*, 2005).

The 'cryptic species', beaked whales and *Kogia* sp., present a challenge for assessment using visual surveys (Barlow, 1999). Both taxa have long dive intervals and relatively short surface intervals and thus have limited availability for observation. Likewise, both groups are difficult to see at the surface and generally dive before they can be approached by the vessel. Hence, sighting conditions and other factors strongly influence the number and types of these species encountered. It is likely that our characterisations of their spatial distribution and habitat preferences are incomplete. For example, during the summer 1998 survey, there was a greater number of *Kogia* sp. sightings in the southern offshore portion of the survey area (Mullin and Fulling, 2003), and *Kogia* are regularly seen stranded along the southeastern US coastline (Waring *et al.*, 2007).

The majority of beaked whales encountered in this survey were in the southern portion of the survey range. We had very few beaked whale sightings north of Cape Hatteras, and those occurred only in deeper water. Beaked whales are well documented to occur north of our survey area along the southern flank of Georges Bank and along the shelf break off the coast of southern New England (Kenny and Winn, 1987; Waring *et al.*, 2001). However, in Waring *et al.* (2001) there was a lack of sightings along the mid-Atlantic shelf break south of approximately 38ºN latitude, which is the northern extent of our survey. These data do suggest that there is a discontinuity in the spatial distribution of beaked whales in the mid-Atlantic region which may correspond to a faunal break in the distribution of this diverse species group which potentially includes four species of the genus *Mesoplodon* along with *Ziphius cavirostris* (Cuvier's beaked whale; Waring *et al.*, 2007).

This study demonstrates that groups of cetaceans are closely associated with distinct hydrographic regimes over broad spatial scales. The separation between continental shelf and slope water masses represent transitions in the underlying physical characteristics of the water column, changes in the composition and density of the prey field, and hence changes in the composition of the cetacean community. The convergence of water masses along the shelf-break in the mid-Atlantic and at Cape Hatteras results in localised increases in the diversity of the cetacean community. The increased surface water primary and secondary production at these boundaries results in very high densities of cetaceans. Evaluating finer scale physical variability and the associated variation in the species composition and density of cetacean species will further improve the understanding of the habitat associations, partitioning of prey resources, and spatial distribution of cetacean communities.

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Abundance estimate of striped dolphins (*Stenella coeruleoalba*) in the Pelagos Sanctuary (NW Mediterranean Sea) by means of line transect survey

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ABSTRACT

To assess cetacean densities in the Pelagos Sanctuary for Mediterranean Marine Mammals, a Marine Protected Area (MPA) specifically designated to protect cetaceans, a survey was carried out in the Ligurian-Provencal Basin (NW Mediterranean) in August 2008. An area of 58,000 km^2 was surveyed in eight days with equally spaced zigzag transects, covering 1,255 km in favourable conditions. Tracklines were designed using Distance 5.0 to allow for homogeneous coverage probability over the selected area. Fifty three sightings of four cetacean species were made: striped dolphins ($n = 37$), fin whales ($n = 12$), sperm whales ($n = 3$) and Cuvier's beaked whales ($n = 1$). Estimates of abundance were obtained using Distance 5.0. The estimated dolphin abundance was 13,232 (CV = 35.55; 95% CI = 6,640–26,368), with a density of 0.23 individuals km^{-1} (CV = 35.55; 95% CI = 0.11–0.45). No fin whale abundance estimate was possible due to the small sample size. The point estimate of the 2008 striped dolphin abundance estimate was almost half of that of a survey conducted in 1992 by Forcada and colleagues (1995) in the same area with comparable effort, platform and methodology (25,614; CV = 25.3; 95% CI = 15,377–42,658); nevertheless, the difference was not statistically significant. These results strongly support the need for further systematic monitoring in the Sanctuary and in the surrounding areas, in order to assess striped dolphin abundance, spatial and temporal trends.

KEY WORDS: ABUNDANCE ESTIMATE; SURVEY-VESSEL; CONSERVATION; EUROPE; STRIPED DOLPHIN

INTRODUCTION

The Pelagos Sanctuary for Mediterranean Marine Mammals, is the world's first high-seas Marine Protected Area (MPA) (Hoyt, 2005). It was established by Italy, France and Monaco in 1999, after a long process that recognised the high productivity of the area, and its unusual cetacean concentrations (Notabartolo di Sciara *et al.*, 2008; 2003). The 87,500km^2 of the Pelagos Sanctuary covers both pelagic and neritic regions, representing areas suitable both for breeding and foraging needs of many of the cetacean species found in the Western Mediterranean Sea (Notabartolo di Sciara *et al.*, 2008). Among these, fin whales (*Balaenoptera physalus*) and striped dolphins (*Stenella coeruleoalba*) are the most common species regularly present in the Pelagos Sanctuary (Forcada and Hammond, 1998).

The area is subjected to a number of potentially severe anthropogenic factors: the recreational importance of the Pelagos coastal regions is responsible for strong tourism pressure and high concentrations of pleasure boats during summer. These elements, coupled with coastal run off and sewage, chemical pollution, ferries and merchant traffic may represent important threats for the biological features of the area (Fossi and Lauriano, 2008; Fossi *et al.*, 2003; Panigada *et al.*, 2008; Panigada *et al.*, 2006). Despite the importance of the region for cetaceans' presence and the management and conservation issues related to the existence of an MPA with such high levels of human pressure, no regular cetacean monitoring programmes have been planned for the Sanctuary. However, recently, the Italian Ministry of the Environment has funded a series of research programmes in order to monitor cetacean presence and abundance in the seas around Italy, and also the whole Pelagos Sanctuary.

Striped dolphins' abundance in the Corso-Ligurian Basin was previously estimated with a line transect survey carried out during summer 1992 (Forcada *et al.*, 1995). Abundance was estimated as 25,614 (CV = 25.34; 95% CI = 15,377–42,658) retrospectively representing the first striped dolphin abundance estimate for the Pelagos Sanctuary. The results were believed to show the relatively good status of striped dolphins after the mass mortality due to morbillivirus in the 1990–92 period (Aguilar and Raga, 1993). Other abundance estimates for the summers of 1996 (Gannier, 1998) and 2001 (Gannier, 2006) in the region provided abundance estimates similar to those reported by Forcada *et al.* (1995); however, differences in area and survey procedures, design and platform, do not allow for a proper comparison.

The general level of habitat degradation over the last 20 years, in addition to direct impacts including disease and bycatch may have negatively impacted the population (Reeves and Notarbartolo di Sciara, 2006). Information on striped dolphin abundance is therefore urgently needed to assess current population status and highlight potential temporal and spatial shifts in distribution. This paper presents information on abundance and densities of striped dolphins in the western portion of the Pelagos Sanctuary, obtained through ship-based visual line transect sampling.

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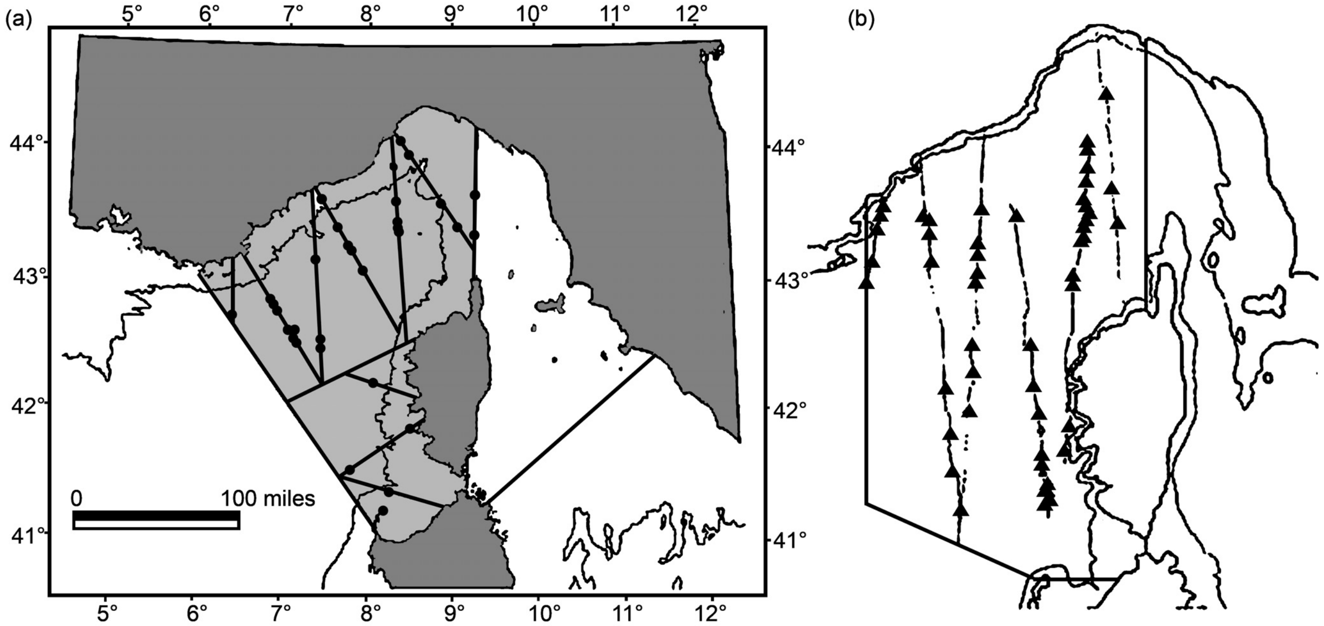

Fig. 1. (a) The study area and the distribution of the cetacean encounters with the tracks lines. (b) The area investigated in 1992 with the transects and the striped dolphin encounters (from Forcada *et al.,* 1995 modified).

MATERIALS AND METHODS

Survey design

The study area was located in the northwestern part of the Ligurian Sea and encompassed 58,000 km^2 of the Pelagos Sanctuary (Fig. 1). The survey design took into account the previous experience of Forcada *et al.* (1995) using a similar platform and comparable methodology. The survey design was selected using the software Distance 5.0 (*http://www.ruwpa.st-and.ac.uk/distance/*; Thomas *et al.*, 2007), thus allowing equal coverage probability.

The time spent at sea was dictated by the available ship time and logistics. The design class was ‘equal spaced zig-zag’ and the study area was divided into two strata of 15,916 (stratum 1) and 42,013 (stratum 2) km^2 respectively, in order to optimise the expected variability in cetacean density between strata and to minimise variability within-stratum (Thomas *et al.*, 2007). The vessel used was the 54m *Arctic Sunrise* provided by Greenpeace International; survey speed was set between 8 and 10 knots (15 and 18.5km h^{-1} respectively). The observation platform was set at 8m above sea level on the main deck, being the highest accessible area for the observers’ team. The observation team consisted of three persons (at least one with specific previous experience in visual surveys); the port and starboard observers searched (with naked eyes) a sector from the trackline to 90°, while the third person was involved in data entry in a laptop computer. Observer teams rotated every 90 minutes. Once a cetacean group was sighted, 7 × 50 binoculars were used to identify species and assess group size. Primary effort (on effort) was maintained under defined conditions of ≤ 3 on the Beaufort scale. The radial angle from the track line to the school was measured with an angle board (Buckland *et al.*, 2001) mounted on the deck fence; the distance was estimated with measuring sticks, following the protocol used for Scans II (SCANS-II, 2008). Sighting data such as radial angle, distance, species and school size estimate were collected at the beginning of the sighting; in order to maximise time on effort, passing mode was used (Dawson *et al.*, 2008), i.e. the vessel did not close with sightings.

Schools sighted while off effort (sea state >3 on the Beaufort scale), were not considered in the density and abundance estimates. Geographical positions were registered with a Global Positioning System (GPS) connected to the computer, equipped with the Logger2000 software[3]. The GPS was set to register position each minute, the computer operator entered navigation data every 15 min and/or every time a change in conditions (i.e. weather, ship speed, course, sighting conditions, on and off effort) occurred.

Data analysis

Given the relatively low number of sightings and thus information on their associated variables, only Conventional Distance Sampling (CDS) could be used to analyse the data (Thomas *et al.*, 2007). Although sightings of all cetacean species seen were recorded, it was possible only to produce abundance estimates for striped dolphins. Different detection functions, given by the combination of the uniform and half normal key functions and the cosine expansion term, were fitted to the data, and the model with the smallest Akaike’s Information Criteria (AIC) values was selected. Responsive movements of the striped dolphin schools have also been taken into account, considering the Q3/Q1 ratio described in Palka and Hammond (2001).

RESULTS

An area of 58,000 km^2 was surveyed in eight consecutive days (3–10 August), with a total of 1,255km of the planned 1,370km covered under favourable conditions (91.6%). A total of 53 sightings of four cetacean species were made (Table 1).

Striped dolphins were found in the offshore area, in both strata. Thirty four out of the 37 striped dolphin sightings were primary sightings and have been used for the

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[3] Logger 2000, *http://www.ifaw.org.*

abundance estimate. Since only three such sightings occurred in stratum 2, abundance estimates were obtained by pooling the strata. The size of the dolphin schools observed ranged from 1 to 35 (mean 7.51, SD = 7.40); the frequency distribution of all the striped dolphin sightings is shown in Fig. 2. In order to estimate the detection function (Fig. 3), sightings were truncated at a perpendicular distance of 800m. From AIC, the best model was a half normal function with cosine adjustment terms. Group size was estimated by regressing the natural log of group size against estimated detection probability (Thomas *et al.*, 2006).

The estimates for the relevant parameters for striped dolphins are given in Table 2. The ratio Q3/Q1 was less than 1 (0.66) suggesting avoidance rather than attraction; nevertheless the ratio was not statistically significant ($\chi^2 = 0.80$, df = 1, $P>0.05$). The total abundance in the surveyed area was 13,232 (CV = 35.6; 95% CI = 6,640–26,368).

Table 1

Summary of species sighted, group size and composition.

Species	*n*	Mean group size [range]
Stenella coeruleoalba	37	7,51 ± 7,396 [1–35]
Balaenoptera physalus	12	1,08 ± 0,288 [1–2]
Physeter macrocephalus	3	1
Ziphius cavirostris	1	1

Table 2

Estimates for striped dolphins.

Sample size:	10 transects	33 encounters		
Estimated parameters:	Point estimate	%CV	LCI	UCI
P	0.404	17.92	0.28	0.58
ESW	324.7	17.92	225.94	466.62
ER	0.026	15.84	0.18	0.37
E(s)	5.64	26.29	3.33	9.56
DS	0.40	23.92	0.25	0.65
D	0.228	35.55	0.11	0.45
N	13,232	35.55	6,640	26,368

CV = coefficient of variation; LCI and UCI = lower and upper 95% confidence intervals; P = probability of observing a dolphin in a defined area; ESW = effective strip width (m); ER = encounter rate (N/L); E(S) = estimated of expected value of group size; DS = estimate of density of groups; D = estimate density of animals (numbers/km^2); *N* = estimate of abundance.

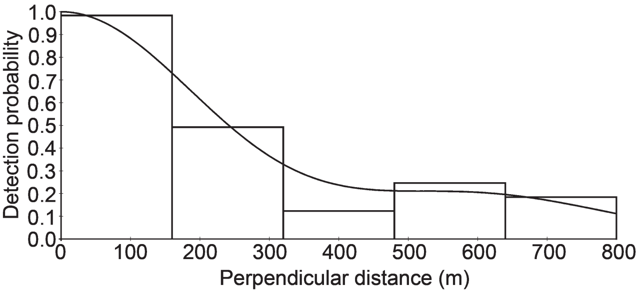

Fig. 3. Detection function with a right truncation at 800m.

DISCUSSION

This study provides an estimate of striped dolphin abundance in the western portion of the Pelagos Sanctuary, 16 years after the first estimate made in 1992 (Forcada *et al.*, 1995). To the extent possible, the recent survey was carried out within the same area, at the same time of the year and used similar methods.

The small number (*n* = 12) of fin whale sightings during the present study precluded estimation of abundance. While this prevents quantitative comparison, the low number is in accord with a general suggestion of reduced fin whale sightings given by whale watching operators in the area. The 1992 fin whale estimate (Forcada *et al.*, 1995) was 901 (CV = 21.8; 95% CI = 591–1,374). Considering that the surveyed area is believed to be one of the major summer feeding grounds for this species in the Mediterranean Sea (Notarbartolo di Sciara *et al.*, 2003; Panigada *et al.*, 2005), the lack of sightings raises some concern.

With respect to striped dolphins, the density estimate of 0.22 dolphins km^{-2} (CV = 35.6; 95% CI = 0.11–0.45) is lower than that of 0.4km^{-2} (CV = 25.3; 95% CI = 0.26–0.73) in the same area presented by Forcada *et al.* (1995). It is also lower than those reported from similar surveys by Forcada and Hammond (1998) for the Ligurian Sea (D = 0.3; CV = 35) and for the Ligurian-Provençal Basin (D = 0.24; CV = 26), although the areas are not identical. It is also lower than estimates provided using quite different survey methods in the Ligurian-Provençal basin by Gannier (1998).

Although the results clearly suggest a decrease in abundance/density in the region between 1992 and 2008, caution is needed when interpreting differences between the 1992 and 2008 density and abundance estimates.

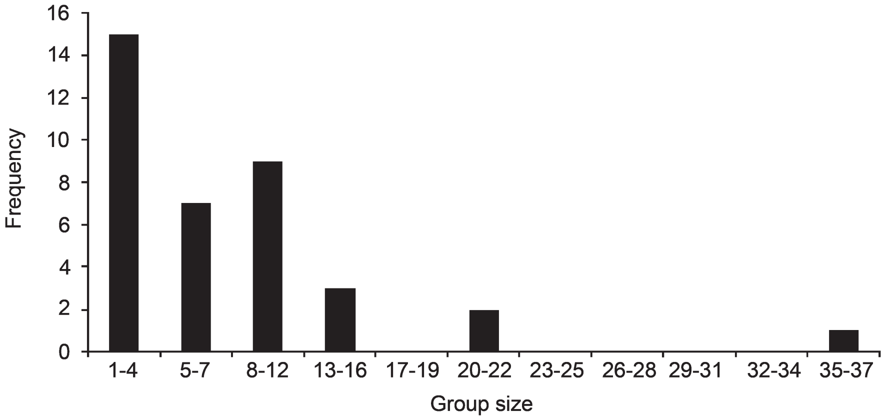

Fig. 2. Frequency distribution of all the striped dolphin groups.

Perhaps the most important general factor relates to the question of stock structure, movements and distribution. The 1992 and 2008 surveys provided abundance estimates for an area which represents just a portion of the striped dolphin distribution range in the western Mediterranean Sea; therefore, these 'local' estimates are subject to the natural temporal and spatial fluctuations in the density distribution of the animals within their full range. Geophysical parameters responsible for the high productivity of the Pelagos Sanctuary, one of the most productive pelagic areas in the whole Mediterranean basin (D'Ortenzio and Ribera d'Alcala, 2008) are of relevance here. For example, studies on the seasonal and inter-annual variability of chlorophyll concentrations *(chl-a)* within the north-western portion of the Pelagos Sanctuary from 1997 to 2004 (Finoia *et al.*, 2007; Manca Zeichen *et al.*, 2008; Notabartolo di Sciara *et al.*, 2008) showed a decrease of the phytoplankton spring bloom patch visible from satellites up to 2003, with the exception of 1999. These analyses showed that the bloom drop, along with a significant reduction of *chl-a* values from 1997 to 2003, might have influenced cetacean food availability, causing their displacement towards the west (i.e. the Gulf of Lyons) – where the phytoplankton bloom is recurrent – with a consequent population decrease within the Pelagos area.

It is clear therefore, that without better information on population structure, such estimates as presented here, even if correct, cannot be used to estimate population level trends.

More specifically with respect to the survey estimates themselves, although the present study was intended to replicate as much as possible the 1992 survey (e.g. ship characteristics, survey speed, area, time of the year and methods), some differences were inevitable given financial and logistical constraints. For example, distance measurements were dissimilar in the two surveys. During the first survey, distance was estimated and corrected based on distance estimation experiments; during the latter, partly to maximise survey time, it was measured using measuring sticks but no experiments to correct potential errors in use were carried out. The potential for bias cannot therefore be evaluated. Similarly, to maximise effort passing mode was employed (it also has some other theoretical benefits) in 2008 whereas the 1992 survey used closing mode. However, group size tends to be underestimated at greater distances (Dawson *et al.*, 2008); if group sizes were underestimated in 2008 then the resultant abundance and density estimates would be negatively biased.

Neither the 1992 or 2008 surveys collected data (e.g. double platform data) (Buckland *et al.*, 2004; Hammond *et al.*, 2002) to allow correction for availability bias (animals may be underwater and not available to be seen) or perception bias (for a number of reasons, observers do not see animals when they are available to be seen). Thus, the most important assumption of line transect surveys, that the probability of seeing animals on the trackline is one (Buckland *et al.*, 2001) could not be assessed; however, the probabilities may well have been different between the two surveys. Similarly, insufficient data were available to adequately address the possibility of differences in the levels of responsive movements between the two surveys. Considering all of the above mentioned issues, strict comparisons are not possible and thus unequivocal conclusions about trends cannot be made.

That being said, during the time between the two surveys, authors have drawn attention to several threats (Aguilar, 2000; Notabartolo di Sciara *et al.*, 2008) that may have had a negative effect on the striped dolphin population in the Mediterranean; a recent Red List assessment proposed that *S. coeruleoalba* in the Mediterranean be considered Vulnerable (Reeves and Notarbartolo di Sciara, 2006).

Disease is one such factor. The morbillivirus epizootic that occurred from 1990 to 1992, for example has been postulated to have perhaps reduced the population abundance to one third of its original level (Aguilar, 2000); in early July 2007 morbillivirus again hit striped dolphins in the Gulf of Valencia (Raga *et al.*, 2008). Whether this recent occurrence was due to the permanence of the virus in the Mediterranean specimens or to a periodic re-entrance (Di Guardo *et al.*, 2009), this, and/or the presence of *Toxoplasma gondii*, which have been reported as a cause of death for striped dolphins in the Mediterranean Sea (Di Guardo *et al.*, 2009) might have had a negative impact on the striped dolphin population. Related to this, toxicological stress was recognised as significant in the 1990–92 die off (Raga *et al.*, 2008) and the exposure to contaminants (organochlorines and PCBs) can negatively affect endocrine functions and reproduction in some marine mammals (Fossi *et al.*, 2003).

More directly, Mediterranean striped dolphins have suffered from high levels of mortality due to incidental capture in fishing gear, leading to the overall declaration of the level of bycatch in pelagic driftnets in the Mediterranean Sea as unsustainable (Perrin *et al.*, 1994). High bycatch rates were reported in all the Mediterranean Sea in the 1990s and despite the European Union driftnets ban since 2001 (Council Regulation n° 1239/98), illegal driftnetting was recently reported within the Pelagos Sanctuary where conventional and/or modified nets targeting tuna-like fish are regularly deployed (Cornax *et al.*, 2006; 2007).

Despite the lack of reliable quantitative information on bycatch levels of striped dolphins in the Mediterranean, there is a general consensus (e.g. Bjørge and Donovan, 1995) in assuming anthropogenic removal levels exceeding 1% of the estimated population size, as a cause of concern. It is not unlikely that such bycatch levels occurred in the Pelagos Sanctuary and surroundings areas, according to the estimate inferred from the Spanish driftnet fishery (Forcada and Hammond, 1998) and from the Moroccan fleet (Tudela *et al.*, 2005).

Despite the uncertainty, the above considerations suggest that the striped dolphins' abundance may have changed in the surveyed area and perhaps at a population level.

It is clear that to properly address the conservation of striped dolphins in the Mediterranean and within the Sanctuary, a vital component is a comprehensive, well-designed monitoring programme (e.g. see the ACCOBAMS[4] survey initiative). In such a context, the recent commitment by the Italian Government to promote systematic monitoring is particularly timely and welcome.

[4] *http://www.accobams.org/index.php?option=com_ontent&view=article&id=1090&Itemid=76.*

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Effect of tourist vessels on the behaviour of the pantropical spotted dolphin, *Stenella attenuata*, in Drake Bay and Caño Island, Costa Rica

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ABSTRACT

Despite the exponential increase in whalewatching activities in Costa Rica, little is known about its biological impact on resident coastal populations of dolphins in the country. Globally, this activity has brought economic benefits to the communities where it is practiced and in some cases, has played an important role in conservation of these mammals. However, when intensively practiced, this activity may significantly affect the animals, since its success depends on following cetaceans for extended periods of time. This study was conducted during the 2004–2005 and 2005–2006 dry seasons, to examine the biological factors associated with this activity in two areas where it is intensively practiced: Drake Bay and Caño Island. Three strip transects were followed within a high (vessel) traffic area. The pantropical spotted dolphin was studied through instant sampling, every two minutes. Sighting density of dolphins accompanied by tourist boats was greater within 3km of the island compared to the average density in the whole study area. Dolphins reacted negatively to those boats that did not follow at least one of the rules of boat handling in the current existing national regulation for whalewatching guidelines. Furthermore, a logistic regression analysis showed that feeding and resting are less likely to occur in the presence of tourist boats. These two behaviours are extremely important and mishandled boats could cause the spotted dolphin to leave this area if these flaws continue. Due to the lack of economic resources and staff from state institutions in Costa Rica, the reinforcement of the Whalewatching Executive Decree 32495 (2005) may be more efficient with 'bottom up' control, where community representatives control their own resources in conjunction with government oversight.

KEYWORDS: WHALEWATCHING; MANAGEMENT; SPOTTED DOLPHIN; BEHAVIOUR; TOURISM; REGULATION

INTRODUCTION

Whalewatching can be defined as any commercial enterprise which provides for the public to see cetaceans in their natural habitat (IWC, 1994). Since about 70% of the global whalewatching activity is done from boats (e.g. Hoyt and Hvenegaard, 2002), an accelerated growth of a tourism activity such as this, is inevitably accompanied by an increase in boat traffic, which in turn is associated with a variety of economic impacts on communities as well as biological impacts on the animals involved (Luck, 2003). Whalewatching, although a non-lethal use of cetaceans (e.g. IWC, 1994), may have negative impacts on the behaviour and health of the populations of these mammals (e.g. Bejder *et al.*, 1999; Bejder *et al.*, 2006; Blane and Jaakson, 1994; Constantine, 2001; Constantine *et al.*, 2004; Corkeron, 1995).

In Costa Rica, commercial whalewatching began in 1990, and remained low until 1998 (Hoyt, 2001). It was not until after that date that an expansion of hotels and tourist activities (including whalewatching) occurred in Drake Bay or 'Bahía Drake' (Rasmussen *et al.*, 2002). However, the marked seasonal climate of this area has led to the intensification of this activity during the dry season (December to April), due to favourable weather conditions. Today, 11 different species of cetaceans are known to occur in Drake Bay (May-Collado *et al.*, 2005; Rasmussen *et al.*, 2002), some considered indisputable target species for whalewatching because of their predictable behaviour and proximity to the coast. These species include the most common dolphin in the area, the pantropical spotted dolphin (*Stenella attenuata*). The objective of this research was to determine if there is an impact of tourism vessels on the behaviour of the pantropical spotted dolphins found in Costa Rica's Southern Pacific.

METHODS

Field methods

Systematic surveys were conducted between Drake Bay and Caño Island, during the summer of 2004–2005 and 2005–2006 following three strip transects in a triangle (Fig. 1). Strip transects were of variable length (29, 25 and 23km) and 1000m wide (500m on each side of the trackline), and were followed for three or four consecutive days every month. Sampling of the area was performed continuously throughout the day, from 6:00hrs to 15:00hrs, alternating the start location each day.

The study area was selected based on its proximity to national protected areas and tourist attractions. Drake Bay is located in the South Pacific Region of Costa Rica. The total sampling area included approximately 160km^2 from Punta Sierpe (8º46'N, 83º39'W) to Punta Llorona (8º38'N, 83º44'W) in the Osa Peninsula, including the Caño Island (8º71'N, 83º89'W). The study area is adjacent to three major tourist destinations in the region: the Térraba-Sierpe Mangrove Wetland, Corcovado National Park and Caño Island Biological Reserve. The marine area within these sites represents rocky reefs and coral communities of high diversity (Guzmán and Cortes, 1989).

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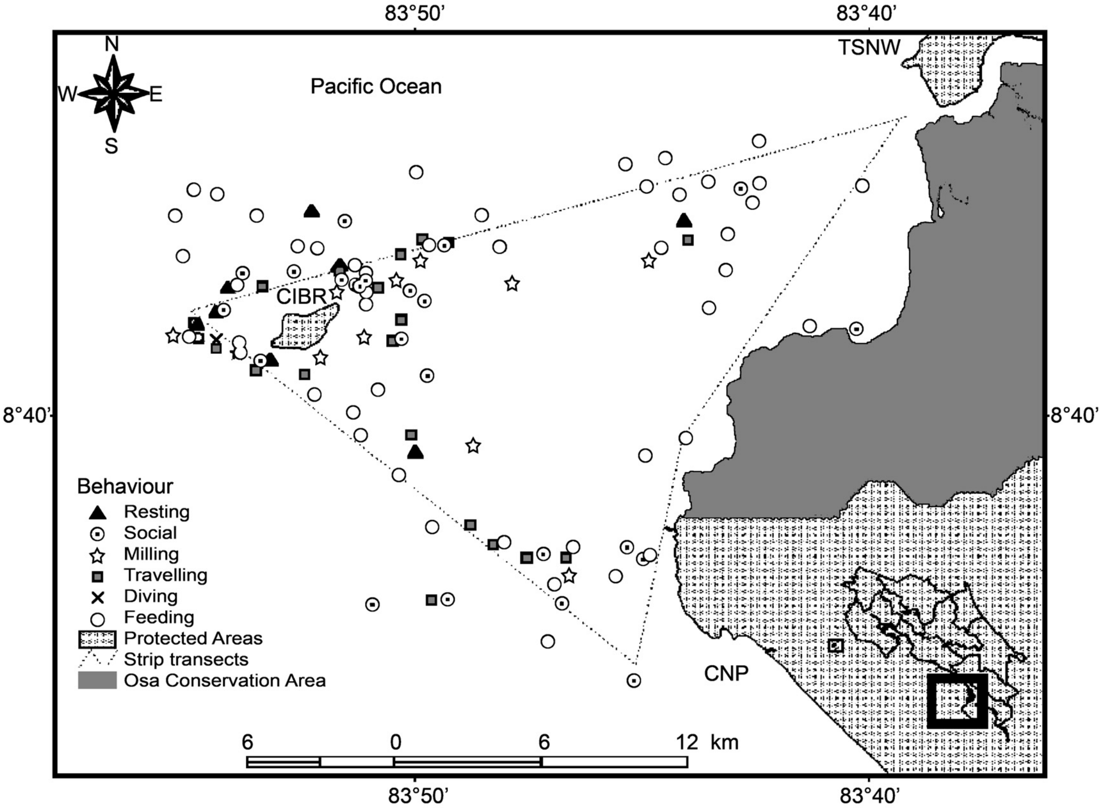

Fig. 1. Location of study site for spotted dolphins' behaviour (2004–2006). Averaged strip transects are joined by Terraba Sierpe National Wetland (TSNW), Caño Island Biological Reserve (CIBR) and Corcovado National Park (CNP) in Costa Rica. Locations of dolphin schools monitored are shown based on the dominant behaviour of each sighting.

An independent research boat (7m, outboard, 4 stroke engine) followed the transects during the two dry seasons between December 2004 and April 2006. The research boat was manoeuvred in order to minimise its potential effects on the dolphins' behaviour. Suggestions on appropriate boat manoeuvre were followed after Constantine *et al.* (2004) and the 32495 Executive Decree[1]. Despite these precautions, the presence of the research boat was also considered as a potential disturbance factor. Whenever a school of spotted dolphins was encountered, data on school size and age-class composition were collected. A focal-follow (see next section) observation was then initiated and data on the number of boats within 300m of the school and the dolphins' behavioural state were collected.

For analysis purposes, observations were divided into 'controlled observations' in presence of the research boat only and 'tourism observations' when one or more boats (besides the research boat) were present in a 300m radius watching the dolphins. Distance measurements were undertaken with the aid of a laser rangefinder (Bushnell Yardage Pro 500).

Behaviour

The dolphins' behavioural state was determined by using a 2-minute focal-school scan sampling methodology and assigning a predominant school activity (Altmann, 1974). An encounter was finalised when reaching a 40 minute limit or whenever the group was lost. A school consisted of any number of dolphins in apparent association, moving in a similar direction and often engaged in similar behaviours (Wells *et al.*, 1999).

The dolphins' behavioural states were assigned to one of the five categories detailed in Table 1, modelled on preliminary observations and adaptations from Shane (1990), Lusseau (2003) and May-Collado and Morales-Ramírez (2005).

Based on articles 13 and 16 of the 32495 Executive Decree (2005, Appendix I)[1], boat manoeuvring of sighted whalewatching vessels was categorised as 'Correct' when all regulations were followed and 'Incorrect' if the vessel: (1) was closer than 50m from the group with the engine running; (2) was closer than 100m when the dolphins were feeding or socialising; (3) remained more than 30 minutes with the same group; (4) interrupted the course of the group; (5) had passengers throwing food, liquid or waste into the sea; (6) generated excessive noise within 100m of the group; (7) approached animals from the front or perpendicular; (8) drove faster than the slowest-swimming dolphin; (9) remained in the place despite disturbance signs; (10) when it was the third vessel to arrive, did not respect the 200m distance from the first 2 boats; and/or (11) had passengers swimming with dolphins.

Furthermore, school reactions were assigned based on previously reported behaviours that were considered stress reactions signs for different cetaceans species in presence of vessels (Baker and Herman, 1989; Berggren, 2001; Blane and Jaakson, 1994; Constantine and Baker, 1997; Corkeron, 1995; Kruse, 1991; Nowacek *et al.*, 2001; Richter *et al.*, 2001; Williams *et al.*, 2002) as follows: avoidance; change of

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[1] Decreto Ejecutivo Nº 32495-MINAE-MOPT-MSP-MAG. 2005. 'Reglamento para la Operación de Actividades Relacionadas con Cetáceos en Costa Rica'. Publicado en La Gaceta, Número 145 del 28 de julio del 2005. 6pp.

Table 1

Definitions of the behavioural states of the spotted dolphin schools during dry seasons (2004–06) in Drake Bay and Caño Island.

Feeding (FEED)

(1) Foraging: individuals dive synchronously for long intervals (2 mins). Steep dives finalising in fluke exposition are observed. Dives are often performed in a common concentric point. Group spacing and direction of movement varies.

(2) Hunting: individuals consuming prey by surface persecutions and circular fast swimming (not following another dolphin). This state involves encircling, aerial behaviour and direct prey catch.

Socialising (SO)

(1) Boat interaction: dolphins voluntarily approach a boat and show boat-riding or 'inspect' the vessel. Swim, get close.

(2) Interactions among individuals of the same group or between groups, manifested by persecutions, rubs, sexual contact, mother-calf interactions and aerial behaviour.

Travelling (TRAV)

Individuals moving faster than the idle speed of the research boat with constant direction, swimming with short, relatively constant dive intervals. Group spacing varies.

Resting (REST)

Individuals moving slower than the idle speed of the research boat with constant direction, swimming with short (<1 min.), relatively constant synchronous dive intervals. Animals are tightly grouped.

Milling (MILL)

Individuals surfacing facing different directions. No net movement. School often changes direction, dive intervals variable but short. Group spacing varies.

Diving (DIVE)

Individuals dive synchronously for longer intervals than previously observed within the observation. Often observed as a potential evasive reaction of the group. Direction of movement varies.

behaviour; increased dive intervals; leaps and tail slapping; change in direction; and increased swimming speed.

Analyses

Chi-squared (χ^2) approximations (PAST version 1.67b; Hammer *et al.*, 2001), were used to compare different behavioural categories and behavioural reactions to boats; in both cases, selected α was <0.05. To estimate tourism boats density in the study area, *Animal Movement* extension (Hooge and Eichenlaub, 1997) from *Arc*View GIS 3.3, (ESRI, 1998) was used within a sampling area of 350km^2. The specified search ratio for each circle was 2km.

Behavioural sampling units consisted of 2-minute scans. In order to compare the relationship between type of boats present (controlled or tourism) and behaviour observed in each school, a logistic regression of binomial response for each behaviour (SAS Institute, 2000, GENMOD procedure with repeated measures) was used. This generalised linear model (GLM) allows an evaluation of the probability of sighting a school in a certain behaviour based on the type of boat present. The *Link function* was a *Logit* function compatible for binomial responses.

The initial autocorrelation due to repeated measures throughout time (2 minute-observation periods within one sighting) was corrected by the repeated measures design. All scans belonging to the same sighting were specified as repeated measures within the model. Rejection criteria for each Ho was based on odds ratio estimates for each regression coefficient and 95% Wald Confidence Limits.

RESULTS

Forty eight hours were spent on dolphin observations. The average observation lasted 19 minutes (SD = 13.7min) and a total of 1,452 2-minute scans samples were collected. Dolphins were accompanied by boats in addition to the research boat in 8% of scans (n = 120). A total of 1,332 controlled scans and 120 tourism scans were collected. Comparisons of dolphin behaviour proportions within controlled observations vs tourism observations showed a significant difference (χ^2 = 32.93, g.l. = 5, $p<0.001$). This suggests that the research boat could be considered as a suitable observation platform to contrast behaviour in the presence of boats other than a carefully driven research boat (Constantine *et al.*, 2004). This does not suggest that the research boat itself had no impact (Nowacek *et al.*, 2001) but that measured changes occurred above the effect of the research boat.

Tourism boats median closeness to dolphin groups was 50m, with any given behaviour the group showed at the time of getting closer to watch it. Almost 60% of the boats registered in whalewatching activities were observed between 8:30 and 9:30hrs. Tourism sightings density (boats/km²) was higher (= 0.17 ± 0.19) within 3km from Caño Island in comparison to average density (= 0.045 ± 0.1) in the whole surveyed area (Fig. 2). Groups showed more negative reactions in presence of vessels that showed an incorrect manoeuvring (Fig. 3; χ^2 = 4.96, g.l. = 1, p = 0.026). Types of reactions presented by the dolphins were, in descending order, the following: behavioural change (29%); superficial tail slapping (19%); increased diving intervals (19%); evasiveness (18%); change of direction (11%); and increased speed (4%).

A negative effect of tourism boats was observed over feeding (FEED) and resting (REST) behaviour of spotted dolphins in Drake Bay and Caño Island. Feeding behaviour is 4.7 times more likely to occur in presence of the research boat than in presence of the tourism boat (Odds ratio = 4.7; Table 2). Resting behaviour was observed exclusively when tourism boat was not present (Fig. 4). On the other side, socialising (SO) behaviour is 4.7 times more likely to occur in presence of tourism boats than in presence of the research boat. There were no significant differences between control and tourist boats for presence of Traveling (TRAV), Milling (MILL) and Diving (DIVE) behaviours.

DISCUSSION

The median distance closeness of boats to dolphin groups within the Bay (50m) suggests the distance established in the

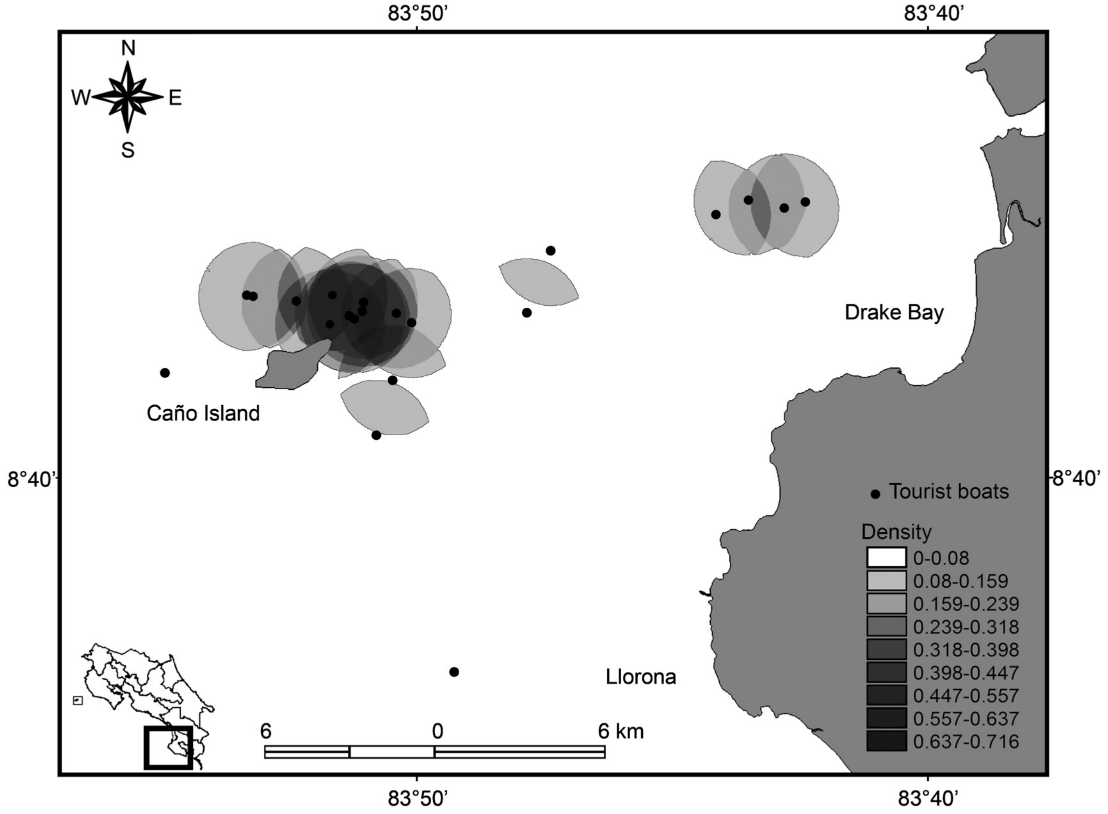

Fig. 2. Tourist boat density (boats/km^2) within study site during dry seasons of 2004–2005 and 2005–2006. Drake Bay and Caño Island, Costa Rica.

Decree ('50m as a minimum distance to the closest dolphin') is a viable instruction to be followed. Nevertheless, this distance measure was estimated including groups involved in feeding and socialising behaviours, in which case the vessels should remain at greater distances for being behaviours that directly affect cetacean group cohesion (Clark and Mangel, 1986; Emlen, 1991). Differentiating among behaviours requires training, so it is to be expected that most of the captains would not be able to identify the different behavioural categories for cetaceans.

Negative reactions from dolphins occurred mostly in the presence of those vessels with Incorrect manoeuvring (Fig. 3 and Appendix 1), which demonstrates the importance of following the regulations exposed in the national decree. The majority of these negative reactions are consistent with what has been cited by other researchers in different parts of the world and with different species of cetaceans (Baker and

Table 2

GLM results for pantropical spotted dolphin observed behaviours, as of present boat types during dry seasons 2004–06 in Drake Bay and Caño Island, Costa Rica.

Behaviour	Odds ratio*	Wald confidence limits 95%		Pr > |Z|
Feeding (FEED)	0.21	0.60	0.076	0.0035
Socializing (SO)	4.68	10.05	2.18	<.0001
Resting (REST)	2.9 E-11	–	–	0
Travelling (TRAV)	1.30	3.09	0.55	<.0001
Milling (MLLI)	0.88	3.15	0.24	0.015
Diving (DIVE)	2.42	8.29	0.71	<.0001

*Odds ratio<1 indicates a lower likelihood of a behaviour to occur in relation to the probability of this same behaviour without tourism boat(s).

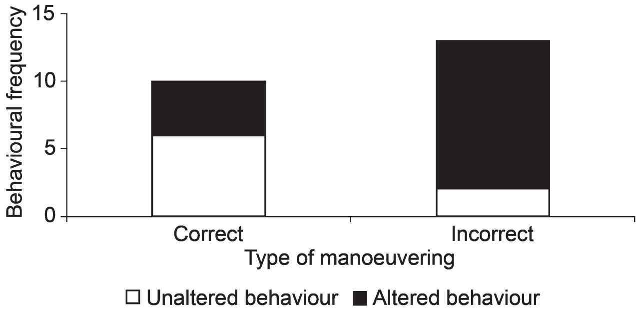

Fig. 3. Frequency of behavioural responses of the spotted dolphin depending on the type of tourist boat maneuvering within Drake Bay and Caño Island, Costa Rica (2004–2005 and 2005–2006). Altered behaviour = negative reaction.

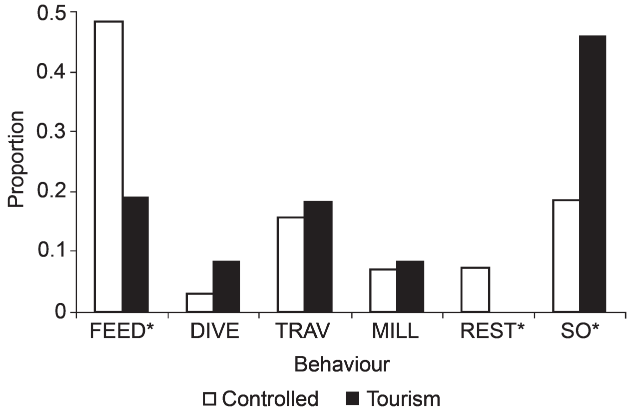

Fig. 4. Proportion of controlled and tourism observations in presence of distinct behaviours of the spotted dolphin in Drake Bay and Caño Island, Costa Rica (2004–2005 and 2005–2006). FEED = Feeding, DI = Diving, TR = Traveling, MI = Milling, RE = Resting and SO = Socialising, * = significant differences as reported by the GLM (95% confidence).

Herman, 1989; Blane and Jaakson, 1994; Constantine and Baker, 1997; Corkeron, 1995; Kruse, 1991; Nowacek *et al.*, 2001; Richter *et al.*, 2001; Williams *et al.*, 2002).

This study has shown that the presence of tourist boats had a negative effect on feeding and resting behaviour of spotted dolphins in Drake Bay and Caño Island. Feeding behaviour is more likely to occur in the presence of the research boat than in the presence of tourist boats. Similarly, Taubitz (2007) observed a trend in bottlenose dolphins to reduce foraging activities in the presence of boats. Few studies have addressed the effect of boats specifically with respect to foraging behaviour and feeding. Allen and Read (2000) found that bottlenose dolphin feeding use declined in primary habitat during high boat density periods and suggested that this may be due either to dolphins wanting to avoid high-traffic areas or in response to prey reaction to this high traffic (Engas *et al.*, 1995; Misund and Aglen, 1992; Mitson and Knudsen, 2003). Another possible explanation for this foraging decline is that the noise of the boat could be masking echolocation signs while dolphins hunt (Au, 2000). Montero-Cordero (2007) found no association between the behaviour of *S. attenuata* and time of day (study performed during the same period of time and within the same sampling area of this research). The latter discards any potential for time of day to be a factor responsible for some of the differences in dolphin behaviour.

It is a warning signal that resting behaviour in Drake Bay did not occur in any of the cases where tourist boats were present. Lusseau (2003) noted a decrease in the time occupied by bottlenose dolphins in resting behaviour in Fiordland, New Zealand. He reported 1% of this behaviour while tour boats were present, in contrast with 11% when only the research boat was present. In the case of Drake Bay, the daytime resting behaviour of spotted dolphins in the presence of exclusively the research boat, represent 8% of their daily activities (Montero-Cordero and Martinez-Fernandez unpublished data), compared to no resting at all in the presence of tourist boats. A decrease in this behaviour due to human disturbance was also observed by Constantine *et al.* (2004) for the bottlenose dolphin. Resting behaviour is fundamentally important for the health of various species of mammals (Bishop, 1999). The impact of reduced resting time for dolphins found in previous studies are still unknown, but other studies in other mammals have demonstrated physiological stress (Fowler, 1999; MacArthur *et al.*, 1979; Tietje and Ru, 1980). An overall reduction of resting will probably result in a reduction of energy reserves, which can affect foraging efficiency, alertness levels and parental care levels (Constantine *et al.*, 2004).

Socialising in dolphins implies visually conspicuous displays. This might partially explain the result of socialisation behaviour being ‘more likely to occur in presence of tourist boats’. Tourist boats will probably spot a group where individuals are jumping or approaching for bow riding (Table 1). Ransom (1998) noted that Atlantic spotted dolphins frequently approached tourist boats without interrupting their socialising behaviour. Nevertheless, several studies have reported cetaceans to increase group cohesion in the presence of vessels (Bejder *et al.*, 1999; Bejder *et al.*, 2006; Nowacek *et al.*, 2001) and in contexts of presumed surprise or threat (Whitehead and Glass, 1985).

Coastal dolphins (e.g. spotted dolphins) tend to live in discrete-area societies with relatively small home ranges, thus tourist boats’ disturbance becomes habitat degradation (Corkeron, 2004). Coastal spotted dolphins appear to maintain coastal populations in Golfo Dulce throughout the year and could be leaving and returning from this gulf with no substantial effort (Acevedo-Gutierrez and Burkhart, 1998; Cubero-Pardo, 1998)). The relative closeness of the Golfo Dulce with Drake Bay means that several of the dolphins found off the coast in Bahia Drake may be part of the resident population in the Gulf but this requires testing through dedicated photo-identification programmes. If it is the case, the same dolphins would be often exposed to repetitive harassment from boats in this area. This could reduce the biological adaptation of a stock when it occurs in the presence of critical behaviours such as diet, rest and play (Scheidat *et al.*, 2004). Moreover, Escorza-Treviño *et al.* (2002) clearly distinguished the genetic structure of spotted dolphins’ population of the Pacific of Costa Rica, when compared with stocks in the rest of Latin America. This suggests a discrete displacement area for this population, within Costa Rican waters. Despite the fact that the ‘tourism observations’ were considerably less than ‘controlled observations’, statistical analysis detected significant negative effects on the behaviour of dolphins, which indicates that the impact of the tourist boats is a real problem in the Drake Bay and Caño Island areas.

Management implications

Good practices in tourism activities also make good business sense, as improved performance can enhance a tour operator’s reputation and recognition in a tourism marketplace that is increasingly showing a preference for responsible products and suppliers (Sweeting, 2008). According to the IWC (2004), a number of options are available for managing the effects of whalewatching on cetaceans. These may be put into practice through regulations, permit conditions, codes of conduct, voluntary codes of practice or through targeted education programmes. Voluntary codes of conduct can be effective where there is good industry cooperation (International Whaling Commission, 2004). However, there are no whalewatching voluntary codes of conduct yet in Costa Rica, but a process has already been started (C. Molina, pers. comm.).

The limited success of regulations imposed by governments might be an indicator of resistance to ‘top-down’ controls on marine activities in rural areas. ‘Bottom up’ regulations, produced by local organisations and those actively involved in whalewatching, have been more accepted by operators in different countries (e.g. Parsons and Woods-Ballard, 2003) than top-down controls. In bottom-up management, operators are actively involved in monitoring and managing the ecotourism industry. By contrast, top-down systems control the anthropogenic use of natural resources through governmental laws or regulations. Environmental management studies in different parts of the world (e.g. Corbelli, 2007; Fraser *et al.*, 2006; Parsons and Woods-Ballard, 2003) have demonstrated a shift towards integrating participatory bottom-up approaches with conventional top-down systems. The latter integration resembles co-management, which implies the sharing of power and responsibility between the government and local resource users (Berkes, 2009).

For this specific case study, the duty of responsibility to monitor and control the responsible whalewatching, could be partially delegated to local tourism organisations or associations, for which scientific and proper legal advice should be provided. Nevertheless, beyond the local results presented here for Costa Rica (Central America), we recognise that the bottom-up approach has been effective in different socio-economic and environmental settings (Fraser *et al.*, 2006). Coastal communities around the world whose captains and guides possess a certain level of experience performing whalewatching activities, technical training and commitment to the environment, could be good candidates for a bottom-up approach.

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Appendix 1

Articles from Costa Rica Executive Decree Nº 32495 (2005).[1]

13.1 Do not get closer than 50 meters from the closest dolphin with the engine running and at least 30 meters with the engine off. Stay at least 100 meters apart from whales and cetaceans larger than 5 meters long.
13.2 Do not get closer than 100 meters from dolphins and 200 meters from whales and cetaceans larger than 5 meters long, when these show feeding or socialising behaviours.
13.4 Do not stay longer than 30 minutes with the same group of cetaceans, even when respecting indicated distances.
13.5 Do not stay longer than 15 minutes with mother-calf couples or solitary individuals and stay at least 100 meters apart from dolphins and 150 meters apart from whales and cetaceans larger than 5 meters long.
13.6 Do not interrupt cetaceans' routes by putting the boat in between animals or splitting a group.
13.7 Do not feed any cetacean species.
13.8 Do not generate excessive noise, like music, any kind of percussion, including noises generated by the engine, at less than 100 meters apart of any cetacean.
13.11 Do not throw any kind of waste, substance or material in watching or conservation areas, taking into account the other regulations about waste deposition in the ocean.
16.1 The dolphin or whale watching approach to moving animals should be from behind and slightly to one side coming from the same direction of the group. Vessels should advance by driving parallel to cetaceans´ traveling direction.
16.2 In presence of cetaceans, vessels should drive slower than the slowest-swimming animal from the group observed.
16.3 Leave the place at low speed if cetaceans show alteration signs.
16.4 A maximum number of 2 vessels around the same cetacean group are allowed. Any other vessel should keep a 200 meter distance apart from the first vessels.
16.5 In presence of solitary individuals or mother-calf couples, vessels should not approach to a closer distance than 100 meters.
16.6 Do not practice activities related to possession, fishing, diving or swimming, aquatic ski, 'jet-ski' or aquatic motorcycles, 'wind-surf', oars, canoes or kayaks in presence of cetaceans.

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[1] Decreto Ejecutivo Nº 32495-MINAE-MOPT-MSP-MAG. 2005. 'Reglamento para la Operación de Actividades Relacionadas con Cetáceos en Costa Rica'. Publicado en La Gaceta, Número 145 del 28 de julio del 2005. 6pp.

Subsistence harvest of beluga or white whales (*Delphinapterus leucas*) in northern and western Alaska, 1987–2006

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ABSTRACT

Four stocks of beluga or white whales (*Delphinapterus leucas)* are hunted by Alaska Natives in northern and western Alaska. These are the Beaufort Sea, eastern Chukchi Sea, eastern Bering Sea and Bristol Bay stocks. Since 1987, the Alaska Beluga Whale Committee has monitored the subsistence harvests of belugas from these stocks. During this 20 year period, the total landed harvest for the four stocks combined (adjusted for years with missing data) ranged from 208 in 1995 to 494 in 1988, with a 20 year average of 323 per year. For individual stocks the average annual landed harvests for 1987–2006 were: Beaufort Sea – 41; Chukchi Sea – 62; eastern Bering Sea – 191; and Bristol Bay – 20. There was no significant long-term trend (p>0.05) in the rate of harvesting for any stock from 1987–2006. Average landed harvests relative to estimated stock size were: 0.1% for the Beaufort Sea (0.4% including belugas harvested from the Beaufort Sea stock by Canadian hunters); 1.7% for the eastern Chukchi Sea; 1.1% for the eastern Bering Sea; and 1.1% for Bristol Bay. The success of beluga harvest monitoring in Alaska is due to the cooperation of beluga hunters from more than 40 small coastal communities who report their harvests to the Alaska Beluga Whale Committee (ABWC). Through the ABWC, beluga hunters have been able to formalise their role in managing their subsistence resources.

KEY WORDS: BELUGA WHALE; WHITE WHALE; WHALING – ABORIGINAL; MANAGEMENT; ARCTIC

INTRODUCTION

Beluga whales (*Delphinapterus leucas*)[1] are an important subsistence resource for coastal residents of Alaska. They appear seasonally near villages and hunting camps from Cook Inlet to the Beaufort Sea where they are hunted with rifles and harpoons, or are caught in nets. Meat and muktuk (or maktaaq, i.e. skin and blubber) are consumed locally, or are shared with friends and relatives in other communities.

Belugas in Alaska are found in five summering concentrations, namely Cook Inlet, Bristol Bay, eastern Bering Sea (Norton Sound/Yukon Delta), eastern Chukchi Sea (Kotzebue Sound and Kasegaluk Lagoon), and the eastern Beaufort Sea. These concentrations have been considered as provisional management stocks for the last 20 years and previous harvests have been evaluated in terms of their impact on these provisional stocks (Frost and Lowry, 1991; Frost and Suydam, 1995; Lowry *et al.*, 1989; Seaman *et al.*, 1988). Recent analysis of mitochondrial DNA has supported treatment of these summering concentrations as separate management stocks (Angliss and Lodge, 2002; O'Corry-Crowe *et al.*, 2002; O'Corry-Crowe *et al.*, 1997). Belugas are also sometimes present and harvested in Kuskokwim Bay, although Kuskokwim Bay belugas have not been assigned to a provisional stock.

Prior to 1988, information on the harvest of belugas in Alaska was obtained primarily by personnel of the Alaska Department of Fish and Game (ADF&G) on an opportunistic basis. Harvest data have previously been published for 1977–1979 (Seaman and Burns, 1981), 1980–1983 (Burns and Seaman, 1988), and 1984–1986 (Lowry *et al.*, 1989). In 1988, the Alaska Beluga Whale Committee (ABWC) was formed, with goals of collecting harvest data and biological samples and identifying and conducting research needed for beluga whale conservation and management (Adams *et al.*, 1993). At that time, the ABWC began to compile and report harvest data on a more complete and regular basis than had been done in the past (Frost, 1999; Frost and Suydam, 1995).

In this paper, we present harvest data collected by the ABWC for the 20 year period from 1987–2006 for the four management stocks in western and northern Alaska and for animals harvested in the Kuskokwim delta. Data are not included from Canadian hunters in the Mackenzie River estuary, they are reported in Harwood *et al.* (2002). Data for Cook Inlet, where hunters and hunting communities do not belong to the ABWC, are not included.

METHODS

Harvest data were obtained from a variety of sources, as follows.

The ABWC held annual meetings in the autumn of each year from 1988–2006, after beluga hunting had ended for the year. At those meetings, hunter representatives reported on the harvest for each village or region. In addition, at least twice each year harvest questionnaires were mailed to village or regional government offices and to several hunters in most beluga hunting villages. Those hunters were individuals who had assisted the ABWC with harvest information in the past. The questionnaires asked for the number of belugas landed and struck and lost by season, as well as for other information about the location and type of hunt.

In the Norton Sound region, the Elim-Shaktoolik-Koyuk Marine Mammal Commission, with some financial support from the ABWC, hired monitors to collect harvest data for these three villages, and also to collect certain measurements

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[1] While not the official IWC common name for this species, 'beluga' is the name used in Alaska.

and samples for the ABWC. At some locations in northern Alaska (Point Lay), the entire beluga whale harvest was observed and sampled by harvest monitors working for the North Slope Borough Department of Wildlife Management in cooperation with local residents. The collection of Bristol Bay harvest data was coordinated by the ADF&G Division of Subsistence, in cooperation with the Bristol Bay Marine Mammal Council and Bristol Bay Native Association. Information from each of these sources about the number of landed belugas was considered reliable.

In addition, information about harvest was obtained through interviews with local residents and from ADF&G and other biologists working along the coast. Numbers obtained through these means were sometimes estimates rather than exact counts, or may not have included all hunters or the entire hunting season. Sometimes we received a minimum estimate of the number taken but other sources indicated that additional whales may have been harvested. In such cases, we used the highest estimate for the reported harvest. Whenever possible, harvest numbers were corroborated by contacting multiple sources for each harvest location.

Struck and lost rates were calculated only when data were reported for both harvested and struck and lost animals for a given year at a particular site. When either the harvest or the number struck and lost was missing or estimated, the struck and lost data were not used in the calculations.

We partitioned harvest data among the four management stocks, based on our knowledge of the seasonal distribution and movements of belugas as well as on genetics information. Harvests south of Bering Strait occurred while belugas were in seasonal concentration areas, and thus were easily ascribed to a particular stock. In Bering Strait and along the Chukchi Sea coast, harvest was assigned to a particular stock based on both where and when it occurred. The Beaufort Sea stock travels through open leads in the sea ice along the coast in spring (March–May) on its way to the eastern Beaufort Sea, and belugas harvested during this period were presumed to belong to the Beaufort Sea stock. This assumption was verified by genetics analyses of whales take at Point Hope, where the majority of such hunting occurs (O'Corry-Crowe *et al.*, 2002; 1997). Western Alaska villages whose spring harvests were attributed to the Beaufort Sea stock included Diomede, Kivalina, and Point Hope. Belugas taken in the Beaufort Sea at Barrow and Kaktovik during summer or moving westward in the fall were also attributed to the Beaufort Sea stock.

During and after breakup of the sea ice (June to mid-August), belugas appear along the Chukchi Sea coast between Kotzebue Sound and Wainwright (Frost and Lowry, 1991; Seaman *et al.*, 1988). All summer harvests in this region were assigned to the eastern Chukchi Sea stock. Villages/towns included in this region were Buckland, Deering, Kotzebue, Noatak, Kivalina, Point Hope, Point Lay, and Wainwright. Mitochondrial DNA analysis suggests that belugas harvested in Kotzebue Sound may differ genetically from other eastern Chukchi Sea belugas (G. O'Corry-Crowe, pers. comm.). However, for the current analysis, we have assumed that Kotzebue animals are part of the eastern Chukchi Sea stock.

For some locations in some years, particularly in the ABWC's early years, no harvest data were available. Since a compilation of harvest data without these sites would underestimate the total harvest, we calculated an 'adjusted harvest' to use in evaluating landed harvest relative to stock size and harvest trend. For all missing data points, we assigned a value equal to the average value for that site for all years for which data were available. Thus, if data were missing for 4 of 20 years at a particular site, the assigned value would be the average of the 16 years with data. The adjusted harvests were used to estimate the percent of a stock that was harvested.

We used a linear model and data adjusted for missing values to examine trend in harvest over the 20 year period. We used standard diagnostics to ensure that errors were normally and identically distributed (e.g., normal probability plots, model specification tests) and that there was no autocorrelation in the data (i.e. Durban-Watson tests). Landed harvest as a proportion of estimated stock size was evaluated by comparing adjusted harvests, by stock, to the best available population estimate for that stock.

RESULTS

Reported landed harvests

Landed harvest data for 1987–2006 were obtained from more than 40 communities from along the Bering, Chukchi and Beaufort Sea coasts of Alaska (Fig. 1). The reported annual landed harvest of belugas (Table 1) averaged 275 (SE = 17.9) and was lowest in 1995 (101) and highest in 1988 (418).

Seven communities harvested animals from the Beaufort Sea stock (Table 1), with an average annual harvest of 39 belugas landed (range 4–85 SE = 4.7). Belugas from the eastern Chukchi Sea stock were harvested by six villages, with an average annual harvest of 62 (range 2–116, SE = 6.4). Fifteen or more communities hunted from the eastern Bering Sea stock, with an annual harvest of 152 (range 31–281, SE = 14.6). Eleven communities hunted from the Bristol Bay stock, with annual harvests averaging 17 (range 6–35, SE = 1.7). Harvest reports were intermittent for the Kuskokwim delta region, but it is clear from the data we received and from discussions with local hunters that few belugas were seen or taken in most years. The average annual harvest over 20 years was 5 (range 0–27. SE = 1.6).

Only four villages reported average annual harvests of 20 or more belugas and 30 villages harvested five or fewer belugas per year. The average annual harvests were dominated by a single village for the Beaufort Sea (Alaska only, Point Hope, 79%) and the eastern Chukchi Sea (Point Lay, 60%) stocks. The harvest was spread more evenly among villages for the eastern Bering Sea and Bristol Bay stocks.

Harvest reporting improved over the 20 year period 1987–2006. For the four management stocks combined (not including the Kuskokwim where data were intermittent), reporting improved from 75% of villages reporting during 1987–1996 to 90% during 1997–2006. Improvement was greatest for the eastern Bering Sea and Bristol Bay stocks (about 70% of villages reporting during the first 10 years and 90% in more recent years). Reporting rates were high over the entire 20 year period for the Beaufort Sea (80%–87%) and eastern Chukchi Sea (93%–97%) stocks. A comparison of reported harvest totals to totals adjusted for missing data suggests that the actual harvest was probably about 20%

Table 1

Reported landed harvest of beluga whales from western and northern Alaska, 1987–2006. Data provided by the Alaska Beluga Whale Committee. nd = no data.

Location	87	88	89	90	91	92	93	94	95	96	97	98	99	00	01	02	03	04	05	06	$\bar{x}$
Beaufort Sea stock																					
Barrow	0	0	1	0	1	0	2	5	0	2	8	1	1	1	1	1	2	1	7	1	2
Diomede	10	3	6	5	3	2	1	0	nd	0	1	4	0	6	0	0	4	0	0	4	3
Kaktovik	0	0	0	10	0	0	0	0	1	0	2	0	0	0	0	0	3	1	0	0	1
Kivalina	nd	5	0	0	0	10	3	3	3	7	0	0	1	43	0	0	0	1	2	0	4
Nuiqsut	nd	nd	nd	nd	nd	nd	nd	nd	0	0	0	1	0	nd	0	0	0	0	0	0	0
Point Hope	40	59	17	16	39	15	79	53	nd	15	32	52	33	16	24	23	34	29	11	0	31
Wales	0	0	2	3	nd	1	nd	1	nd	nd	nd	1	nd	nd	nd	0	nd	nd	nd	nd	1
TOTAL	**50**	**67**	**26**	**34**	**43**	**28**	**85**	**62**	**4**	**24**	**43**	**59**	**35**	**66**	**25**	**24**	**43**	**32**	**20**	**5**	**39**
Eastern Chukchi Sea stock																					
Buckland	7	17	0	31	0	4	0	0	1	5	1	1	0	1	18	2	0	0	0	0	4
Deering	0	0	0	nd	nd	nd	0	0	0	2	0	0	nd	0	nd	0	0	0	0	0	0
Kivalina	0	1	0	1	1	0	0	0	0	0	1	0	0	1	0	3	0	0	0	0	0
Kotz/Noatak	2	8	37	6	11	5	6	7	4	68	7	4	2	0	9	4	0	1	1	2	9
Point Lay	22	40	16	62	35	24	77	56	31	41	3	48	47	0	34	47	36	53	41	29	37
Wainwright	47	3	0	0	5	20	0	0	0	0	4	38	3	0	23	37	38	0	1	0	11
TOTAL	**78**	**69**	**53**	**100**	**52**	**53**	**83**	**63**	**36**	**116**	**16**	**91**	**52**	**2**	**84**	**93**	**74**	**54**	**43**	**31**	**62**
Eastern Bering Sea stock																					
Alakanuk	nd	nd	nd	14	nd	10	nd	10	nd	nd	7	14	nd	9	13	nd	nd	32	37	15	16
Elim	nd	27	22	8	14	2	16	5	10	27	21	13	9	30	20	13	9	22	17	11	16
Emmonak	3	nd	1	nd	nd	12	15	20	nd	nd	20	20	8	30	30	40	30	nd	40	30	21
Golovin	nd	5	13	0	0	1	2	1	0	0	1	4	0	0	2	0	0	0	0	5	2
Hooper Bay	nd	5	nd	nd	10	nd	nd	40	5	35	17	6	6	39	69	46	8	3	29	33	23
Kotlik	nd	nd	nd	1	nd	9	40	15	5	2	8	4	13	11	6	12	nd	nd	13	12	11
Koyuk	15	54	30	45	55	3	20	8	0	8	6	6	4	0	13	17	3	9	5	3	15
Nome/Brevg	0	0	0	0	0	nd	nd	1	0	1	0	0	0	2	nd	nd	0	nd	3	2	1
Nunam Iqua	nd	nd	nd	25	nd	11	nd	nd	nd	nd	2	1	nd	3	12	13	10	7	9	2	9
St. Michael	nd	50	25	22	5	5	17	1	5	14	8	16	13	8	21	13	2	3	6	5	13
Scammon	nd	nd	nd	7	15	5	nd	nd	nd	6	3	13	9	12	12	nd	11	11	6	7	9
Shaktoolik	nd	30	15	12	17	5	12	9	10	4	17	16	13	nd	40	12	10	26	13	14	15
Stebbins	10	30	22	20	46	7	12	6	12	6	20	17	16	15	18	21	20	8	9	9	16
Unalakleet	nd	35	14	12	17	3	2	nd	8	3	5	14	20	29	21	38	5	4	4	10	14
White Mt.	nd	1	1	0	0	0	0	nd	0	0	1	0	2	0	6	4	4	7	5	4	2
Other	3	6	12	2	10	6	nd	nd	nd	nd	7	nd	1	nd	nd	5	nd	0	53	1	9
TOTAL	**31**	**243**	**155**	**168**	**189**	**79**	**136**	**116**	**55**	**106**	**143**	**144**	**114**	**188**	**281**	**234**	**112**	**132**	**249**	**163**	**152**
Kuskokwim																					
Eek	nd	3	4	0	7	7	nd	2	nd	nd	nd	nd	nd	0	0	0	0	0	0	nd	2
Goodnews	nd	1	nd	0	0	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	0	nd	nd	nd	0
Newtok	nd	nd	nd	nd	nd	nd	0	nd	nd	nd	0	nd	nd	0	0	1	0	0	nd	nd	0
Quinhagak	nd	5	4	nd	15	8	nd	4	nd	0	0	nd	4	0	0	0	0	0	0	0	3
Toksook	0	0	0	0	0	nd	nd	nd	nd	2	0	0	0	0	0	3	0	0	nd	8	1
Tuntatuliak	nd	1	3	nd	nd	nd	nd	0	nd	nd	nd	nd	nd	nd	nd	nd	0	nd	nd	nd	0
Other	5	3	2	nd	5	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	2	1	3
TOTAL	**5**	**13**	**13**	**0**	**27**	**15**	**0**	**6**	**0**	**2**	**0**	**0**	**4**	**0**	**0**	**4**	**0**	**0**	2	**9**	**5**
Bristol Bay Stock																					
Aleknagek	nd	nd	1	nd	nd	nd	3	1	1	1	0	0	2	1	2	2	0	2	3	1	1
Clark's Point	1	nd	1	4	nd	nd	4	3	0	3	2	2	2	3	4	4	0	0	nd	0	2
Dillingham	2	3	nd	nd	nd	nd	3	5	1	8	4	1	3	5	8	1	8	4	7	4	4
Egegik	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	1	0	0	0	0
Igiugig	nd	nd	nd	nd	nd	3	3	1	1	0	0	0	0	1	nd	0	0	3	0	2	1
Iliamna	nd	nd	nd	nd	3	3	0	1	1	0	0	0	0	0	2	0	0	0	0	2	1
Levelock	3	23	6	10	10	5	9	2	1	4	2	0	3	7	2	1	0	0	2	2	5
Manokotak	3	nd	nd	6	6	nd	11	4	1	2	2	2	2	3	4	1	10	4	3	5	4
Naknek	nd	nd	nd	nd	nd	1	2	0	0	0	1	1	1	1	nd	nd	nd	1	1	4	1
Newhalen	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	1	nd	nd	nd	0	3	0	1
Non-local	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	2	2	2	2	nd	2
Togiak	nd	0	0	0	0	0	0	0	0	0	0	0	0	2	1	nd	0	0	0	0	0
TOTAL	**9**	**26**	**8**	**20**	**19**	**12**	**35**	**17**	**6**	**18**	**11**	**6**	**13**	**24**	**25**	**11**	**21**	**16**	**21**	**20**	**17**
All stocks	**173**	**418**	**255**	**322**	**330**	**187**	**339**	**264**	**101**	**266**	**213**	**300**	**218**	**280**	**415**	**366**	**250**	**234**	**335**	**228**	**275**

higher than reported during 1987–1996, and about 7% higher during 1997–2006. The difference was greatest for stocks with the most missing data. Reported harvests were probably about 30% underestimated for the eastern Bering Sea stock and 25% for the Bristol Bay stock during 1987–1996, but less than 10% underestimated for both during 1997–2006.

Harvest trends and harvest relative to population size

There was no statistically significant long-term trend ($p>0.05$) in the rate of harvesting for the Beaufort Sea, eastern Chukchi Sea, eastern Bering Sea and Bristol Bay beluga stocks from 1987–2006 (Table 2; Fig. 2). For each stock, residuals from the linear models used to estimate trend were normally distributed, homoscedastic, and devoid of serial autocorrelation ($p>0.05$), indicating that linear models were appropriate for examining trends.

Average annual landed harvests of Alaskan beluga stocks during 1987–2006 ranged from 0.1% to 1.7% of estimated stock size (Table 2). The Alaskan harvest of the Beaufort Sea stock was <0.2% of the estimated stock size. The average

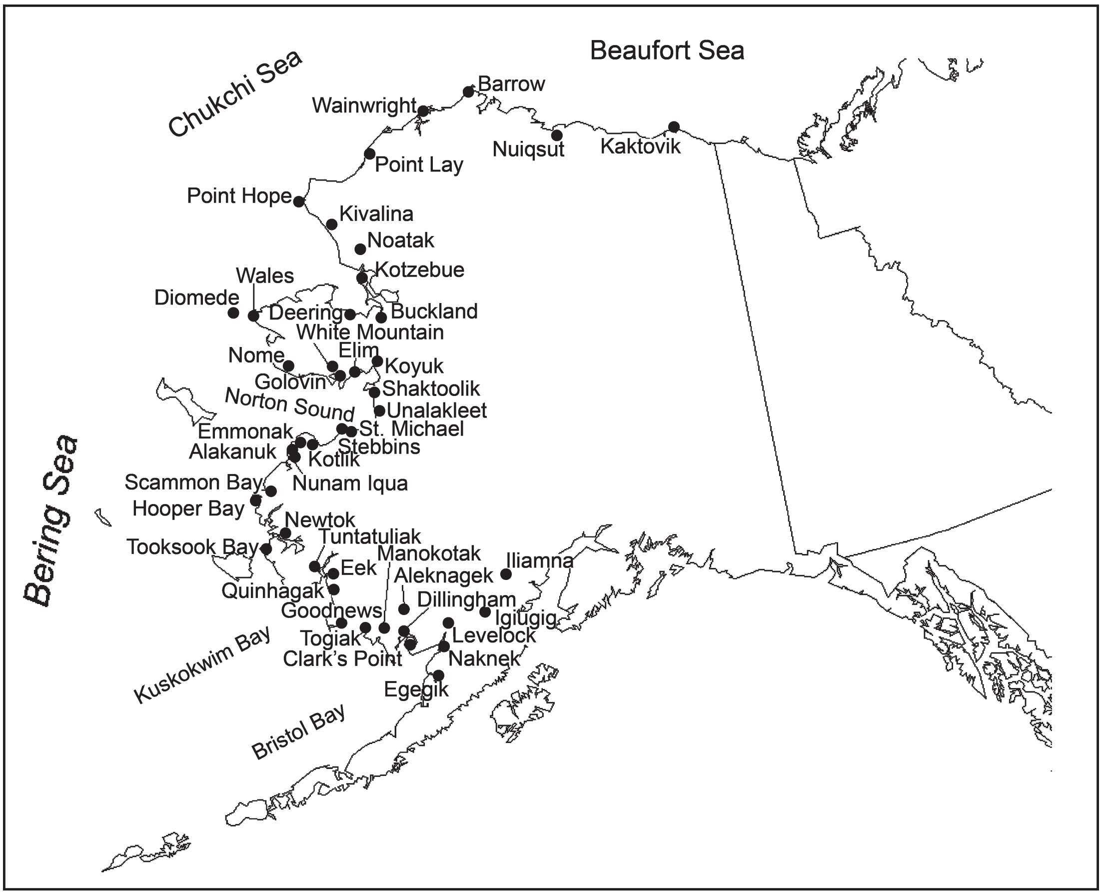

Fig. 1. Map of Alaska showing communities where beluga harvest data were collected.

adjusted harvest for the eastern Bering Sea and Bristol Bay stocks was 1.1% of estimated stcok size and never exceeded 1.9% in any year. Annual harvest for the eastern Chukchi Sea stock averaged 1.7%, although in a few years it was as high as 3.1%.

Struck and lost

Struck and lost information was not reported for all sites and years. Information was poorly reported in most years for most communities hunting from the Beaufort Sea and eastern Bering Sea stocks so we do not report it. Data for the eastern Chukchi Sea and Bristol Bay stocks are summarised below for 1997–2006 only since data before then were sporadic.

Struck and lost information was available 83% of the time for the six villages harvesting from the eastern Chukchi Sea stock, with an average struck and lost rate for 1997–2006 of 7%. Sixty percent of the eastern Chukchi Sea harvest occurred at Point Lay, where the hunt usually occurs in very shallow water (<2m) near town. Starting in 1996 Point Lay hunters established guidelines that encouraged hunters to harpoon every beluga before it was shot. Their struck and lost rate since 1996 has been less than 2%. Struck and lost information was available for Bristol Bay villages 63% of the time. The average struck and lost rate for the 11 Bristol Bay villages was 9%.

DISCUSSION

Landed harvest as a proportion of estimated stock size

A marine mammal population is thought to be able to reach or maintain its optimum sustainable population if human-induced mortalities are kept below one half of the maximum net productivity rate. Two percent has been considered a

Table 2

Raw and adjusted annual harvest (average, range) for four Alaska stocks of beluga whales relative to stock size during 1987–2006. Stock identity and abundance are unknown for Kuskokwim belugas. Data were provided by the Alaska Beluga Whale Committee.

	Stock estimate	Raw landed (range)	Reported % of stock	Adjusted (range)	Adjusted % of stock
Beaufort[a]	39,258[b, c]	39 (4–85)	0.10 (0.01–0.22)	41 (6–86)	0.11 (0.02–0.22)
E. Chukchi	3,710[b, d]	62 (2–116)	1.68 (0.05–3.13)	62 (2–116)	1.68 (0.05–3.13)
E. Bering	18,142[b]	152 (31–281)	0.84 (0.17–1.55)	191 (103–309)	1.05 (0.57–1.70)
Kuskokwim	?	5 (0–27)		8(1–27)	
Bristol Bay	1,888[b, e]	17 (6–35)	0.90 (0.32–1.85)	20(6–36)	1.08 (0.32–1.85)
TOTAL		275 (101–418)		323 (208–494)	

[a] Does not include harvests by Canadian hunters when this stock is in the Mackenzie River estuary; [b] Angliss and Lodge (2002); [c] Harwood *et al.* (1996); [d] Lowry *et al.* (1999); [e] Frost *et al.* (2002).

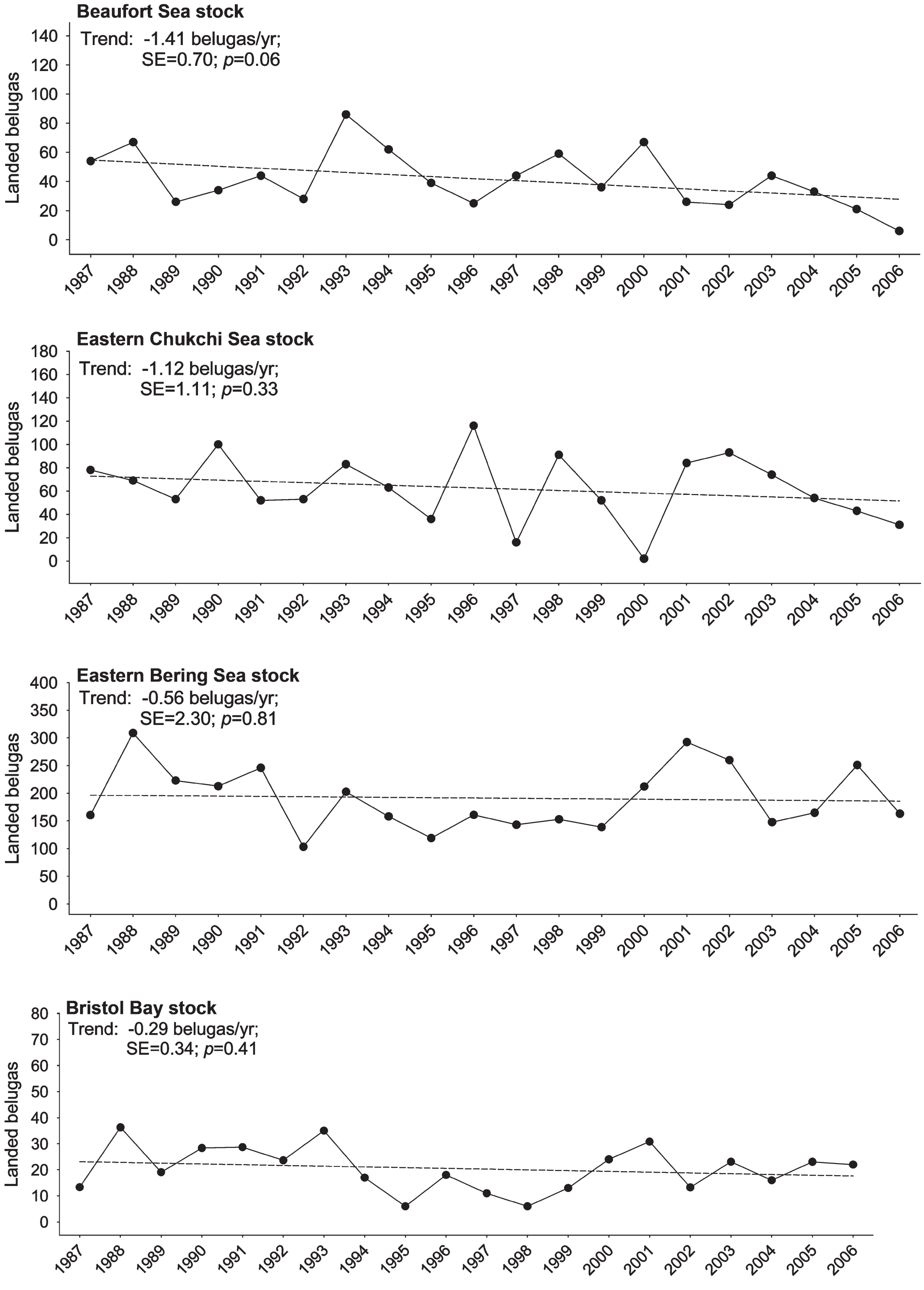

Fig. 2. Adjusted annual harvests from four stocks of beluga whales in northern and western Alaska, 1987–2006. Data are from the Alaska Beluga Whale Committee.

reasonable estimate for cetaceans such as belugas (Wade, 1998) although it has been suggested that removal rates of >1% can be problematic for some small cetaceans (Reeves and Brownell, 2009). However, even 2% for belugas may be conservative in some situations. Lowry *et al.* (2008) reported an estimated rate of increase for beluga whales of 4.8%/year (95% CI = 2.1%–7.5%) in Bristol Bay, Alaska during 1993–2005 (a 65% population increase overall). The increase was concurrent with landed harvests of 1%–2% per year, as well as some fishing related mortality and predation by killer whales. All of the long-term average landed harvests for the Beaufort Sea, eastern Chukchi Sea, eastern Bering Sea and Bristol Bay beluga stocks are <2% of the estimated stock size.

There is large annual variation in the harvest from each management stock, due to a combination of factors including accessibility of belugas, weather, sea ice conditions and activities of the hunters. Thus, annual harvests from a particular stock can range from a few percent to almost double the long-term average. For this reason, the ABWC recommended to NOAA in 1996 that harvests be averaged over moving five year periods when assessing harvest sustainability. During 1987–2006, the average landed beluga harvest did not exceed 1.9% of the estimated stock size during any five year period for any stock, and was usually much less.

Harvest data reported here for the Beaufort Sea stock (Table 1) do not include harvests by Canadian beluga hunters in the Mackenzie River estuary where average landed harvest for 1990–1999 was 111 plus a loss rate of 11.3% (Harwood *et al.*, 2002). When Canadian and Alaskan average harvests were combined, the total landed harvest represents an estimated 0.4% of the Beaufort Sea stock.

Annual landed harvest for the eastern Chukchi Sea stock averaged 1.7% over 20 years, although in a few years it was as high as 3.1%. We do not consider these occasional higher harvests to be of concern since five year average harvests as well as the long-term harvest were always <2%. Furthermore, the population size used in the calculations for this stock was likely an underestimate. During 1998, five belugas were satellite tagged in the eastern Chukchi Sea. When aerial surveys were conducted 5–9 days after tagging, only one of the tagged belugas was located in the area counted (Lowry *et al.*, 1999; Suydam *et al.*, 2001).

Struck and lost

Hunters are often reluctant to provide information about the number of animals that are struck and lost during hunting. Recently however there has been increased awareness of this issue. Senior hunters are encouraging young and new hunters to be careful how and where they hunt so as to minimise hunting-related loss. Communities are discussing hunter guidelines that would encourage hunting in shallow water where belugas are more easily retrieved. The overall ABWC Management Plan, as well as regional management plans for Bristol Bay and northeastern Norton Sound include provisions encouraging the reduction of struck and lost rates (Adams *et al.*, 1993; ABWC unpublished).

The struck and lost information we did obtain for 1997–2006 suggests that the number of belugas struck and lost is not problematic relative to estimated stock size for the Bristol Bay and eastern Chukchi Sea stocks. The reported struck and lost rate for Bristol Bay was 9%. Whether or not the reported rates for the Bristol Bay stock were underestimated, the harvest there was clearly sustainable since the population increased by 65% over the period during which the harvest data were collected.

The overall reported struck and lost rate for the eastern Chukchi Sea stock was 7%, and it was an even lower 2% at Point Lay. Information for Point Lay was considered accurate since biologists usually observed the hunt, research aircraft often flew over the lagoon on days following the harvest and could observe any lost carcasses, and research boats transited the area during and following the harvest. The very shallow water in the hunt area, the drive nature of the hunt, and hunting practices requiring that each beluga be harpooned before it was shot all probably contributed to the very low struck and lost rate. Although researchers were not present at other Chukchi Sea villages during the harvest, hunting conditions and methods were similar in several, suggesting that low reported stuck and lost rates were reasonable.

There are no reliable struck and lost data for the Alaskan harvests of the Beaufort Sea stock. However, the landed harvest for Alaska and Canada combined is less than 0.4% of a population exceeding 39,000. Similarly, the eastern Bering Sea stock landed harvest is about 1% of a population exceeding 18,000.

Data quality and the ABWC

The ABWC has collected harvest data since 1987 (Frost, 1999; Frost and Suydam, 1995). Data for this 20 year period are generally more complete and include many more villages than data available prior to 1987 (Burns and Seaman, 1988; Lowry *et al.*, 1989; Seaman and Burns, 1981). Since 1987, the ABWC has seen consistent improvements in the quality of the harvest data. Harvest reports have been obtained from about 45 communities since 2000, compared to 21 in 1987. On average, data were missing for 25% of the communities during the ABWC's first 10 years (1987–1996) and only 9% since then.

In the early years, many harvest reports were of an approximate number or range of belugas taken. It was sometimes unclear whether the reported harvest represented both spring and fall hunts or only the most recent harvest. Rarely was it known whether the belugas were taken by net or by shooting, which can be an important factor when estimating the number of animals lost. At present, for most villages data are reported separately for spring, summer and fall harvests, and for belugas taken by net and by shooting. Comments frequently accompany the data about unusual occurrences, beluga diet, or general abundance.

Coverage has been adequate for all communities hunting from both the Beaufort (79% of communities reporting 1987–1996 and 87% 1997–2006) and eastern Chukchi sea stocks (93% of communities reporting 1987–1996 and 97% 1997–2006). This is in large part because of the North Slope Borough Department of Wildlife Management's regular involvement in harvest monitoring in their region, dedicated village harvest monitors, and because ADF&G biologists have a long-term history of working with beluga hunters there.

The quality and regularity of harvest data for the eastern Bering Sea stock has improved since the initial formation of the ABWC. During the first 10 years that the ABWC collected harvest data (1987–1996), there were no harvest reports for an average of 31% (13%–73%) of the eastern Bering Sea communities in any one year. During the subsequent 10 year period (1997–2006), data were missing on average from only 11% (0%–19%) of the communities in any year. Few Yukon delta villages reported harvests initially, but reporting has improved markedly since then with the addition of more village representatives at ABWC meetings.

Harvest reports have been intermittent for Kuskokwim villages. In part, this reflects the intermittent occurrence of belugas in the area and consequently the infrequent hunting of belugas. According to local residents, belugas were common in Kuskokwim Bay and the lower Kuskokwim River in the early 1900s, but stopped using the area regularly in the 1940s (Frost and Lowry, 1991). During April–August 1988, frequent sightings of up to several hundred belugas were made in Kuskokwim Bay for the first time in many years (Frost and Lowry, 1991). We also received reports of harvests by several communities that year (Table 1). Since then, there have been infrequent reports of both sightings and harvests. Many Kuskokwim villages have responded to ABWC questionnaires saying that belugas are rarely seen and seldom, if ever, hunted. Therefore, less effort has been devoted to obtaining harvest information from this region.

ADF&G's Division of Subsistence, in cooperation with the Bristol Bay Marine Mammal Council, began to monitor the beluga harvest in Bristol Bay in 1994. Since then, there has been nearly complete reporting for all villages in all years. From 1987–1996, 70% of all villages reported harvest data. Since 1996, 93% have reported.

The existence of the ABWC and its agreement with NOAA for the cooperative management of the beluga subsistence hunt has enhanced the amount and quality of harvest data collected since 1988. In addition, the ABWC has raised awareness of the need for better abundance estimates, genetics studies and biological information about belugas to ensure data for management of sustainable harvests. Through the ABWC, beluga hunters have been able to formalise their role as partners in managing their subsistence resources. They have become active participants in recommending and conducting research needed for effective management.

Future work

Estimates of stock size are required for any evaluation of the impact of harvest on population status. Currently, adequate estimates exist for the Bristol Bay, eastern Bering Sea and Beaufort Sea stocks. However, there is no adequate estimate for the eastern Chukchi Sea stock. The existing estimate of about 3,700 is thought to be quite low. To address this deficiency, the ABWC convened a workshop in March 2010 to develop a detailed design for a future Chukchi Sea assessment effort to begin in 2011.

The ABWC continues to stress the importance of obtaining accurate and complete harvest data, including the number of belugas struck and lost.

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The beluga hunters of Alaska and hunter delegates to the Alaska Beluga Whale Committee contributed the harvest data presented in this report. Without their cooperation and assistance, it would not be possible to compile accurate harvest information and to obtain samples necessary for biological investigations of belugas in Alaska. Many have answered questionnaires and taken the time to report information about belugas in their area. Their assistance is much appreciated and has been essential to the success of the harvest monitoring program. This study was funded in part by NOAA grants NA17FX2593, NA27FX0258-01, NA37FX0267, NA47FX0498, and NA57FX0368 and NA67FX0197 to the Alaska Beluga Whale Committee and was supported by in kind contributions of the Alaska Department of Fish and Game and the North Slope Borough. Kawerak, Inc., the Elim-Shaktoolik-Koyuk Marine Mammal Commission, the Association of Village Council Presidents and the Bristol Bay Native Association have contributed through the hiring of Natural Resource Specialists to collect harvest data and samples, and by providing additional travel funds for some hunter representatives to attend annual meetings and workshops. The manuscript was improved by reviews by Lloyd Lowry (University of Alaska Fairbanks); John Citta (Alaska Dept. Fish and Game); and Doug DeMaster (US National Marine Fisheries Service).

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Cetacean distribution in the coastal waters of the Sultanate of Oman

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ABSTRACT

Small boat surveys were conducted between 2000 and 2003 in three main regions of Oman's coastal waters: Muscat, the Gulf of Masirah and Dhofar. Survey data were analysed to calculate relative abundances of the seven most frequently encountered species in these areas. These include (in order of frequency) bottlenose dolphins (*Tursiops sp.*), long-beaked common dolphins (*Delphinus capensis*), humpback whales (*Megaptera novaeangliae*), spinner dolphins (*Stenella longirostris*), Indo-Pacific humpback dolphins (*Sousa chinensis*), Bryde's whales (*Balaenoptera sp.*) and Risso's dolphins (*Grampus griseus*). Other species observed include false killer whales (*Pseudorca crassidens*), blue whales (*Balaenoptera musculus*), rough-toothed dolphins (*Steno bredanensis*) and unidentified beaked whales. Encounter rates per distance searched were plotted by 0.1 x 0.1 degree grid cell, giving an indication of relative abundances and key areas of habitat used by each of the seven most frequently encountered species. These plots demonstrate that the nearshore areas of the Gulf of Masirah, as well as the coastal waters of Dhofar, are areas of concentration for the Arabian Sea's recently designated Endangered subpopulation of humpback whales, as well as Indo-Pacific humpback dolphins, which are considered Near Threatened on the IUCN Red List of Threatened Species.[1] The results presented here provide valuable baseline data for future research and help to inform conservation management efforts that are required to address the highly vulnerable status of the humpback whale and Indo-Pacific humpback dolphin populations in question.

KEY WORDS: ARABIAN SEA; GULF OF OMAN; DISTRIBUTION; HABITAT; BRYDE'S WHALE; BLUE WHALE; HUMPBACK WHALE; SPERM WHALE; BOTTLENOSE DOLPHIN; COMMON DOLPHIN; SPINNER DOLPHIN; RISSO'S DOLPHIN; INDO-PACIFIC HUMPBACK DOLPHIN; ROUGH-TOOTHED DOLPHIN

INTRODUCTION

The Sultanate of Oman is a rapidly developing country on the Arabian Peninsula, with a landmass of approximately 300,000km^2 and over 3,240km of coastline (Al-Oufi, 2003 p.149). Politically and economically isolated and devoid of modern industry or infrastructure prior to 1970 (Kechichian, 1995), the country subsequently relied heavily on hydrocarbon resources which remain the primary economic driver. The economy is now diversifying, with emphasis on tourism, fisheries exports and port services (Ministry of National Economy, 2003). These sectors can lead to significant coastal and nearshore development pressures. It has been estimated that more than 80% of the country's population of over 2.3 million lives within 20km of the coast (Ministry of National Economy, 2003).

The oceanography of the Sultanate of Oman is complex. Coastal upwelling during the northeast and southwest monsoon seasons creates nutrient-rich 'temperate' marine conditions in an otherwise tropical marine climate (Banse, 1987; Burkhill, 1999; Kindle and Arnone, 2001; Sheppard *et al.*, 1992). During peak southwest monsoon months (July and August), sea-surface temperatures can drop to 16–17°C (Sheppard *et al.*, 1992; Wilson, 2000). High nutrient levels in upwelled waters result in phytoplankton blooms and high productivity. Along the Arabian Sea coast of Oman productivity increases tenfold from less than 0.1g C m^{-2} d^{-1} during the southwest inter-monsoon period to above 1.1g C m^{-2} d^{-1} (Brock and McClain, 1992). This level of productivity is expected to support an abundance of cetacean prey (e.g. Papastavrou and Van Waerebeek, 1997) for a range of species with documented occurrence in Omani waters (e.g. Alling *et al.*, 1982; Baldwin, 1997; Ballance *et al.*, 1996; Gallagher, 1991; Papastavrou and Salm, 1991; Salm, 1991; Salm *et al.*, 1993). Baldwin *et al.*'s (1999) review of cetaceans in Arabian waters used this literature and other incidental/opportunistic sightings and strandings data to provide the most comprehensive published overview to date. However, with the exception of Ballance *et al.* (1996), which focused on offshore environments only, previous studies do not provide information on relative or absolute abundance, and limited data on habitat preference or ecology. As the rate and scale of development and associated human activities increase in Oman, it becomes more important to define habitats, and assess threats within those habitats, in order to design optimal management and conservation strategies (e.g. Bannister *et al.*, 1996; Evans and Hammond, 2004).

Geographic Information Systems (GIS) can be used to map cetacean distribution and abundance in relation to physical and environmental factors such as depth, slope, sea surface temperature and chlorophyll-a concentrations (e.g. Baumgartner *et al.*, 2001; Cañadas *et al.*, 2002; Davis *et al.*, 2002; Moses and Finn, 1997; Smith *et al.*, 1986; Waring

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[1] IUCN. 2010. IUCN Red List of Threatened Species. Version 2010.4. [Available from: *http://www.iucnredlist.org*].

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et al., 2001). These studies have, with varying degrees of statistical significance, determined correlations between environmental factors (particularly depth and slope) and cetacean distribution. Here we use GIS to analyse data collected during small boat surveys conducted in coastal waters of Oman. Observed distribution and relative abundance of a number of cetacean species are analysed to identify key areas for the most frequently encountered species and to provide a baseline against which to compare future research efforts in the face of increased coastal development and habitat degradation.

METHODS AND MATERIALS

Field surveys

Small boat surveys were conducted between January 2000 and October 2003 in three main locations; the Gulf of Masirah, Dhofar, and Muscat. Timing and locations of surveys are shown in Table 1 and survey tracks from these surveys are depicted in Figs 1a–c. Surveys were always conducted in the Gulf of Masirah in October and November, in the Dhofar region in February and March and monthly in the Muscat region throughout most of the three-year study period. One additional 3-day survey was conducted in Ras al Hadd in March–April 2001. Rough seas and fog generated by the SW Monsoon prevented small boat surveys along the Arabian Sea coast during the summer months (May–September). Survey effort was further constrained by the availability of funding and personnel, as all work was conducted on a volunteer basis. Furthermore, surveys in the Dhofar and Gulf of Masirah area were conducted with an aim to collect distributional, photo-ID, and biopsy data from the Arabian Sea's subpopulation of humpback whales, designated as Endangered by the IUCN in

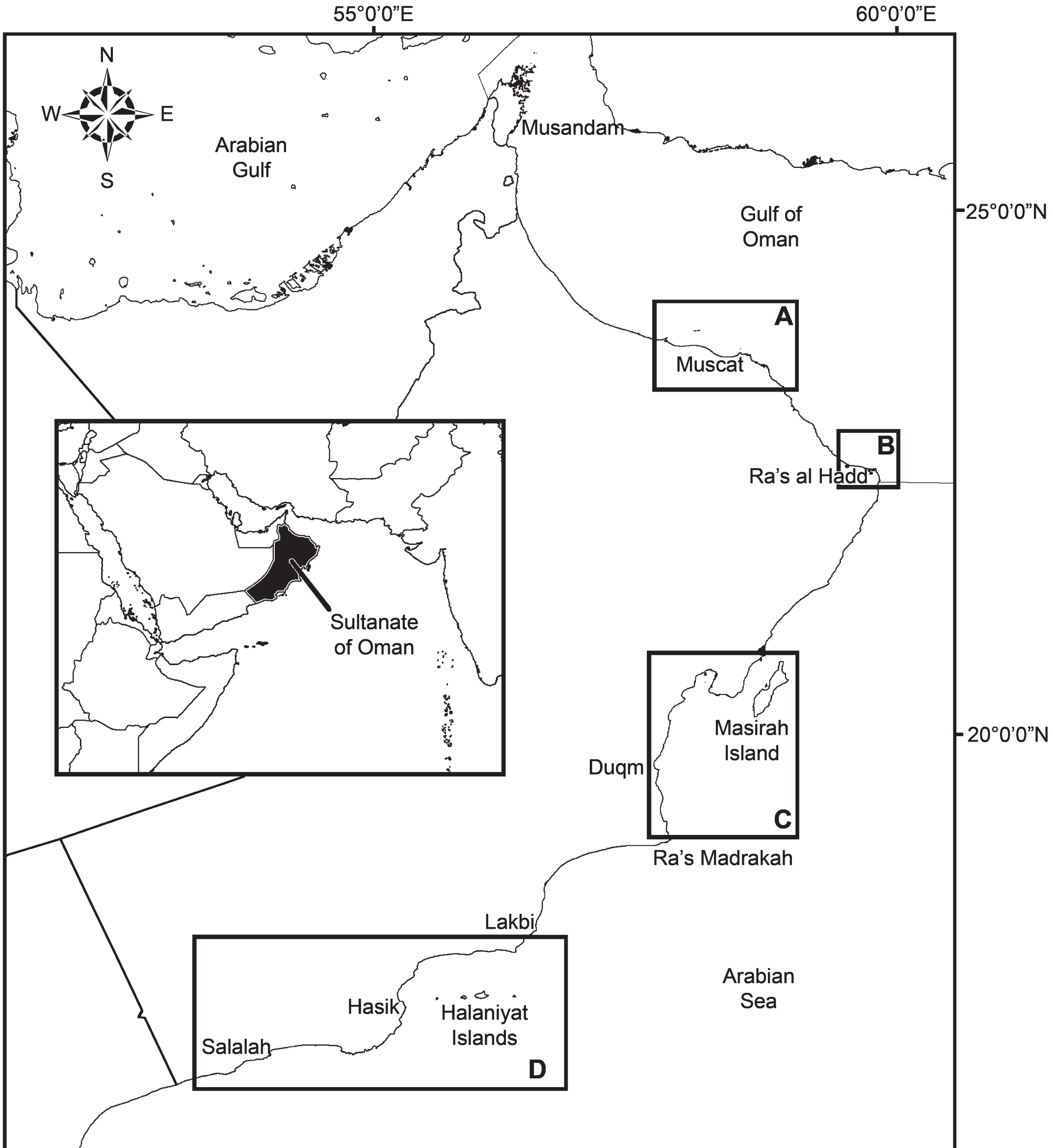


Fig. 1. Northern Indian Ocean and Arabian region highlighting Oman and the four main study regions (A–D) along the coast. A: Search effort in the Muscat region from 2001–2003 – on effort tracks = 2,264km. B: Ras al Hadd (not shown in detail as it is only 3 days of survey effort) – on effort tracks = 200.8km. C: The Gulf of Masirah (GoM) – on effort tracks = 2,555.24km. D: The Dhofar region – on effort tracks = 3819.71km.

2008 (Minton *et al.*, 2008). As such, areas of known or suspected humpback whale distribution (based on historical data and anecdotal reports) were targeted. However, within those general survey areas, tracks were designed to provide as much coverage of the area as possible and in as even a manner as possible, without taking into consideration the specific location of previous sightings.

The majority of surveys were conducted from a 6.5m rigid-hulled inflatable boat (RIB), powered by two outboard engines. Pre-determined survey tracks were plotted on bathymetric charts, and generally followed an irregular saw-tooth pattern along the coast. These were designed to cover different depth ranges within the logistical and safety limitations imposed by vessel size and nightly mooring opportunities. Exceptions to this general survey plan included surveys in January and February 2000, when two observers conducted opportunistic searches from vessels in transit around the Hallaniyat Islands (17.50°N, 56.00°E), maintaining records of effort and survey tracks, and during 15–17 October 2000, when a survey was conducted using a 5.5m fibreglass fishing skiff powered by a 25hp engine in the northern portion of the Gulf of Masirah (20.33°N, 58.25°E).

Surveys were conducted in ‘closing mode’, with search effort suspended when animals were sighted. Survey speeds ranged from 12 to 15 knots. Observers standing at deck level or seated on an A-Frame 3m above the sea surface scanned by eye areas forward of the beam. All observer activities were logged to the nearest minute allowing for post survey stratification of effort types. Weather conditions were recorded hourly and search effort was suspended in Beaufort sea-states of 4 or higher. Positional data were recorded using Garmin 12 or 12XL GPS units. Tracks were logged, with the

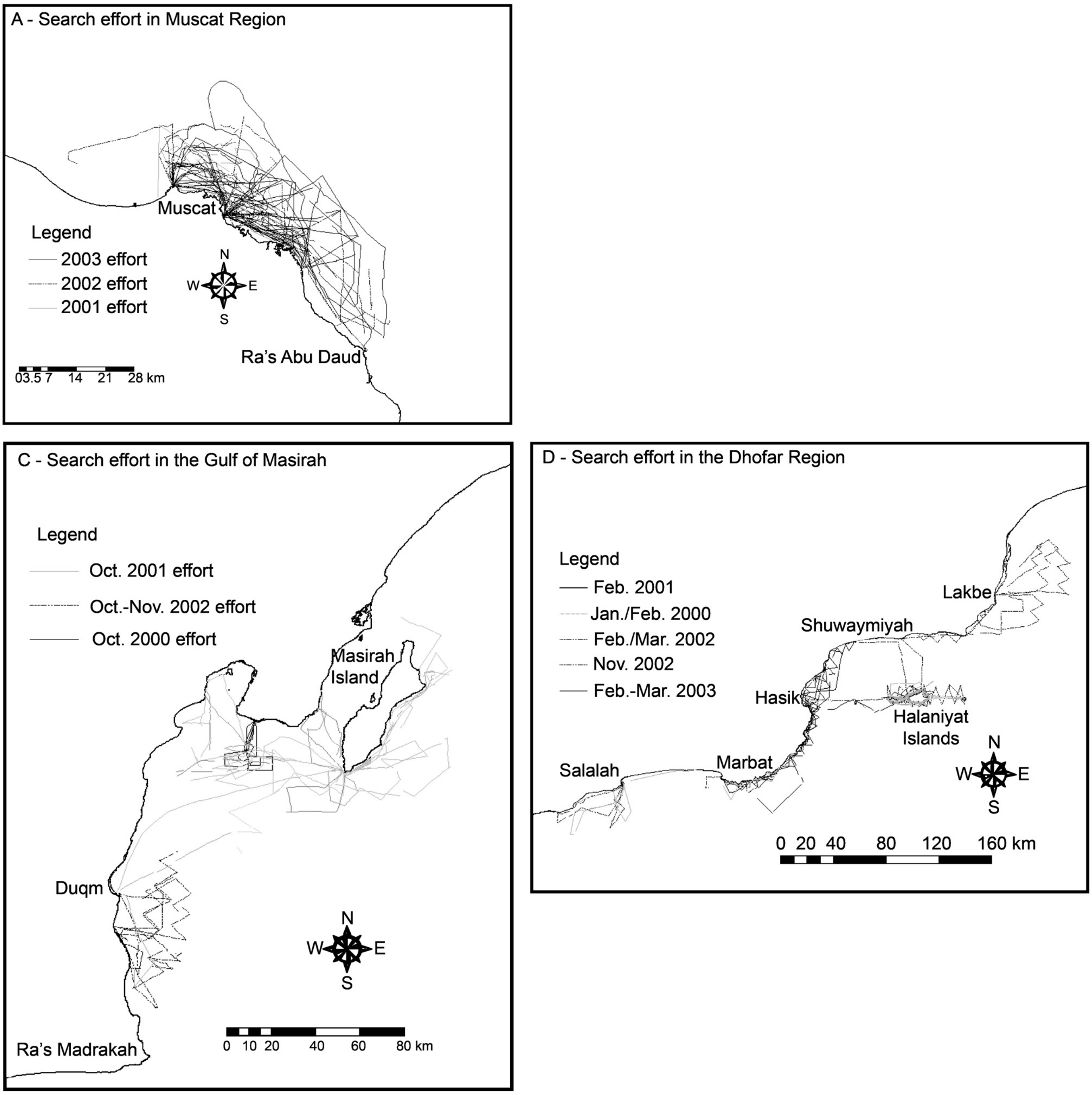


Fig. 1 (continued).

Table 1

Dates and locations of small boat surveys in Oman.

Survey area	Survey dates	Effort hours*
Muscat		
Monthly surveys	15 Mar. 2001–15 Jul. 2003	**104.21**
Dhofar		
Hallaniyat Islands	15–24 Jan. 2000/8–21 Feb. 2000	63.5
Dhofar	9–22 Feb. 2001	34.26
Dhofar	10 Feb.–2 Mar. 2002	62.37
Hasik Bay	24–26 Jun. 2002	4.32
Sharbitat and Hallaniyats	17–20 Nov. 2002	36.83
Dhofar	24 Feb.–19 Mar. 2003	116.31
Dhofar (Hasik only)	15–17 May 2003	2.17
Total		**319.76**
Gulf of Masirah		
N Gulf of Masirah	15–17 Oct. 2000	11
Gulf of Masirah	4–27 Oct. 2001	83.15
Gulf of Masirah	24 Oct.–16 Nov. 2002	58.2
Total		**152.35**
Other areas		
Ras al Hadd	30 Mar.–2 Apr. 2001	**8.13**
Shore-based observations		
Duqm	10–13 Jun. 2001	**25**

*Effort indicates time spent actively searching for whales and excludes time spent working with whales, in transit, or on breaks.

vessel's position recorded every 30–45 seconds, and these and other positional data were imported into ArcView® (3.2a) for viewing and analyses at the end of each day. Depth was measured using a boat-mounted Raytheon L365 fishfinder (range to 650m), or by referencing British Admiralty ARCS charts (see below).

Analysis

All sightings data made in the field were collated in an MS Access database (the Oman Cetacean Database, OMCD), along with other records of cetaceans from the region. For the purpose of this study, sightings and other data within the OMCD were classified into five effort categories:

Type 1 – sightings made while the vessel was on track during optimal search effort;

Type 2 – sightings recorded during surveys when at speed or with compromised observer effort;

Type 3 – sightings recorded while off effort during surveys or sightings made by the authors with no associated effort, as well as shore-based observations (February 2000, June 2001) and seismic survey data (e.g. Baldwin, 1997);

Type 4 – other incidental or dedicated sighting records from reliable third parties (incl. sightings associated with images, Ballance and Pitman, 1998; Mikhalev, 1997; Reeves *et al.*, 1991 and pers. obs. M.D. Gallagher 1970–1998; Salm *et al.*, 1993); and

Type 5 – incidental reports with detailed descriptions that support a species ID but cannot be confirmed by images.

Only Type 1 sightings were used in the calculation of encounter rates and relative densities in this study. Type 2–4 sightings were used in some behaviour and group composition analyses (see Minton *et al.*, in press) and also offer an additional source of data on species distribution outside survey areas and times. Type 5 sightings were taken into account when choosing survey areas, which were intended to target areas where humpback whales were known to occur, but were not included in any analyses.

Encounter rates were calculated for Type 1 sightings in three different ways: (1) Number of sightings per hour of search effort; (2) estimated number of individuals per 100km of survey trackline searched; and (3) number of cetaceans sighted per decimal degree searched in each 0.1 decimal degree (dd) × 0.1 dd cell. Grid cell size was determined as a compromise between accuracy in classifying habitat characteristics within grid cells, and the need for sufficient encounters within each cell to yield usable results (e.g. Hamazaki, 2002). On-effort portions of survey tracks were imported into ArcGIS and converted into shape files, one for each day's effort. These were plotted, and overlaid with a grid of 0.1 × 0.1 dd cells (approximately 11 × 11km). The geo-processing 'intersect' and 'dissolve' functions of ArcGIS were then used to calculate the total distance (in decimal degrees) surveyed on-effort in each cell.

Type 1 sightings data were imported into ArcGIS from the OMCD. The 'spatial join' function of ArcGIS was used to calculate the total number of groups and number of individuals in each cell for the most frequently encountered species (see Table 2 for species list). All Bryde's whale sightings are referred to as *Balaenoptera sp.* in this paper pending further analysis to determine species (e.g. Best, 2001; Sasaki *et al.*, 2006; Wada *et al.*, 2003).

Digitised depth files were generated for each of the four survey areas by creating points of known depth soundings from rasterised nautical charts (British Admiralty ARCS series, enabled with ARCS for GIS software – Intelliscan®). Kriging functions of ArcGIS Spatial Analyst were then used to interpolate these depth files and generate depth rasters with a mask applied to exclude terrestrial surfaces from grid cells overlapping the coast.

A further analysis was attempted by assigning each grid cell a depth and slope value, and plotting encounter rates against these to test for statistically significant relationships. However, initial approaches that would account for spatial auto-correlation did not yield statistically significant results, and are therefore not discussed in detail here but are the subject of ongoing analyses to be presented elsewhere.

Similarly, attempts were made to include statistical analysis of encounter rates in relation to remotely sensed chlorophyll-a and sea surface temperature data. The data obtained included 8-day averages of chlorophylla-a and SST during all the periods covered by our surveys. Initial inspection of the data revealed high seasonal and inter-annual variation in values for both of these parameters. For this reason the sample sizes of encounter rates per grid cell in relation to the time scale of the remotely sensed data (even if averaged for each survey) were too small to allow for meaningful statistical analysis.

RESULTS

Encounter rates and relative abundance

Surveys conducted between January 2000 and October 2003 comprised 585 hours and 8,840km of search effort. A total of 448 sightings of cetaceans were made, of which 304

Table 2

Number of on-effort (type 1 only) encounters/sightings and individual animals sighted per region, and encounter rates (for groups and individuals) per hour and per 100km searched.

Survey area		Muscat (monthly surveys 2000–03)	Dhofar (2000–03)	Gulf of Masirah (2000–03)	Ras al Hadd Apr. 2001	Total/average
Effort hours*		104.21	319.86	152.35	8.13	584.55
Total distance searched (km)		2,264.50	*3,819.71*	2,555.24	200.83	8,840.28
Bottlenose dolphins	Sightings	8	59	10	1	78
Tursiops sp.	Individuals	252	1,428	608	4	2,292
	Sighting/hour	0.08	0.18	0.07	0.12	0.11
	Individuals/100km	11.13	37.39	23.79	1.99	18.57
Common dolphins	Sightings	22	36	2	8	68
Delphinus capensis	Individuals	7,672	1,076	600	1465	10,813
	Sighting/hour	0.21	0.11	0.01	0.98	0.33
	Individuals/100km	338.79	28.17	23.48	729.47	279.98
Spinner dolphins	Sightings	32	2	2	4	40
Stenella longirostris	Individuals	8,130	420	200	1,375	10,125
	Sighting/hour	0.31	0.01	0.01	0.49	0.20
	Individuals/100km	359.02	11.00	7.83	684.65	265.62
Humpback dolphins	Sightings	0	29	5	0	34
Sousa chinensis	Individuals	0	234	116	0	350
	Sighting/hour	0.00	0.09	0.03	0.00	0.03
	Individuals/100km	0.00	6.13	4.54	0.00	2.67
Risso's dolphins	Sightings	3	3	0	0	6
Grampus griseus	Individuals	365	64	0	0	429
	Sighting/hour	0.03	0.01	0.00	0.00	0.01
	Individuals/100km	16.12	1.68	0.00	0.00	4.45
Bryde's whales	Sightings	5	1	4	0	10
Balaenoptera sp.	Individuals	7	2	6	0	15
	Sighting/hour	0.05	0.00	0.03	0.00	0.02
	Individuals/100km	0.31	0.05	0.23	0.00	0.15
Humpback whales	Sightings	0	33	23	0	56
Megaptera novaeangliae	Individuals	0	51	34	0	85
	Sighting/hour	0.00	0.10	0.15	0.00	0.06
	Individuals/100km	0.00	1.34	1.33	0.00	0.67

*Effort indicates time spent actively searching for whales and excludes time spent working with whales, in transit, or on breaks.

(68%) were Type 1 sightings. Encounter rates of sightings per hour of search effort, and estimated individuals per 100km searched are detailed in Table 2 and encounter rates of individuals in relation to distance searched per 0.1 dd × 0.1 dd cell are depicted in Figs 2a–g.

Fig. 3 shows the encounter rates (number of sightings per hour of search effort) for each survey region based on a compilation of all the survey effort in each region between January 2000 and October 2003. Additional species that were infrequently encountered on-effort were not considered. Similarly, encounter rates for the three-day Ras al Hadd survey held in March–April 2001 are not included in the figures.

Additional species observed

Sperm whales (*Physeter macrocephalus*) were encountered only once on-effort during surveys in the Dhofar region. Type 1–4 sightings recorded through 2003 included large groups of up to 25 animals, comprising mostly females and/or juveniles and only a few males. Of the 35 type 1–4 records of this species, 31 occurred in waters of 100m depth or greater. False killer whales (*Pseudorca crassidens*) were recorded only once on-effort during surveys, but were represented by 25 Type 1–4 sightings in the OMCD. Sightings were concentrated in the Muscat area and the Ra's Madrakah-Dhofar region. Reported group sizes ranged from 5 to 150, with calves confirmed on three occasions.

Only one blue whale (*Balaenoptera musculus*) was observed on effort, but the OMCD included a total of four confirmed records of this species up to October 2003, three from the Gulf of Oman (Muscat area), and one on-effort from Dhofar. The Muscat sightings occurred in the months of November and December, while the Dhofar sighting occurred in February. During all four sightings, the animals (three singletons and one trio) were observed to be milling in the same general area and diving for 3–10 minutes between surface intervals. Blue whales observed in Oman were all estimated to be under 20m in length.

Rough-toothed dolphins (*Steno bredanensis*) were observed only twice on effort during the period covered by these surveys. One of these sightings, a mixed group of Risso's and bottlenose dolphins was only identified as having included rough-toothed dolphins two years after the sighting when photographs were more closely examined.

Species associations

The most frequently observed association between cetacean species was that between common dolphins and spinner dolphins, which were observed in mixed groups in every survey area, but with highest frequency in Muscat ($n = 12$), where they were also observed feeding together. Common dolphins were the only species with which spinner dolphins associated, but common dolphins were also observed with bottlenose dolphins ($n = 2$). Bottlenose dolphins were seen in association with other species including Risso's dolphins ($n = 3$), Risso's and rough-toothed dolphins ($n = 2$), humpback whales ($n = 1$), and Indo-Pacific humpback dolphins ($n = 2$). One antagonistic inter-specific interaction

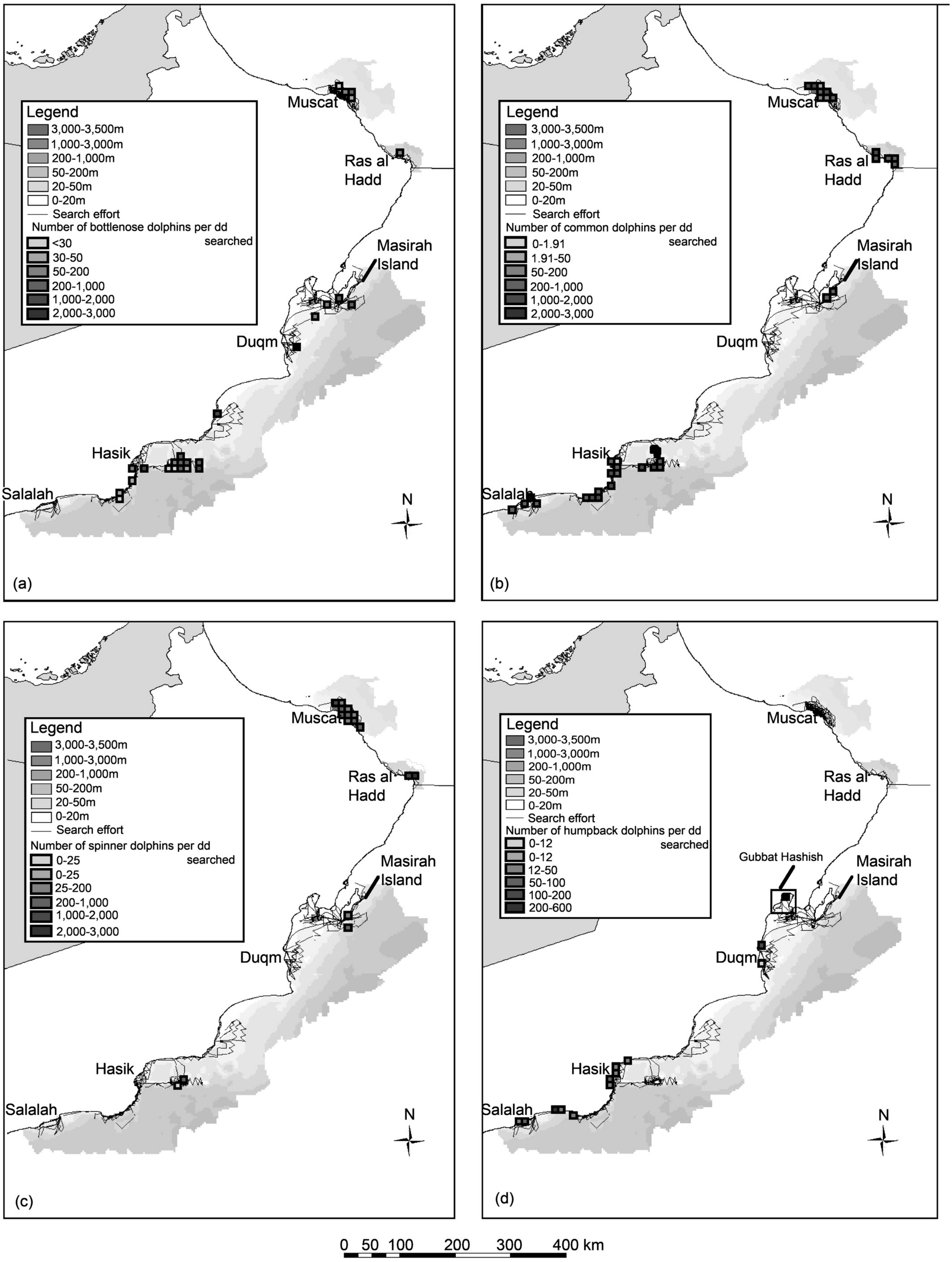
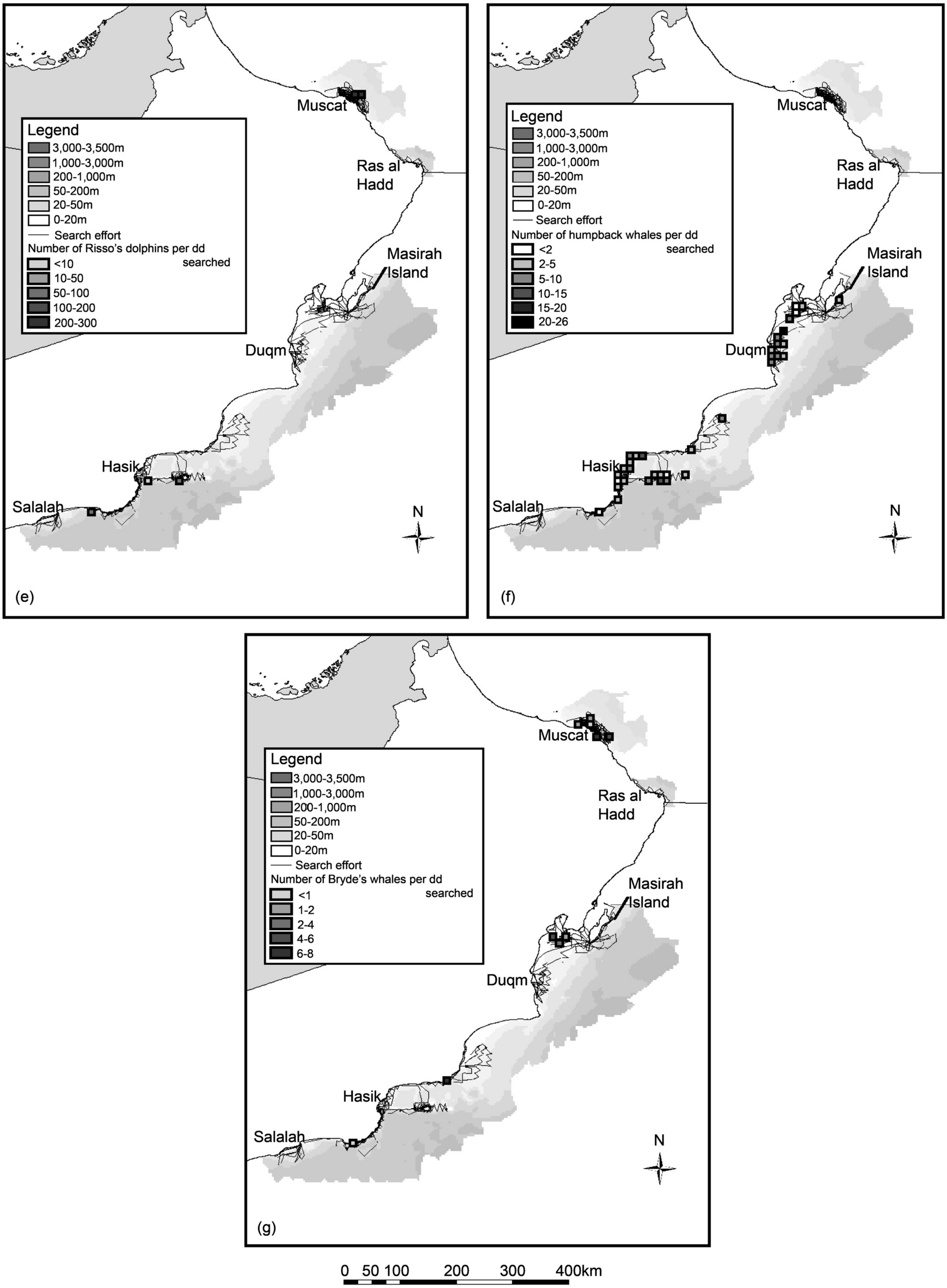


Fig. 2. Relative encounter rates per 0.1 × 0.1 decimal degree grid cell for the seven most frequently encountered species: (a) bottlenose dolphins, (b) common dolphins, (c) spinner dolphins, (d) humpback dolphins, (e) Risso's dolphins, (f) humpback whales, (g) Bryde's whales. Encounter rates were calculated as the total number of animals encountered in the grid cell divided by the distance searched (decimal degrees) in the grid cell.



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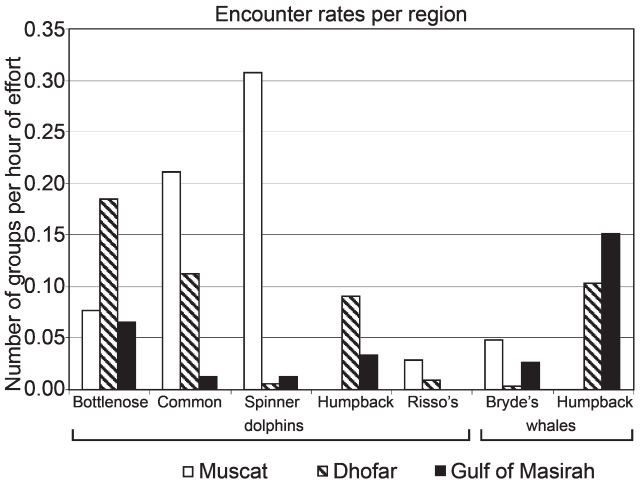

Fig. 3. Encounter rates (number of sightings per hour of search effort) in each region, for the most frequently sighted species.

was observed, when a group of 40 bottlenose dolphins were seen to harass a single Indo-Pacific humpback dolphin in Hasik, Dhofar, surrounding it, then body-slamming and biting it repeatedly over a period of thirty minutes.

DISCUSSION

Regional differences in relative abundance

The differences between survey areas in relative abundance of various species (highlighted in Table 2 and Figs 2a–g, as well as Fig. 3) are most likely linked to each species' demonstrated associations with specific depth and slope classes. Encounter rates for continental slope and deep water species such as spinner and Risso's dolphins (Baird, 2009; Perrin, 2009) were generally higher in the Muscat and Ras al Hadd regions, where a greater proportion of search effort was dedicated to grid cells in these categories. Conversely, nearshore and continental shelf species, such as Indo-Pacific bottlenose (Wang and Yang, 2009) and humpback dolphins (Parra and Ross, 2009) and humpback whales (Clapham, 2000) were encountered with greater frequency in the Gulf of Masirah and Dhofar, where a greater proportion of search effort was spent within the 200m isobath. However, there may be other factors influencing longshore distribution along the coast, such as upwelling-driven differences in water temperature and productivity, which were unfortunately beyond the scope of this study.

Sighting probability is also known to vary according to species' group size, body size, dive durations and surface behaviour (e.g. Mullin and Fulling, 2004), so it is likely that long-diving and/or cryptic species are under-represented in this study. Nonetheless, distributions for species reported here generally support those reported by Baldwin *et al.* (1999) and those determined for the same species in other regions. These are discussed in greater detail on a species-by-species basis below.

Bottlenose dolphins

Bottlenose dolphins were the most frequently encountered species across surveys. Relatively high encounter rates across all three major survey regions (Table 2; Fig. 2a), indicate that *Tursiops sp*. are distributed throughout both the Gulf of Oman and Arabian Sea coasts of Oman. Observations of this species included at least two readily distinguishable forms, likely representing the two recognised species, *T. truncatus* and *T. aduncus* (Reeves *et al.*, 2002).

Bottlenose dolphins observed in the Muscat and Ras al Hadd survey areas were large (with some individuals estimated to exceed 3m in length), stocky, heavily scarred, and blunt-nosed. They were most often encountered in deeper offshore waters and were associated with other deep-water species, especially Risso's dolphins (3 out of 8 survey sightings). These groups most likely represent *T. truncatus* (Peddemors, 1999; Wang *et al.*, 2000).

Bottlenose dolphins observed in nearshore regions of Dhofar (representing all but one of the 59 Dhofar sightings) were generally smaller (averaging an estimated 2m in length, with no individuals exceeding 2.5m), exhibiting a long slender rostrum, pronounced dark cape and prominent speckling on the ventral surface of adults when exposed. They were most often found within 1km of shore in depths averaging 9.4m (SD 5.2) if the deepest sighting (120m) in Dhofar is excluded from the sample. In the shallow waters of the Gulf of Masirah, bottlenose dolphins were also smaller, more slender and had long rostra, although the dorsal capes were not as pronounced as those in Dhofar. The preference for nearshore and/or shallow waters demonstrated by bottlenose dolphins in Dhofar and the Gulf of Masirah is more in keeping with the reported habitat preference for *T. aduncus*, which is in water depths of less than 30m (e.g. Findlay *et al.*, 1992; Wang and Yang, 2009).

Although the Dhofar and Gulf of Masirah bottlenose appear morphologically to be *T. aduncus* (Perrin *et al.*, 2007; Wang *et al.*, 2000), planned genetic analysis of biopsies taken from nearshore groups of bottlenose dolphins in Dhofar, as well as a large number of samples that have been collected from beach-cast specimens all along the coast of Oman, may help to shed light on the taxonomy and possible population divisions of bottlenose dolphins in Oman. Even so, taxonomy of bottlenose dolphin populations worldwide is confused (e.g. Hoelzel *et al.*, 1998; Kingston and Rosel, 2004), and it may be some time before species and/or sub-species are clearly defined.

Common dolphins

Common dolphins were the second most frequently encountered species across surveys, with an indicated continuous distribution along both the Gulf of Oman and Arabian Sea coasts of Oman. Recent morphometric analysis of skulls collected from the Arabian Region and elsewhere indicates that common dolphins in Oman are likely to represent the long-beaked form, *D. capensis tropicalis* (Jefferson and Van Waerebeek, 2002). However, these authors suggest a clinal distribution for *D. capensis capensis* and the sub-species *D.c. tropicalis*, the latter being most prominent (longest rostral length) off the Indian Subcontinent, with rostral lengths tapering toward the *D. c. capensis* form toward the east coast of Africa. It is therefore possible that some sightings off Oman represent *D. c. capensis* rather than the *tropicalis* form. Jefferson and Van Waerebeek (2002) also suggest that there may be some hybridisation between the two forms in regions of overlap. Group size ranged from 1 to 3,000, with 51 of the 68 on-effort sightings exceeding 100 individuals. Calves were observed in both the Dhofar and Muscat regions between December and February. Fig. 2b illustrates how encounter rates for this species peaked at the edge of the continental shelf in the 50–200m depth category. This is consistent with findings in other parts of the world (e.g. Cañadas and Hammond, 2008).

Spinner dolphins

Spinner dolphins were the fourth most frequently encountered species, with the highest encounter rates in the Muscat area (Fig. 2c). Morphologically, spinner dolphins observed off Oman may represent at least two forms or subspecies, with one form being slightly larger with a clear tripartite pattern, and the other being smaller with a less distinct pattern and more elusive habits. Morphometric analysis of skeletal material supports the hypothesis that Oman hosts a distinct form or subspecies, only slightly larger than the dwarf form described from Thailand (Van Waerebeek *et al.*, 1999), but the skeletal samples in the 1999 analysis could not be linked to external colouration on live animals, and neither field data nor genetic evidence are yet able to confirm the distribution or population identity of different forms in Oman.

Spinner dolphin distribution in the present study is consistent with that reported by Baldwin *et al*. (2000; 1999), but included sightings on the Arabian Sea coast of Oman (where Baldwin *et al.* reported no occurrence). As illustrated in Fig. 3, encounter rates for this species were much higher in the Muscat region (0.31 groups per hour) than in Dhofar or the Gulf of Masirah (both less than 0.01 groups per hour). Ballance and Pitman (1998) found spinner dolphins to be the most commonly encountered dolphin species in the deeper offshore areas areas of Oman covered by their study, and the higher encounter rates in the Muscat area may reflect the deeper, more steeply sloping nearshore coastline in that region. Estimated group size ranged from 20 to 700 individuals. Calves were observed in both the Dhofar and Muscat regions in the months of October and February.

Indo-Pacific humpback dolphins

Indo-Pacific humpback dolphins were encountered frequently along the Arabian Sea Coast of Oman, but not at all on the Gulf of Oman Coast (Fig. 2d). Type 3–4 sightings indicate a discontinuous distribution of this species, with one concentration found in the coastal waters around the Musandam Peninsula (Northernmost region of Oman and including the Straits of Hormuz – not included in our surveys), and another concentration south of Ras al Hadd on the Arabian Sea Coast. The strong preference for shallow inshore waters displayed by humpback dolphins in Oman is in keeping with distributions and habitat preferences reported in other parts of this species' range (e.g. Jefferson and Karczmarski, 2001; Karczmarski *et al.*, 2000; Parra, 2006).

Surveys conducted from 2000 through March 2002, contained little near-shore effort, while those conducted in Autumn 2002 and February–March 2003 included several days of dedicated near-shore effort. Encounter rates for humpback dolphins increased significantly during the latter surveys. Survey data and relative abundance calculations (Fig. 2d) indicate that certain areas are important for this species, including the shallow nearshore waters of the northern end of the Gulf of Masirah (particularly the 'Ghubbat Hashish' in the Northwest corner of the gulf), nearshore areas immediately to the North of Duqm, Hasik Bay in Dhofar, and the nearshore areas to the southwest of Salalah. The presence of small calves and the direct observation of feeding behaviour in all of these areas indicate that these are important feeding and breeding habitats.

The distribution, ecology, and taxonomy of this species is discussed in greater detail in Baldwin *et al.* (2004). Their reported sightings, together with those of this study, represent some of the largest group sizes ever reported for this species, with up to 100 individuals observed in a single aggregation. Additionally, Oman appears to hold the record body length for this species at 3.14m (a beach-cast male examined in December 2001).

Rough-toothed dolphins

Rough-toothed dolphins were not known to occur in Oman's coastal waters prior to 1998, when a previously misidentified partial skull was re-identified as *S. bredanensis* (Van Waerebeek *et al.*, 1999). Balance *et al*. (1996) recorded this species far offshore in their 1995 survey, but the two sightings made during this survey and a mass stranding of bottlenose and rough-toothed dolphins that occurred near Ras al Hadd in January 2002 (Collins *et al.*, 2002), are the first documented sightings of this species in Oman's coastal waters.

Risso's dolphins

Records of Risso's dolphins span both the Gulf of Oman and Arabian Sea coasts of Oman. This species was generally sighted in deeper waters further offshore (Fig. 2e), which agrees with distributions reported elsewhere in the world (e.g. Baumgartner *et al.*, 2001; Cañadas *et al.*, 2002; Hamazaki, 2002). Estimated group size of all Type 1–4 sightings ranged from 12 to 800, and on at least two occasions the species displayed a 'tail up' behaviour' where a large portion of the group would be stationary in the water, with tails exposed, for several minutes at a time.

Sperm whales

Soviet whaling fleets took a total of 954 sperm whales from the Arabian Sea between 1963 and 1967 (Mikhalev, 2000). While distribution maps show that few whales were taken off Oman in comparison to the Gulf of Aden and offshore waters at lower latitudes, some catches were attributed to the Arabian Sea coast of Oman. Mikhalev reported that foetus lengths from 121 pregnant females indicated that sperm whales in the region were adhering to a Northern Hemisphere breeding cycle, though his conclusions were based on a postulated 11–12 month gestation period for this species, contrary to the 15-month period accepted by most other researchers.

It is interesting to contrast the paucity of recent sightings of this species with the findings of Ballance and Pitman (1998) who found sperm whales to be the most frequently encountered cetacean species of their 1995 survey. This discrepancy is likely due to the concentration of nearshore effort in our surveys in Oman compared with the predominantly offshore nature of the 1995 Ballance and Pitman survey, a theory supported by the fact that all the

Type 1–4 records of this species in Oman are from water depths 100m or greater.

Humpback whales

Seasonal distribution, habitat use and ecology of this species are discussed in greater detail in Minton *et al.* (in press), and the discussion here will be limited only to the present study's implications for relative abundance and possible habitat preferences. With 56 on-effort sightings, humpback whales were the third most frequently encountered species on surveys. Surveys were designed to maximise encounters with humpback whales, targeting areas of suspected abundance on the Arabian Sea coast (the Gulf of Masirah and Dhofar).

The nearshore distributions demonstrated in Fig. 2f concur with the habitat preferences reported by Hamazaki (2002), who classified humpback whales as a 'North Atlantic Shelf Species' preferring depths of less than 400m. Moore *et al.* (2002) also found that humpback whales were more likely to occur on the 'middle' shelf, near the 50m contour. The concentration in nearshore/island areas in Oman is the most likely reason that this species was not observed at all by Ballance and Pitman (1998), whose survey concentrated on deeper offshore regions.

Although a few opportunistic sightings and anecdotal evidence indicate that humpback whales can be found in the Gulf of Oman, survey data reveal that the species is more abundant off the Arabian Sea Coast of Oman, and additionally suggest that the targeted survey areas, the Gulf of Masirah and the 'Kuria Muria Bay' of the Dhofar region (the area surrounding the Halaniyat Islands), are of particular importance for this population. Consistently high chlorophyll-a values in the Gulf of Masirah indicate high levels of productivity (Brock and McClain, 1992; Brock *et al.*, 1998; Marine Science and Fisheries Center Oman, 2001) and it seems likely that this region is an important feeding ground for humpback whales throughout the year. Variation in encounter rates between survey years and their possible relationship to seasonal and annual variations in sea surface temperature and chlorophyll-a concentrations are also discussed further in Minton *et al.* (in press).

Bryde's whales

Recent genetic analysis of sloughed skin samples and tissue samples collected from beach-cast whales, suggests the majority, if not all, Bryde's whales sighted in Oman to date are currently considered to be of the inshore form of Bryde's whale (*Balaenoptera edeni*) (T. Collins, unpublished data). Confirmed sightings have been recorded in every month except July, suggesting the species is resident off the coast of Oman (Baldwin *et al.*, 2000; Mikhalev, 2000). This species was heavily hunted in the Arabian Sea between 1963 and 1966 (Mikhalev, 2000). Full stomachs observed in these Soviet catches, coupled with direct observations of feeding activity and mother-calf pairs during our surveys indicate that the coastal waters of Oman may serve as both a breeding and feeding ground for this species as well as for humpback whales. Fig. 2g demonstrates how sightings of this species were limited to nearshore shallow waters, less than 50m depth, and how encounter rates were highest in the Gulf of Masirah and the southern portion of the Muscat survey areas. In Fig. 2g the southernmost sighting was a blue whale (*B. musculus*) while all others are likely to be the inshore form of Bryde's whale.

Blue whales

Blue whales were heavily hunted by the Soviet Union between 1963 and 1966, with a total of 1,294 whales taken from the Arabian Sea (Mikhalev, 1996; Mikhalev, 2000). These catches were identified as pygmy blue whales (*B. musculus brevicauda*) (Mikhalev, 1996; Mikhalev, 2000). Mikhalev (2000, p.149) provided a breakdown of catches per region within the Arabian Sea, but the 'Aden-Omani' region is combined ($n = 106$), and it is not clear how many of these animals came from the Omani coast. Mikhalev's (2000, p.144) distribution maps show three main areas of concentration within the Arabian Sea, including the Gulf of Aden, offshore from the southeastern tip of the Indian continent, and just below the equator offshore from Somalia.

Blue whale sightings recorded by Ballance and Pitman (1998) were concentrated around Sri Lanka and the Maldives. Anderson *et al.* (1999) reported strandings and sightings of blue whales in Maldivian waters, while Small and Small (1991), Alling *et al.* (1982), and Eyre (1995) all reported on sightings of blue whales during their surveys in the Northern Indian Ocean. It is unclear whether the paucity of sightings of this species off the coast of Oman is due to their prey preferences or reduced numbers from heavy Soviet whaling in the mid 1960's. The distribution of blue whales throughout the Southern Hemisphere and Indian Ocean is discussed in greater detail in Branch *et al.* (2007).

Other species

Additional species are known to occur in Oman, but were not encountered during surveys, and are therefore not described in detail here. These include the pantropical spotted dolphin (*Stenella attenuata*) (Baldwin *et al.*, 2000; Ballance and Pitman, 1998), the striped dolphin (*Stenella coeruleoalba*) (Baldwin *et al.*, 2000), the pygmy killer whale (*Feresa attenuata*) (Alling, 1986; Baldwin *et al.*, 2000), the melon-headed whale (*Peponocephala electra*) (Van Waerebeek *et al.*, 1999) and the killer whale (*Orcinus orca*) (Baldwin *et al.*, 2000). Reports of minke whale (*Balaenoptera acutorostrata*) and fin whale (*Balaenoptera physalus*) sightings off the coast of Oman in Baldwin *et al.* (1999) are not supported by photographic evidence, and the presence of these species in the region has since been suggested as highly unlikely (Baldwin, 2003). In addition, the Soviet catch data for the Arabian Sea referred to blue and Bryde's whales only (Mikhalev, 2000; Yukhov, 1969), and it seems unlikely that they would have overlooked minke or fin whales if they had been present in the region. It is possible that past sightings of minke and fin whales represented vagrants within the Arabian Sea, but it is more likely that they were misidentified.

Problems of spatial and temporal scale

Hamazaki (2002) discusses the limitations and potential biases of analysing cetacean distribution derived from cetacean surveys as a means of identifying critical or preferred habitats. One limitation is the sighting data themselves. Statistical analyses of relationships between cetacean sightings and habitat characteristics make the assumption that sightings are made in the cetaceans' preferred habitats. However, determining whether cetaceans are in their preferred habitat when sighted or in transit between one preferred habitat and another is not always possible.

Furthermore, while depth and slope are constant habitat characteristics that do not change from one survey period to the next, other environmental characteristics, such as chlorophyll-a concentrations and sea surface temperature (SST) can be highly variable between survey periods. Some studies have assumed that average seasonal values will not vary significantly over several decades (e.g. Gregr and Trites, 2001), or between survey years (e.g. Hamazaki, 2002), and have used averaged values of remotely sensed data from one particular period (deemed 'most seasonally typical') as a basis of comparison against several years' worth of sightings data. In light of the considerable inter-annual variability shown in oceanographic characteristics of the Arabian Sea (e.g. Brock and McClain, 1992), as well as in the SST and chlorophyll-a obtained for the periods of this study, this approach is not possible for Oman.

Furthermore, while instantaneous data may be available for the time of survey sightings, cetaceans may not respond to instantaneous changes of ocean conditions. There may be a significant time lag between an upwelling or algal bloom and a subsequent increase in cetacean prey availability. This time lag may affect each cetacean species differently, necessitating a detailed understanding of the life cycle and feeding preferences of a cetacean species' prey base (e.g. Baumgartner *et al.*, 2003a; Baumgartner *et al.*, 2003b) in order to make accurate assumptions. Such extensive knowledge of prey preferences and prey characteristics is not yet available for any cetacean species in Oman.

In Oman, baleen whales are documented to feed only on fish (Mikhalev, 2000), and very little is known about the preferred prey of other cetacean species. Stomach content analyses of beach-cast and by-caught cetaceans will help to yield more information on prey preferences, and consequently help direct future research on habitat preferences (Ponnampalam *et al.*, 2007).

CONCLUSIONS

More extensive surveys of Oman's coastal and offshore waters are required to obtain more representative seasonal coverage and to include regions that were not accessible during the three years of survey effort discussed here. It is likely that future surveys will identify additional (seasonal) habitats of key importance to different cetacean species. Alternative survey methodologies, such as ship-based or aerial line-transect surveys, are also required in order to obtain absolute rather than relative abundance estimates for cetacean species in Oman. However, continued surveys geared toward collecting data on relative abundance will still yield valuable information on distribution and population trends, vital for future management and conservation efforts. In particular, analysis of relative abundance for the most frequently encountered species in this study indicates that the nearshore areas of the Gulf of Masirah (particularly the Ghubbat Hashish in the northern Gulf, and Duqm Bay), and Dhofar (particularly Hasik Bay) are of importance for Indo-Pacific humpback dolphins, while the waters just slightly further offshore from these same areas, with the addition of the Hallaniyat Islands, appear to comprise critical habitat for the Arabian Sea subpopulation of humpback whales, designated by the IUCN as Endangered in 2008 (Minton *et al.*, 2008). This population's ecology is discussed in greater detail in Minton *et al.* (in press). These areas should be considered a priority for future cetacean conservation and management measures.

Research to date has shown that at least four cetacean species occurring in Oman (spinner dolphins, Indo-Pacific humpback dolphins, humpback whales and blue whales), may be undescribed sub-species and/or isolated breeding stocks (Branch *et al.*, 2007; Pomilla *et al.*, 2006; Rosenbaum *et al.*, 2002; Rosenbaum *et al.*, 2009; Rosenbaum *et al.*, 2006; Van Waerebeek *et al.*, 1999). For these, and other as yet less studied species, continued research to refine population affiliations further and to obtain absolute abundance estimates and trend data is critical in a country which is undergoing rapid population expansion and development. Growth in hydrocarbon, shipping, fisheries, port/harbour and tourism sectors, all place direct pressure on cetaceans and their habitats. Since 2003, there have been notable increases in such pressure, including that in areas noted here as important habitat for some species. Oman's coastline already includes some of the busiest shipping lanes in the world (Ghose, 2010), a trend which is likely to increase significantly as a large industrial port is currently under construction in the area around Duqm on the Arabian Sea coast – the area coinciding with the highest encounter rates for humpback whales. High speed ferry services are being introduced in various parts of Oman, including in the Dhofar region where a service will run to and from the Hallaniyat Islands (Vaidya, 2008), where the only confirmed sightings of humpback whale mother-calf pairs were made and humpback whale song was recorded frequently. Furthermore, the number of registered fishing vessels has increased rapidly, more than doubling from just over 5,500 in 2006 to over 11,000 in 2008 (Ministry of National Economy, 2009). Without more extensive data and urgent management intervention, the possibly isolated and unique populations of cetaceans in Oman may not be sufficiently protected from these increasing threats.

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Odontocetes of the Southern Ocean Sanctuary

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ABSTRACT

Twenty-eight odontocete species were identified as occupying sub-Antarctic and Antarctic habitat covered by the 1994 IWC-established Southern Ocean Sanctuary. Toothed whales evidently play an important part in the Antarctic polar ecosystem. Twenty-two species are autochthonous in showing a regular, apparently year-round, presence in the Sanctuary: *Physeter macrocephalus, Kogia breviceps, Orcinus orca, Globicephala melas edwardii, Pseudorca crassidens, Lagenorhynchus cruciger, Lagenorhynchus obscurus, Lissodelphis peronii, Cephalorhynchus commersonii, Cephalorhynchus hectori, Tursiops truncatus, Delphinus delphis, Phocoena dioptrica, Hyperoodon planifrons, Berardius arnuxii, Ziphius cavirostris, Tasmacetus shepherdi, Mesoplodon layardii, Mesoplodon traversii, Mesoplodon grayi, Mesoplodon bowdoini* and *Mesoplodon hectori.* Six species are considered vagrants into the Sanctuary*: Kogia sima, Grampus griseus, Steno bredanensis, Mesoplodon peruvianus, Mesoplodon densirostris* and *Mesoplodon mirus*. However, vagrant status of these three mesoplodonts is only provisionally assigned, considering that improved knowledge of diagnostic features of beaked whales should, as in recent years, continue to facilitate at-sea identification. Two species are considered as having a 'contiguous' range (records less than 2° north of Sanctuary boundaries): *Mesoplodon ginkgodens* (at 39°S) and *Mesoplodon mirus* (at 38°24'S). The habitual southern range of at least four odontocetes extends significantly farther poleward than expected. *G. melas edwardii* is regularly encountered south of the Antarctic Polar Front, much like *M. grayi* which is known to reach the Ross Sea ice edge (*ca*. 67°S). *Z. cavirostris* and *L. obscurus* cross the Polar Front occasionally. The distribution of *M. peruvianus* and *M. traversii* and their relation to SST are unclear. Their southernmost records, 42°31'S and 44°17'S respectively, may either be extralimital or, more likely, reflect ordinary austral range. Temporally non-aligned distribution patterns of *Hyperoodon planifrons* in Antarctic and South African waters may suggest stock segregation.

KEY WORDS: DISTRIBUTION; MOVEMENTS; ODONTOCETES; BEAKED WHALES; ANTARCTIC; HABITAT; SOUTHERN HEMISPHERE; SANCTUARY; SURVEY-VESSEL

INTRODUCTION

The odontocetes of the Southern Ocean are relatively poorly known compared to the baleen whales. Commercial whaling in the Southern Ocean during the 20th century largely concentrated on baleen whales and the sperm whale. Catches of other odontocete species, particularly of the southern bottlenose whale (*Hyperoodon planifrons*) and Arnoux's beaked whale (*Berardius arnuxii*) often collectively referred to as 'bottlenose whales' and the killer whale *Orcinus orca* were much smaller and conducted on a largely opportunistic basis (e.g. Klinowska, 1991; Mitchell, 1975a; 1975b). The ecology of odontocetes within the Southern Ocean ecosystem is very different (due to the greater variety of their prey species) and more complex than the baleen whales whose diets are dominated by krill (Euphausiidae). Thus it is likely that odontocetes will respond very differently to physical and biological factors. In particular, the killer whale is unique in terms of its predatory interactions on other cetacean species, particularly minke whales (e.g. Branch and Williams, 2006; Jefferson *et al.*, 1991).

In addition to direct takes, human activities may also impact on odontocetes in different ways to baleen whales in the Southern Ocean. Whereas fishing removals of krill (*Euphausia superba*) are currently relatively low compared to biomass, fishing mortalities for some fish species have been high (Constable *et al.*, 2000). Some odontocetes also show a very high degree of site affiliation and may spend their entire lives within a very limited geographical area. Such limited ranges have been a consideration in designating Marine Protected Areas (MPAs) specifically for odontocetes in other regions, e.g. northern bottlenose whales (*Hyperoodon ampullatus*) in the Gully, Nova Scotia (Hooker *et al.*, 2002) and several populations of bottlenose dolphins (e.g. Evans and Pascual, 2001).

On 26 May 1994 the International Whaling Commission (IWC) at its 47th Annual Meeting, in Puerto Vallarta, Mexico, voted[8] for the creation of a whale sanctuary in the Southern Ocean. This provided for a prohibition on commercial whaling, to be reviewed at successive 10 year intervals, with the first review completed in 2004 (IWC, 2004). The northern boundary of the Southern Ocean Sanctuary (further 'the Sanctuary') was set at 40°S except between two longitudinal sections, one (50°W–130°W) in the eastern South Pacific and western South Atlantic, where the northern boundary was set at 60°S thus 'cutting out' South American waters, and the other (20°E–130°E) where the northern boundary was set at 55°S (which is the southern boundary of the existing Indian Ocean Sanctuary). The

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[8] A Schedule amendment of the International Convention for the Regulation of Whaling (ICRW) requires a 75% majority vote.

present paper[9] aims to summarise and update information on distribution for each odontocete species inhabiting the Sanctuary as a first step towards implementing the recommendation from the IWC Scientific Committee for systematic 'inventory' programmes (IWC, 2005).

The most comprehensive set of cetacean surveys in the Southern Ocean are the IWC/IDCR (International Decade of Cetacean Research) and IWC/SOWER (Southern Ocean Whale and Ecosystem Research) programmes which have involved an annual effort since 1978 (Matsuoka *et al.*, 2003). In addition, the increase in multi-disciplinary research cruises in the Sanctuary since 1994 has yielded new data on the distribution of odontocete species from opportunistic observations, visual and acoustic surveys (Gillespie, 1997; Leaper *et al.*, 2000; Leaper and Scheidat, 1998; Pierpoint *et al.*, 1997; Rendell *et al.*, 1997; Thiele, 2002; 2004; 2000; Thiele and Gill, 1999; Thiele *et al.*, 1997; 2001; 2002; 2003). While it is still not always possible to identify beaked whales to species level, the multi-disciplinary data allows examination of some of the factors that may relate to odontocete distribution. Only a handful of earlier papers discussed the distribution of ziphiids and other odontocetes of the Southern Ocean in a broader perspective (Baker, 1990; Brownell, 1974; Goodall and Galeazzi, 1985b; Kasamatsu *et al.*, 1988; Kasamatsu and Joyce, 1995; Lillie, 1915; Liouville, 1913; Miyazaki and Kato, 1988; Nishiwaki, 1977; Ohsumi *et al.*, 1994; Paulian, 1953; Sapin-Jaloustre, 1953). Abundance estimates and associated caveats for the most frequently sighted odontocete species (Leaper *et al.*, 2008) were reviewed at a joint workshop of the IWC and the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR) in 2008 (IWC and CCAMLR, 2010).

MATERIAL AND METHODS

The occurrence of odontocetes within the Southern Ocean Sanctuary was reviewed and grouped in two categories: autochthonous (regular, probably year-round presence) and vagrant species (with three or less confirmed records[10]). Species that have been found in waters less than 2° latitude north of the Sanctuary boundaries were classified as 'contiguous occurring'. Three items were addressed for each species, each largely limited to information applicable to the study area: (i) systematics and populations; (ii) distribution in the Southern Ocean Sanctuary; and (iii) conservation status and exploitation. Conservation status designations are updated, for CITES up to 23 June 2005 and for CMS up to 25 November 2005 (8th Conference of the Parties, Nairobi[11]). IUCN status follows the 2008 IUCN Red List of Threatened Species[12].

The IDCR/SOWER dataset represents a large annual survey effort in the region and a potential source of abundance estimates. These cruises surveyed a different longitudinal sector of the Southern Ocean south of 60°S in each season, resulting in a total of three full sets of circumpolar surveys where each sector was surveyed at least once. These surveys are referred to as CPI, CPII and CPIII, covering the periods 1978/79–1983/84, 1985/86–1990/91 and 1991/92–2003/04 respectively. Some circumpolar abundance estimates for odontocetes have been generated using these data (Branch and Butterworth, 2001; Kasamatsu and Joyce, 1995). However, both papers note a number of caveats to their estimates. These caveats include, uncertainty in the proportion of animals directly on the trackline that are detected ($g(0)$), uncertainty in identification to species level (primarily a concern for beaked whales), and responsive movement (primarily a concern for hourglass dolphins). Kasamatsu and Joyce (1995) used a model of diving behaviour to estimate $g(0)$ for sperm whales (0.32), beaked whales (0.27), killer whales (0.96) and pilot whales (0.93). There are currently limited data from the Southern Ocean to refine these estimates or estimate $g(0)$ directly for these species. Branch and Butterworth (2001) noted that in the three sets of circumpolar surveys only 5%, 60% and 71%, respectively, of the beaked whale sightings were identified to species level. These changes in the attention given to species identification of beaked whales will have particular importance for estimates and distribution patterns of the less common species. Changes in the ice edge, latitudinal coverage and timing of the surveys also need to be considered when interpreting changes in distribution or abundance. The timing of surveys from 1994/95 to 2000/01 was later than in earlier years. Unpublished data from the IDCR/SOWER cruises, distributional data spanning the seasons 1978/79–2003/04, and unpublished data from the 'Southern Ocean Cetacean Ecosystem Program' (SOCEP) and the IWC's Southern Ocean Collaboration Working Group Program (IWC SOC) and associated cruises are included in this review. Although largely focussed on baleen whales, all cetacean species are recorded during these surveys. All odontocete records from these programmes collected up to the 2003/04 season were included in this review.

The SOCEP programme has been funded by the Australian Government since 1995/96 in direct response to the declaration of the IWC Southern Ocean Sanctuary. Its primary objective is to conduct visual survey, tissue biopsy, individual photo-identification and passive acoustic studies on cetaceans in the Sanctuary, alongside multidisciplinary research aimed at understanding the dynamics and variability in Antarctic marine ecosystems. Visual cetacean SOCEP surveys were conducted in East Antarctica (60°E–150°E) from 1995/1996 to 2002/2003. Data collected on this programme in the 1995/96 season have been published in Gill and Thiele (1997), Thiele and Gill (1999), Nicol *et al.* (2000) and Thiele *et al.* (2000). For a listing of the seventeen SOCEP survey cruises between July 1995 and March 2003, see Van Waerebeek *et al.* (2004).

The IWC commenced collaborative research with CCAMLR (IWC/SO GLOBEC/CCAMLR) in the Southern Ocean during the 1999/2000 austral summer (Hedley *et al.*, 2001; Reilly *et al.*, 2000). This initial cruise included a dedicated passive acoustic survey for odontocetes from one vessel (Leaper *et al.*, 2000). In 2001 a multi-year series of collaborative research cruises between the IWC and a

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[9] An earlier version was presented to the Scientific Committee at IWC/56 as a background document to facilitate the 2004 IWC Southern Ocean Sanctuary review discussions (Van Waerebeek *et al.*, 2004).

[10] Exceptionally, despite a single record, *Mesoplodon traversii* is considered autochthonous, because the record is one of only three known for the species and originates from deep inside the Sanctuary (*ca.* 475km, at Chatham Islands).

[11] *http://www.cms.int/bodies/cop_mainpage.htm*

[12] *http://www.iucnredlist.org*

Table 1

Odontocete sightings south of 60°S during IWC-SOC surveys 2000/2001 to 2003/2004. Sightings of *Orcinus orca* were not identified to the level of the A, B, C and D morphotypes known from Antarctica (Pitman and Ensor, 2003; Pitman *et al.* 2007) and likely represent an aggregate of different species.

Voyage	GMT date	Lat south	Long west	Species code	Species	Group size best
LMG 0201A	28/02/02	60.074	63.053	5	*P. macrocephalus*	1
NBP0202	30/04/02	68.902	69.688	10	*Orcinus orca*	4
NBP0204	11/08/02	68.735	76.092	10	*Orcinus orca*	1
LMG 0302	24/02/03	68.556	70.798	10	*Orcinus orca*	12
LMG 01-03	29/03/01	68.261	70.988	10	*Orcinus orca*	10
NBP0202	12/05/02	68.220	69.812	10	*Orcinus orca*	3
LMG 0201A	18/02/02	68.049	69.389	10	*Orcinus orca*	25
LMG0203	30/04/02	67.928	69.223	10	*Orcinus orca*	10
LMG0203	11/05/02	67.693	69.319	10	*Orcinus orca*	12
LMG0203	11/05/02	67.649	69.485	10	*Orcinus orca*	6
LMG0203	19/04/02	67.450	67.746	10	*Orcinus orca*	12
LMG 01-03	05/04/01	67.412	67.815	10	*Orcinus orca*	30
LMG 0302	25/02/03	67.207	67.724	10	*Orcinus orca*	4
LMG0203	14/05/02	64.843	63.921	10	*Orcinus orca*	4
NBP0104	27/08/01	64.735	63.071	10	*Orcinus orca*	3
LMG 0302	03/03/03	64.256	62.733	10	*Orcinus orca*	22
LMG 01-03	23/03/01	64.093	61.808	10	*Orcinus orca*	6
LMG 01-03	23/03/01	63.725	61.339	10	*Orcinus orca*	8
LMG 01-03	07/04/01	65.840	65.184	11	*Ziphiidae*	1
LMG 01-03	23/03/01	64.174	61.856	11	*Ziphiidae*	4
LMG 0302	20/02/03	65.980	71.052	13	*L. cruciger*	3
LMG 01-03	01/04/01	65.502	70.297	13	*L. cruciger*	4
Polarstern ANTXVIII5b	17/04/01	65.136	70.969	13	*L. cruciger*	10
MG 0201A	09/02/02	60.632	62.850	13	*L. cruciger*	2
NBP0202	12/04/02	60.574	65.198	15	*Unidentified dolphin*	1
LMG 0201A	28/02/02	61.627	62.442	38	*Mesoplodon sp.*	
LMG 0201A	28/02/02	60.309	62.978	38	*Mesoplodon sp.*	5
LMG 0201A	28/02/02	60.076	63.049	38	*Mesoplodon sp.*	3

number of nations began with the Southern Ocean GLOBEC programme. The cruises are multidisciplinary and comprise passages for deployment of moorings, line transect surveys over a constant grid, and process studies at selected locations, within the Western Antarctic Peninsula study region in the vicinity of Marguerite Bay; and in the Ross and Weddell Seas. Their objective is to define the influence of spatial and temporal variability in the physical and biological environment on cetacean distribution (IWC, 2000, p.346) Visual surveys, passive acoustic monitoring and tissue biopsy collection were conducted by IWC SOC observers and collaborating passive acoustics scientists (see table 2 in Van Waerebeek *et al.*, 2004 for more cruise details). Odontocete sightings south of 60°S collected during IWC SOC surveys 2000/2001 to 2003/2004 are listed in Table 1.

The Antarctic Convergence and West Wind Drift are here substituted by the synonymous but currently preferred terminology of Antarctic Polar Front (Orsi *et al.*, 1995) and Antarctic Circumpolar Current (ACC), respectively. South Island and North Island refer to New Zealand. Species status designations under the IUCN Red List (*http://www.iucnredlist.org*), CITES (*http://www.cites.org/eng/app/appendices.shtml*) and CMS (*http://www.cms.int/documents/appendix/Appendices_COP9_E.pdf*) are valid as of February 2010. Frequently used terms are abbreviated as SST (sea surface temperature), NZ (New Zealand), SH (Southern Hemisphere) and ESU (evolutionary significant unit).

AUTOCHTHONOUS ODONTOCETES

We found 22 species of odontocetes as being autochthonous in the Sanctuary, as outlined in the following species accounts.

Sperm whale *Physeter macrocephalus* (Linnaeus 1758)

Systematics and populations

No subspecies are described, geographical morphological variation is minimal and mtDNA is remarkably homogeneous (Dufault *et al.*, 1999; Machin, 1974; Whitehead, 2002). Since 1973, sperm whales of the Southern Hemisphere have been divided in nine stocks or 'divisions'. The boundaries of some of these divisions were called into question but no conclusive assessments were made (Donovan, 1991). It is highly unlikely that management stocks defined by such boundaries would reflect biological population structure.

Distribution in Southern Ocean Sanctuary

The distribution of sperm whales in the Southern Ocean (Fig. 1) is better documented than for other odontocetes. Of the great whales, sperm whales were second only to fin whales in terms of the numbers of individuals (over 400,000) taken by 20th century whaling operations (Clapham and Baker,

Plate 1. Sperm whale (*Physeter macrocephalus*) (PHE52190013). Photo credit: Paul Ensor. All photos by Paul Ensor were collected using camera equipment provided by Canon NZ Community Sponsorship Programme.

2001) in the Southern Hemisphere. Sperm whales are also relatively easy to detect and identify when at the surface and so there are considerable data on distribution from sightings surveys, although abundance estimates are still complicated by the long dive times of the species. More recently, passive acoustic techniques have proven effective for sperm whale surveys in the Southern Ocean (Gillespie, 1997; Leaper *et al.*, 2000). Globally, the sperm whale is known as a deep water species and similar distribution patterns in relation to water depth and bottom topography are seen in the Antarctic (Kasamatsu *et al.*, 2000). Kasamatsu and Joyce (1995), reviewing data from sightings surveys conducted between 1976/77 and 1987/88, reported highest encounter rates in the Indian Ocean sector with highest densities in the area bounded by 62°–66°S, 90°–120°E and south of 66°S, 150°–180°E. These results are consistent with more recent data from acoustic surveys where Gillespie (1997) reported densities some 2–3 times greater for the area 62°–66°S, 80°–125°E compared to the densities reported by Leaper *et al.* (2000) for the Scotia Sea.

Tynan (1998) used historic catch data to show the influence of the Southern Boundary of the Antarctic Circumpolar Current on sperm whale distribution. The circumpolar distribution appears to follow the Southern Boundary, with sperm whales concentrating at higher latitudes in the Indian Ocean than the South Atlantic and tracking the increasing southern penetration of the Southern Boundary between 20°E and 60°E. Tynan noted that regions in which sperm whales occurred in greatest numbers in the 1950s lie along or to the north of the Southern Boundary and suggested that sperm whales migrate southward as far as the poleward extent of Upper Circumpolar Deep Water. Thiele *et al.* (2000) supports these findings with data from a large scale survey (80°–150°E) with concentrations of sperm whales found along the Kerguélen Plateau. SOCEP data also shows concentrations of this species near frontal zones and eddies associated with the area south of the Southern Boundary, and also well south of this zone in association with the shelf slope and other areas of complex bathymetry (D. Thiele, unpublished data). Data collected on East Antarctic SOCEP surveys (1995/96–2003/04) also show sightings concentrated at 60°E–117°E and in January, and the southernmost record from these surveys is 66°32'S, 64°30'E (D. Thiele, unpublished data).

South of 66°S, Kasamatsu and Joyce (1995) reported high densities of sperm whales between 150°–180°E with sightings as far south as 74°S in the Ross Sea. Thus concentrations of sperm whales do occur to the south of the Southern Boundary.

The distribution of female sperm whales is generally limited to the tropics and warm temperate waters at latitudes less than about 40°S. It is likely that sperm whales within the Sanctuary are predominantly male and that females are limited to northern waters approaching the 40°S boundary. Gaskin (1973) found that the proportion of females decreased southwards abruptly at about latitude 44°S in the Tasman Sea and at about 46°–47°S to the east of New Zealand (NZ). Mass strandings of sperm whales on the west coast of Tasmania (41°S–43°S) in 1998 were predominantly female, indicating the presence of some female groups south of 40°S (Evans *et al.*, 2002). The only report of a female south of the Antarctic Polar Front was of a single whale caught off South Georgia (Matthews, 1938). Sperm whales at Kaikoura, NZ (42°25'S, 173°43'E) were dominantly males with only occasional encounters with nursery groups (Childerhouse *et al.*, 1995). Gaskin (1973) found that, like the female population, male sperm whale density also decreases southwards – the density between 50° and 60°S appeared to be less than 25% of that between 30° and 50°S. Gaskin related the distributional and seasonal changes to

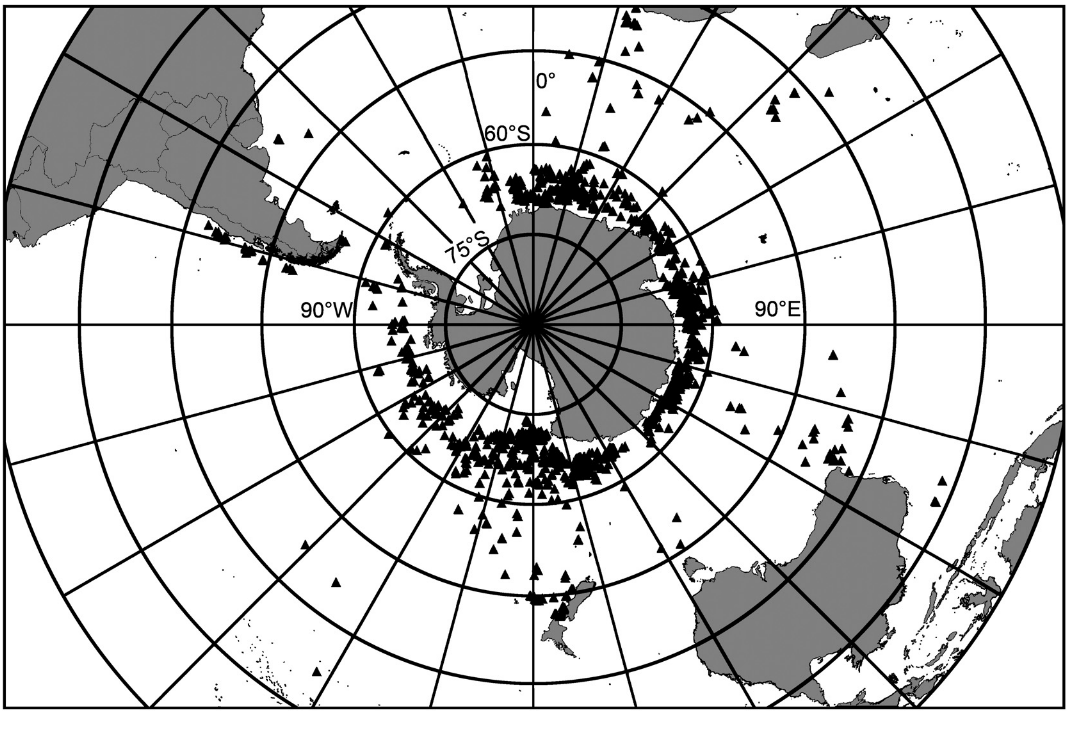

Fig. 1. Sightings of sperm whales from IDCR/SOWER cruises (black triangles).

optimal conditions (upper level sea temperatures) for squid schooling. Although data from sightings surveys generally indicate school sizes of one (Kasamatsu and Joyce, 1995), data from acoustic surveys show that sperm whales frequently form aggregations of several individuals within an area of a few square kilometres (Gillespie, 1997; Leaper *et al.*, 2000).

Migration and seasonality

Assessing the winter distribution of all whales in the Southern Ocean is hampered by the lack of survey effort, however it is generally believed that sperm whales move to lower latitudes in winter. For instance, Gambell (1967; 1972) and Best (1979) report on the seasonality in sightings and catches in the Durban and Donkergat whaling grounds respectively. Sperm whales are suggested by these authors to show a northward movement in autumn and a southward movement in spring. Kasamatsu and Joyce (1995) found that overall sperm whale numbers in Antarctic waters increased during November and December to a peak in early January. Analysis of squid beaks from stomachs of males caught off Durban and Donkergat showed that Antarctic squid species were present in stomachs between May and September suggesting that male sperm whales were moving north over these months (Clarke, 1980). However, Antarctic squid beaks were only present in the stomachs of large and medium-sized sperm whales, no Antarctic squid beaks were found in the stomachs of small males. Sperm whales are certainly present in the northern waters of the Sanctuary during winter months. Ashford *et al.* (1996) report the presence of sperm whales off South Georgia in April/May at 53°30'S and Thiele and Gill (1999) found them at 44°S, 146°E in July. Sperm whales are also present off Kaikoura, NZ, throughout the winter.

Conservation status and exploitation

Sperm whales were exploited much later in the Antarctic than at lower latitudes, and prior to 1933 annual takes were less than 100 animals. However, catches rose quickly due to the success of the deep-sea pelagic fleets and by 1939 annual catches were around 2,500. After a reduction in catches in the early 1940s due to the war, whaling increased again in the 1950s with average annual takes of around 6,000 sperm whales up until zero catch limits were introduced in the Southern Hemisphere from the 1981/82 season. Kasamatsu and Joyce (1995) give an estimate of 28,100 (CV 0.18) sperm whales south of the Polar Front in January based on IDCR sightings data between 1978/79–1987/88. Branch and Butterworth (2001) give estimates of 5,400 (CV 0.38), 10,000 (CV 0.15) and 8,300 (CV 0.16) for the first 1978–1984), second (1985–1991) and third (1991–1998) IDCR–SOWER circumpolar sighting surveys respectively. These latter estimates are not corrected for $g(0)$, although Whitehead (2002) applying a correction for $g(0)$ suggested around 12,000 sperm whales south of 60°S. This estimate, while approximate, does appear consistent with both visual and acoustic survey data. Assuming the area for the Southern Ocean south of 60°S to be 1.8548×106 km^2 (Whitehead, 2002), the range of estimates of sperm whale abundance (5,400–10,000 whales) calculated by Branch and Butterworth (2001) would correspond to densities of between 0.29–0.54 sperm whales per 1,000km^2. Whitehead's (2002) $g(0)$ correction gave an average density for south of 60°S of 0.65 sperm whales per 1,000km^2. These estimates compare well with density estimates from acoustic surveys. Gillespie (1997) estimated densities of between 0.50–0.73 sperm whales per 1,000km^2 for the sector 80°E–126°E which is believed to be a relatively high density area, while Leaper *et al.* (2000) estimated 0.13 sperm whales per 1,000km^2 for the Scotia Sea, which based on the results of circumpolar surveys (Kasamatsu and Joyce, 1995) is believed to be a relatively low density area. Species status designations are Vulnerable (IUCN Red List) and Appendix I (CITES and CMS).

Plate 2. Pygmy sperm whale (*Kogia breviceps*) stranded at Whangara, E. coast of North Island, NZ. Photo credit: Alan Baker.

Pygmy sperm whale *Kogia breviceps* (de Blainville, 1838)

Systematics and populations

No subspecies are described. Recent molecular genetic research suggests limited intraspecific population structure (Plön, 2004; Plön *et al.*, 2003). Nonetheless, further genetics work and a global study of geographic variation in cranial morphology is recommended.

Distribution in the Sanctuary

Cosmopolitan, but not in polar waters. In the Sanctuary it is known only from stranded specimens from South Australia and Tasmania (Baker, 1983; Bannister *et al.*, 1996), and from New Zealand where (until 1990) 212 strandings occurred between 42°S and 38°S (Baker and van Helden, 1990). Seasonality has not been studied in any detail.

Conservation status and exploitation

No population estimates exist and no data specific for our study area could be sourced, however incidental mortality in fishing gear and ingestion of plastic debris may be a problem (Reeves *et al.*, 2003; Stamper *et al.*, 2006). Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Killer whale *Orcinus orca* (Linnaeus, 1758)

Systematics and populations

Lillie (1915, p.121) suggested polymorphism in Antarctic killer whales by claiming that 'high-finned whales' seen in the pack ice are 'probably only a variety of *Orcinus orca*, or possibly a new species of that genus.' The taxonomy of

Plate 3a. Type A killer whale (PHE090121-084). Photo credit: Paul Ensor.

Plate 3b. Type B killer whale (PHE090121-067). Photo credit: Paul Ensor.

Plate 3c. Type C killer whale (PAO060215-167). Photo credit: Paula Olson.

Plate 3d. Type D killer whale (PAO061226160). Photo credit: Paula Olson.

Orcinus is currently under revision, and at least two (*O. orca* and *O. nanus*; Mikhalev *et al.*, 1981), perhaps three species (with *O. glacialis*; Berzin and Vladimirov, 1982) require recognition. Much uncertainty was generated by the loss of the holotype and paratype specimens of *O. nanus* and *O. glacialis*. Pitman and Ensor (2003) and Pitman *et al.* (2007; 2011) added substantial evidence to the case for speciation, documenting four morphologically and ecologically distinct forms in Antarctic waters that do not appear to mingle in schools, and are not thought to interbreed, despite geographic range overlap. Geographic variation in vocalizations between Ross Sea and Northern Hemisphere killer whales (Awbrey *et al.*, 1982) was also congruent with morphological heterogeneity. The Northern Hemisphere form *O. orca* is thought to be the cosmopolitan species.

Distribution in Southern Ocean Sanctuary

Killer whales occur throughout Antarctic waters (Fig. 2), with highest numbers observed (January) close to the northern edge of the pack ice (Brownell, 1974; Budylenko, 1981; Kasamatsu and Joyce, 1995; Kasamatsu *et al.*, 2000; Mikhalev *et al.*, 1981). Hundreds were seen as south as 78°S, 170°E 'at the farthest point of open water to the South' (Wilson, 1907). The three forms (named A, B, C) of killer whale present in the Antarctic during summer show signs of both parapatric and partially overlapping distribution patterns and ecological traits (Pitman and Ensor, 2003). Recently, Pitman *et al.* (2011) described a fourth monotype (D) from Antarctic waters.

IWC SOC data indicate the presence of killer whales inside fjords of the western Antarctic Peninsula, and presence south to 69°S (Fig. 3). SOCEP surveys found killer whales mostly in 60°E–110°E often in ice. Distribution seems to be concentrated near (but not confined to) shelf and shelf slope areas.

Winter observations in the pack ice were documented by Taylor (1957) and Thiele and Gill (1999). Pitman and Ensor (2003) suggested that type A is an open-water species which migrates to lower latitudes during the winter and possibly that type B also migrates. Type B and C killer whales, but not A, have been found within the pack ice in winter. Type A are believed to prey primarily on Antarctic minke whales, type B primarily on seals and type C primarily on fish, such as Antarctic toothfish (Pitman and Ensor, 2003). The C type is one of the smallest killer whales known, which has been studied in the dense pack ice of the southern Ross Sea (Pitman *et al.*, 2007). Recent mtDNA evidence suggests the three pheno- and eco-types are also genetically distinct (LeDuc *et al.*, 2008).

A marked increase in the density of killer whales in the Durban whaling grounds at the height of the winter whaling season was recorded by Findlay *et al.* (1992). It is unknown if such movements reflect migration of killer whales in association with the breeding migrations of baleen whales. One specimen (type to be confirmed) taken in the Durban whaling grounds, South Africa, had remains (vibrissae and nails) of at least three elephant seals within its stomach contents (Findlay *et al.*, 1992), suggesting some migration as the closest elephant seal colony is some 2,000km from Durban. Killer whale distribution data were reviewed at the 2007 IWC Scientific Committee meeting but it was noted that the factors responsible for spatial variation in distribution were not understood (IWC, 2008). During the review, information from localised studies was provided for Macquarie Island (Morrice, 2007), Terra Nova Bay in the Ross Sea (Fortuna *et al.*, 2007), and the Antarctic Peninsula (Dalla Rosa *et al.*, 2007).

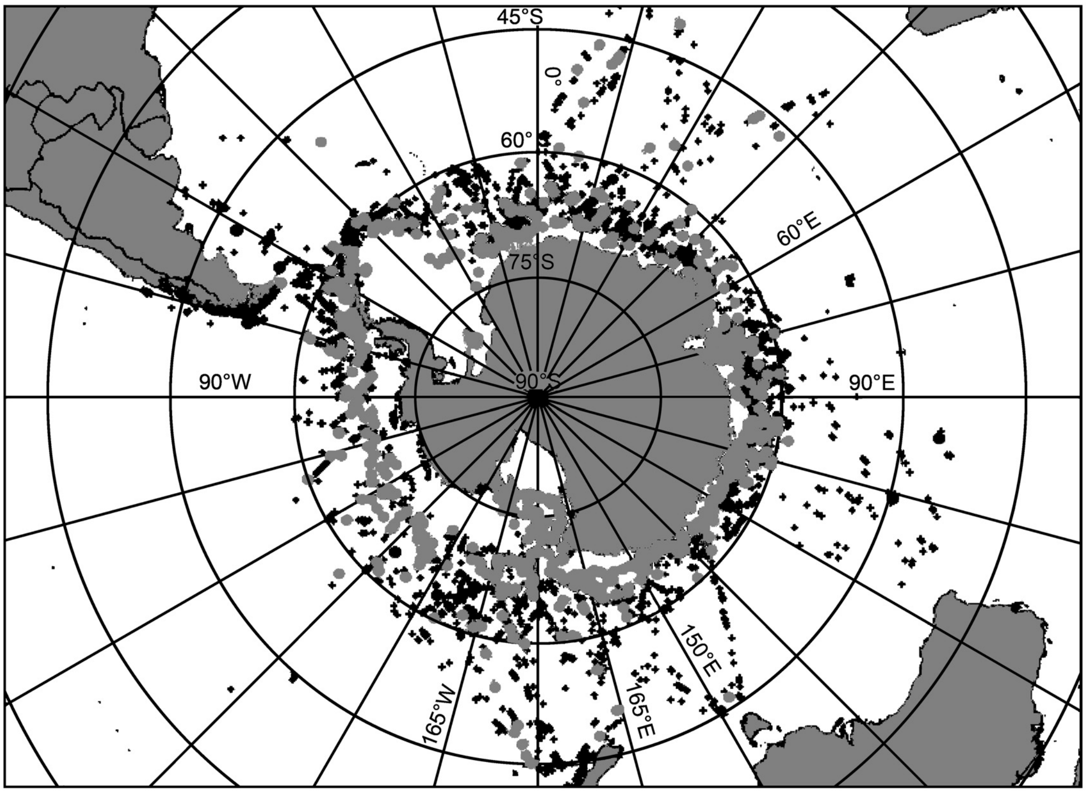

Fig. 2. Sightings of killer whales from IDCR/SOWER cruises (grey circles). Crosses indicate all sightings to give indication of survey effort.

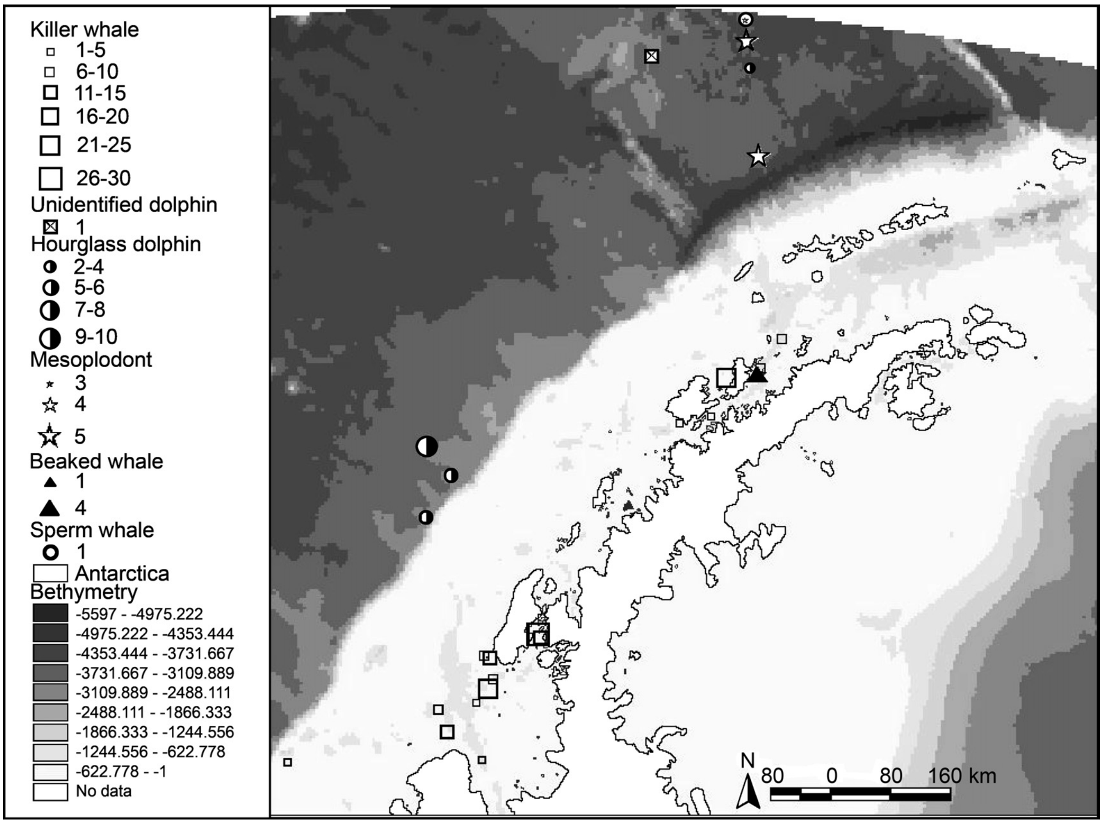

Fig. 3. Odontocete sightings during IWC SOC surveys off the Western Antarctic Peninsula.

Conservation status and exploitation

Killer whales were not a primary target for the pelagic whaling fleets in the Southern Ocean. However, Chrisp (1958) reported whaling fleets shooting killer whales because of perceived competition for large whales. Killer whale catches by the former Soviet Union were usually less than 10 per annum between 1947 and 1966 but were over-reported in several years. Thus, in that period the total reported catch by the *Slava* was 331 whereas the real catch was 57 (Centre for Russian Environmental Policy, 1995). The catching of killer whales by Soviet whaling ships increased dramatically in 1979/80. The USSR reported a total of 906 killer whales (447 males and 459 females) taken between 18 January 1980 and 21 March 1980 (Ivashin, 1981) compared to a total take of 738 between 1953/54 and 1978/79 (Mikhalev *et al.*, 1981). The killer whales were taken from 140°E–60°E. That year the IWC Scientific Committee (IWC, 1981b) recalled its recommendation of the previous year that the USSR be urged by the Commission to take no more than 24 killer whales from Antarctica in 1979/80 and noted that the Commission did not follow this recommendation. Referring to a complicated stock structure and insufficient evidence on which to base geographical stock boundaries, the Committee then recommended that 'catch limits for Antarctic killer whale stocks be zero' (IWC, 1981a).

The Commission considered the Scientific Committee recommendation first in its Technical Committee which agreed to recommend an addition of a new sentence to what was then Schedule Paragraph 9(d) (now paragraph 10(d)), i.e. the moratorium on factory ship whaling for species other than minke whales. No party has filed an objection, so the Paragraph is binding on all parties. The text proposed by the Technical Committee was adopted by the Commission by consensus: 'This moratorium applies to sperm whales, killer whales and baleen whales except minke whales'. Catches of killer whales by the Soviet Union ceased after 1980.

Kasamatsu and Joyce (1995) gave an estimate of 80,400 (CV 0.15) killer whales south of the Polar Front in January based on IDCR sightings data between 1976/77–87/88. Branch and Butterworth (2001) give estimates of 91,000 (CV 0.34), 27,000 (CV 0.26) and 25,000 (CV 0.23) for the IDCR-SOWER CPI, CPII, CPIII, sightings surveys respectively. These estimates are associated with a number of caveats, and may be particularly sensitive to changes in the location of the ice edge and the proportion of animals south of the ice edge. One possible explanation for the much higher abundance estimates for killer whales reported in Branch and Butterworth (2001) for CPI compared to CPII and CPIII, was that one survey vessel followed the ice edge for some of the earlier surveys.

Occasional interactions between killer whales and longline fisheries for Patagonian toothfish (Ashford *et al.*, 1996) could lead to incidental mortality, although no examples are documented. Visser (1999) reported ship strikes on killer whale, including one individual within the Sanctuary area (42°S).

Conservation status designations for *O. orca* are Data Deficient (IUCN Red List) and Appendix II (CITES, CMS).

Southern long-finned pilot whale *Globicephala melas edwardii* (A. Smith, 1834)

Systematics and populations

Davies (1960) assigned subspecific status to the Southern Hemisphere long-finned pilot whales which he nominated *G. melas edwardii* (A. Smith, 1834). This remains unchallenged. Otherwise no population structure is documented. External characters which distinguish *G. melas* from *G. macrorhynchus* are hard to ascertain if not approached closely. Southern range boundaries for *G. macrorhynchus* are imprecisely known and many sightings may in fact be presumed to be long-finned pilot whale on mere latitudinal considerations. We recommend explicit indication of diagnostic features as to allow re-evaluation of data. Stranding records of *G. macrorhynchus* on the south coast of South Africa may reflect southward movement within the warm southerly flowing Agulhas Current of the region (Findlay *et al.*, 1992).

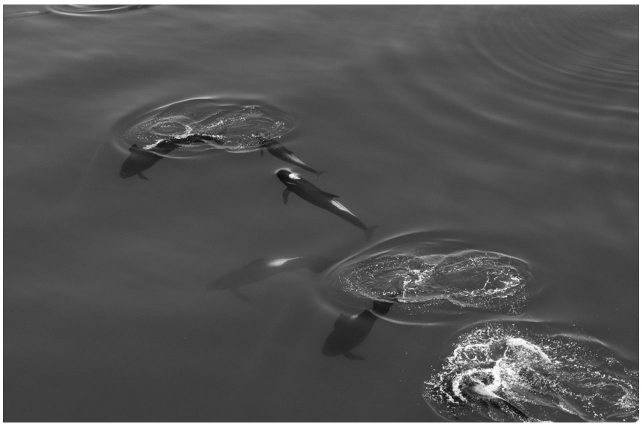

Plate 4. Southern long-finned pilot whale (*Globicephala melas edwardii*) at 36°20.20'S, 108°26.90'E on 19 Feb 2009 (PHE0902019-0370). Photo credit: Paul Ensor.

Distribution in Southern Ocean Sanctuary

G.m. edwardii is found throughout the Southern Ocean in cold currents (Antarctic Circumpolar, Humboldt, Falkland and Benguela Currents), north of the Antarctic Polar Front (Goodall and Galeazzi, 1987). Recent data show it to occur also south of the Polar Front. Several groups were sighted in the Scotia Sea, off South Georgia, Elephant Island, South Shetlands, and South Orkneys in summer (Brownell, 1974; Goodall and Macnie, 1998; Hanson and Erickson, 1985). However, none were encountered during IWC SOC surveys off the Western Antarctic Peninsula over the two years, in any season. Also, SOWER/IDCR surveys encountered only a single group between 40°W and 75°E, south of 45°S, indicating a lowest density area (Fig. 4). During SOCEP surveys, pilot whales were seen near ice as far south as 63°16.8'S and sightings concentrated from 90°E–110°E and 130°E–150°E generally off the shelf and at the base of the steep shelf slope, appropriate habitat for its main prey, cephalopods (Clarke and Goodall, 1994). Often observed in close association with minke whales, hourglass and dusky dolphins (Goodall and Galeazzi, 1987; Goodall and Macnie, 1998).

Strandings have occurred in Tasmania (Davies, 1960; 1963; Guiler, 1978; Scott, 1942), North and South Islands of New Zealand, Auckland Islands (Baker, 1977; 1999); South Orkneys and South Georgia (Goodall and Macnie, 1998), the latter being the southernmost specimen record in the Atlantic sector. In the Indian Ocean, a carcass was retrieved from Heard Island (Guiler *et al.*, 1987), just north of the Sanctuary. Based on observations from whaling vessels, Nishiwaki (1977) depicted the circumpolar southern distribution boundary consistently south of the Polar Front at about 56°S, with two southern dips to *ca.* 65°S (north of the Ross Sea and off the Antarctic Peninsula). Kasamatsu *et al.* (1988) in six IWC/IDCR cruises registered 26 schools (1,578 animals) south of 58°S. All but one sighting (200 animals in Area IV) was made away from the ice-edge.

Peaks in encounter rates, longitudinally in IDCR/SOWER cruises, were found at 90–100°E (E. Indian Ocean), 170–160°W (South Pacific) and smaller peaks at 120°–130°E, 110°–120°W and 40°–50°W (Kasamatsu and Joyce, 1995). An apparent distribution gap is reported at 54°–58°S in the South Atlantic-Indian Ocean sector, but no such gap was seen in the South Pacific sector. Highest encounter rates are reported for the second half of January (Kasamatsu and Joyce, 1995). No clear seasonality was identified, but neither discounted (small sample size). Pilot whales were seen in Antarctic waters only in mid and late summer (December and March) and Kasamatsu and Joyce (1995) reported no sightings south of 50°S in winter, but survey effort is mimimal then. The southernmost winter sighting (18 June; 25 animals) is at 55°27'S, 68°44'W, in Drake Passage (Goodall and Macnie, 1998).

Southernmost summer sightings are at 64°S (Kasamatsu and Joyce, 1995), near Scott Island (67°S,179°W) and in the central Pacific sector at 68°S, 120°W (Brownell, 1974), in the Atlantic at 67°41'S,05°44'W (SST = –1.0°C) (Kasamatsu *et al.*, 1988) and a group of six spotted NW of the Ross Sea at 66°33'S, 140°40'E. These lend credibility to a 1909 sighting at 69°53'S by Liouville (1913). Summer sightings south of the Polar Front below 60°S are nothing unusual, even with calves (Ensor *et al.*, 2009; 2008; 1999; Goodall and Macnie, 1998).

Conservation status and exploitation
Exploitation in the Southern Hemisphere has been sporadic and very low (Mitchell, 1975a; 1975b). No direct exploitation is known to occur at present in Sanctuary waters. Long-finned pilot whale is considered relatively scarce in Antarctic waters (Hanson and Erickson, 1985; Sapin-Jaloustre, 1953), and sometimes the species is not recorded over a complete summer survey (IWC SOC data; Thiele *et al.*, 2000). However, Hanson and Erickson (1985) note observations of large schools and that overall abundance would be higher than expected based on just number of schools sighted. Kasamatsu and Joyce (1995) give a January estimate of 200,000 (CV 0.35) long-finned pilot whales south of the Polar Front, based on IDCR sightings data between 1976/77–87/88, but note several caveats to this estimate in discussion. No estimates are provided by Branch and Butterworth (2001) due to the overall paucity of sightings and high variability in the number of sightings between surveys.

Status designation of *G. melas* includes Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (*G.m.edwardii*) (CMS).

False killer whale *Pseudorca crassidens* (Owen, 1846)

Systematics and populations
Pseudorca is a monotypic genus originally described as fossil. Geographical variation in cranial features has been described by Kitchener *et al.* (1990), and a subspecies, *P. crassidens meridionalis* (Flower, 1885) was erected by Deraniyagala (1945) for Indo-Pacific populations. Rice (1998) considered, however, that, in the absence of suitable defining characteristics, recognition of any subspecies would be premature.

Distribution in the Southern Ocean Sanctuary
In the Pacific and Indian Oceans, *P. crassidens* is known from New Zealand, Tasmania, South and Western Australia

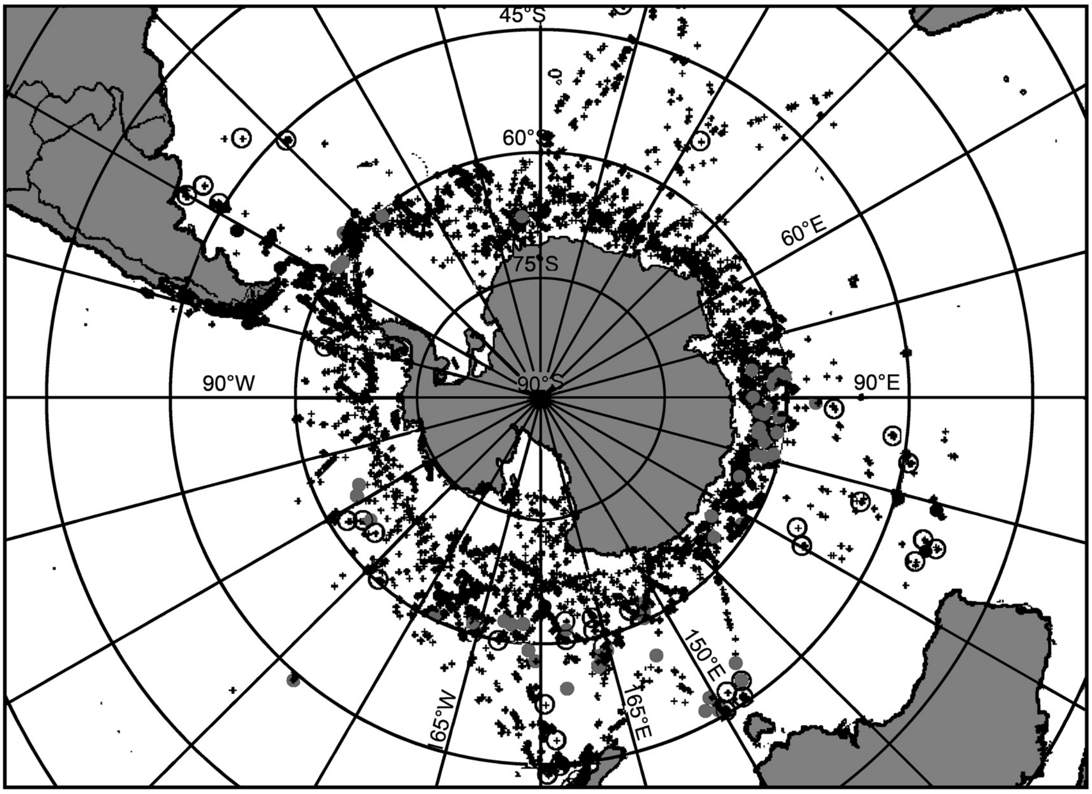

Fig. 4. Sightings of pilot whales from SOWER cruises (grey circles identified as long-finned, open circles as 'pilot whale'). Crosses indicate all sightings to give indication of survey effort.

Plate 5. Hourglass dolphin (*Lagenorhynchus cruciger*) (PHE52190006). Photo credit: Paul Ensor.

(Baker, 1999; Bannister *et al.*, 1996). North of the Sanctuary, it also occurs in South African waters (Findlay *et al.*, 1992) and in the eastern Magellan Strait at 52°27'S, southern Chile (Alonso *et al.*, 1999). In the Atlantic Ocean, Bastida and Rodriguez (2005) report occurrences in Argentina's Patagonia and Tierra de Fuego.

Conservation status and exploitation

No commercial exploitation occurs, although the false killer whale is occasionally captured for subsistence food in tropical regions, e.g. West Africa (Ofori-Danson *et al.*, 2003). Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Hourglass dolphin *Lagenorhynchus cruciger* (Quoy and Gaimard, 1824)

Systematics and populations

Taxonomic history is comprehensively reviewed by Goodall *et al.* (1997a). *L. obscurus* and *L. australis* were equivocally synonymised with *L. cruciger* (e.g. Bierman and Slijper, 1947; Liouville, 1913) until Fraser (1966) demonstrated that the name covers the hourglass dolphin only. Hence pre-1966 *L. cruciger* records can be either of three SH species, unless supported by diagnostic evidence. Discovery of *L. cruciger* is often (Goodall *et al.*, 1997a; Rice, 1998) attributed to Quoy and Gaimard (1824) thanks to an unmistakable description and a rough sketch, but no specimen. True (1889) and Robineau (1990) argued that d'Orbigny and Gervais (1847) should be considered the original species description since these authors described a first type specimen collected SE of Cape Horn. No subspecies or population structure are recognised and the species is most likely panmictic.

Distribution in the Southern Ocean Sanctuary

Pelagic, deep water, circumpolar on both sides of the Antarctic Polar Front and northward in cool currents associated with the Antarctic Circumpolar Current; from about 45°S to fairly near the pack ice. The southernmost sighting was at 67°38'S in the South Pacific (Goodall, 2002; Goodall *et al.*, 1997a). D'Orbigny and Gervais (1847) stated that they found *Delphinus cruciger* from '57°S–76°S', with the southernmost latitude presumably a misprint. SOWER/IDCR data suggests a dearth of sightings south of 60°S and east of the Antarctic Peninsula as far as 70°E, despite extensive effort (Fig. 5).

During IWC SOC surveys, hourglass dolphins were seen south to 66°S, off the Western Antarctic Peninsula. In the SOCEP surveys, they were concentrated on the shelf slope and the outer edges of steep bathymetry (like Kerguélen Plateau) mostly around KP in the west of the study area (75°E–115°E). Most sightings between 115°E–150°E were in the vicinity of the shelf and shelf slope, the southernmost group was encountered at 64°31.2'S.

The northern range boundary of *L. cruciger* is unclear, but unsupported reports at Atico (16°13'S,73°39'W) in southern Peru (Heintzelman, 1981), 'about 25°S' (Scheffer and Rice, 1963) and 36°14'S,52°43'W off the Rio de la Plata, Uruguay (Nichols, 1908) are not credible without authentication. All could have been dusky dolphins, *L. obscurus*. An alleged sighting off Valparaiso at 33°40'S, 74°55'W (Clarke, 1962) likely also involved dusky dolphins and its naming as *L. cruciger* may simply be the result of the pre-1966 instability in nomenclature, as explained higher[13]. The northernmost substantiated records in the Southeast Pacific Ocean are a 163cm specimen captured some 170km west of Isla Esmeralda in southern Chile at 49°S,78°W (Nichols, 1908), and another at 53°13'S,106°20'W in the South Pacific (N. Miyazaki, in Brownell and Donahue, 1999). A sighting in the Argentine basin at 42°24'S, 42°28'W on 14 November 1912 (Murphy, 1947) is unauthenticated. An alleged specimen from Tasmania (Guiler, 1978) was re-identified as *Lissodelphis peronii* (Van Waerebeek, 1993b).

L. cruciger is often reported from around South Georgia (e.g. Matthews, 1977). Gaskin (1968, in Brownell, 1974) recorded a number of sightings from southeast of the Chatham and Antipodes Islands (50°–60°S) and Gaskin (1972) claimed a winter record from Kaikoura, New Zealand, at 42°20'S, 174°05'E in 1963, without supporting evidence. Goodall *et al.* (1997a) listed four specimens from the east coast of South Island, NZ, including a skull in the Otago Museum. An alleged sighting in the western Gulf of Aden, off Somalia were referred to as 'unusual dolphins' that most closely resembled hourglass dolphins (Small and Small, 1991). Van Waerebeek *et al.* (1997) reviewing warm-water *Lagenorhynchus* records suggested that another, perhaps unrecognised, *Lagenorhynchus*-like delphinid may be involved.

Wilson (1907) reported *L. cruciger* from the outer zone of the pack-ice at 55°–60°S and about 135°E (Australian sector) in summer (November–January). Lillie (1915) believed it confined to a comparatively narrow band just north of the pack ice, and observed it from December–April at 55°–65°S and from 157°E to 88°W (*n* = 9; South Pacific sector). However, neither author offered any authentication. Bierman and Slijper (1947) reported *Lagenorhynchus wilsoni* (junior synonym of *L. cruciger*) from the SE Atlantic at 48°59'S,06°36'E (*n* = 4) and a school at 46°52'S,08°30'E in the Bouvet sector (Area III), with SST respectively 3.7 and 5.5°C. As their drawing confirms *L. cruciger*, these can be

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[13] Robert Clarke (1962), who clarified that in 1896 Philippi recorded the species he sighted as '*Tursio obscurus* Gray' [a synonym for dusky dolphin *L. obscurus*, see Gray, 1866], logically applied the pre-1966 used name '*L. cruciger*' for any southern *Lagenorhynchus*, including dusky dolphin *L. obscurus*. On his survey Clarke did otherwise not report any *L. obscurus*, one of three most common cetacean species off Chile (Van Waerebeek, 1992a; 1992b).

considered the northernmost positive records of the species. Southernmost sightings include one in the northern Ross Sea, near Scott Island (66°36'S,177°51'E) (G. Joyce photo in Leatherwood *et al.*, 1983) and another at 67°38'S,179°57'E in the South Pacific (Miyazaki and Kato, 1988).

Fraser (1966) examined a specimen taken at 56°20'S, 40°09'E, south of the Prince Edward Islands, and Stahl (1982) reported two February sightings from south of the Crozet Islands at 47°44'S (north of the Sanctuary) and 55°04'S. Thiele *et al.* (2000) reported ten of eleven sightings, offshore in the east section of eastern Antarctica (80–150°E) (10 sightings, 49 animals between 63°–64.3°S). All except one occurred at, or in close proximity to, the Southern Boundary of the ACC and/or the Antarctic Divergence (AD), or between these features in regions where the two oceanographic regions were separated. Preference was shown for areas associated with frontal zones and eddies.

Kasamatsu and Joyce (1995) found hourglass dolphins mainly in the northernmost areas of the Antarctic, especially in the Indian Ocean and South Atlantic sectors. They penetrated farthest south (67°S) between 150°E–150°W in the South Pacific and were not seen south of 66°S in the South Atlantic and Indian Ocean sectors. Longitudinal gaps appeared at 80–150°W and 0–40°W. The distribution pattern is apparently similar to that of the long-finned pilot whale (Kasamatsu and Joyce, 1995). Hourglass dolphins rarely strand in the Sanctuary and few specimens exist in collections, namely from South Island, NZ and Livingston Island, South Shetlands (Fraser and Noble, 1968; Goodall *et al.*, 1997a).

Seasonality in Antarctic waters is suggested by an increase in encounter rates in February, perhaps corresponding to an increase in SST which peaks in March; either linked to thermoregulatory factors (*cf.* small body size) or prey availability (Kasamatsu and Joyce, 1995). Thiele and Gill (1999) came upon two groups in winter (July 1995) in open water (55°01.8'S,141°00.6E and 57°14.2'S,139°51.9'E) of, respectively, 1.4°C and 1.1°C SST. Prey has included more or less digested small fish such as Myctophidae (Best, 2007; Goodall, 1997; Nichols, 1908) and squid (Ash, 1962; Clarke and Goodall, 1994) including Onychoteuthidae and Enoloteuthidae (Goodall, 1997).

Conservation status and exploitation

Some have been harpooned for food and a few for research purposes, but otherwise no exploitation of hourglass dolphin existed (Ash, 1962; Brownell and Donahue, 1999; Fraser, 1964; 1966; Nichols, 1908). Nichols (1908), who took two, stated 'their flesh tastes somewhat like meat, somewhat like fish, and is a very welcome break in a diet composed chiefly of salted and canned foods'. At least one specimen was incidentally caught in an experimental Japanese drift net fishery for squid around 53°13'S,106°20'W (N. Miyazaki, in Brownell and Donahue, 1999). No other incidental captures are reported. The January population in Antarctic waters was estimated at 144,300 (CV 0.17) (Kasamatsu, 1993; Kasamatsu and Joyce, 1995). Branch and Butterworth (2001) presented some data on hourglass dolphin abundance also from the IDCR-SOWER surveys, including the second and third circumpolar, but did not consider these estimates reliable. They noted large potential biases due to attraction to the vessel but also that the estimates had little biological meaning because the main distribution for the species was outside of the survey area. No conservation problem is identified (Goodall *et al.*, 1997a; Reeves and Leatherwood, 1994; Reeves *et al.*, 2003). Status designations: Least

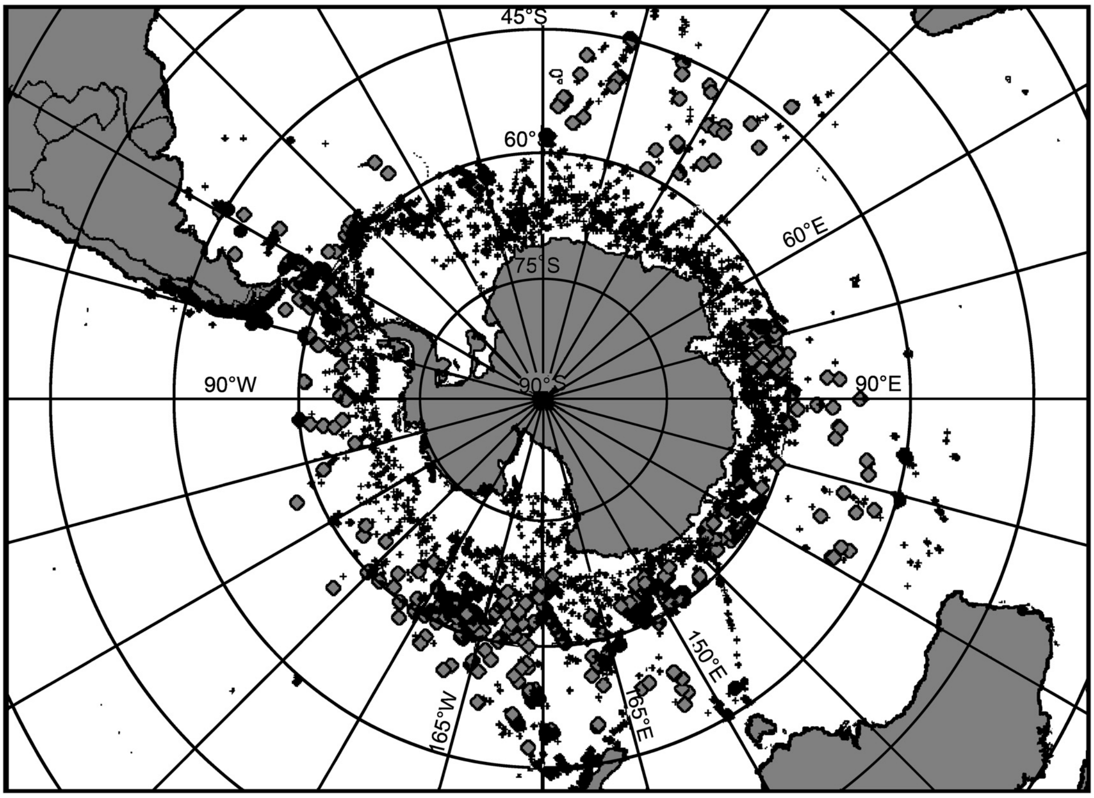

Fig. 5. Sightings of hourglass dolphins from IDCR/SOWER cruises (grey diamonds). Crosses indicate all sightings to give indication of survey effort.

Plate 6. Dusky dolphin (*Lagenorhynchus* obscurus), Kaikoura, NZ (PHE14032010-0013). Photo credit: Paul Ensor.

Concern (IUCN Red List); Appendix II (CITES) and Not listed (CMS).

Dusky dolphin *Lagenorhynchus obscurus* (Gray, 1828)

Systematics and populations

Van Waerebeek (1992a; Van Waerebeek, 1993a) found significant geographic variation in morphology, some considered at subspecific level, but refrained from describing subspecies pending additional evidence. Rice (1998) named three subspecies (for South America, South Africa and New Zealand) without offering diagnoses or types. Molecular genetic analyses support divergent SE Pacific and Atlantic lineages, and reveal an intricate phylogenetic pattern. Peruvian dusky dolphins form a distinct ESU (Cassens *et al.*, 2003; 2005). A probable hybrid of *L. obscurus* with *L. peronii* was photographed off Argentina (Yazdi, 2002)[14] and a skull from Peru suggested hybridism with *Delphinus capensis* (Reyes, 1996).

Distribution in the Southern Ocean Sanctuary

Van Waerebeek *et al*. (1995) comprehensively reviewed *L. obscurus* global distribution. Gill *et al.* (2000) demonstrated its presence off Tasmania and southern Australia by 13 sightings and a stranded mother/calf pair. Van Waerebeek *et al*. (1995) reported a few sightings south of Gough Island (40°20’S,09°54’W) in the mid-Atlantic. Baker (1977) indicated Campbell Island (52°30’S,169°10’E) as the southernmost range for the New Zealand stock. Lillie (1915, p.122) reported its southernmost sighting (7 February 1913) at 51°56’S,168°02’E, and stated that it does not seem to occur further south than about 58°S, however no substantiation is available for this pre-1966 record. Nishiwaki (1977), repeating the 58°S, stated that *L. obscurus* stays 300–500km to the north of the convergence (Antarctic Polar Front). However, two Drake Passage sightings south of the Polar Front, one at 57°50’S and another at 60°29’S, north of the South Shetland Islands (Goodall *et al.*, 1997b; fig.9) seem to contradict this.

A group of two dolphins sighted by KPF at 60°33.65’S, 054°35.70’W off Elephant Island in 13 March 1990 were ‘similar to a gray *Lagenorhynchus* and definitely not hourglass dolphin colouration’.

Thiele *et al.* (1997) encountered dolphins closely resembling dusky dolphins and ‘distinctly unlike hourglass dolphins’ at 64°28’S,120°02’E on 1 March 1997. If *L. obscurus* ventures south of the Polar Front, it remains a rare event. During SOCEP surveys, the species was seen to 48°S. Although dusky dolphins can move over great distances, there is no evidence of regular oceanic migration (Cassens *et al.*, 2003; Cassens *et al.*, 2005; Van Waerebeek *et al.*, 1995). Movements around New Zealand are related to SST (Gaskin, 1968).

Conservation status and exploitation

Unlike the Peru and Argentina populations (Dans *et al.*, 2003; Van Waerebeek, 1992a), the New Zealand/Australia stock is not known to be subjected to any exploitation. If catches occur off South Africa they would be small and difficult to demonstrate as catches are illegal.

Status designations for the dusky dolphin are ‘Data Deficient’ (IUCN Red List) and Appendix II (CITES and CMS).

Southern right whale dolphin *Lissodelphis peronii* (Lacépède 1804)

Systematics and populations

No subspecies or populations are named. Suggested conspecificity with northern right whale dolphin *L. borealis* (Honacki *et al.*, 1982) is unfounded, although all-black individuals have been sighted (e.g. photo p.261 in Jefferson *et al.*, 2008). *L. peronii* is cranially (Van Waerebeek, 1993b) and genetically (cytochrome b gene, LeDuc *et al.*, 1999) very close to *L. obscurus* with which it can hybridize in the wild (Yazdi, 2002), as well as close to the other southern *Lagenorhynchus* and *Cephalorhynchus* species (100% bootstrap support for subfamily *Lissodelphininae*).

Distribution in the Southern Ocean Sanctuary

Brownell (1974) called *L. peronii* a ‘marginal Antarctic species’, with associated SST range, 1.4–13.8°C (Kasamatsu *et al.*, 1988). Circumpolar, from about 40°S–55°S (Baker, 1981; Jefferson *et al.*, 1994) but extending much further north in cold currents, with a northernmost record at Pucusana (12°28’S,76°48’W), Peru, in the Humboldt Current system (Van Waerebeek *et al.*, 1991). It is fairly common off South Island, New Zealand, in the Tasman Sea and waters directly south of Australia (Gaskin, 1968; 1972). Kasamatsu *et al.* (1988) reported only three groups (none with calves) south of 58°S: two in Area VI south of the Antarctic Polar Front (60°39’S,154°14’W and 61°20’S,163°27’W) and one in Area I (58°09’S,67°17’W). All sightings were mixed species schools. The southernmost specimen, according to Goodall and Galeazzi (1985a), is a skull collected from the South Shetland Islands. During SOCEP, four sightings were recorded from 47°S–49.23°S and 131°E–137°E, with ‘like hourglass’ sightings south of 65°S between longitudes 85°E and 145°E. One was a large group of 75 animals, but other groups were small (2–10 animals).

L. peronii is present off Namibia having been recorded in summer (eight sightings between November and January),

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[14] Black-white colouration pattern, elongated body shape expresses *L. peronii* phenotype; presence in dusky dolphin school, dorsal fin (diminutive and caudad position) and hint of flank patch points to affinity with *L. obscurus.*

autumn (two sightings in April) and winter (two sightings in August) although this possible summer seasonality may reflect observer effort bias (Findlay *et al.*, 1992). Further sightings from this region were reported by Rose and Payne (1991) and include one just south of the South African/Namibian border. There is no evidence of any significant migratory movement, however this may be due to a lack of dedicated research.

Conservation status and exploitation

Bennett (1840) wrote that 'our crew never lost the opportunity of harpooning them, as we esteemed their flesh a delicacy'. Apart from some bycatches, anthropogenic mortality in the Sanctuary is thought to be minimal; mortality in gillnet fisheries off Chile may not be insignificant (Van Waerebeek *et al.*, 1991).

Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Commerson's dolphin *Cephalorhynchus commersonii* (Lacépède 1804)

Systematics and populations

Rice (1998) recognised two subspecies[15], coinciding with the population off South America, including Drake Passage (Sanctuary waters), and the population from the Kerguélen Archipelago (Paulian, 1953; Robineau, 1989a; 1990). Their plausible origin and radiation is discussed by Pichler *et al.* (2001). Robineau *et al.*(2007) described and documented a new subspecies from Kerguelen Islands as *C. commersonii kerguelenensis*.

Distribution in the Southern Ocean Sanctuary

Commerson's dolphins are distributed mostly, with highest abundance, in nearshore areas outside the Sanctuary, i.e. east coast of southern South America, Peninsula Valdéz south to Tierra del Fuego and near Burdwood Bank. It is fairly common in the Magellan Straits and Falklands Islands and off Chile, south of 50°S. The northernmost range in the SW Atlantic is unclear but probably close to 41°26'S (Goodall, 1994; Goodall *et al.*, 1988). There is only one area of reported occurrence within the Sanctuary, i.e. the southern stratum of Drake Passage (Dawson, 2002): the two southernmost sighting records were south of the Antarctic Polar Front, NW of Livingstone Island, South Shetlands (n = 1, 61°59'S,63°05'W, 27 Feb. 1966; n = 2, 61°50'S,63°17'W, 27 Feb. 1966); a third sighting ($n = 5$) is from 58°10'S, 67°58'W on 28 Feb.1966; and a fourth ($n = 2$) is reported from 'Drake Passage' on 5 December 1972 (Aguayo, 1975; Aguayo and Torres, 1967). These are the only published sightings from that area and, as Brownell and Praderi (1985) pointedly stated, these sightings are 'extremely puzzling', and were it not for an unmistakable photograph (p.1124, Aguayo, 1975) they might be questioned. Numerous scientific vessels make observations in Drake Passage but have not reported any Commerson's dolphins since. To add to the debate, Ellis (1982) saw 'a film of 15 or so Commerson's dolphins pacing a U.S. Coast Guard icebreaker in the Antarctic'. This film should be re-examined, but it would seem hard to confuse Commerson's dolphin with any other cetacean.

Unsubstantiated reports of *C. commersonii* off South Georgia (Hart, 1935; Matthews, 1931) were questioned (Brown, 1988; Brownell and Praderi, 1985), but seeped widely into the literature (e.g. Brownell, 1974; Fisher and Hureau, 1985; Leatherwood *et al.*, 1983; Strange, 1992). If Commerson's dolphin makes occasional excursions offshore and south into Drake Passage it could enter the Scotia Sea off South Georgia. The former existence of a third, relict population off islands in the Scotia Sea is another possibility. Immediately north to the Sanctuary, a reproductively isolated, both morphologically and genetically distinct, population resides in waters of the Kerguélen archipelago (48°30'–50°S, 68°30'–70°45'E) (de Buffrenil *et al.*, 1989; Goodall, 1994; Paulian, 1953; Pichler *et al.*, 2001; Robineau, 1989a; 1990; Stahl, 1982), now recognised as a new subspecies (Robineau *et al.*, 2007). A sighting of a presumed vagrant of unknown origin at the southern African continental shelf (de Bruyn *et al.*, 2006), suggests a less restricted distribution than hitherto believed. This and rare Drake Passage sightings could point to irregular wandering movements offshore and south, or east, from Patagonia, under (indeterminate) favourable oceanographic conditions. Conceivably, a higher population size half a century ago may have led to more frequent 'extralimital' records. Some individuals from the Kerguélen population may perhaps cross into the Sanctuary's Indian Ocean sector.

Conservation status and exploitation

Although some localised abundance estimates have been made (Leatherwood *et al.*, 1988; Lescrauwaet *et al.*, 2000; Venegas, 1996) these are only for small areas of the Strait of Magellan which is outside the Sanctuary. No abundance data exist within the Sanctuary area. Good estimates are lacking for the level of catches off southern South America (Reeves *et al.*, 2003), and no captures are registered in the Sanctuary. A few specimens were taken for research purposes at Kerguélen (Angot, 1954; Paulian, 1953). French observers on board Russian trawlers fishing on the Kerguélen plateau did not report any incidental dolphin kills (D. Robineau, in Reeves and Leatherwood, 1994). At least 18 dolphins were taken in Tierra del Fuego for captive display purposes (Goodall *et al.*, 1988). Status designations are Data Deficient (IUCN Red List), Appendix II (CITES) and Appendix II (CMS) only for the South American population.

Hector's dolphin *Cephalorhynchus hectori* (P.J. Van Beneden, 1881)[16]

Systematics and populations

A complete taxonomic history with synonymies of Hector's dolphin, endemic to New Zealand, was given by Baker (1978). Since then, mtDNA analysis has identified four regional populations (Pichler, 2002), followed by a study of

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[15] However without presenting names, diagnoses or types.

[16] *non* 'P.J. van Bénéden' (erroneous spelling) *in* Hershkovitz (1966), Rice (1998), and many others. Pieter-Jozef Van Beneden (1809–1894), a Flemish professor of zoology at the Katholieke Universiteit van Leuven, Belgium. Confusingly, his son Edouard Van Beneden, professor of zoology at Luik University, also published on cetacean taxonomy (described *Sotalia brasiliensis*), therefore given name initials are required.

morphological variation showing consistent differences between the North and South Island populations which resulted in Baker *et al.* (2002) describing the North Island population (north of 40° S) as a new subspecies, *C. hectori maui*, and relegated the three South Island populations (south of 40°S; 'East coast', 'West coast' and 'South coast') to the subspecies *C. hectori hectori*.

Distribution in the Southern Ocean Sanctuary
Hector's dolphin occurs around most of South Island (except for the southern fiords), thus within the Sanctuary; in contrast, the entire range of Maui's dolphin falls north of it. No indications exist of any significant seasonal or migratory movements. Satellite tagging has been trialed to track the movements of Hector's dolphin. Both subspecies are exclusively neritic, most often seen within 0.5 n.miles from shore (Baker *et al.*, 2002; Brager, 1999; Dawson and Slooten, 1988), but they range to at least 18 n.miles offshore (ANB, pers. obs.). Local populations occupy relatively small geographic ranges, e.g. of approximately 33 n.miles (Brager, 1999). Three genetically distinct regional populations are distributed on the eastern, southern and western coasts of the South Island (Pichler, 2002).

Conservation status and exploitation
This species is occasionally taken in trawl fishing operations (Baker, 1978; Mitchell, 1975b), gillnets (Dawson, 1991; 2002; Mörzer Bruyns and Baker, 1973; Slooten and Lad, 1991; van Bree, 1972) or killed by boat collisions (Stone and Yoshinaga, 2000). The North Island Maui's dolphin *C. hectori maui* and the South Island Hector's dolphin *C. hectori hectori* require separate management (Baker *et al.*, 2002). Total abundance for this species is low at an estimated 7,300 animals. Incidental takes in inshore gillnets are the biggest threat to its survival (Dawson, 1991; 2002; Slooten and Lad, 1991). The IUCN Red List status is 'Endangered' (*C. hectori*). The northern Maui's dolphin may number fewer than 100 individuals and is listed as 'Critically Endangered'. The species is listed on Appendix II (CITES) and Appendix I (CMS).

Common bottlenose dolphin *Tursiops truncatus* (Montagu, 1821)

Systematics and populations
Of the two species of bottlenose dolphin widely recognised in the Southern Hemisphere, only *T. truncatus* is eurythermic enough to penetrate the northern sectors of the Sanctuary. The Indo-Pacific bottlenose dolphin *T. aduncus* (Ehrenberg, 1833) occupies solely tropical and warm temperate latitudes. Inshore and offshore *T. truncatus* 'ecotypes' constitute unnamed ESUs, distinct at least at subspecies level.

Distribution in Southern Ocean Sanctuary
Common bottlenose dolphins are cosmopolitan although absent from polar waters (Marcuzzi and Pilleri, 1971; Wells and Scott, 2002). Within the Sanctuary the species occurs around Tasmania and New Zealand where it is common in the Bay of Islands, Hauraki Gulf, and Marlborough Sounds (Baker, 1972; 1983; Bannister *et al.*, 1996) and in Fiordland (Haase and Schneider, 2001). De Boer *et al.* (1999) reported a group of six bottlenose dolphins near Hobart (depth 36m; 20°C). Contiguous to the Sanctuary, specimen records exist from the Falkland Islands (Strange, 1992) and Bahía San Sebastian, southern Patagonia, Argentina (Goodall, 1978; Goodall and Galeazzi, 1985a; RNP Goodall, e-mail to KVW, 29 September 2003). The southern distribution boundary is only vaguely known, but 55°S (Bannister *et al.*, 1996) seems plausible. Suggestions of 48°S (map in Wells and Scott, 2002) and 45°30'S (Haase and Schneider, 2001) are disproved by the Falklands records (51°–52°S), but (sub)Antarctic waters proper lie beyond its range (Brownell, 1974; Kasamatsu and Joyce, 1995; Miyazaki and Kato, 1988; authors, pers. obs.), presumably related to SST and prey distribution.

Plate 7. Short-beaked common dolphin (*Delphinus delphis*), SW of South Africa, 24 Feb 1998, 37°20.15'S, 15°10.24'E. Photo credit: Paul Ensor.

No seasonal movements are known in the Sanctuary, however *T. truncatus* can be seasonally migratory in temperate waters, such as at its northern range in the NW Atlantic (Bannister *et al.*, 1996; Wells and Scott, 2002), and the same may hold true for its austral range.

Conservation status and exploitation
No population estimates for the Southern Ocean and no examples of exploitation are identified. A rare mass stranding[17] of 21 animals occurred in Delaware Bay (41°08'S, 173°28'E), South Island, in March 1982, all of which were rescued (Robson, 1984). Status designations include Least Concern (IUCN Red List), Appendix II (CITES), and unlisted populations (CMS).

Short-beaked common dolphin *Delphinus delphis* (Linnaeus, 1758)

Systematics and populations
Bannister *et al.* (1996) reported that two forms of common dolphin exist in South Australia (contiguous to the Southern Ocean Sanctuary) but 'it is not known whether these represent the short- or long-beaked types'. Jefferson and Van Waerebeek (2002) tentatively assigned all available *Delphinus* specimens from South Australia to *D. delphis,* which was in agreement with molecular genetic analysis (White, 1999).

Distribution in Southern Ocean Sanctuary
Common dolphins frequent coastal waters all around New Zealand (Baker, 1972) but their austral range is unclear. The

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[17] Likely a tidal trapping (ANB).

short-beaked form is recorded from Tasmania and southern Australia. *D. delphis* is not reported from territories in the Antarctic or Heard and Macquarie Islands (Bannister *et al.*, 1996). It is not known to be migratory off Australia and New Zealand.

Conservation status and exploitation

Designations include Least Concern (IUCN Red List), Appendix II (CITES) and unlisted population(s) (CMS).

Spectacled porpoise *Phocoena dioptrica* (Lahille, 1912)

Systematics and populations

Barnes (1985) created a new genus for the spectacled porpoise, *Australophocoena.* However, Lahille's (1912) recognition of close morphological affinity with other *Phocoena spp*. is supported by mtDNA cytochrome b analysis (Rosel *et al.*, 1995). No subspecies or population structure is defined, but the relative rostrum length of a skull from Auckland Island (50°45'S,166°06E) was shorter than in other regions (Perrin *et al.*, 2000).

Distribution in Southern Ocean Sanctuary

Showing a circumpolar distribution (Baker, 1977; Goodall and Schiavini, 1995; Sekiguchi *et al.*, 2006), the spectacled porpoise appears particularly common in the southwestern South Atlantic off Tierra del Fuego (Goodall and Cameron, 1979) and, based on IDCR/SOWER sightings (Fig. 6), in the Antarctic sector south of NZ and Tasmania (Sekiguchi *et al.*, 2006). Within the Sanctuary, stranded specimens are reported from South Georgia (Burton, 1997; Fraser, 1968; Strange, 1992), Bruny Island, Tasmania (K. Evans in Brownell and Clapham, 1999), Macquarie Island (Fordyce *et al.*, 1984), South Island of New Zealand (Baker, 1999) and Auckland Islands (Baker, 1977). It has been seen near Antipodes Islands (Kasamatsu *et al.*, 1990) and Auckland Islands (M.Cawthorn in Goodall and Schiavini, 1995). In the Indian Ocean, the spectacled porpoise was recorded at Heard and Kerguélen Islands (Sanctuary contiguous areas). Apparent concentration of records near subantarctic islands is possibly due to observer bias (Bannister *et al.*, 1996). The most boreal record is from southern Brazil at 32°S.

Plate 8a. Spectacled porpoise (*Phocoena dioptrica*) (PE080111-0388). Photo credit: Paul Ensor.

Plate 8b. Spectacled porpoise (*Phocoena dioptrica*) (PE080111-0517). Photo credit: Paul Ensor.

No seasonal movements are confirmed, but sightings made far offshore between 54°S–59°S (IWC, 1991) suggest that there may be some movement across the open ocean, however not necessarily seasonal. In the western South Atlantic the spectacled porpoise seems to breed in spring.

Conservation status and exploitation

Population abundance is unknown, but it is considered 'rare' from small group size ranging from 1–3 animals and low encounter rates (Ensor *et al.*, 2001; Goodall and Schiavini, 1995; Sekiguchi *et al.*, 2006). Kasamatsu *et al.* (1990) concluded, possibly prematurely in view of the small number of sightings, that the spectacled porpoise does not form large schools. A similar claim about Burmeister's porpoise *Phocoena spinipinnis* was dismissed after a school of *ca.* 150 individuals was sighted off Peru (Van Waerebeek *et al.*, 2002). Hamilton (1952) indicated that 'the species [*P.dioptrica*] had been taken off South Georgia'. A live-stranded animal in South Georgia was eaten; moreover according whaling captains these porpoises were sometimes shot for food (Brownell and Clapham, 1999; Fraser, 1968). It also used to be hunted by fishermen from Uruguay and southern Chile (Goodall and Schiavini, 1995; Praderi and Palerm, 1971). Some mortality may occur by entanglement in driftnets and other nets, set, lost or discarded in international waters at higher latitudes (Bannister *et al.*, 1996), and in bottom and midwater trawls (Reeves *et al.*, 2003).

Conservation status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Beaked whales *Ziphiidae*

During SOCEP and IWC SOC surveys beaked whales could not be identified to species because most individuals were too distant from the vessel for diagnostic features to be seen. Implementation of closing mode for long-diving species such as beaked whales is problematic due to the relatively short duration of their surfacing bouts. Experience during the IWC/IDCR and IWC/SOWER programmes indicate that some of the best opportunities for identifying and documenting such species are the occasions when they are detected close to the vessel irrespective whether the vessel is in passing or closing mode (e.g. Van Waerebeek *et al.*, 2005). SOCEP found ziphiids mostly on the shelf slope and near ice. During IWC SOC surveys, three mesoplodont

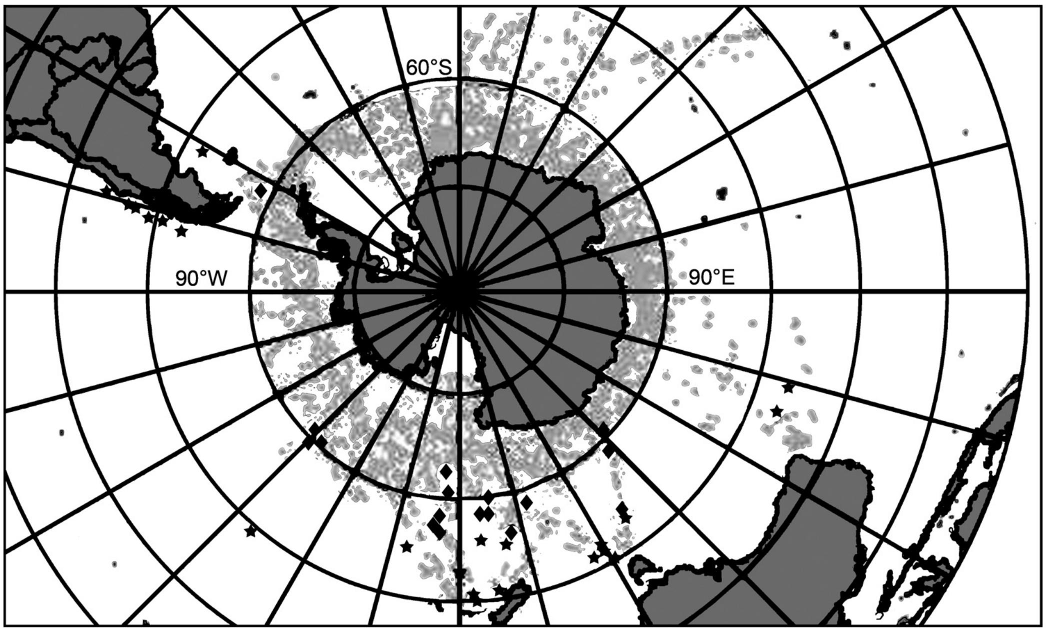

Fig. 6. Sightings of spectacled porpoise (black diamonds) and southern right whale dolphins (black stars) from IDCR/SOWER cruises.

sightings were recorded across Drake Passage, and unidentified beaked whales were seen in Gerlache Strait and in the fjords of the Western Antarctic Peninsula (see Fig. 2).

Southern bottlenose whale *Hyperoodon planifrons* (Flower, 1882)

Systematics and populations

No subspecies are named and no population structure is documented (Mead, 1989b; Mitchell, 1975b), however mtDNA control region sequences of two specimens from New Zealand differed 4.12%, considerably higher than the normal interspecific variation of 2% in other beaked whales (Dalebout *et al.*, 1998). Intraspecific variation in colouration pattern may be mostly ontogenetic and sexual, but geographic variation is not ruled out (Van Waerebeek *et al.*, 2005). A global phylogenetic study is necessary.

Plate 9a. Southern bottlenose whale (*Hyperoodon planifrons*) (PE080124-0594). Photo credit: Paul Ensor.

Plate 9b. Southern bottlenose whale (*Hyperoodon planifrons*) (PE080124-0753). Photo credit: Paul Ensor.

Distribution in Southern Ocean Sanctuary

Extensive circumpolar distribution (Fig. 7), from the Antarctic continent north to Dampier Archipelago, NW Australia at *ca.*20°35'S,116°39'E where the holotype was found (Flower, 1882). In the Sanctuary, strandings are known from the South Island, NZ (Baker, 1999) and at 54°19'S on South Georgia (G.J.B. Ross, pers. comm. to ANB, October 2001). Aerial survey sightings of 'like-bottlenose whale' off Durban, South Africa, during whaling operations show strong seasonality with peaks in February and October (but no data for November–January) (Findlay *et al.*, 1992; Sekiguchi *et al.*, 1993). It remains unclear whether or not the February peak results from a general late summer movement northward out of the Antarctic, and alternatively perhaps two parapatric stocks exist. Ross (1984) noted an apparent summer seasonality of this species in South African waters. Findlay *et al.* (1992) found bottlenose whales of the southern African subregion to be confined to temperate and subtropical waters of the Agulhas Current and few records within the cold temperate Benguela system. Although these data may be biased by the paucity of offshore sighting survey effort in the Benguela system, the records from the warm temperate and subtropical Agulhas Current (in relation to the Southern Ocean distribution patterns) suggest some stock segregation or migration. Nemoto *et al.* (1980) report evidence of the skin diatom *Cocconeis ceticola* on a specimen of *H. planifrons* from South African waters, while Sekiguchi *et al.* (1992) recorded beaks from Antarctic squids in the stomachs of animals from South African waters.

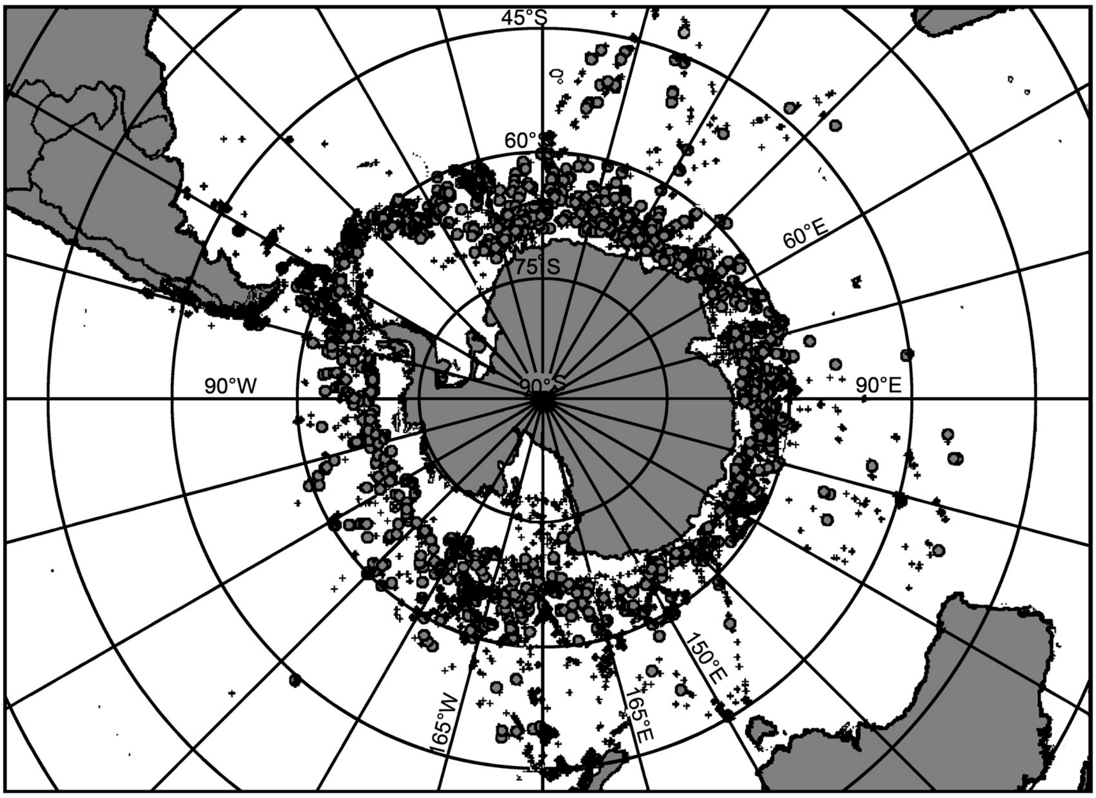

Fig. 7. Sightings of southern bottlenose whales from IDCR/SOWER cruises (grey circles). Crosses indicate all sightings to give indication of survey effort.

Conservation status and exploitation

Southern bottlenose whale 'is sometimes encountered by whalers and killed' (Mitchell, 1975a). At South Georgia, one bottlenose whale was landed in 1912–13; at South Shetlands seven were landed in 1911–13, two in 1923–24 and three in 1925–27 (Harmer, 1928; Risting, 1922). Mead (1989b) published a 1923 photograph of an adult specimen on a flensing platform, reportedly at a South Georgia whaling station. In contrast, Harmer (1928) did not report any South Georgia catches for 1923, but for the South Shetland Islands. Fraser (1945) studied a male caught near South Georgia in an indeterminate year. A few specimens were taken by Soviet whalers for research purposes (Tomilin and Latyshev, 1967; Zemskii and Budylenko, 1970) while Japanese whalers took 42 specimens in Antarctic waters, from 1970–1982 (Kasamatsu *et al.*, 1988). Before 1990, population estimates and even relative abundance data of *H. planifrons* were lacking (Mead, 1989b). Sekiguchi *et al.* (1993) indicated it as the second most frequently encountered cetacean in high latitudes, probably based on the claim by Kasamatsu *et al.* (1988) that southern bottlenose whales account for more than 90% of ziphiid sightings. More recently, Kasamatsu and Joyce (1995) suggested there to be about 600,000 beaked whales in the Antarctic during summer months, of which the majority southern bottlenose whales, based on IDCR sightings data between 1976/77–87/88, with a correction for g(0). However, the majority of the data for this estimate comes from the first circumpolar survey when only 5% of beaked whales were identified to species level (Branch and Butterworth, 2001). Branch and Butterworth (2001) give estimates of 72,000 (CV 0.13) and 54,000 (CV 0.12) southern bottlenose whales for the second and third IDCR-SOWER circumpolar surveys respectively. These estimates were not corrected for g(0) and have a number of caveats mentioned in the discussion.

Status designations: Least Concern, (IUCN Red List), Appendix I (CITES) and Not listed (CMS).

Arnoux's beaked whale *Berardius arnuxii* (Duvernoy, 1851)

Systematics and populations

Duvernoy (1851) described Arnoux's beaked whale from a 975cm male stranded in Akaroa, New Zealand, the skull of which is curated at the Muséum national d'Histoire naturelle, Paris (Robineau, 1989b). No subspecies, populations or stocks are defined. The species is reviewed by Balcomb (1989), Klinowska (1991), Mead (2002) and Mitchell et al. (1981).

Plate 10. Arnoux's beaked whales (*Berardius arnuxii*). Photo credit: Meike Scheidat.

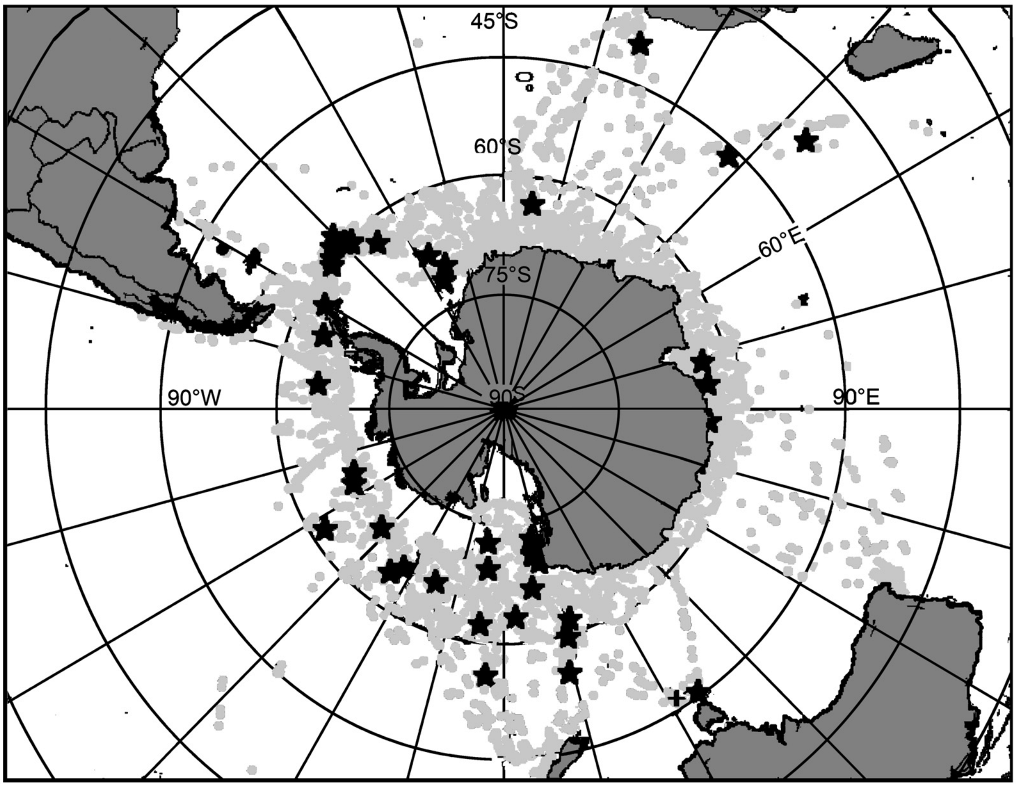

Fig. 8. Sightings of Arnoux's beaked whale (black stars) from IDCR/SOWER cruises.

Distribution in Southern Ocean Sanctuary

Arnoux's beaked whale has a wide circumpolar distribution (Fig. 8), from the Antarctic pack-ice north to approximately 34°S (Reeves *et al.*, 2003). Specimens are reported from New Zealand, Tasmania, South Georgia, South Shetland Islands and sightings from the Tasman Sea (Baker, 1999; Balcomb, 1989; Bannister *et al.*, 1996; Fraser, 1937; Jefferson *et al.*, 1993). Sighted predominantly in Antarctic and subantarctic regions, and frequently close to the ice edge, with many reports along the Antarctic Peninsula and in the Weddell Sea (Hobson and Martin, 1996; Lichter, 1986; McCann, 1975; Stonehouse, 1972; Taylor, 1957). Multiple sightings are reported along Victoria Land coast, western Ross Sea, as far south as the McMurdo Sound ice edge (Ponganis *et al.*, 1995), and at 66°56'S,61°54'E off Kemp Land, east Antarctic coast (Rogers and Brown, 1999).

B. arnuxii occurs both north and south of the Antarctic Polar Front, but there is no information available on seasonal shifts or migration. All stranding records from the South African coast are from summer (Findlay *et al.*, 1992). It is recorded among sea ice, even in winter when trapped (Taylor, 1957).

Conservation status and exploitation

No abundance estimates exist, but it is notably rare compared to the sympatric southern bottlenose whale. The usual dive duration is 15–25min, but *B. arnuxii* can stay submerged for an hour, which complicates sightings-based surveys. Rarely taken in the Antarctic (Mitchell, 1975b). A specimen of *B. arnuxii* was caught outside Deception Island, South Shetlands (Fraser, 1937), another ice-trapped individual was probably killed by gun (Taylor, 1957).

Status designations are Data Deficient (IUCN Red List), Appendix I (CITES) and Not listed (CMS).

Cuvier's beaked whale *Ziphius cavirostris* (G. Cuvier, 1823)

Systematics and populations

No subspecies are currently recognised but, like most other cosmopolitan cetaceans, it is likely that significant population structure exist in this ziphiid, and research is needed. For instance, the status of a named Indo-Pacific subspecies (*Z. cavirostris indicus* Van Beneden, 1863) deserves verification. Rice (1998) dismissed it without any useful argument.

Distribution in Southern Ocean Sanctuary

Until recently, Cuvier's beaked whale was said to be found in all oceans and major seas except in the polar regions (Heyning, 1989). Since Goodall and Galeazzi (1985b) first mentioned 'sightings south of 60°S' several others demonstrated Antarctic penetration. Two observations south of the Antarctic Polar Front in Areas I and VI (Kasamatsu *et al.*, 1988), and three summer sightings were reported at 63.7°S,90°E; 63.3°S,119.6°E and 64.6°S,128.5°E[18] (mean group size = 2) (Ensor *et al.*, 1999). In February 2004, a small group of Cuvier's beaked whales was encountered at close range in Drake Passage, en route to Livingston Island (AMLR 2004 Weekly Report 7, unpublished data). The evidence for Balcomb's (1989) 'from the Antarctic continent and ice edge (78°S) north to about 34°S' is unclear. Southernmost specimens are from the Falkland Islands (at 693cm the largest known specimen; Heyning, 1989), SE of Auckland Islands at 52°08'S (Baker, 1977), New Zealand mainland near Cook Strait (41°24'S) (Baker, 1990), and Tasmania (Guiler, 1978). No migratory movements are known.

[18] Approximate locations read from published chart.

Conservation status and exploitation

No abundance estimates for this offshore, deep-diving species are available for any region in the Southern Hemisphere. Cuvier's beaked whale has not been subjected to any fishery in the Southern Hemisphere (Klinowska, 1991; Mitchell, 1975a; 1975b). Some incidental mortality may occur in high-seas gillnet fisheries. The species appears to be exceptionally vulnerable to acoustic trauma (Reeves *et al.*, 2003). Status designations are Least Concern (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Shepherd's beaked whale *Tasmacetus shepherdi* (Oliver, 1937)

Systematics and populations

Shepherd's beaked whale (or Tasman beaked whale) is almost exclusively known from about 42 strandings in New Zealand, southern Australia, Argentina, Juan Fernández Islands and Tristan da Cunha (Best *et al.*, 2009; Pitman *et al.*, 2006). There are no confirmed sightings. No subspecies nor stocks are discerned, their study made difficult by the scarcity of specimens.

Distribution in Southern Ocean Sanctuary

Probably circum-global in temperate waters of Southern Hemisphere (Mead, 1989c; Rice, 1998), associated with cooler waters from 33°S to at least 53°50'S (Klinowska, 1991; R.N.P. Goodall, pers. comm. to ANB, October 2000). Occurrence further south into (sub-) Antarctic is likely. Most strandings in the New Zealand region are from South Island, Stewart and Chatham Islands, (Rice, 1998); six strandings have occurred in summer (November–March) and one in winter (August), however conclusions on seasonality are premature. Putative sightings were reported in the western South Atlantic (53°45'S, 42°30'W) (Laughlin, 1996) and off Christchurch, on the East coast of South Island (Watkins, 1976), besides three others, but none are convincingly supported (Pitman *et al.*, 2006).

Conservation status and exploitation

Although Shepherd's beaked whale seems ichthyophagous, it is not bycaught in any fisheries (Mitchell, 1975a; 1975b; Northridge, 1984). An oceanic species feeding in deep water, it would be unlikely to suffer any major interactions or habitat problems (Klinowska, 1991; Northridge, 1984). Nonetheless its conservation status remains unknown (Reeves *et al.*, 2003) as reflected in its status designations which include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Strap-toothed beaked whale *Mesoplodon layardii* (Gray, 1865)

Systematics and populations

Largest of mesoplodonts, also known as Layard's beaked whale. No subspecies, populations or stocks are described. May be confused with another large tusked mesoplodont, *M. traversii*, whose external characteristics and behaviour are unknown. At sea, teeth of males *M. traversii* and *M. layardii* may hardly be distinguishable, and it is hoped that the bold colouration pattern of the latter will remain diagnostic once the *M. traversii* colouration will be documented.

Plate 11. Strap-toothed beaked whale (*Mesoplodon layardii*) (ILB061230596). Photo credit: Isabel Beasley.

Distribution in Southern Ocean Sanctuary

Distributed throughout the southern oceans in cold temperate waters (Fig. 9). Specimens have stranded in southern Australia, Tasmania, southern New Zealand (Dixon, 1980; Mead, 1989a; Rice, 1998), and South Georgia[19]. In the past few years a fair number of sightings are reported in Antarctic waters. Southernmost specimens include a rostrum found on Macquarie Island at 54°30'S (re-identified by Baker and van Helden, 1999) and a broken calvaria from Heard Island (53°S, 73°30'E), situated just north of the Sanctuary (Guiler *et al.*, 1987). Most strandings of *M. layardii* occur between 33°S and 53°S (G.J.B. Ross, pers. comm. to ANB, October 2001). Migratory movements cannot be evaluated due to the limited number of confirmed sightings, although the marked seasonality of records from South Africa (January to June) (Findlay *et al.*, 1992) is suggestive of a possible northward shift in distribution during autumn and winter.

Conservation status and exploitation

No catches of *M. layardii* are reported. Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Spade-toothed whale *Mesoplodon traversii* (Gray, 1874)

Systematics and populations

No subspecies, populations or stocks are known. The holotype, a mandible with large teeth, from the Chatham Islands was incorrectly assigned to *M. layardii* for over 125 years. *M. traversii* was resurrected after re-examination of the holotype at the Museum of New Zealand, as well as a White Island specimen (van Helden *et al.*, 2002). A morphological linkage with Bahamonde's beaked whale *Mesoplodon bahamondi* (Reyes *et al.*, 1995) from Chile supported by mtDNA analysis (van Helden *et al.*, 2002), recognised *M. bahamondi* as a junior synonym of *M. traversii*. At sea, this species may potentially be mistaken for *M. layardii* in view of the lack of known distinguishing features and great similarity in male tusks.

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[19] The skull and four ribs of an adult female *M. layardii* are curated at the Museum of South Georgia and were examined and photographed by KVW in December 2003. The *ca.* 5m specimen, found beached at Larsen Harbour, was donated to the museum by Philippe Poupon, skipper of the yacht *Fleur Austral*.

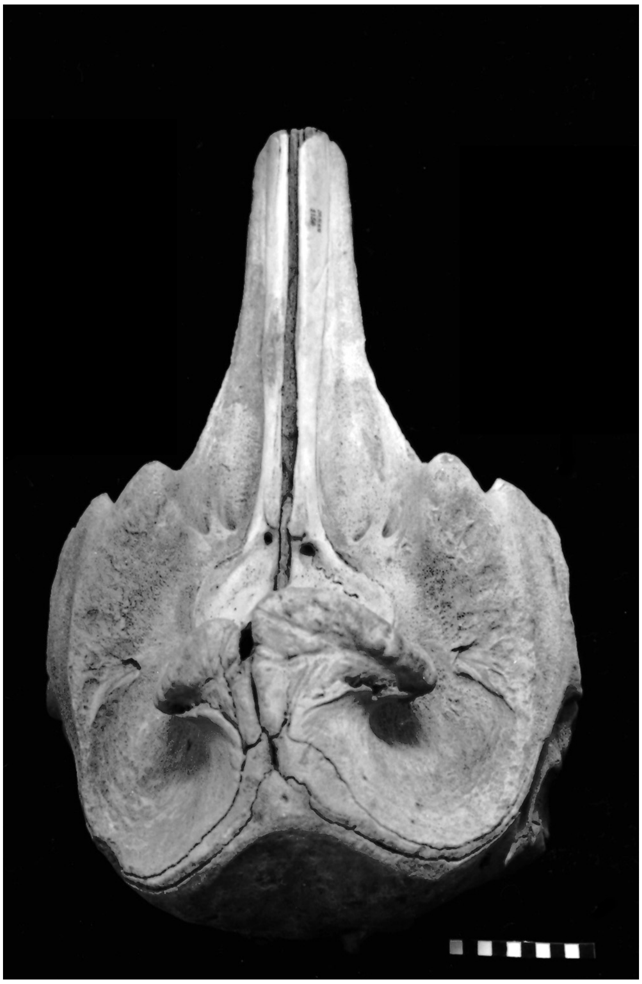

Plate 12. Spade-toothed beaked whale (*Mesoplodon traversii*), dorsal view of holotype skull *M. bahamondi* (MNHN, Santiago). Photo credit: Julio C. Reyes. Scale = 10cm.

Distribution in Southern Ocean Sanctuary

Spade-toothed whale is the least-known of all living species of cetaceans with only three recognised, skeletal, specimens (Reyes *et al.*, 1995; van Helden *et al.*, 2002): one each from Robinson Crusoë Island, Juan Fernández Islands (33°37'S,78°53'W), White Island, NZ (35°31'S,177°11'E) and the only Sanctuary record, from Pitt Island, Chatham Islands (44°17'S,176°15'W). This medium- to large-sized mesoplodont (5.5m estimated adult size; Reyes *et al.*, 1995) may possibly visit (sub-)antarctic waters. No data exist on migratory movements; the three specimens stranded in an indeterminate season.

Conservation status and exploitation

No catches are reported. Conservation status designations of spade-toothed whale are Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Gray's beaked whale *Mesoplodon grayi* (von Haast, 1876)

Systematics and populations

No geographic variation or subspecific division is reported. A North Sea stranding, the only extralimital record from the Northern Hemisphere, is genetically similar to *M. grayi* specimens from New Zealand (M. Dalebout, pers. comm. to ANB, 2001).

Plate 13. A juvenile/female *Mesoplodon* sp., without visible teeth at 55°29.44'S, 19°32.47'E on 31 Dec 2005 (PHE051231), possibly a strap-toothed beaked whale *M. layardii*, or spade-toothed beaked whale *M. traversii*. Photo credit: Paul Ensor.

Distribution in the Southern Ocean Sanctuary

The long-held view was that Gray's beaked whale has an austral circumglobal distribution in temperate or cold temperate waters between 30°S–45°S (e.g. Marcuzzi and Pilleri, 1971; Mead, 1989a; Pitman, 2002; Rice, 1998; Ross, 1979), but not further south. Indeed, most strandings happen between 35°S and 45°S (G.J.B. Ross, pers. comm. to ANB, October 2001). In New Zealand, *M. grayi* is the second commonest single strander after *Kogia breviceps*, with some 180 recorded specimens from both North and South Islands[20]. These strandings and the absence of regular inshore sightings at sea, indicate an offshore population close to New Zealand's east coast. It is known from Tasmania (Baker, 2001; Bannister *et al.*, 1996; Nichols, 1986)[21] .

Pitman (2002) summarised *M. grayi* distribution as 'circumglobal in temperate waters of the southern hemisphere', however the same paper also features a photo taken in Antarctic waters by Richard A. Rowlett. Ohsumi *et al.* (1994) indicate a sighting of Gray's beaked whale, made during a JARPA (Japanese whale research program under special permit) expedition at *ca.* 62°30'S,150°E (from map) in the Australian Antarctic basin. De Boer *et al.* (1999) sighted two Gray's beaked whales near the Balleny Islands, also off the Ross Sea ice edge (*ca.* 67°S; 7–20 January 1999) in 950m of water and 2°C. *M. grayi* also occurs in the Scotia Sea, south of the Polar Front (e.g. Table 2). A 25-year history of IWC/IDCR and IWC/SOWER cruises, till 2003, yielded 31 groups sighted, with mean group size of 3.1 (SD 1.85) animals, and of which 11 groups were sighted south of 60°S, with 65°40'S,014°60'E (sic) the southernmost record (see Dalebout *et al.*, 2004, table 2). Clearly, circumpolar, (sub-)antarctic waters are part of the normal range of *M. grayi* (Fig. 9).

Goodall and Galeazzi (1985a) referred to 53 stranding observations or specimens of *M. grayi* recorded from the tips of the southern continents, the southernmost being Tierra del Fuego at *ca.* 54°S.

Mead (1989a), followed by Ohsumi *et al.* (1994), discussed specimen NMNZ612 in the Museum of New Zealand which he said was collected from an unknown locality in the Antarctic; Mead adds 'it is difficult to attach much importance to this record because *M. grayi* has been

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[20] New Zealand Whale Stranding Database, Museum of New Zealand, Wellington, courtesy A.L. van Helden.

[21] A presumed *M. bowdoini* specimen in the Tasmanian Museum (A748) was re-assigned to *M. grayi* (Baker, 2001).

Table 2
New sightings of (like-) Gray's beaked whale in the Scotia Sea by two authors (KVW and RL), observed from the RRV *James Clark Ross*.

Identification	Date	Location	Group size	Depth\ SST (°C)	Comments
M. grayi	4 Feb. 2003	61.394166°S 31.194° W	6 (5–7)	3,956m 1.16°C	Long, white rostra sticking above sea surface; no teeth seen.
Like *M. grayi*	8 Feb. 2003	56.85648°S 31.59967°W	4 (4–5)	3,608m 2.64°C	In one individual: long rostrum seen before whitish head surfaced; body grey-brown, no scars seen on 3.5–5m body

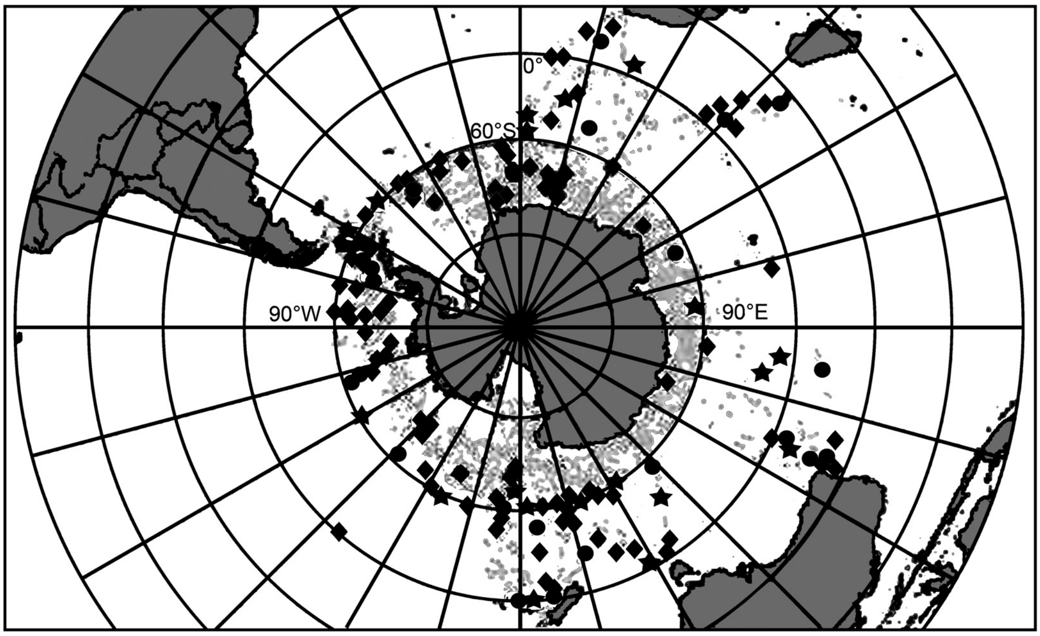

Fig. 9. Sightings of Gray's beaked whale (black stars), strap-toothed beaked whale (black circles) and Shepherd's beaked whale (black diamonds) from IDCR/SOWER cruises.

known to stray before'. However, the origin of this specimen is of particular interest here. Anton van Helden, Collection Manager (marine mammals) at the Museum of New Zealand questions Mead's (1989a) conclusion (in litt. to KVW, 8 April 2003) and believes the specimen is from the Chatham Islands[22]. No information on migration is available, but strandings between 30°S and 50°S occur most frequently from December through March, suggesting a nearshore movement in summer (ANB, unpublished data). The occurrence of early foetuses in May, near-term foetuses in September, and mother with calves in January–February indicates summer breeding in the New Zealand region (ANB, unpublished data).

Conservation status and exploitation

It is the only one of two mesoplodonts[23] known to have mass-stranded: 25 came ashore on the Chatham Islands in 1873 (von Haast, 1877). No information is available on abundance. No catches have been reported in the Southern Ocean. Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Andrews' beaked whale *Mesoplodon bowdoini* (Andrews, 1908)

Systematics and populations

No subspecies, populations or stocks have been designated. At one time suggested to be possibly conspecific with *M. stejnegeri* and *M. carlhubbsi* from the North Pacific (IWC, 1989; Mead, 1989a; Orr, 1953), *M. bowdoini* is now firmly confirmed as a distinct species both morphologically and by molecular genetics (Baker, 2001; Dalebout *et al.*, 1998).

Distribution in Southern Ocean Sanctuary

No confirmed at-sea sightings, and only 35 specimens (strandings) are registered. In New Zealand, stranded specimens are documented from South Island, Stewart, Chatham and Campbell Islands (Andrews, 1908; Baker, 2001; Mead, 1989a). A rostrum was retrieved also from Macquarie Island (Baker, 2001), at 54°30'S the southern-most authenticated record. The northern-most record is from Bird Island, Western Australia, at 32°12'S, 115°40'E.

Within and beyond the Sanctuary, this species is circumpolar, having been recorded also from southern Australia, Tristan da Cunha and the Falkland Islands (Baker, 2001). A presumed specimen from Tasmania (Guiler, 1967)

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[22] *M. grayi* specimen NMNZ612 was donated to the Colonial Museum (now Museum of New Zealand, Wellington) by Lady Kinsey of Christchurch. The next specimen in the register, a mandible of *T. shepherdi*, listed as coming from the Chatham Islands, is from the same donor. The previous record is a southern right whale earbone that is listed as 'Antarctic ?'. A ditto mark is below this associated with the *M. grayi* specimen, however this has been crossed out (probably a product of the transfer of information from the old register). Anton van Helden concludes that, in his opinion 'the most likely origin of the *M. grayi* specimen is the Chatham Islands'.

[23] The other species is *Mesoplodon layardii*, with one known mass stranding in South Africa (Findlay *et al.*, 1992).

was re-identified as *M. grayi* (Baker, 2001). Nothing is known of migratory movements.

In the New Zealand region, calving period is thought to be summer/autumn (Baker, 2001; Klinowska, 1991).

Conservation status and exploitation

No captures have been recorded. In the absence of an abundance estimate, the conservation status of Andrews' beaked whale is unknown. Species status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Hector's beaked whale *Mesoplodon hectori* (Gray, 1871)

Systematics and populations

The systematic status of *M. hectori* has attracted a considerable amount of discussion (Dalebout *et al.*, 2002; Fraser, 1950; Mead and Baker, 1987; Moore, 1960; Ross, 1970). MtDNA polymorphism analysis suggests no subspecies in samples from New Zealand and Australia (M. Dalebout, pers. comm. to ANB, 2001).

Distribution in Southern Ocean Sanctuary

Hector's beaked whale is limited to the Southern Hemisphere. Four presumed specimens from the Northern Hemisphere (California) represent a new species, Perrin's beaked whale *Mesoplodon perrini* (Dalebout *et al.*, 2002; 1998). External features of *M. hectori* are known from only three documented live individuals. A male and female stranded alive in the Buenos Aires Province (Cappozzo *et al.*, 2005) and one juvenile individual, confirmed genetically, was sighted porpoising off southwest Australia (see photos in Best, 2007, p.121 and Jefferson *et al.*, 2008, p.121). Only some 40 stranded specimens have been positively identified, 15 of these originate from within the Sanctuary circumpolar, and the remainder were recorded slightly north of the Sanctuary, to *ca.* 34°S. The largest sample ($n = 16$) is from New Zealand, 13 of those within the Sanctuary, followed by Argentina, Tasmania, South Africa and Chile (Cappozzo *et al.*, 2005; Goodall, 1978; Mead and Baker, 1987). Four specimens have stranded on the Falkland Islands (Scheffer and Rice, 1963; R.N.P. Goodall, pers. comm. to ANB, October 2000). The southernmost specimen record is from Navarino Island, Tierra del Fuego at *ca.* 55°07'S, 67°05'W (R.N.P. Goodall, pers. comm. to ANB, April 2003) and, according to Rice (1998) it appears circumglobal in temperate waters of the Southern Hemisphere. Nothing however suggests *M. hectori* would not occur in (sub-)antarctic waters. With only recent, limited information on diagnostic external features, it may have gone unnoticed so far as unidentified small beaked whales. The seasonal nature of stranding records (December through April in New Zealand) suggests an inshore movement in summer. A mother with calf recorded in early April at Stanley (40°45'S, 147°19'E), northern Tasmania, and in January in both northern Argentina and New Zealand, would be concordant with a summer calving season.

Conservation status and exploitation

Hector's beaked whale is not exploited, but its conservation situation is indeterminate. International conservation status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

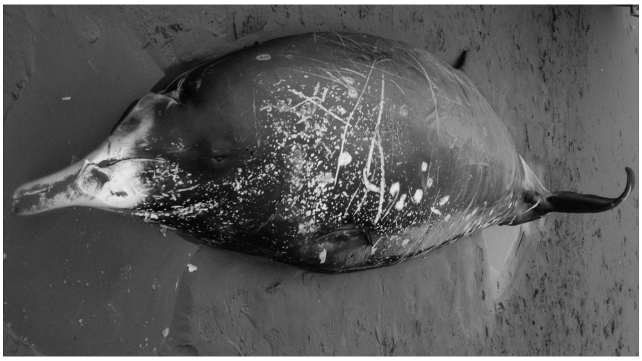

Plate 14. Adult male Gray's beaked whale (*Mesoplodon grayi*) stranded at Canterbury, New Zealand, 19 Sept 2005. Photo credit: Al Hutt.

ODONTOCETES VAGRANT INTO THE SANCTUARY

Six species of odontocetes were determined to be vagrant into the Southern Ocean Sanctuary as outlined in the following species accounts.

Dwarf sperm whale *Kogia sima* (Owen, 1866)

Systematics and populations

No subspecies are recognised, however molecular genetic research suggests significant population structure (Chivers *et al.*, 2005; Plön *et al.*, 2003). Further work should also encompass a study of cranial variation world wide, considering that ample museum specimens are available.

Distribution in Southern Ocean Sanctuary

Cosmopolitan, but not in polar waters (McAlpine, 2002). In the Sanctuary, it is known only from stranded specimens in Tasmania (Bannister *et al.*, 1996; Guiler, 1978). One of us (DT) collected a female with full-term foetus at Cloudy Bay, Bruny Island (*ca.* 43°20'S, 147°19'E), Tasmania. In contiguous waters, strandings are known from South Australia (Bannister *et al.*, 1996), South Africa (Findlay *et al.*, 1992) and three records from northern New Zealand (Baker and van Helden, 1990; ANB, unpublished data) *K. sima* is more coastal than pygmy sperm whale, and thought to prefer warmer water (McAlpine, 2002). Information is lacking on migration in and around Sanctuary waters.

Conservation status and exploitation

No population estimates or other data specific to the study area exist; however incidental mortality in fishing gear and ingestion of plastic debris may be problematic (McAlpine, 2002; Reeves *et al.*, 2003). Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS)

Lesser beaked whale *Mesoplodon peruvianus* (Reyes, Mead and Van Waerebeek, 1991)

Systematics and populations

Also named Peruvian beaked whale or pygmy beaked whale. No subspecies or populations are identified, however *M. peruvianus* from the eastern and western Pacific, and from Southern and Northern Hemispheres may belong to different stocks. Sightings of 'Mesoplodon sp. A' in the offshore eastern Pacific, tentatively assigned to *M. peruvianus* by Pitman and Lynn (2001) based on remotely estimated body size and adult male tooth position remain to be confirmed as such by specimens or molecular genetics and until then can not be 'identified' as *M. peruvianus*.

Distribution in the Southern Ocean Sanctuary
Lesser beaked whale specimens have been reported from the temperate eastern Pacific including Peru, Chile, Mexico (Reyes *et al.*, 1991; Sanino *et al.*, 2007) and California, USA (Dalebout *et al.*, 2007). One 372cm physically mature individual stranded at Kaikoura (42°31'S,173°30'E), New Zealand (Baker and van Helden, 1999), the hitherto southernmost record for the species, locating it firmly within the Sanctuary. Pitman and Lynn (2001) referred to the latter as 'almost certainly an extralimital record', perhaps prematurely, taking into account that few positive sightings of *M. peruvianus* exist and that recent records in coastal waters around 29°S in north-central Chile, including a specimen (Sanino *et al.*, 2007) have moved the known range in the eastern Pacific 14° latitude to the South. The true austral distribution range of lesser beaked whales remains uncertain, and it could extend further south into the Sanctuary.

Little is known on seasonality and even less on migration. Most fresh specimens in Peru were landed during summer months (Reyes *et al.*, 1991; K.Van Waerebeek and J.C. Reyes, unpublished data), possibly related to inshore movements linked to reproductive behaviour or prey availability. Apparent correlation with SST may be spurious, indeed in the Humboldt Current system the low SST year round is only weakly linked to seasons.

Conservation status and exploitation
Off Peru, lesser beaked whales are incidentally taken with some regularity (Reyes *et al.*, 1991; K. Van Waerebeek and J.C. Reyes, unpublished data). A skull retrieved from a beach in northern Chile showed two bullet lesions (Sanino *et al.*, 2007; Van Waerebeek *et al.*, 1999). No abundance estimates, nor precise distribution data are on hand, hence it is listed as Data Deficient in the IUCN Red List, Appendix II (CITES) and Not listed (CMS).

Dense-beaked whale *Mesoplodon densirostris* (de Blainville, 1817)[24]

Systematics and populations
Also referred to as Blainville's beaked whale. No subspecies or populations have been described, but given its global distribution (see Pastene *et al.*, 1990), studies of geographic variation are needed. Besharse (1971) studied individual cranial variation.

Distribution in Southern Ocean Sanctuary
At the southern end of its circumglobal distribution in low and mid-latitudes, contiguous to the Sanctuary, dense-beaked whale is known from strandings in South Africa (Findlay *et al.*, 1992), New Zealand (at 39°19'S,176°57'E, 90 Mile Beach, Northland and Tangoio, Hawke Bay), just north of the Sanctuary boundary (Baker and van Helden, 1999), and southern Chile at 41°28'S,73°00'W (Bannister *et al.*, 1996; Guiler, 1966; Pastene *et al.*, 1990). The only record firmly within the Sanctuary boundaries is a specimen from Tasmania, at 40°50'S (Bannister *et al.*, 1996; Guiler, 1966). A mesoplodont rostrum from Macquarie Island was thought to be a vagrant far beyond the supposed normal range' (Bannister *et al.*, 1996), however this specimen was re-identified as a large male *M. layardii* (Baker and van Helden, 1999).

No evidence of migratory movements. Dense-beaked whale's normal warm-water distribution suggests that it penetrates northernmost Sanctuary strata (circa 40°S) predominantly in summer months. Its most boreal incursion is off Nova Scotia, Canada, at 45°N (Pastene *et al.*, 1990), and austral latitudinal penetration is likely comparable.

Conservation status and exploitation
No abundance estimates are available. One adult male killed by fishermen near Puerto Montt, southern Chile, was rendered for its oil (Pastene *et al.*, 1990). Status designations are Data Deficient (IUCN Red List), CITES Appendix II and Not listed (CMS).

Risso's dolphin *Grampus griseus* (G. Cuvier, 1812)

Distribution in the Southern Ocean Sanctuary
Risso's dolphin normal distribution includes warm and temperate seas. Off Australia south to 39°S (Victoria) (Bannister *et al.*, 1996). The species has been recorded from within the Sanctuary in New Zealand waters by Baker (1974), who reported two strandlings on the northern coast of Cook Strait at 41°17'S,174°54'E. An adult Risso's dolphin, the celebrated 'Pelorus Jack', accompanied ships across Admiralty Bay, Marlborough, New Zealand (40°58'S, 173°51'E) between 1880 and 1912 (Baker, 1974). Off southern Patagonia, Risso's dolphin has been encountered as far south as Punta Catalina, Magallanes, Chile (Venegas and Sielfeld, 1978) situated at 52°33'S, 68°46'W, and Puerto Roca, Isla de los Estados, Argentina (54°45'S, 63°53'W) (Goodall, 1986) so it may penetrate considerably farther south into the Tasman Sea than the Cook Strait record suggests.

Conservation status and exploitation
Listed as Least Concern (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

Rough-toothed dolphin *Steno bredanensis* (Lesson, 1828)

Distribution in Southern Ocean Sanctuary
Typically a warm-water species, the rough-toothed dolphin is known from the Sanctuary only at 41°30'S,174°03'E, Cloudy Bay, east coast of South Island, New Zealand, where two specimens stranded together in June 1990 (ANB, unpublished data). *S. bredanensis* has occasionally stranded in cold-water upwelling influenced areas. One specimen was retrieved from a beach in northern Chile (Van Waerebeek and Guerra, 1988), still the only positive record for Chile, while a second specimen was found in a collection in northern Namibia (assumed by Findlay *et al.*, 1992 to be a local specimen). Both rough-toothed dolphins most likely accidentally penetrated these cool areas from warm offshore waters. Ross (1984) noted that the specimens attributed to the Cape of Good Hope may have been collected elsewhere by vessels en route to Europe via the Cape Colony.

Conservation status and exploitation
Steno bredanensis is classified as Least Concern (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

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[24] de Blainville, Henri Marie Ducrotay (1777–1850), French physician. *Non* 'Blainville' (Hershkovitz, 1966; *et alii*).

ODONTOCETE OCCURRING CONTIGUOUS TO THE SANCTUARY

Two species, both beaked whales, have been encountered contiguous to the Southern Ocean Sanctuary, i.e. less than 120 n. miles north of its northern boundaries. It is plausible that at some point these species will be found within the Sanctuary.

Ginkgo-toothed beaked whale *Mesoplodon ginkgodens* (Nishiwaki and Kamiya, 1958)

Distribution in Southern Ocean Sanctuary

The ginkgo-toothed beaked whale is known from three stranding events on the New South Wales coast, Australia, at *ca.*37°S (Bannister *et al.*, 1996), and one on the west coast of the North Island of New Zealand at 39°S (A.L. van Helden, pers. comm. to ANB, 2003). We conclude that *M. ginkgodens* inhabits the Tasman Sea.

A skull initially assigned to this species, collected at White Island, New Zealand (37°31'S,177°11'E) (Baker and van Helden, 1999) was re-identified as *M. bahamondi* (Reyes *et al.*, 1991), junior synonym of *M. traversii* (see van Helden *et al.*, 2002).

Conservation status and exploitation

Status designations for *M. ginkgodens* are Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

True's beaked whale *Mesoplodon mirus* (True, 1913)

Systematics and populations

No subspecies or populations are described, however divergent Southern and Northern Hemisphere ESU may exist taking into consideration the lack of records in equatorial waters.

Distribution in the Southern Ocean Sanctuary

In the Southern Hemisphere, just a few strandings are known, including from areas contiguous to the Southern Ocean Sanctuary, namely South Africa (Findlay *et al.*, 1992; Ross, 1969) and southern Australia at 38°24'S (Bannister *et al.*, 1996; Dixon and Frigo, 1994). Reports from Tasmania and western Australia (Bannister *et al.*, 1996) have to date not been substantiated. No evidence of migratory movements exists.

Conservation status and exploitation

Status designations include Data Deficient (IUCN Red List), Appendix II (CITES) and Not listed (CMS).

DISCUSSION

The design of the majority of cetacean surveys in the Southern Ocean has been focused on those species subject to commercial exploitation although many surveys recorded sightings of all species. The Southern Ocean Sanctuary was established in 1994 and in recent years, there has been a much greater emphasis on multi-disciplinary research programmes. Many of these have provided data resulting in an enhanced basic knowledge of the distribution of odontocetes in the Southern Ocean. Nonetheless, many species remain very poorly known but this review shows that odontocete diversity south of the Antarctic Polar Front is higher than previously thought and that several species venture significantly further south.

Beaked whales

Despite hundreds of biologists navigating in Antarctic waters for two centuries, until recently, the only two ziphiids recognised from the Antarctic Ocean have been the southern bottlenose whale and Arnoux's beaked whale (e.g. Brownell, 1974; Kasamatsu and Joyce, 1995; Miyazaki and Kato, 1988; Nishiwaki, 1977; Ponganis *et al.*, 1995), easily identified due to large size and highly visible bulbous head. Other species of beaked whales are not uncommon, but less conspicuous. As recently as Nishiwaki (1977), *M. layardii, M. grayi* and *M. hectori* were considered of a temperate-cold, but not a cold-water, distribution. Goodall and Galeazzi (1985a) first suggested that 'a species of Mesoplodon' may reach the South Shetland Islands.

Until fairly recently, many research cruises pooled mesoplodonts under ziphiid whales (e.g. Nishiwaki *et al.*, 1999; most SOCEP cruises) as the positive identification of several beaked whale species, considering their often still poorly described diagnostic external features, was not considered feasible at sea. Also, the few stranding records south of 45°S may reflect the paucity of land surfaces and any associated human populations. Most subantarctic islands are uninhabited and only visited during dedicated expeditions for purposes not including searching for stranded cetaceans. Despite these limitations, evidence presented in this paper suggests that mesoplodonts are widely distributed throughout (sub-)Antarctic seas. *M. layardii, M. grayi* and unidentified mesoplodonts (Tables 2 and 3) are not uncommon, especially considering their lengthy and deep diving and inconspicuous surface behaviour which makes them difficult to detect. Positive data are still too scarce to establish the southern distribution range for several other mesoplodonts, including two smaller *M. hectori* and *M. peruvianus,* and four larger species *M. bowdoini, M. traversii, M. mirus* and *M. ginkgodens*, but any of these could occur in the (sub)antarctic. Mesoplodonts continue to surface in unexpected areas. For instance, (boreal) polar distribution was recently revealed for Sowerby's beaked whale *Mesoplodon bidens* (Carlström *et al.*, 1997; Lien and Barry, 1990).

Mesoplodonts previously identified as like strap-toothed beaked whales (and perhaps even some positive sightings) may conceivably cover also some spade-toothed beaked whales *M. traversii* (Gray, 1874) taking into consideration that external features of the latter are unknown and no sightings have been reported. The tusk-size teeth of adult male *M. traversii* are so similar to these of *M. layardii* that the two were confused for over a century (see van Helden *et al.*, 2002). Future morphological data hopefully should pin-point discriminating features.

While earlier some authors denied a distribution of Cuvier's beaked whale in polar regions (Heyning, 1989; Moore, 1963; Rice, 1998), their occurrence south of the Antarctic Polar Front is now well established.

If many earlier surveys in (sub-) Antarctic waters did not reveal Mesoplodon spp., it may be readily explained by a bias in research effort focused on large whales[25] and a lack of confidence in beaked whale identification leading to lumping species as ziphiids (e.g. Kasamatsu and Joyce, 1995; Miyazaki and Kato, 1988; Ohsumi *et al.*, 1994). This

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[25] Closing on ziphiids is rarely attempted during surveys.

does not exclude a potential austral shift in mesoplodont latitudinal distribution over time. Increasingly experienced observers, the use of high magnification binoculars, and enhanced dedicated research effort will continue adding positively identified sightings and shed light on the issue. Rapid advances in digital camera and lens technology over the last decade has facilitated obtaining high resolution images of diagnostic features at sea.

Status

It is obvious from the above review that the abundance and status of odontocete populations in the Southern Ocean Sanctuary are poorly known. Branch and Butterworth (2001) calculated estimates for sperm whale, killer whale, and southern bottlenose whale from the IDCR/SOWER surveys, but noted important caveats surrounding all of these estimates. For deep-diving species such as sperm whale and southern bottlenose whale, the assumption of $g(0) = 1$ is not realistic and will result in an unquantified but possibly substantial negative bias. For hourglass dolphins and southern right whale dolphins in particular, estimates are also complicated by responsive movement because these delphinids are known for approaching vessels in order to bowride. SOCEP data suggest long-finned pilot whales and killer whales often approach vessels with active, bottom-mounted pingers (D. Thiele, unpublished data).

Kasamatsu and Joyce (1995) had previously calculated abundance estimates for sperm whales, killer whales, long-finned pilot whales, hourglass dolphins, and all beaked whales combined (due to small sample sizes) from the IDCR surveys between 1976/77 and 1987/88. This included data from the first (1978/79–1983/84), and part of the second circumpolar survey. An estimate of $g(0)$ was made for each species based on a model of diving behaviour and this was used to correct the abundance estimates. For several of the species including sperm whale, long-finned pilot whale and hourglass dolphin the area south of 60°S covered by the IDCR/SOWER surveys has only limited overlap with their known latitudinal range in the Southern Ocean. The different latitudinal and longitudinal coverage of the circumpolar surveys has complicated comparisons between them. Branch and Butterworth (2001) did not find reliable evidence of any trends in odontocete numbers.

Trophic relationships

Some studies have attempted to examine the prey consumption by odontocetes in the Southern Ocean (Kasamatsu and Joyce, 1995) and particularly the consumption of cephalopods relative to pinnipeds and seabirds (Hindell *et al.*, 2003; Santos *et al.*, 2001). These comparisons are complicated by the exact area considered, assumptions about factors such as energy requirements that affect estimates of total prey consumption and the data used for abundance estimates. Kasamatsu and Joyce (1995) estimated that beaked whales consumed around three times more prey mass than sperm whales, whereas the range of values estimated by Santos *et al*. (2001) for sperm and beaked whales overlapped but with sperm whales having the higher values. Regardless of the exact data used, it is clear that odontocetes play an important role in the Southern Ocean ecosystem. Understanding the feeding ecology of odontocetes is further complicated by the lack of data on cephalopods. There have only been limited exploratory fisheries for squid in Antarctic waters and most population data that exist have been derived from remains, especially beaks, in the gastro-intestinal tracts of higher predators. Estimates of the proportion of diet consisting of squid compared to other invertebrates and fish may be biased by squid beaks remaining undigested for longer than other prey items (IWC and CCAMLR, 2010).

Deep-water squid, thought to be regurgitated at the surface by sperm whales form part of the diet of several species of albatross. Clarke *et al.* (1981) concluded from an examination of wandering albatross (*Diomedea exulans*) diet that twenty-two of the species found in albatross regurgitations have also been identified from sperm whale stomachs: only three species were not found in sperm whale stomachs. It would be hard to imagine how else an albatross could catch such squid. Clarke and Prince (1981) found less overlap with sperm whale diet in grey-headed albatross (*Thalassarche chrysostoma*) and black-browed albatross (*Thalassarche melanophris*), finding one large *Ancistrocheirus sp*. squid beak that was likely regurgitated, but the remainder of species found were thought to be caught at the surface. Thus it is possible that sperm whales play a significant role in the feeding ecology of albatrosses, particularly the wandering albatross.

When CCAMLR was first negotiated as part of the Antarctic Treaty System the initial objective agreed by the Antarctic Treaty Consultative Parties in 1977 was to ensure that exploitation of krill would not inhibit the recovery of whale and seal populations. However, it is also non-krill dependent predators, such as odontocetes, that may have been affected by recent fisheries, including the collapse of the marbled rockcod (*Notothenia rossii*) in the early 1970s and, some stocks of the Patagonian toothfish (*Dissostichus eleginoides*) within the CCAMLR area (Constable *et al.*, 2000). Squid fisheries also have a high potential to impact on odontocetes, most especially on beaked whales, many of which seem strictly teuthophagous, as well as on sperm whales. Following declines in catches of *Ilex argentinus* in the southwest Atlantic, the ommastrephid squid (*Martialia hyadesi*) is a likely candidate for further exploitation (Rodhouse, 1997). *M. hyadesi* is widely distributed in the sub-Antarctic Scotia Sea and in considering an ecological approach to the potential fisheries management for this species, Rodhouse (1997) included sperm whale, southern bottlenose whale and long-finned pilot whale as significant predators. The southern elephant seal (*Mirounga leonina*) is also a major predator on squid and amongst the pinnipeds probably occupies the closest ecological role to sperm whales and beaked whales in terms of diet and diving behaviour and off-shore foraging patterns. Hindell *et al.* (2003) suggested that southern elephant seals account for between 19–36% of the total Antarctic consumption of cephalopods by sperm whales, beaked whales, seals and seabirds combined. In the Indian Ocean sector, southern elephant seal numbers declined between the 1950s and 1980s. Although food availability has been regarded as an explanation for the decline (e.g. McMahon *et al.*, 2003) an alternative suggestion is that the decline may have been due to predation by killer whales (Branch and Williams, 2006).

A workshop on ecosystem models in the Antarctic held by CCAMLR and IWC in 2008 noted that the lack of information on absolute and relative abundance for squid severely limits the ability to include this component in ecosystem models (IWC and CCAMLR, 2010). Thus predicting the effects of changes in environment or prey abundance on odontocetes will be especially difficult.

There has generally been insufficient data to examine relationships in distribution patterns between odontocetes and other cetaceans in the Southern Ocean. Most of the relationships examined involve killer whales. Results from the IDCR/SOWER surveys indicate a strong correlation between observed densities of killer and minke whales with densities of both species being highest close to the ice edge (Branch and Butterworth, 2001; Branch and Williams, 2006). Leaper *et al.* (2000) noted an association in occurrence between sperm whales and killer whales from line-transect data in the Scotia Sea. Other authors have also reported observations of the two species together (e.g. Mikhalev *et al.*, 1981; Nolan *et al.*, 2000). However, analysis of data from circumpolar surveys demonstrated temporal variations in density suggesting 'different migration patterns by species, especially between sperm whale and killer whale' (Kasamatsu and Joyce, 1995). This may indicate that correlations between sperm whale and killer whale distribution patterns are limited to localised areas.

From the above, it becomes clear that a significant amount of new information on Odontocete spatial and temporal distribution in the Southern Ocean, and their ecological interactions, has become available over the past decade, but our knowledge remains patchy. This could optimally be addressed by more directed research effort, instead of relying mostly on incidental encounters during cruises with research protocols targeting baleen whales.

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