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Monitoring cetaceans in European waters

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ABSTRACT

1. Monitoring spatial and temporal patterns in cetacean abundance involves a variety of approaches depending upon the target species and the resources available. As a first step, the collection of incidental sightings or strandings information aids the construction of a species list and a rough measure of status and seasonal variation in abundance. These often make use of networks of volunteer observers although the wide variation in abilities and experience means that special attention must be paid to training and to data quality control. More robust monitoring of numbers requires quantification of effort and some correction for factors that influence detectability, such as sea state.

2. The presence of cetaceans may be recorded visually, or indirectly by acoustics. Each has advantages and disadvantages, and their applicability may vary between species. The use of fixed stations tends to allow sustained monitoring at relatively low cost but coverage is limited to the immediate vicinity. For more extensive coverage, mobile platforms are necessary. Platforms of opportunity such as ferries, whale-watching boats, etc. are often used to survey areas at low cost. These may allow repeat observations to be made over time, but with no control over where the vessel goes, it is typically not possible to sample wide areas, thus limiting abundance estimation.

3. Line transect surveys using dedicated platforms allow representative coverage of areas from which abundance estimates can be made (either using indices or absolute measures derived from density estimation). Assumptions relating to detectability and responsiveness need to be addressed and various methods (such as two-platform surveys) have been developed to accommodate these.

4. For some cetacean species, mark-recapture methods can be applied using photo-identification of recognizable individuals. Again, a number of assumptions are made, particularly relating to recognizability, representativeness of sampling and capture probabilities. Capturing, on film, as many animals in the population as possible helps to reduce the problem of heterogeneity of capture probabilities. Mark-recapture methods require at least two sampling occasions. If multiple sampling is employed, either open or closed population models can be used.

5. Measuring population change represents a particular challenge for mobile animals such as cetaceans. Changes in ranging patterns may have a large impact on abundance estimates unless very large areas are adequately covered. Power analysis is a useful method to indicate the ability of the data to detect a trend of a given magnitude. Increasingly, spatial modelling using GLMs and GAMs is being used to provide a better understanding of the biotic and hydrographic factors influencing cetacean distribution.

Keywords: acoustics, distribution, dolphins, line transect surveys, marine mammals, photo-identification, population change, whales

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INTRODUCTION

Animal populations may change in size and distribution over time for a wide variety of reasons. Monitoring those changes and then identifying the causes for them forms the core of conservation research. Information on spatial and temporal variation in cetacean abundance, as with other taxa, is essential to determine both whether management actions are necessary and the effectiveness of any actions that are taken. Such information must be interpreted in the light of other information on population structure, on direct (e.g. harvesting and capture in fishing gear) and on indirect (e.g. pollution and disturbance) anthropogenic effects. Examples of how different types of information are of value to meet management objectives include:

1. Information on *trends in abundance* is useful both for identifying populations for which there is concern and for monitoring whether management actions taken are working.

2. Information on *absolute abundance,* in conjunction with information on population structure, direct and indirect removals, and productivity, can identify populations for which management action is required.

3. Information on *geographical and temporal distribution* guides us in determining whether there are predictable areas and times of concentration that can be used to focus conservation measures in relation to human activity (e.g. by-catch reduction measures; disturbance by shipping, tourism, etc.). It may also highlight times and areas of special significance for various stages in the life cycle, such as calving or mating.

In this paper, we review the various approaches that are currently being used for monitoring cetacean populations, highlighting the strengths and limitations of each and giving examples, drawn where possible from within Europe. Given that available resources are often limited, we make some judgements on their relative cost-effectiveness and indicate the species for which a particular approach might be most appropriate. We do not have the space to go into detail on any one methodology, but have given some key references for further reading. General texts not cited below to which the reader is referred include Garner *et al*. (1999), Hammond (1987, 1995, 2002) and, for acoustic surveys, Gordon & Tyack (2002). In addition, the European Cetacean Society held a workshop on monitoring by visual and acoustic surveys, and this discusses a number of practical issues that need to be considered for meaningful results (Evans, 1990b).

A number of approaches are generally adopted towards the collection of information on cetacean distribution and status. These are detailed in the following sections ordered broadly from lowest costs but yielding least information to highest costs but yielding most information.

Various types of platform can be used within each approach: they can be fixed observation points such as headlands, islands or oil rigs; or they can be mobile survey platforms, including aircraft and a wide variety of vessels. Surveys can be very basic or sophisticated capable of yielding indices of abundance or absolute abundance measures. Most attention is given to the more developed visual methods but acoustics are also considered.

Before considering the various approaches to monitoring, it is important to clarify the questions one is trying to answer – is it to investigate spatial patterns of usage of an area (population distribution), or to identify changes in abundance (population status), or changes in life history parameters (fecundity, mortality), or a combination of these? Equally important is prior thought on what is being monitored: the population across its entire range, a population inhabiting a particular area, or usage of that area? Since cetaceans are mobile and wide-ranging, frequently crossing national boundaries, there are great advantages for collaborative efforts to involve neighbouring countries. However, in practice, most studies will be concentrated upon smaller areas. For that reason, careful consideration should be given to the possibility that local status changes observed may simply reflect a shift in the distribution of that population.

FIXED STATIONS AND PLATFORMS OF OPPORTUNITY

Incidental records for preliminary information on status and distribution

For regions about which little is known, the collection of incidental sightings or strandings information tends to be the first step to developing a species list and some rough measure of status and seasonal variation in abundance. It provides no quantitative measure for assessing population change and is often difficult to interpret without information on effort and sightability, but it yields basic data at low cost, and can be useful in drawing attention to geographical areas or seasons for cost-effective targeting using more refined survey methodology. It may also reveal gross distributional changes over time. For rare species, unless they occur in predictable locations, it may be the primary source of information.

Several European countries have either regional or national schemes for reporting strandings and sightings. The nature of strandings data is that although they provide important information on life history parameters (growth, age and size at sexual maturity, longevity, reproductive rates, and seasonality of reproduction), as well as other biological aspects such as pathology, taxonomy, genetics, diet, and contaminant loads, they tend to be crude measures of status and distribution. Even where effort has been quantified, changes in numbers of bodies washed ashore may reflect increased general mortality, mortality caused by a particular factor, increased population size, or a change in distribution. A good example of this is the sperm whale *Physeter macrocephalus*; the recent increase in reported strandings in North-west Europe has been attributed to a range of causes from an actual population change through to increased pollution and sound disturbance (Jacques & Lambertsen, 1997).

Sightings data come from a wide variety of platforms which may be coastal land-based observation points such as headlands, vessels of many different types, or light aircraft/helicopters. There tends to be greater heterogeneity of observers since the general public are involved as well as specialists. This means that for data to be of any value, special emphasis has to be placed on ensuring that they are of high quality and that species identification is correct. Once started, and operating satisfactorily, it is important that these schemes continue for a long time otherwise the data have limited value. In the British Isles, such a scheme has been operating since 1973, with a network of observers providing information on the overall status and distribution of the 28 cetacean species that have been recorded locally (Evans, 1976, 1980, 1992, 1998; Evans *et al*., 1986; Table 1). Even without corrections for varying effort or sightings conditions, they yield broad-scale information on status and distribution at low cost. Simple compilation of sightings plotted on a grid-based scale may reveal overall distribution patterns without recourse to expensive survey effort. This is illustrated by maps of white-beaked dolphin *Lagenorhynchus albirostris* and common dolphin *Delphinus delphis* sightings constructed more than 10 years ago (Fig. 1), compared with more sophisticated maps derived from a larger and more recent data set with corrections for effort and sea state (Fig. 2). A 'snapshot' survey may have difficulty detecting the presence of a species in those areas where it is rare. Thus, the SCANS survey in July 1994 had no sightings of harbour porpoises *Phocoena phocoena* in the Channel, but casual sightings reveal the species to be present in small numbers, mainly in the western portion and in winter months. The larger the number of observers and broader their coverage, the lower the influence that potential biases may have.

Sightings data, held for example in an Access database, can be linked to other data sets

Table 1. List of cetacean species recorded in UK waters

13 species are regular Harbour porpoise *Phocoena phocoena*, short-beaked common dolphin *Delphinus delphis*, white-beaked dolphin *Lagenorhynchus albirostris*, Atlantic white-sided dolphin *Lagenorhynchus acutus*, bottlenose dolphin *Tursiops truncatus*, Risso's dolphin *Grampus griseus*, long-finned pilot whale *Globicephala melas*, killer whale *Orcinus orca*, northern bottlenose whale *Hyperoodon ampullatus*, sperm whale *Physeter macrocephalus*, minke whale *Balaenoptera acutorostrata*, fin whale *Balaenoptera physalus*, and humpback whale *Megaptera novaeangliae*
Seven species are occasional Striped dolphin *Stenella coeruleoalba*, beluga *Delphinapterus leucas*, Cuvier's beaked whale *Ziphius cavirostris*, Sowerby's beaked whale *Mesoplodon bidens*, True's beaked whale *Mesoplodon mirus*, sei whale *Balaenoptera borealis*, and blue whale *Balenoptera musculus*
Eight species are vagrant Fraser's dolphin *Lagenodelphis hosei*, melon-headed whale *Peponocephala electra*, false killer whale *Pseudorca crassidens*, narwhal *Monodon monoceros*, Blainville's beaked whale *Mesoplodon densirostris*, Gervais' beaked whale *Mesoplodon europaeus*, pygmy sperm whale *Kogia breviceps*, and northern right whale *Eubalaena glacialis*

(effort and environmental information), and using GIS packages such as ARC-INFO and ARCVIEW, they may yield grid-based plots of distribution that can then be compared with remote sensing data and other sources of environmental information. A Joint Cetacean Database has been established in the UK, holding three main databases: European Seabirds At Sea (recording cetaceans alongside seabird observations) held by JNCC; Sea Watch Foundation; and the Sea Mammal Research Unit (SCANS Survey). These have been used to produce a cetacean distribution atlas (Reid, Evans & Northridge, 2003).

Quality control

The desire to recruit larger numbers of observers for widening coverage both in space and time should not be compromised by a greater heterogeneity in the quality of the data gathered. This is a major challenge requiring much effort, and is often given too little attention. The first need is for correct species identification. Records should be accompanied by good descriptions and/or pictures. For the former, it is best to use standardized forms which direct observers to record the most salient information that allows independent verification (examples of such forms can be downloaded from the website: http://www.seawatchfoundation.org.uk).

Photographs allow a more objective evaluation of a sighting record, but heavy reliance upon them could introduce a bias in relative numbers since some species are easier to photograph or identify from photos than others. Some species (e.g. harbour porpoise) are not amenable to being photographed every time they are seen, being inconspicuous, only showing a small portion of themselves above the surface and for very short periods. In those, and other circumstances, the use of video can be very useful, more so than still photographs, although it is unlikely to replace the latter entirely because of its greater expense and generally more limited scope of the lenses. If still photographs or video are used routinely to accompany sightings records, it may be helpful to use a data back or take a picture of the recording form directly after photographing the animal so as to avoid mixing up records. Nowadays, developments in photographic technology (accurate exposure control, fast autofocus zoom lenses, motor drives, etc.) have made it feasible for persons without much experience to take usable photographs for species identification. The rise in digital photography has further aided this by enabling people to check immediately whether the pictures they have taken are adequate.

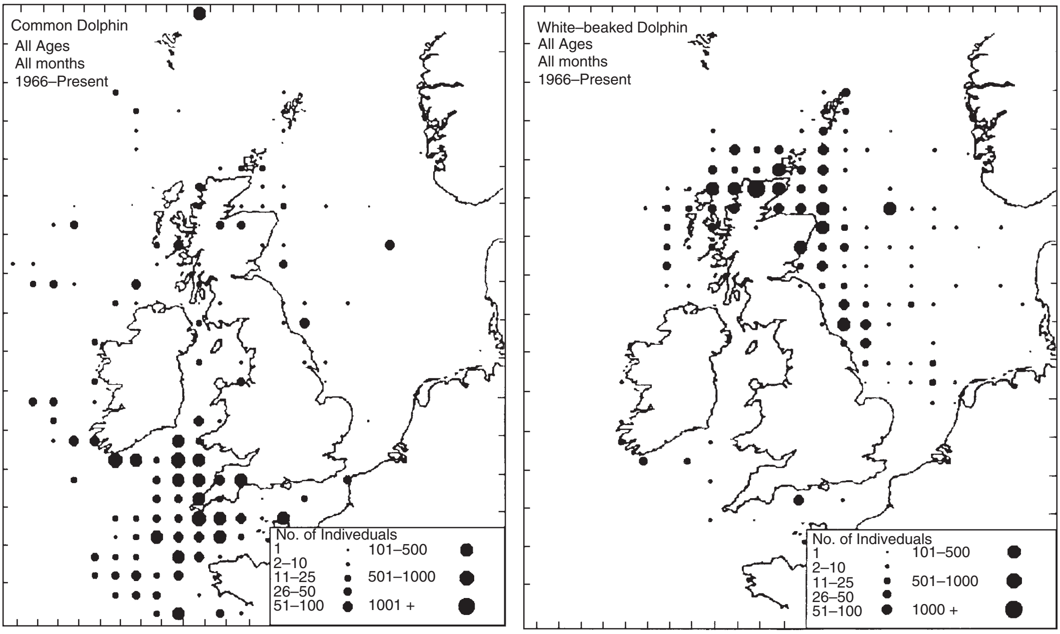

Fig. 1. Distribution maps of white-beaked dolphin *Lagenorhynchus albirostris* and common dolphin *Delphinus delphis*, from incidental sightings (from Evans, 1990a).

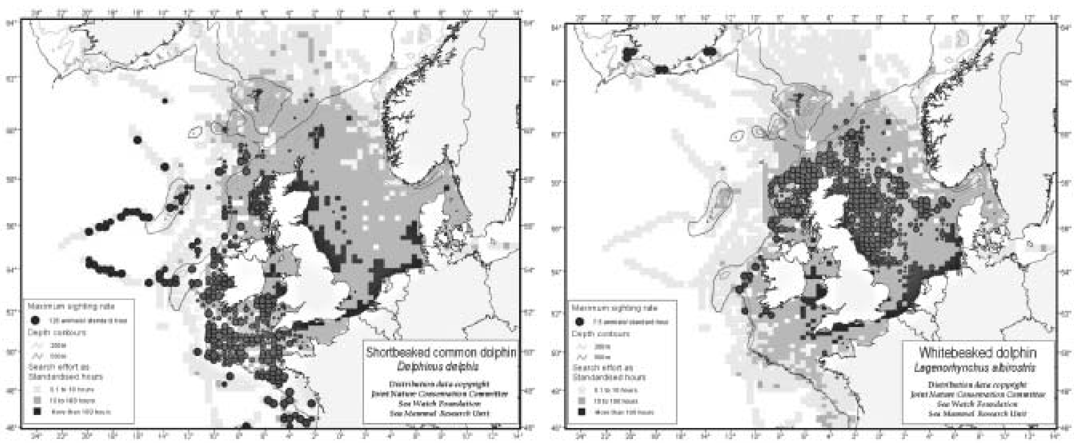

Fig. 2. Distribution maps of white-beaked dolphin *Lagenorhynchus albirostris* and common dolphin *Delphinus delphis*, from effort-related sightings (from Reid, Evans & Northridge, 2003).

Data should only be used for which there is confident species identification. It is important that one is not afraid to discard data if reliability or accuracy is in doubt.

Although observers may vary in their abilities to record sightings of cetaceans accurately, most improve with practice. This will be achieved in part by the experience they themselves acquire from repeated sightings, but can be aided further by suitable identification aids. A number of illustrated guides exist (see for example booklets by Evans, 1995 for European Seas, and Cresswell & Walker, 2001 for the Bay of Biscay and English Channel; and books by Kinze, 2003 for the North Atlantic, and Reeves *et al*., 2002 for worldwide coverage). Many organizations have also produced identification charts or posters (in the UK and Ireland, see those produced by Sea Watch Foundation, Irish Whale & Dolphin Group, and Whale & Dolphin Conservation Society), whilst Sea Watch Foundation has produced a pack of 80 slides (depicting 30 species) and a 40-minute video (depicting 24 species) aimed specifically at identification of European species.

Two obvious biases arise when using a network of observers over a wide area. These are that effort is likely to be greater in some areas compared with others, and that it will be greater at particular seasons. Human populations are not evenly distributed and there is a tendency for people to congregate in particular areas during holidays, especially when weather is most favourable. Coverage will generally be greatest in inshore waters, and this should obviously be borne in mind when interpreting distribution patterns.

Effort should be made to extend coverage evenly over as wide an area as possible, and to do likewise for coverage throughout the year. This may require extra effort targeted on particular localities and times of the year, using either one's own personnel or making use of potential observers who as part of their job are operating over a large area (e.g. fisheries and coastguard aircraft patrols) and/or throughout the year (e.g. oceanographic research vessels and ferries). In those cases, training procedures and critical evaluation using photographs and/or descriptions become ever more important.

Systematic recording of effort-related data from fixed stations

Changes over time in the status of a species may not be reflected in distributional changes. They will usually require more precise monitoring of numbers in the population and this in turn requires quantification of effort and some correction for factors that influence detectability.

Fixed stations have a number of advantages over line transects: the data collected are easier to standardize; they are generally cheaper to undertake and so can be made at greater frequencies; and there are no additional complications of movement of the observer that can affect sightability. A major disadvantage is that the area of coverage is limited, generally to marine areas immediately adjacent to land, although fixed stations such as oil and gas platforms may be available in offshore areas. On the other hand, acoustic devices, particularly Porpoise Detectors (PODs) and PopUps, are especially useful for providing information on usage of a particular area, and can do so on a regular or continuous basis at relatively low cost (Tregenza, 1998; Gillespie & Chappell, 1998; Clark & Charif, 1998; Swift *et al*., 2002). At the same time, autonomous units can be deployed to record both vocal activity and related environmental parameters such as current strength, sea temperature and salinity.

Regular land-based watching for defined periods of time has frequently been used to identify coastal areas important for particular species and to determine variation in numbers both seasonally and over the longer term. Thus standardized watches at 50 sites around mainland Shetland at a similar time over four summers indicated that porpoises mainly occurred on the east coast with concentrations at particular locations (Evans, Weir & Nice, 1997). Temporal variation in abundance indices has been described for several species and areas (see for example Evans, 1992). Similarly, acoustic monitoring using PODs has highlighted shifts in the use by porpoises at different coastal sites in Pembrokeshire, with increased activity in autumn at certain locations at the same time as that of bottlenose dolphins *Tursiops truncatus* in the area (Fig. 3).

Watches should be standardized as much as possible. This may be difficult when it is not known which factors are influencing the presence of cetaceans in the observation area. For

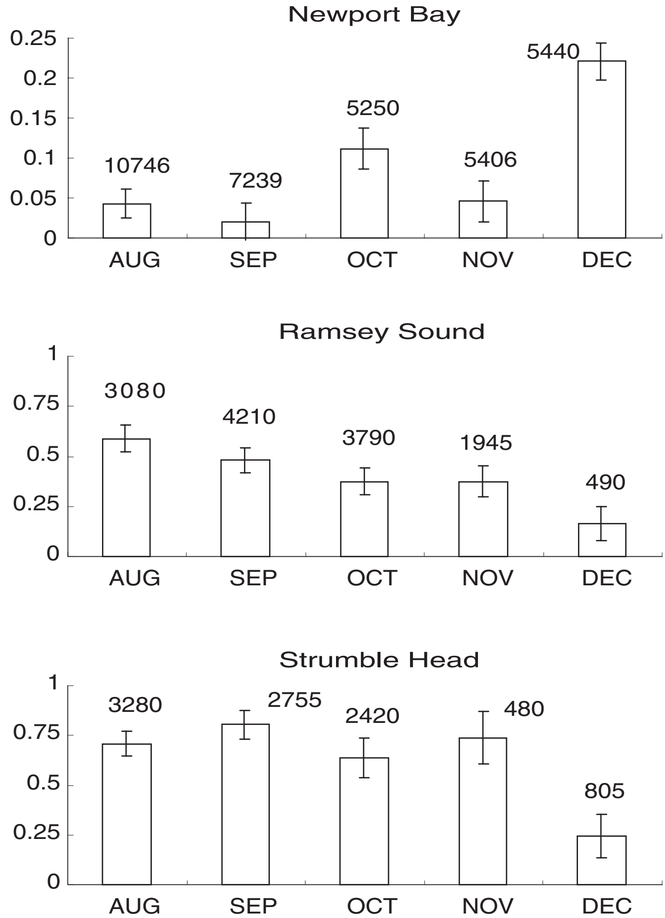

Fig. 3. Monthly trends in acoustic monitoring indices for harbour porpoises *Phocoena phocoena* off Pembrokeshire (Newport Bay, Ramsey Sound and Strumble Head) using PODs (from Pierpoint, Baines & Earl, 1999). The proportion of minutes with acoustic porpoise detections is shown for Newport Bay (±95% CI). The mean proportion of standardised visual scan samples with porpoises is shown for the other two sites (±1SE). Monitoring effort is given in minutes.

example, watches might be repeated at a particular time of day, but the presence of a species be determined by the state of the tide which obviously varies its timing from day to day. The relationship between the presence or particular activity of a species and tidal state may vary from location to location depending upon the nature of the currents. A more precise measurement would be of current strength and direction rather than tidal state alone (since at least some species make use of the inflow of water to an area to replenish food resources). If the opportunity arises to examine closely these relationships at a particular site, then that is all to the good. However, in practice, in most cases such problems can be alleviated if there are sufficient watches to perform statistical analyses of the effects of different variables.

A most important variable affecting the detectability of cetaceans is the weather. Sea state and wind (which obviously affects sea state) especially determine the number of sightings, whilst for whales in particular, glare may be important also. Surveys should generally not be made in sea states above Beaufort scale 2, in other words as soon as white caps appear upon waves, since this markedly reduces the detectability of surfacing cetaceans. This applies particularly to a species such as the harbour porpoise that is relatively inconspicuous and often solitary. Since sea states are generally higher in offshore areas and in winter, sightings rates in such areas and in the winter months are likely to be biased underestimates. An illustration of this can be provided from a study conducted in the Shetland Islands where

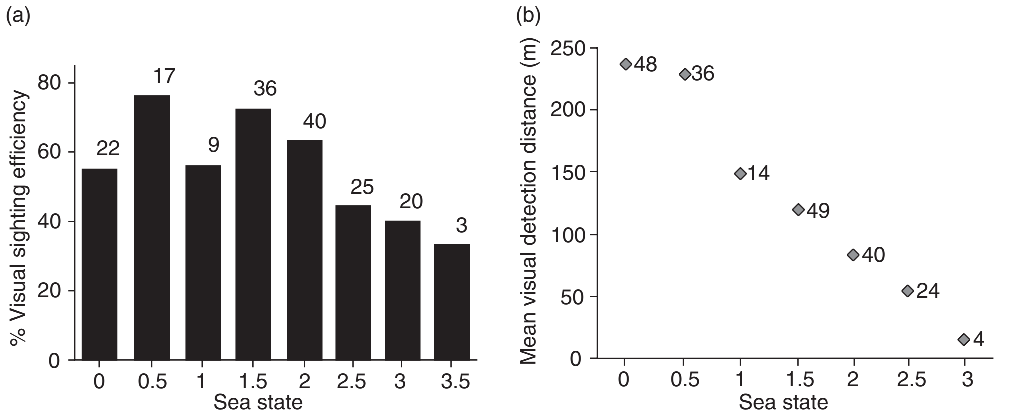

Fig. 4. Effects of sea state upon porpoise sightability from a vessel (from Evans & Chappell, 1994): (a) percentage of 5-minute periods when porpoises detected acoustically were also seen; (b) mean visual detection distances for different sea states.

porpoises were surveyed both visually and acoustically from a vessel, and a comparison made of the sightings frequency and detection distances at different sea states with acoustic registrations (using a towed hydrophone and porpoise click detector) (Fig. 4). As soon as sea state rose above zero (flat calm), there was a reduction in the sightings of porpoises (this relationship is likely to be platform specific and so it would be unwise to extrapolate to other platforms, particularly from land-based cliff-top observation points to small vessels).

Beaufort scale is more useful as a measure than wind speed alone because it takes account of conditions elsewhere which may affect the amount of swell on the sea. Analyses of watches made during various sea states allow some assessment of their effects, and enable one to calculate appropriate correction factors. This can be applied by taking the ratio of the mean sightings rate at a particular sea state to the mean of sightings rate at sea state 0 (c_n), using the following equation:

$$c_n = sr_n / sr_0 \qquad \text{(Equation 1)}$$

where sr_n is the mean sightings rate at sea state = n, and sr_0 is the mean sightings rate at sea state = 0.

With respect to the effects of sea state, acoustic methods for monitoring have advantages over visual ones because cetacean vocalizations tend to be independent (or, at least, more independent) of sea conditions. This is particularly useful with a species such as the harbour porpoise that vocalizes a lot (by high frequency echolocation clicks).

Other variables that may affect sightings rates include the number of observers present, the speed of the observation platform (which can vary from 5 to 6 knots in sailing vessels to 30 knots or more in fast ferries and speed boats), the eye height of the observer, and the observer's experience and ability to spot animals.

For visual monitoring, choosing the duration and frequency of observation bouts is a difficult procedure since it requires a balance between the maximization of sightings and what can practically be achieved with available resources. If day-to-day variation is greater than variation within a day, then it will be best to go out little and often. Watches over extended periods will determine the length of time necessary within an observation bout before a plateau in sightings is reached. However, this is likely to vary not only between areas but also seasonally, so such initial tests will need to involve several replications.

When using fixed stations to monitor status changes, it is important to keep in mind that one is monitoring the occurrence of animals in a particular restricted area and not the population at large. Even then, small changes in distribution can dramatically affect what is seen, although this problem is reduced by having a number of sites distributed over the area (easier for coastal locations than offshore waters). If broader geographical coverage is required, it may be preferential to combine watches from fixed stations with offshore line transects.

For many purposes, a population size estimate may not be necessary; instead, an abundance index applied to a prescribed area may be quite sufficient so long as numbers are calculated per unit effort, variables that might affect detectability of animals are measured, and observation conditions are standardized as much as possible.

Platforms of opportunity

Many groups in Europe conducting surveys of cetaceans use platforms of opportunity – ferries, oceanographic or fisheries research vessels, oil exploration guard vessels, whale-watching boats, etc., which cross areas of sea often on a routine basis as part of their work. They do so primarily to minimize costs. Some specialist vessels that may also carry additional specialized equipment for oceanographic/hydrographic monitoring can be particularly valuable platforms for surveys to understand factors affecting distribution and abundance. Indices of abundance are obtained either in terms of time spent in observation or distance travelled, and analyses can be conducted which also take into account viewing conditions such as sea state. They are a cost-effective means of providing wide coverage over protracted periods. The major limitations are that there is rarely any control over the routes taken or the speed of the vessel and the vessel typically cannot divert from its track to confirm species identity or school size.

Two examples of the use of platform of opportunity data for examining spatio-temporal patterns of abundance are fin whales *Balaenoptera physalus* recorded from a ferry across the English Channel into the Bay of Biscay (Brereton, Williams & Williams, 2000), and minke whales *Balaenoptera acutorostrata* recorded from a whale-watching vessel in the Sea of Hebrides (Leaper *et al.*, 1997).

Different platforms can vary markedly in viewing area, platform height and speed, and this may affect sightings rates. This is not a problem if data are being compared over time within a platform or platform grouping, but may become so if comparisons are made between platform types. On a ferry, for example, observers may be forced to watch over an area of 90° to port or starboard, or 180° forward or 180° back. Where the observation conditions can be controlled, most surveys have been undertaken by watching an area of 180° forward, and not including (at least in the survey data set) any sightings made outside this range. On large ferries, where often this is not possible, independent observers can watch on port and starboard sides. If conditions are right, the collected data may be appropriate for line transect analysis to estimate absolute abundance (see below).

However, before embarking upon this, it is important to recognize and understand the reasons for and requirements of such an analysis. In particular, there is no point in trying to conduct a complicated survey to estimate absolute abundance if: (i) that is not why the work is being done, and (ii) it will not be possible to meet the data requirements for this analysis.

Even if an abundance survey is not the objective, accurate location of sightings is desirable for analyses of sightings rates in relation to bathymetry, substrate, and oceanographic features using GIS.

As noted previously, there are many advantages to using acoustic devices to detect vocalizing animals, particularly from platforms of opportunity. For most species, they are usually more efficient at detecting cetaceans; they are relatively independent of viewing conditions, generally less affected by weather, and can operate throughout 24 hours. If automated detection is employed (or recordings made and analysed by several operators), the data collected are more homogeneous, being less susceptible to variability in skills/experience between operators (Chappell, Leaper & Gordon, 1996; Gillespie & Chappell, 1998; Gordon & Tyack, 2002).

There are some disadvantages to the use of acoustics, however. First, they rely upon animals vocalizing, and if those are silent during particular activities or at certain seasons, they will not be detected. Thus, for example, in the eastern North Atlantic, large baleen whales are detected vocally mainly during mating and calving seasons (September–March) (Clark & Charif, 1998; Charif & Clark, 2000), but sightings surveys have shown them to be present in the region regularly in summer (Reid, Evans & Northridge, 2003). Second, the relationship between vocalization rates and absolute abundance remains unclear in most cases, although work is underway to try to achieve at least some form of semi-quantitative assessment. Finally, the vocalizations of some dolphin species can be difficult to distinguish from one another, although better discrimination techniques are being trialled by various groups. Other disadvantages in some situations include the costs of equipment and its deployment, equipment maintenance requirements, and the need to minimize and/or distinguish other sounds present in the marine environment (not least being engine noise from the survey vessel).

DEDICATED SURVEY PLATFORMS

If a platform can be dedicated to surveying for cetaceans, it becomes possible to select a sampling design that is representative, although there may still be financial and logistical constraints. Dedicated surveys usually require significant resources, which in turn may pose limitations on their spatial and temporal extent. Vessels or aircraft can be used as dedicated platforms; in some situations the latter may be more cost effective although associated oceanographic and acoustic data cannot be collected.

Examples of dedicated visual surveys abound. Many of these are conducted at low cost either by serving as training platforms (e.g. Rosen *et al*., 2000) or in conjunction with whale-watching operations (Boran, Evans & Rosen, 1999). Others may be conducted where the target is another marine taxon, such as seabirds (Northridge *et al*., 1995). Dedicated acoustic surveys are increasingly being undertaken, sometimes in conjunction with visual surveys. Small-scale efforts have taken place in Britain and Ireland, particularly in offshore waters along the Atlantic continental shelf edge (see for example Aguilar, Rogan & Gordon, 2002), but most efforts of this nature in Europe have focused upon the Mediterranean (see for example Gordon *et al*., 2000).

Most importantly, dedicated visual surveys allow the full application of line transect sampling methods.

Line transect sampling

The idea behind line transect sampling is to estimate the density of the target species in strips sampled by surveying along a series of transects, and to extrapolate this density to the entire survey area. The calculated number is therefore an estimate of abundance in a defined area at a particular time. If this is not the information required, then line transect sampling may not be the best method to use.

In line transect sampling, the distance to each detected animal is measured, and these distance measurements are used to estimate the effective width of the strip that has been searched. This is necessary because the probability of detecting an animal decreases the further away it is from the transect line. A special case is a strip transect in which it is assumed that all animals are detected out to a given distance from the survey platform. The only way to determine if this is a valid assumption is to measure distances. For this reason, distance measurements should always be collected. Animals occur in groups in many cetacean species so the target for detection in a line transect survey is often a school. Data on the number of animals in each school must also be collected.

The equation that relates density to the collected data is:

$$\hat{D} = \frac{n\bar{s}}{2\,esw\,L} \qquad \text{(Equation 2)}$$

where $\hat{D}$ is density (the hat means that it is an estimated quantity), n is the number of separate sightings of animals (or schools), $\bar{s}$ is mean school size, L is the total length of transect searched, and *esw* is the estimated effective strip half-width. The quantity 2 *esw* L is thus the area of the strip that has been searched. The effective strip half-width is estimated from the perpendicular distance data for all the detected animals. It is essentially the width at which the number of animals detected outside the strip equals the number of animals missed inside the strip, assuming that everything is seen at a perpendicular distance of zero. The assumption that every school is detected on the transect line itself is an important one, to which we shall return later. Texts describing line transect sampling in more statistical detail include Buckland *et al*. (2001) and Hiby & Hammond (1989).

Another assumption is that animals do not move prior to detection. Cetaceans, of course, do move. How important is this? Random movement with respect to the survey platform causes a positive bias in abundance estimates but this bias is small so long as the survey platform travels quickly relative to the animals. This will always be true for aerial surveys but care must be taken with shipboard surveys. A survey speed of 10 knots is typically taken as a minimum. Movement in response to the survey vessel can be more of a problem. Aerial surveys are again immune to this but it is not uncommon for cetaceans to be attracted to survey ships or to avoid them. The obvious solution is to search sufficiently far ahead of the vessel that animals do not respond before they are detected, as is done in surveys for oceanic dolphins in the eastern tropical Pacific (Wade & Gerrodette, 1992; Barlow, Gerrodette & Forcada, 2001). Alternatively, there are methods for accounting for responsive movement (Buckland & Turnock, 1992; Borchers *et al*., 1998; Palka & Hammond, 2001).

Other practical assumptions are that animals are correctly identified to species and that, if detections are of schools of animals, school size can be measured or estimated accurately. Care needs to be taken to minimize the chance of violating these assumptions.

The basic concept is easy to understand but to conduct an effective line transect survey for cetaceans requires consideration of a number of important questions. Is the available survey platform appropriate? Can the area of interest be surveyed representatively? Can the necessary data be adequately collected? Are important assumptions of the method likely to be violated and, if so, what can be done about this? Are there appropriately experienced and trained people available for collecting and analysing the data? The answers to parts of these questions will be entirely practical matters that depend on the particular surveys being planned. But there are some guiding principles that can help.

Survey design

The basic requirement for a line transect survey is that it provides representative coverage of the area for which an abundance estimate is desired. This is because, typically, estimated density in the sample strip is assumed to be representative of the whole area so that it can be extrapolated simply by multiplying sample density by survey area. A survey design giving representative coverage is one that gives each point in the area an equal probability of being sampled; this is necessary because animals are not distributed randomly in space. Common designs are sets of equally spaced parallel lines or a regular zigzag pattern, starting from a random point along one edge of the survey area. Of course, a particular realization of a survey design will cover only a limited amount of the area. But before the random starting point is chosen, a transect could pass through any point. This is why the random starting point is essential for the design.

There are some general practical points needing attention when designing a survey. Transects should as far as possible run perpendicular to any density gradient; for example, coastal surveys typically have transects that run more or less perpendicular to the shore line. It is good practice to avoid surveying during times of directed population movement, such as a migration. A survey progressing in the same direction as a migration may survey the same high (or low) density patches repeatedly; whilst progress against the movement may lead to patches of animals being missed. Surveying across the direction of migration is probably the best compromise. Such considerations depend on prior knowledge, which is not always available.

For some methods of analysis, coverage is not required to be equal, but the probability that each point in the survey area will be sampled must be known. To generate this kind of design requires a computer program; one is available as part of the DISTANCE 4 software (Thomas *et al.*, 2002; and see below). Such designs are especially useful for surveying irregularly shaped areas, for which ensuring equal coverage probability is problematic. It is also possible to estimate abundance from data that were not generated from a designed survey, by using spatial modelling (see below). This is an important development but does not obviate the need for a proper survey design in a dedicated survey.

Generally, which design is most appropriate will depend on the topography of the area, the amount of resources available, logistical limitation on the survey platform, prior knowledge of the area and the animals, the intended method of analysis, the desired precision of the abundance estimates, and so on. The final choice should depend on balancing all the above and on a generous application of common sense.

Data collection

The primary data required for estimating the abundance of cetaceans using line transect sampling are the distance searched along the transect, and the perpendicular distance to and size of each detected school of the target species. In shipboard surveys, perpendicular distance is usually calculated from radial sighting distance and angle. In aerial surveys, it can be measured directly using an inclinometer. Accurate radial distance and, especially, angle data are at the heart of a good estimate of density. Most shipboard surveys use angle boards to increase accuracy and avoid rounding angles to convenient values (which can cause particular problems in analysis). Wooden vessels can use digital compasses.

Measuring distance at sea is notoriously difficult; Gordon (2001) has described a photographic method to assist this, but this may not be equally applicable for all species or in all situations. If such direct methods cannot be used, it is important to compare estimated distances with actual ones measured directly by radar or GPS in distance estimation experiments. Whether such experiments are used for training and improvement, or to calculate correction factors, needs to be carefully considered. Laser range-finding binoculars may also be used as a training tool. Data on sighting distance and angle will allow density to be calculated using Equation 2. However, there are other important data that are usually desirable.

The probability of detection of a school of cetaceans from the survey ship or aircraft depends on many things including the ability and experience of the observers, the height and characteristics of the survey platform, the weather, school size, and the behaviour of the animals, and it is good practice to record this information either in association with a sighting or with the survey effort. Depending on the size of the survey, there may be sufficient data to investigate the effects of some or all these on detection probability. School size and sea state regularly turn up as important factors in analyses.

Standard line transect sampling assumes that all targets present on the transect line will be detected with certainty (see above). This is clearly not the case with cetaceans, with the result that estimates of abundance calculated in this way will be biased downwards by an unknown amount. The accepted way to take this into account on shipboard surveys is to use data collected from two observation platforms on the same vessel. The basic idea here is that a school seen by one of the platforms acts as a 'trial' for the other platform which may or may not also see it. This allows estimation of the proportion of detected schools. There are a number of methodological variations in how this can be done including two-way independence in which both platforms set up 'trials' for the other (Palka, 1995), and one-way independence in which one platform searches further ahead of the other one and tracks schools until they are seen (or missed) by the other platform (Buckland & Turnock, 1992; Borchers *et al.*, 1998). But all require two teams of observers. Clearly this is only possible on survey ships of a certain size.

Estimating the proportion missed on the transect line is more difficult with aerial surveys. Interested readers can refer to Forney, Barlow & Carretta (1995), Hiby & Lovell (1998), and Hiby (1999).

The importance of training to ensure correct data collection cannot be overemphasized. Line transect surveys are expensive. Proper training of observers, distance estimation experiments, and even an experimental survey in advance take up relatively few resources but can make the difference between success and failure.

Analysis

Fortunately for all semi-quantitative biologists, there exists the excellent analysis program DISTANCE 4 (Buckland *et al.*, 2001; Thomas *et al.*, 2002). As well as providing a framework for estimating detection functions and other calculations, DISTANCE 4 also designs surveys (see above) and has a comprehensive data management capability so that it can be used from beginning to end in generating estimates of abundance and their precision. This not only takes a lot of the boredom and pain out of data analysis but also enables all users to get the most from their data. In particular, because the best analysis is rarely a standard recipe, the program allows a lot of flexibility and the user must be aware of the implications of his/her analytical choices. It goes without saying that a certain minimum knowledge is required; no one should ever use a software package for analysis who does not understand the underlying principles. The creators of the program offer training courses that will allow even the most reluctant biologist to generate statistically robust survey designs and abundance estimates (http://www.creem.st-and.ac.uk/).

DISTANCE currently does not incorporate the facility to analyse two-platform data. Nor does it therefore deal with responsive movement. Recent papers describing methods for accomplishing this include Borchers (1999), Laake (1999) and Palka & Hammond (2001).

Example

In 1994, a large-scale line transect survey was conducted to obtain accurate and precise estimates of abundance of the harbour porpoise and other small cetaceans in the North Sea and adjacent waters (Hammond *et al*., 2002). The study, known as SCANS, utilized nine ships and two aircraft to cover an area of about 1 million km^2 in summer (Fig. 5a). New methods were developed for the collection and analysis of shipboard (Borchers *et al*., 1998) and aerial survey (Hiby & Lovell, 1998) data. These had to take into account the fact that harbour porpoises are not only difficult to detect, so that detection on the transect line was far from certain, but may also respond to survey ships before they were detected. Each survey ship therefore used two observation platforms and in the aerial survey blocks, two aircraft flew in tandem so that the new methods could be applied. Porpoises were found distributed over most of the survey area (Fig. 5b) and the final estimate of 341 000 ($CV = 0.14$) harbour porpoises in the survey area is precise and should not be biased as a result of missed animals on the transect line or responsive movement. Such large-scale surveys are expensive and take a lot of organization. They should not be undertaken lightly or frequently.

MARK-RECAPTURE USING PHOTO-IDENTIFICATION DATA

The basic idea behind mark-recapture methods is that, initially, a sample of individuals is captured, marked and released (n_1). On a subsequent occasion, a second sample of individuals is captured (n_2) of which a number are already marked (m_2). The proportion of individuals that are marked in the second sample can be equated with the proportion in the population at large (N).

$$\frac{m_2}{n_2} = \frac{n_1}{N} \qquad \text{(Equation 3)}$$

Because the numbers of animals captured and marked each time is known, this allows population size to be estimated.

$$\hat{N} = \frac{n_1 n_2}{m_2} \qquad \text{(Equation 4)}$$

This simple two-sample estimator is known as the Petersen estimator or Lincoln index. The same idea applies when samples of animals are captured, marked and released on multiple occasions; this will be elaborated below.

Mark-recapture methods were initially developed, and have mostly been used, for studies in which individual animals are physically captured and marked in some way (painting, branding, mutilating, tagging), released and then physically recaptured. Indeed, the initial use of these methods with cetaceans was to mark large whales with so-called Discovery tags (metal bolts about 30 cm long) by firing them into the blubber and recovering them when the animal was butchered after being harpooned in harvesting operations. More recently, mark-recapture methods have been used with individual recognition data on cetaceans obtained via photo-identification to provide information on movements and population parameters (Hammond, 1986, 1990a). The required data to estimate population size are representative sets of good quality photographs of the well-marked parts of individual animals for two or more sampling occasions. Each of these elements is covered in more detail below.

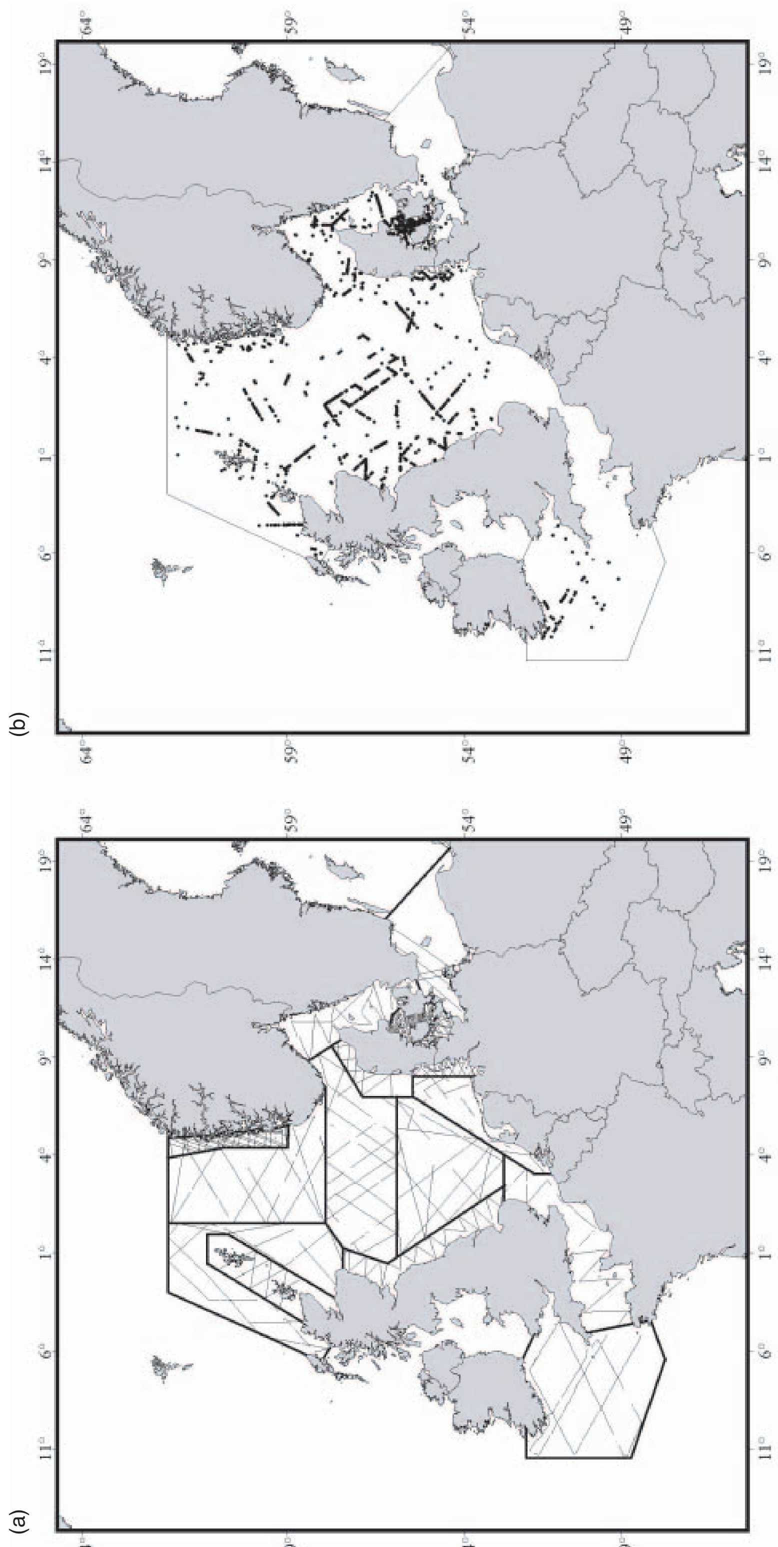

Fig. 5. (a) SCANS Survey tracks and (b) Harbour porpoise *Phocoena phocoena* sightings (from Hammond *et al.*, 2002).

There are some advantages of photo-identification over traditional sources of mark-recapture data, the most obvious being that the animal does not have to be physically captured or marked. Another is that marks cannot be lost as can tags (although there is the potential for some natural markings to change). On the negative side, natural markings are more difficult, and take more time, to recognize than, for example, a number on a brand or a tag, and not all species have adequate markings.

As with all statistical models, a number of fundamental assumptions are made about the data. One of these is that a marked animal will always be recognized if it is seen again; this is often translated into marks not being lost and not changing. In fact marks can change as long as this does not affect recognition. Another assumption is that samples of individuals must be representative of the population being estimated. This is to ensure that the proportion marked in a sample is a valid estimate of the proportion in the population. If marked animals did not mix fully with the rest of the population between sampling occasions, they violate this assumption. Mark-recapture models also assume that marking an animal does not affect its probability of recapture. With photo-identification this should be assured.

The final and most important assumption is that every animal in the population should have the same probability of being captured within any one sampling occasion. If this assumption is violated, this is known as heterogeneity of capture probabilities, and the estimate of population size will be biased downwards. If the reader needs convincing of this, imagine a population of 1000 animals in which 500 have a probability of capture of 0.5 and 500 have a probability of 0.1. Use the Petersen estimator (Equation 4) to calculate estimates of population size for the two groups separately (as if they could be distinguished), i.e. $n_1 = 500 \times 0.5$, $n_2 = 500 \times 0.5$, $m_2 = n_2 \times 0.5$ for the first group, etc. Now do the same calculation but combine data for the two groups for n_1, n_2 and m_2 as if they could not be distinguished. It is the combination of different proportions marked in the two groups that causes the bias. In the extreme, if some animals have zero probability of capture they will simply not be included in the population estimate at all. In a real study, ensuring equal probability of capture for all individuals will never be possible. Analysis can address this in some cases but the aim should be to minimize the problem in the field (see below).

Survey design

Mark-recapture methods estimate the number of animals in a population of individuals that mix together. Obtaining a representative sample of data therefore means sampling individuals representatively. Note the difference between this and the situation for line transect sampling, which estimates the number of animals in an area and for which a representative sample of the area should be obtained. Capturing as many animals as possible, i.e. making average capture probability as high as possible, is a good way to get close to representative samples and minimize the problem of heterogeneity of capture probabilities (Hammond, 1986). The point here is that if you have captured a large proportion of the animals, it does not matter how difficult they were to capture – you have got them anyway. How well this can be achieved is a matter of the size and extent of the population and the amount of resources available.

Data collection

The basis for a good photo-identification study is good photographs, good enough so that animals that are considered marked will be recognized with certainty if seen again later. This partly depends on how well marked animals in the population are and how much variability there is in these markings. Well-known examples of good markings for photo-identification

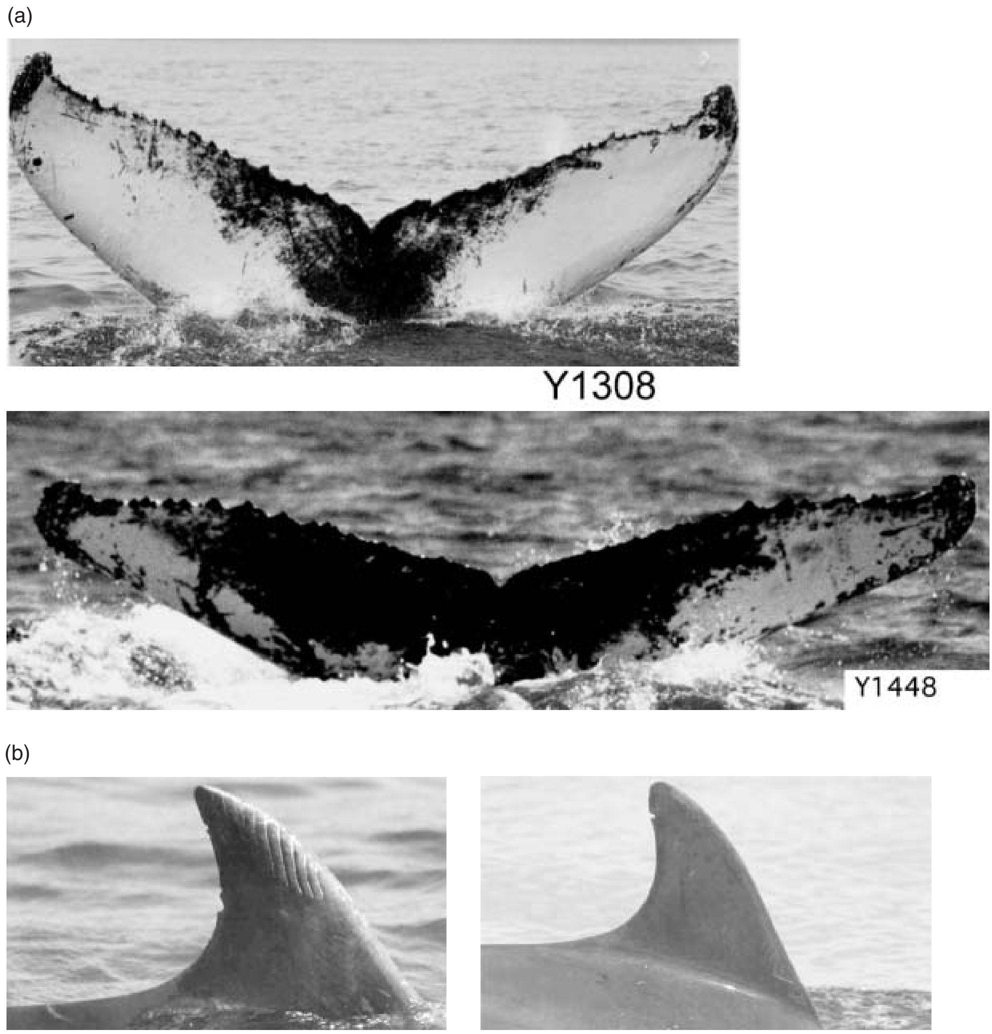

Fig. 6. (a) Humpback whale *Megaptera novaeangliae* tail flukes and (b) bottlenose dolphin *Tursiops truncatus* fins used in Photo-ID studies. Humpback individuals have unique patterns on the undersides of their tail flukes whilst bottlenose dolphins frequently have nicks along the trailing edge of their dorsal fins. (Photos of humpback tail flukes from the YoNAH database – Smith *et al*., 1999; and of bottlenose dolphins from the University of Aberdeen & University of St. Andrews east coast of Scotland database, and are copyright to those institutions).

are the ventral surface of humpback whale tail flukes and the nicks and notches in the dorsal fins of bottlenose dolphins (Fig. 6).

Not all animals in the population may bear distinguishing markings. For example, bottlenose dolphins acquire nicks as they get older so younger animals are typically not well marked. Data on the proportion of well-marked animals in each school encountered can be used to estimate the proportion of identifiable animals in the population (Wilson, Hammond & Thompson, 1999).

Good photographs come from good photographers. But even the most experienced photo-identification practitioner will not always take top quality photographs. In most studies, photographs are graded according to quality, and only the best are used in estimating population size. Including poor quality photographs may lead to animals not being recognized on recapture; the effects of this are described below.

Analysis

Photo-analysis precedes statistical analysis in mark-recapture studies using photo-identification. New photographs should be graded for quality and then matched to the existing database (catalogue) of marked animals. Matches are recaptures. It is good practice to be conservative in matching to eliminate the possibility of false positives. However, this may lead to some missed matches – false negatives. Missing matches leads to overestimation of population size. Stevick *et al*. (2001) used genetic identity (Palsbøll *et al*., 1997) as a control in a double-marking experiment to estimate the rate of missed photographic matches in humpback whales and developed a method to use this information to correct population estimates. The false negative error rate was found to increase markedly with poorer photo quality.

Mark-recapture methods require at least two sampling occasions. In this simplest case the Petersen estimator (Equation 4), or a variant of it, is the only choice. It is a closed population model because it assumes that population size is closed to births, deaths, immigration and emigration, i.e. it does not change over the period of study. This will rarely be the case but if either deaths or births can be assumed to be negligible, the estimate will be unbiased at the time of the first or second sample, respectively.

If the study has multiple sampling occasions, a time series of estimates can be obtained and there is more flexibility in analysis; open or closed population models can be used (Hammond, 1986). If the sampling occasions are close together in time, so that population size can be assumed not to change, closed population models can be used. One implementation (program CAPTURE – Otis *et al*., 1978; Rexstadt & Burnham, 1991) has models that can take account of heterogeneity of capture probabilities. If animals are believed to emigrate temporarily from the study area, there are methods for taking this into account in analysis (Whitehead, 1990).

Examples

The resident population of bottlenose dolphins along the coast of eastern Scotland (centred in the Moray Firth) has been the focus of a photo-identification study for many years (Wilson, Thompson & Hammond, 1997) and abundance has been estimated for the early years of the study (Wilson, Hammond & Thompson, 1999). Apart from factors already mentioned above, a number of important practical issues had to be addressed. Dorsal fin markings can appear different from each side, so separate but linked databases were maintained and separate estimates made for left- and right-side photographs. Types of identifying marks were found to last different lengths of time so a set was identified that was known to last at least 1 year. Data from each survey (= sampling occasion) for the period May – September for each of 3 years were analysed using a closed multisample model (using the program CAPTURE). Despite attempts to minimize the problem, heterogeneity of capture probabilities was evident in the data, so this option in program CAPTURE was implemented. The best abundance estimate of 129 ($CV = 0.15$) was an average of left and right side estimates for the year with most representative coverage. This study is a good illustration of some of the practicalities involved in estimating cetacean population size from photo-identification data using mark-recapture models.

A recent study in Cardigan Bay, West Wales used both line transect techniques and mark-recapture analysis of photo-ID data alongside one another, to estimate the abundance of bottlenose dolphins inhabiting the candidate Special Area of Conservation (Evans *et al.*, 2002). During May – September 2001, the former method gave an estimate of 135 bottlenose dolphins (95% CI 85–214) whilst mark-recapture analysis indicated that 215 animals (95% CI 179–290) actually inhabited the area. As noted above, the two are not measuring the same quantity. The former is an estimate of the average number of dolphins in the area during the study period. The latter is an estimate of the number of animals using the area during the study period and thus includes animals spending part of their time outside the study area.

MEASURING POPULATION CHANGE

At its most simplistic, measuring population change involves comparing two or more estimates of abundance made at different times. A series of estimates might warrant the fitting of a curve and the estimation of a rate of change. It is important that abundance estimates used in a trend analysis are relevant and comparable. For example, estimates for relatively small areas may not be good indicators of the size of a population that occupies a much larger area, especially if it is highly mobile, as cetacean populations tend to be. Changes in ranging patterns caused by variations in environmental conditions could have a large impact on abundance estimates made in such small areas and therefore on estimates of trend. Using generalized additive models (GAMs), Forney (2000) found that environmental variables were only partly successful in reducing variability in sighting rates and trends over time for short-beaked common dolphins *Delphinus delphis* and Dall's porpoise *Phocoenoides dalli*. Other studies where the size of the area in which abundance has been estimated may influence the validity of an estimated trend in population size include Hammond (1990b) and Chaloupka, Osmond & Kaufman (1999) for humpback whales *Megaptera novaeangliae*, and Wilson *et al.* (in review) for bottlenose dolphins.

Another important consideration in terms of the usefulness of the result is the length of the series and the precision of the abundance estimates, and whether any estimated population change is statistically significant at a given probability. A power analysis will indicate the ability of the available (or planned) data to detect a trend of given magnitude (Gerrodette, 1987). For example, if analysis shows that an estimated trend is not statistically significant, it is important to know the power of the test to show a significant trend. It would be a mistake to infer that lack of a significant trend implied no change, when the power of the test was low, as it will be for short time series and/or variable abundance estimates.

Power analysis can also be used to assess the length of a time series of abundance estimates, or the CV of those estimates, necessary to detect a trend of given magnitude. An approximate way of making these calculations is given by Gerrodette (1987):

$$r^2n^3 \geq 12CV^2(z_{\alpha/2} + z_{\beta})^2 \qquad \text{(Equation 5)}$$

where r is the annual rate of population change, n is the number of abundance estimates, CV^2 is the squared coefficient of variation of estimated abundance, $\alpha/2$ is the one-tailed probability of making a Type I error (i.e. accepting a trend when one does not really exist), β is the probability of making a Type II error (i.e. not accepting a trend when one does exist), and z is the standard normal variate. The salient point is that it is very important to use power analysis as an aid in interpreting the results of a trend analysis.

Wilson *et al.* (1999) used power analysis to assess the likely time scale and frequency of surveys necessary to detect given rates of population change, given the variability in the data, for bottlenose dolphins in the Moray Firth. Thompson *et al.* (2000) combined a similar analysis with a population viability analysis to argue for a precautionary approach to the conservation of this population. Turnock & Mizroch (2002) have also explored the effect of survey frequency on ability to detect trends in abundance.

For larger, more diverse, data sets, more complex methods of trend analysis are possible. Forney (1999) used an analysis of covariance model to test for a trend in abundance of harbour porpoises off central California, while accounting for the effects of sea state, cloud cover and area. A rate of decline of 5.9% per year was estimated, which was not significant at the 10% probability level, but a power analysis showed that the test had low power to detect trends of less than 10% per year.

Bravington, Northridge & Reid (1999) used data from ships of opportunity to investigate trends in relative abundance of harbour porpoises over space and time in the North Sea. There are a number of issues to address in such analyses, including the difficulty of combining diverse data sources, and the need for robust statistical methods capable of taking account of particular attributes of the data. Details are beyond the scope of this review.

MODELLING PATTERNS OF ABUNDANCE

Recent developments in statistical modelling allow abundance to be modelled not only as a function of perpendicular distance but also as a function of physical and environmental covariates, such as latitude, longitude, distance from land or ice, depth, bottom topography, sea surface temperature, etc. These methods have the potential to generate better abundance estimates through the incorporation of covariates when estimating density (Thomas *et al*., 2002). Perhaps more importantly though, they allow sample density to be extrapolated to un-sampled parts of the survey area using the estimated relationship between density and the physical and environmental covariates, so long as that information exists for the un-sampled areas. This allows a density surface to be created over the entire survey region. Abundance can be estimated by numerically integrating under the density surface for the whole survey region or any defined area within it.

Hedley, Buckland & Borchers (1999) describe two methods for estimating density as a function of physical and environmental covariates: one based on dividing the cruise track into small segments and modelling the number of schools in each segment; the other based on modelling the intervals between sightings. The methods were applied to data on minke whales in the Southern Hemisphere, and GAMs were used to estimate abundance over defined areas. Similarly, Marques & Buckland (in press) used GAMs to model encounter rates and mean school size of dolphins in the eastern tropical Pacific as a function of spatially and temporally referenced covariates and to estimate abundance.

But the density surface itself may provide valuable insight into areas that are important for a given species, perhaps aiding the process of developing Special Areas of Conservation under the EU Habitats Directive, or other areas where protection measures might be implemented. A feature of these methods is that they do not require a random survey design. They are thus particularly useful for the analysis of data from ships of opportunity which, if analysed using conventional methods, could otherwise lead to biased estimates. Note, however, that good coverage of the region of interest is still needed to obtain robust estimates of abundance using these methods.

CONCLUSIONS AND FUTURE DIRECTIONS

It would be both impractical and unwise to suggest that one methodological approach be used over all others. Each has its advantages and disadvantages (Table 2), and the approaches may frequently complement one another in providing a more complete picture of the status and distribution of a particular cetacean species. Before embarking upon a monitoring

Table 2. Summary of advantages and potential disadvantages of different approaches to cetacean monitoring

Category	Advantages	Potential disadvantages
Survey techniques		
Visual	*For estimation of absolute abundance*	
	Data collection and analysis methods that take potential problems into account are well established	Need to take account of animals missed on the transect line and any responsive movement
		Labour intensive and expensive
		Limited temporal coverage
		Need sufficient data to estimate detection function
Visual	*For estimation of relative abundance*	
	Not labour intensive and relatively cheap	Need to account for sighting efficiency varying with distance from vessel, observer abilities, group size, sea conditions, platform type
	Wide spatial and temporal coverage possible	Estimation of group size
	Minimum equipment requirements	Responsive movement of animals
	Suitable for platforms of opportunity	For platforms of opportunity – little or no control over survey design
Acoustic	Not labour intensive	Relies upon animals being vocal
	Less affected by sea conditions 24-hour coverage possible	Methods to relate sounds to abundance of animals are not well developed
	Easier to standardize and automate data collection	Requires specialist data collection equipment
	Suitable for platforms of opportunity	Ideally requires quiet vessels
		For platforms of opportunity – little or no control over survey design
Photo-ID	Not labour intensive and relatively cheap	Only applicable for species with long-lasting identifiable natural marks
	Abundance estimation through mark-recapture methods	Natural marks must be unique, recognizable and not change
	Additional information on life history (birth and death rates, movements)	Definition of population being estimated not always clear
		Heterogeneity of capture probability
Survey platforms		
Headland/ installation	Non-intrusive	Limited to small detection area
	Usually inexpensive	Information that requires close proximity to animals is hard to collect
	Not labour intensive	
Vessel	Ocean going vessels can cover wide areas over long periods	Large vessels are expensive and may be labour intensive to operate
	Ancillary information (environmental and biological) can be collected	Small vessels are limited to coastal areas
Aircraft	Can cover large areas quickly	Collection of ancillary information limited
	Can make efficient use of windows of good weather	Logistical limitations
	Not labour intensive	Expensive to charter but little flying time may be required

programme, it is prudent to determine precisely what information can be gained and what limitations exist, and then conduct a cost-benefit analysis of the various options available. The type of platform, level of sophistication of survey, and detection method should be considered in each case, and the most appropriate ones identified.

Large-scale SCANS-type line transect surveys are designed to estimate absolute abundance over a wide area and are therefore too expensive to be conducted more frequently than, say, every 10 years. At this frequency they are not able to give information on short-term changes in population size, and neither do they provide information on fine scale distribution. Other data sets are needed for this (see below). It is most appropriate to conduct a SCANS-type survey in summer when the weather is better but a survey in the month of January (or perhaps even June or August) may show a very different species distribution to that in July. This might not be important if the entire range of the population was being surveyed, but that will rarely be feasible. An example of the conservation management implications can be illustrated with the short-beaked common dolphin. If this species undergoes seasonal movements onto the European continental shelf, then a July population estimate for that region may be very different from one at another time of year. If the species experiences a significant fisheries by-catch during a different season to that from which the population estimate was derived, it may be difficult to determine what proportion of the population is being removed by fisheries activities.

Line transect surveys could probably be conducted much more cheaply using smaller vessels, but then one must consider whether the larger amounts of data then available would be more than compromised by any logistical restrictions on data collection (smaller vessels generally mean lower platforms, slower speeds, and restrictions on the extent to which the vessel can operate for long periods away from port). In some cases, the use of aircraft may be the most cost-effective means of survey. Those situations need to be considered on a species-by-species (or even local population) basis as well as seasonally.

Given that fisheries by-catch is one of the most important management issues facing at least some cetacean species (e.g. harbour porpoise) in Europe, large-scale line transect surveys of the region clearly have a major role to play. However, for proper interpretation, there is a need for additional information: the structure and geographical limits of the population, seasonal changes in distribution, and some understanding of fine scale distribution.

For a number of conservation management applications, it may not be necessary to have absolute abundance estimates. After all, for most other animal taxa, relative abundance indices are mainly used to measure population trends. Using GIS, spatial modelling and novel statistical frameworks (Bravington *et al*., 1999; Durban, 2002), there is potential for using such data for examining spatio-temporal distribution patterns and for trend analysis. In the former context, it would be helpful if we could be more predictive of the biotic and hydrographic factors primarily influencing cetacean distribution by use of habitat models. This could inform our survey design protocols, and might allow more refined extrapolation from sample surveys. There may also be scope to calibrate relative abundance estimates with absolute abundance estimates.

In a number of situations, acoustic surveys may be an appropriate means of monitoring population trends. Before this can be used with the same confidence as visual surveys, further consideration of a number of issues is required. These include: relationships between vocalization rate and density (which may not be linear), the feasibility of regular detection, species discrimination, and the costs of different types of equipment and their deployment. In some cases, direct calibration with sightings data may be possible. Acoustic monitoring needs to be considered in greater depth for ways in which it might be developed alongside more traditional survey methods.

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