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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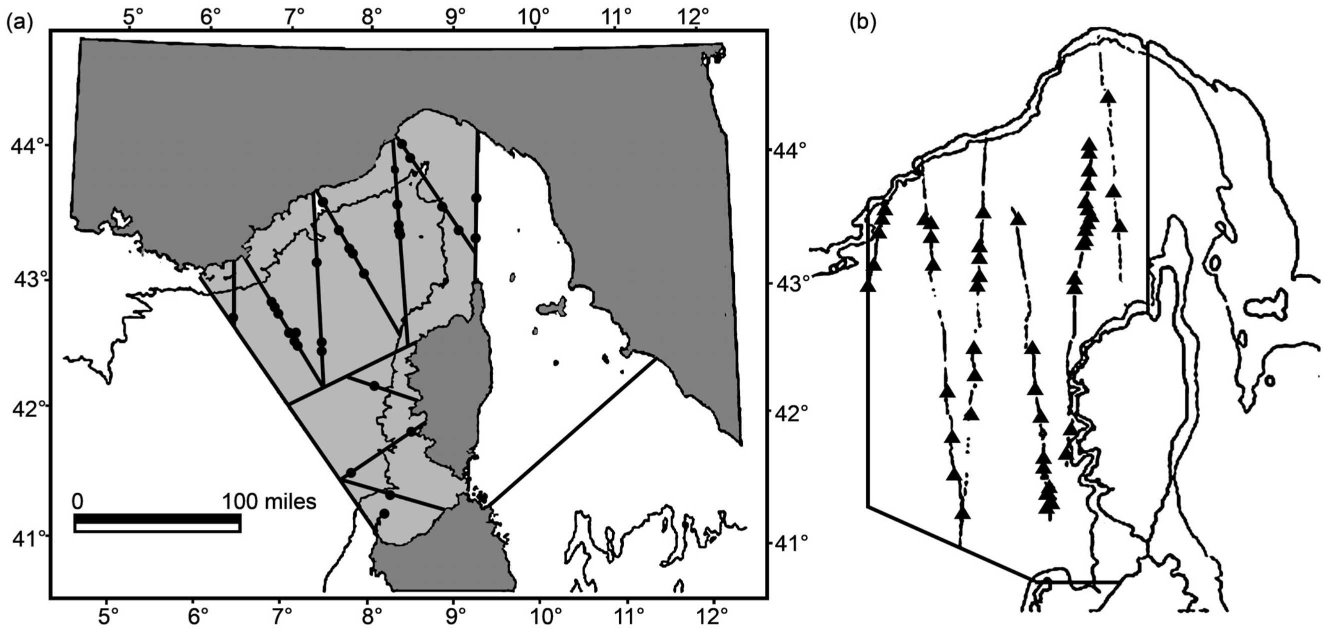

Fig. 1. (a) The study area and the distribution of the cetacean encounters with the tracks lines. (b) The area investigated in 1992 with the transects and the striped dolphin encounters (from Forcada *et al.,* 1995 modified).

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

[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 (χ^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.

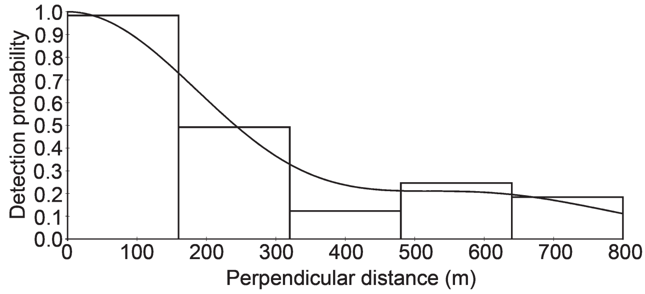

Fig. 3. Detection function with a right truncation at 800m.

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.

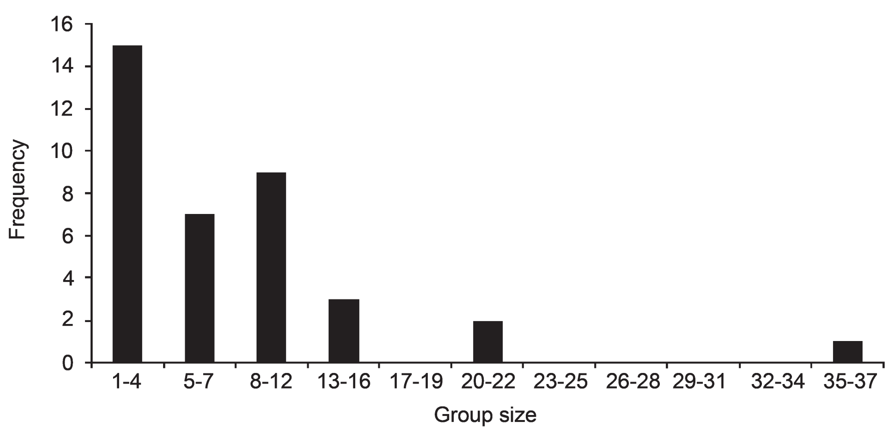

Fig. 2. Frequency distribution of all the striped dolphin groups.

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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