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PII: S0967-0645(17)30141-8
DOI: http://dx.doi.org/10.1016/j.dsr2.2017.04.018
Reference: DSRII4239

To appear in: *Deep-Sea Research Part II*

Received date: 22 January 2016
Revised date: 11 April 2017
Accepted date: 24 April 2017

Cite this article as: Simone Panigada, Giancarlo Lauriano, Greg Donovan, Nin Pierantonio, Ana Cañadas, José Antonio Vázquez and Louise Burt, Estimatin cetacean density and abundance in the Central AND WESTERN Mediterranea Sea through aerial surveys: implications for MANAGEMENT, *Deep-Se Research Part II,* http://dx.doi.org/10.1016/j.dsr2.2017.04.018

ESTIMATING CETACEAN DENSITY AND ABUNDANCE IN THE CENTRAL AND WESTERN MEDITERRANEAN SEA THROUGH AERIAL SURVEYS: IMPLICATIONS FOR MANAGEMENT

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Abstract

Systematic, effective monitoring of animal population parameters underpins successful conservation strategy and wildlife management, but it is often neglected in many regions, including much of the Mediterranean Sea. Nonetheless, a series of systematic multispecies aerial surveys was carried out in the seas around Italy to gather important baseline information on cetacean occurrence, distribution and abundance. The monitored areas included the Pelagos Sanctuary, the Tyrrhenian Sea, portions of the Seas of Corsica and Sardinia, the Ionian Seas as well as the Gulf of Taranto. Overall, approximately 48,000 km were flown in either spring, summer and winter between 2009-2014, covering an area of 444,621 km^2. The most commonly observed species were the striped dolphin and the fin whale, with 975 and 83 recorded sightings, respectively. Other sighted cetacean species were the common bottlenose dolphin, the Risso's dolphin, the sperm whale, the pilot whale and the Cuvier's beaked whale. Uncorrected model- and design-based estimates of density and abundance for striped dolphins and fin whales were produced, resulting in a best estimate (model-based) of around 95,000 striped dolphins (CV=11.6%; 95% CI=92,900–120,300) occurring in the Pelagos Sanctuary, Central Tyrrhenian and Western Seas of Corsica and Sardinia combined area in summer 2010. Estimates were also obtained for each individual study region and year. An initial attempt to estimate perception bias for striped dolphins is also provided. The preferred summer 2010 uncorrected best estimate (design-based) for the same areas for fin whales was around 665 (CV=33.1%; 95% CI=350–1,260). Estimates are also provided for the individual study regions and years. The results represent baseline data to develop efficient, long-term, systematic monitoring programmes, essential to evaluate trends, as required by a number of national and international frameworks, and stress the need to ensure that surveys are undertaken regularly and at a sufficient spatial scale. The management implications of the results are discussed also in light of a possible decline of fin whales abundance over the period from the mid-1990s to the present. Further work to understand changes in distribution and to allow for improved spatial models is emphasized.

1. Introduction

Population parameters such as abundance and density are essential to provide science-based advice on conservation and management issues, both in terms of determining priorities for action and evaluating the success or otherwise of those actions. Such information is also necessary to abide to national and international regulations. Within the European Union (EU), the monitoring of such parameters for large vertebrates is a legal obligation according to EU statutory instruments (Hovestadt and Nowicki, 2008); the EU Habitats Directive (Council Directive 92/43/EEC) requires the monitoring of the favourable conservation status (FCS) of the species of Community interest and their habitats, and to report of this status every 6 years (Hammond et al., 2013, 2002). Similarly, the more recent EU Marine Strategy Framework Directive (MSFD) requests “a description of the population dynamics, natural and actual range and status of species of marine mammals in Europe’s waters” and to act in order to achieve a Good Environmental Status (GES) by 2020 across Europe’s marine regions.

The process of collecting information on a species abundance and density within a given area and across years is an inherently complex task. In the case of cetaceans, it is often hampered by a lack of good information on population structure, the wide seasonal distribution and range of the species, their ability to travel long distances within and between seasons based, amongst the others, on the availability of resources and the potential sources of human disturbance. A small-scale approach is often not practical to fully understand the processes in place, identify conservation priorities and to implement conservation measures and evaluate their effectiveness. This is especially true when considering species whose conservation status is particularly vulnerable or uncertain, as it is for most of the regular cetaceans occurring in the Mediterranean Sea (IUCN, 2012). Therefore, a collaborative effort at the regional, rather than local, scale is necessary.

In this context, the Parties of the Agreement for the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic Area (ACCOBAMS) have acknowledged the need to develop a synoptic survey of cetacean populations across the entire Region, known as the ACCOBAMS Survey Initiative (ASI) (available online at www.accobams.org), currently planned for the summer of 2018.

In accordance with the need to systematically monitor cetacean populations, between 2009 and 2014 the Italian Ministry of the Environment has sponsored a series of multispecies aerial surveys to monitor megavertebrates in the seas around Italy and to assess their density, abundance and distribution, as well as to identify potential critical habitats for the species of interest. Aerial surveys implementing line transect distance sampling methodology (Buckland, 2004; Buckland et al., 2001) were chosen over traditional vessel-based surveys due to their cost-effectiveness and their validity to provide robust estimates (e.g. Panigada et al., 2011).

Panigada et al., (2011), Lauriano et al., (2011; 2014) and Notarbartolo di Sciara et al. (2015) have reported on the seasonal and annual abundance, as well as density and distribution, for the fin whale (*Balaenoptera physalus*), the striped dolphin (*Stenella coeruleoalba*), the common bottlenose dolphin (*Tursiops truncatus*), the loggerhead turtle (*Caretta caretta*) and the giant devil ray (*Mobula mobular*) for a large portion of the Central and Western Mediterranean Basin, including the Pelagos Sanctuary for Mediterranean Marine Mammals (Notarbartolo di Sciara et al., 2008), the Central and Southern Tyrrhenian Seas and portions of the Seas of Corsica and Sardinia. Lauriano et al., (2017) further report on obtaining fishery-independent surface abundance and density estimates of swordfish (*Xiphias gladius*) from aerial surveys in the Central Mediterranean Sea. Previous large scale abundance estimate for the western Mediterranean Sea are only available for the fin whale and the striped dolphin (Forcada et al., 1996).

Understanding the conservation status of a given species requires regular monitoring and in the case of cetaceans it may take several surveys to reliably estimate population trends (e.g. Jewell et al., 2012; Taylor et al., 2007). An unusually high mortality of striped dolphins in the Central Tyrrhenian Sea in 2013 (Casalone et al., 2014; Profeta et al., 2015) prompted another series of aerial surveys in these waters to investigate possible trends in striped dolphin abundance in the area, as well as to provide further information on the current abundance and distribution of cetaceans and other species of conservation concern in a large sector of the Mediterranean Sea. The target area is not only of ecological importance but also greatly impacted by human activities, such as high levels of chemical pollution, marine traffic and plastic debris (Cózar et al., 2015; Fossi et al., 2004; Lauriano et al., 2014a; Marsili et al., 2001; Panigada et al., 2006; Panti et al., 2011).

This paper reports on the results of the aerial survey programme carried out (Table 1 and Figure 1) so far and discusses their conservation and management implication, as well as the potential benefits for the future ACCOBAMS synoptic survey.

2. Material and Methods

2.1. Study area

The surveys monitored a large portion of the Central and North-Western Mediterranean Sea, extending for about 444,600 km^2 and encompassing narrow continental shelves, slope waters and bathyal plains with depths over 2500 metres. A complex coastal morphology, bottom topography and both surface and deep water circulation, along with high volumes of potentially detrimental human activities, make it one of the most dynamic and complex areas of the entire Mediterranean.

Table 1 and Figure 1 synthesise the details about the study regions in terms of strata, surface area, total transect length surveyed and coverage.

Table 1. Details of each study region: surface area (*A*), season and %year surveyed, surveyed transect length (L) and percent coverage of the study area monitored by the survey (given by $100 * \left(\frac{2wL}{A}\right)$ where *w*, the half strip width, was assumed to be 0.8 km). Locations of the regions are shown in Fig. 1

Description	Region	*A* (km^2)	Season	Year	*L* (km)	Estimated % Coverage
Ionian Sea	E	97,326	Spring	2010	5,999	9.8
Central Tyrrhenian Sea	C	93,216	Summer	2010	6,111	10.5
				2013	6,141	10.5
				Total	12,252	21.0
Pelagos	A	88,266	Summer	2009	8,502	15.4
				2010	5,681	10.3
				Total	14,183	25.7
Sea of Sardinia	B	54,789	Summer	2010	3,437	10.0
South Tyrrhenian Sea	D	111,024	Winter	2010	5,723	8.2
				2014	6,388	9.2
				Total	12,111	17.5

TOTAL	**444,621**	**47,982**	**16.7**

2.2. Survey design and data collection

Survey design followed the standard line-transect distance sampling approach (Buckland, 2004; Buckland et al., 2001), while protocols for data collection followed those used in previous aerial surveys in the region (Lauriano et al., 2011; 2014; Panigada et al., 2011; Notarbartolo di Sciara et al., 2015).

The surveys took place between 2010 and 2014; in 2010 the Ionian Sea was monitored during the spring, the Sea of Sardinia, the Pelagos Sanctuary and the Central Tyrrhenian Sea were surveyed during the summer, whereas the Southern Tyrrhenian Sea was monitored during the winter of 2010 and again in winter 2014. Finally, in summer 2013, the survey in the Central Tyrrhenian Sea was repeated.

Considering that surveys were conducted in different periods across the entire study area, there is the potential for seasonal animal movements across adjacent areas to bias the final estimates (Skaug et al., 2004). Very limited knowledge exists on striped dolphins' movement in the Mediterranean region, with indications that daily average movements are of moderate amplitude (e.g. Gannier, 1999) while seasonal movements are currently hypothesized but not supported by comprehensive evidence (e.g. Cotté et al., 2010). For fin whales, long to short distance movements have been reported for the Mediterranean (e.g. Caruso et al., 2015; Castellote et al., 2008, 2012; Geijer et al., 2016) but knowledge on potential routes, daily and seasonal patterns, whilst scant has started to emerge recently (Cotté, 2009; Panigada et al., in review). Given the evident gap of information existing on the species dispersion and movement patterns as well as daily and seasonal ranges, it is difficult at this stage to include this knowledge in the analysis (e.g. Keeping and Pelletier, 2014). Furthermore, given that aerial survey allows the monitoring of large areas in relatively small time windows, with a much greater speed of movement than the target species, in the present work we have considered the effect of animal displacement to be negligible on the final estimates.

With the exception of the Ionian Sea, each study region was divided into strata based upon published existing knowledge of species occurrence and distribution. A set of systematic, parallel transect lines, with a random start point to ensure equal coverage probability, were generated within each stratum using the software Distance 6 (Thomas et al., 2010). Transects within each stratum were oriented perpendicular to the coast and the isobaths to account for possible cross-shelf density gradients (Thomas et al., 2007). Accordingly, 12 strata and 216 parallel transects equally spaced at 15 km and totalling 44,273 km were designed.

Data collection was carried out by using a twin-engine high-wing aircraft (Partenavia P-68). For the period 2010 to 2012, it was equipped with one set of bubble windows to allow direct observation of the trackline. For the 2013 and 2014 surveys the aircraft was equipped with two sets of bubble windows to consequently use two sets of observers in 'double-platform' mode (Borchers et al., 1998; Nichols et al., 2000) allowing for the estimation of 'perception bias' (see below). All observers used digital voice recorders and *Suunto* clinometers. The latter was used to measure the declination angle to each sighting; this, along with plane altitude was used to estimate the perpendicular distance of each sighting to the trackline (Laake et al., 1999). Flight altitude and ground speed were kept constant at 750 feet (229 m) and 100 knots (185 km/h), respectively.

For each sighting, the observers and a dedicated data logger collected sighting-related information, such as the species, group size and composition, the declination angle to the animal/group of animals when abeam, the general behaviour of animals at the surface and signs of any reaction to the approaching aircraft. Positional data (latitude and longitude) were recorded automatically using a GPS during the whole survey as well as for each sighting. Environmental and weather related information including sea state (Beaufort scale), water turbidity, glare, cloud coverage and a subjective measure of the sighting conditions (Gómez de Segura et al., 2006) were collected at the start and end of every transect, as well as at any time when the conditions changed. General conditions were also defined based on an assessment of overall sighting conditions by the observers (taking into account all factors including sea state, turbidity, precipitation, etc.). Information on other factors, including human activities, marine debris, and other large vertebrates, were also collected when appropriate and could be done so without interfering with the main objective of the survey.

All the surveys were conducted in passing mode (i.e. the aircraft did not deviate from the trackline when a sighting was made, e.g. Dawson et al., 2008) unless it was necessary to fly over very large groups of animals to obtain reliable estimates of school size or confirm species, or to obtain the same information for those animals or group of animals, regardless of the group size, far from the track-line. When this occurred (very rarely), search effort was resumed at exactly the same point on the trackline it had been suspended. Any additional sightings ('secondary' sightings) made after leaving the predetermined trackline, although recorded, were not considered for the abundance and density estimates.

2.3. Analytical methods

2.3.1. *Design-based estimation*

Survey effort and associated sightings recorded in acceptable conditions were included in the statistical analysis. When there was a sufficient number of sightings for a given species, its abundance and density were estimated via distance sampling methods (Buckland et al., 2001), using the dedicated software Distance v6.2 (Thomas et al., 2010). In conventional distance sampling (CDS), only perpendicular distance from the trackline to the animal is used as a covariate in the detection function (used to model probability of detecting the animals) and thus to estimate the effective strip half width, *esw* (or, more strictly, the effective half strip width, as it only refers to one side of the trackline). The inclusion of additional explanatory variables in the estimation of the detection function (multiple covariate distance sampling - MCDS - Marques and Buckland 2004) was also investigated. The additional explanatory covariates, treated as factors, were: sea state, an assessment of overall sighting conditions by the observers (taking into account all factors including sea state, turbidity, precipitation, etc.) and individual observer.

To estimate the expected group size, the size-bias regression method (i.e. a regression of the logarithm of recorded group size against detection probability) was used if the regression was significant at an alpha-level of 0.15 (Buckland et al., 2001). If it was not significant, the mean of the observed groups was used.

Sightings were pooled in different combinations to obtain different detection functions in order to obtain density estimates for study regions, years and seasons. There were sufficient sightings of striped dolphins such that five detection functions were fitted to sightings from:

(a) summer 2010 (regions A, B and C);

(b) Central Tyrrhenian Sea (C) in summer 2013;

(d) Southern Tyrrhenian Sea (D) in winter 2010;

(d) Southern Tyrrhenian Sea (D) in winter 2014, and

(e) Ionian Sea (E) in spring 2010;

For fin whales, two groupings were created:

(a) summer all years (A, B and C) and

(b) Pelagos Sanctuary (A) in summer.

The use of a pooled detection function in each case is justified by the level of consistency among the surveys in terms of observers, aircraft and data collection protocols. Estimates of encounter rates and expected group sizes were stratified by year, region and season where appropriate.

Exact perpendicular distances (i.e. not binned) were used in the detection functions. Where deemed necessary to avoid a long tail in the estimated detection function, perpendicular distances were right truncated, following the recommendations of Buckland et al. (2001). For the summer surveys of striped dolphins, a distance of 800m was deemed appropriate after visual inspection and looking at the diagnostics of several truncation options; no truncation was considered necessary for Ionian (spring) and South-Tyrrhenian (winter). In the case of fin whales, after inspection the only truncation distance (1100m) was applied to the Pelagos study region in summer.

The selection of the detection function model was based on the minimisation of the Akaike Information Criterion (AIC, Akaike, 1974), and examination of the qq-plot and the goodness of fit tests (chi-square, Kolmogorov-Smirnov and Cramer-von Mises). The default estimators in Distance (Thomas et al., 2010) were used to estimate variance.

2.3.2. *Model-based estimation*

A grid of cells covering the whole study area, characterized according to several available spatial and environmental variables (Cañadas and Hammond, 2008; 2006) was created. A total of 26,944 cells were generated with a resolution of 17 km^2. The covariates to be tested in the model were: (a) spatial (latitude and longitude), (b) fixed (depth, distance to coast, distance to the 200, 1000 and 2000m isobaths, slope, and aspect), and (c) dynamic (seasonal and annual averages of sea surface temperature, *SST*, and chlorophyll concentration, *CHL-a*). Depth was extracted from ETOPO (http://ngdc.noaa.gov/mgg/gdas/gd_designagrid.html), and its derivates were obtained from it using ArcGis. SST and CHL-a were obtained from SeaWiFS and MODIS-Aqua sensors and the SST of MODIS-Terra and MODIS-Aqua. SST and CHL-a have wide coverage and both are available synchronously every day (at the scale of the processes involved, i.e. within 12 h) at a medium resolution (geo-projected data at 4.6 km for MODIS and 9.2 km for SeaWiFS; Druon et al., 2012). Spatial covariates were obtained as the latitude and longitude of the centre point of each grid cell. Cell depth was calculated as the mean depth of all ETOPO data points within each cell. The depth derivates covariates were calculated with the Surface and the Euclidean Distance Tools in the Spatial Analyst Tools, for the resolution of the grid cells.

All on-effort transects were divided into segments of mean= 3.1 km (max= 5.9 km) with homogeneous effort conditions. It was assumed that there was little variability in physical and environmental features within each segment, as they were split to fit each in a grid cell. Therefore, each segment was associated with the values of the covariates of the specific cell in which it fell.

Using the count of groups in each segment as the response variable, the abundance of groups was modelled using a Generalized Additive Model (GAM) with a logarithmic link function, and a Poisson error distribution. In the model-based estimation, the *esw*, obtained from the detection function described previously, was used to correct for the effective area searched (length of the segment by two times –two sides of the trackline- the *esw*) and included as an offset in the models, and thus taking into account the probability of detection into the spatial models, according to the covariates selected in the detection function, if any, apart from distance. For details and equations please refer to Notarbartolo di Sciara et al. (2015).

Given the difference between observed and expected mean group sizes, group sizes were not modelled spatially, because these models use the original observed group sizes without considering the expected group sizes after size-bias regression. Therefore, the expected group size was used as mean group size stratified per each sub-area to avoid a size bias in the spatial models.

Models were fitted using the R package '*mgcv'* version 1.7-22 (Wood, 2006). The 'best' models were manually selected (through inclusion/dropping terms) using three diagnostic indicators: (a) the Generalised Cross Validation score (GCV, Wood 2000), (b) the percentage of deviance explained; and (c) the probability that each variable was included in the model by chance.

The estimated abundance of animals for each grid cell was calculated as the product between its predicted abundance of groups and stratified mean expected group size. The point estimate of total abundance was obtained by summing abundance estimates in all grid cells in a given period of time.

Finally, to obtain the coefficient of variation (CV) and percentile based 95% confidence interval (95% CI), 400 non-parametric bootstrap re-samples were applied to the modelling process, using day as the re-sampling unit; 400 replicates has been found to be sufficient in similar studies (e.g. Cañadas and Hammond, 2008; 2006). In each bootstrap replicate, the degree of smoothing of each model term was selected by the statistical package, thus incorporating some of the model selection uncertainty in the variance. For the final CV (global and stratified by block), the delta method (Oehlert, 1992; Seber, 1973) was used to combine the CVs from the detection function, from the model, and from the stratified expected, or mean, group size (depending on which one was used).

Datasets for the model-based estimations were organized in a different way given the spatial aspect of these models, but always using the corresponding *esw* for each dataset from their detection functions (Tables 2 and 3). For striped dolphins, five spatial models were created: (1) Ionian Sea in spring 2010; (2) all areas in summer 2010 (Western Sardinia, Pelagos Sanctuary and Central Tyrrhenian Sea); (3) Central Tyrrhenian Sea in summer 2013; (4) Southern Tyrrhenian Sea in winter 2010; (5) Southern Tyrrhenian Sea in winter 2014.

For fin whales, a single spatial model was created for summers 2009-2013 in Pelagos, West Sardinia and Central Tyrrhenian, due to small sample size.

2.3.3. Mark-recapture distance sampling

2.3.3.1. Estimating the probability of detection on the trackline

One of the fundamental assumptions of CDS and MCDS is that all objects on the trackline are detected, i.e. detection at zero perpendicular distance, known as $g(0)$, equals 1 (Buckland et al., 2001). However, this assumption will be violated when an animal, or a group of animals, are at the surface but the observer fails to detect and report them (*perception bias*) and when the animal or group of animals are under water and therefore cannot be detected by the observers (*availability bias*). The installation of two sets of bubble windows in the survey plane in 2013, allowed a mark-recapture distance sampling (MRDS) approach to be implemented in order to estimate perception bias (Laake and Borchers, 2004).

A new team of observers was able to sit behind the original team and search simultaneously and independently being aurally and visually separated. Therefore, animals detected by one observing team could also be independently detected by the other team (termed a duplicate). Modelling the proportion of duplicates (given detection by the other team) as a function of perpendicular distance, and other covariates, allows the probability of detection by each observing team and the probability of detection by at least one of the teams to be estimated (Laake and Borchers, 2004). Ignoring distance and other covariates reduces this model to the Petersen estimator and this reduced form is used for fin whales where sample size is small (see later).

Estimates of perception bias for the front team, obtained from the 2013 survey, in principle would allow results from earlier surveys to be corrected for perception bias. The same observers were used for all surveys; hence, while recognizing that the weather was different for each survey, applying correction factors from the 2013 surveys to earlier ones seems justified,. Availability bias can be substantial on aerial surveys due to the relative high speed of the plane when compared to that of surfacing or diving animals which can possibly remain undetected when the plane passes overhead. Availability bias for striped dolphin has been reported to be negligible or absent (Gómez de Segura et al., 2006). It is likely, therefore, that striped dolphins are more or less continuously available at the surface. For fin whales availability bias has been reported to be very small (Williams et al., 2006); however, we do not include an estimate of this bias here therefore the resulting abundance will be an estimate of available animals.

2.3.3.2. Duplicate identification

Several issues based on how duplicates are identified can arise, leading to potentially biased abundance and density estimates. These issues can be due to:

- differences in group sizes recorded by the two teams;
- differences in the time of the recorded detection and, consequently, differences in the estimation of the position of the animal or group of animals;
- differences in the estimation of the angle to the animal or group used to calculate the perpendicular distances.

During the MRDS analysis, bias could be introduced through the mis-identification of duplicates, therefore, only the duplicates identified with certainty were used for the analysis. After a literature review (e.g. Borchers et al., 2013, 2006, 1998a, 1998b; Buckland et al., 2010, 2007; Heide-Jørgensen et al., 2007; Laake et al., 2008, 2011; Laake and Borchers, 2004; Marques, 2004; Okamura et al., 2012, 2003; Skaug et al., 2004), the following three criteria were defined to identify duplicates:

- a maximum time difference between detections of 9.5 seconds (according to Lauriano et al., 2014),
- a difference between the declination angles within ± 5 degrees, and
- similar recorded group size.

A provisional duplicate was identified when at least two of the three criteria matched. Nonetheless, each detection was then carefully double-checked using field notes and other relevant information (e.g. photographs and audio recordings). If teams recorded different group sizes for a duplicate, the maximum value was used; experience has shown that the number of individuals in a group of cetaceans is more likely to be under-estimated than over-estimated. For large groups, when search effort was suspended to confirm school size, field estimates were verified using the photographs taken while circling over the group.

2.3.3.3. Estimation of detection probability

A mark-recapture distance sampling approach was used to estimate the probability of detection (Laake and Borchers , 2004). The observing teams acted independently and because the plane was moving so fast there was unlikely to have been responsive movement between detection by one team and the other team. Therefore, the form of the model chosen assumed point independence (detections between the two teams are assumed to be independent only at the point where perpendicular distance was zero i.e. on the trackline; Laake and Borchers, 2004). To fit an independent observer (IO) point independence model, two subsidiary models are required: a mark-recapture (MR) model and a distance sampling (DS) model. The DS model is fitted to all unique detections, assuming that the intercept at perpendicular distance zero was one and is similar to the CDS and MCDS models described previously. The MR detection function model allows the probability of detection by each team and the probability of detection by at least one of the teams to be estimated. From this model the probabilities of detection on the trackline (i.e. at distance zero) can be estimated In MRDS, the probability of detection on the trackline by at least one of the teams is used to adjust the intercept of the DS detection function to obtain an overall probability of detection corrected for perception bias. The probability of detection on the trackline by the front team is used here to adjust density and abundance estimates from earlier surveys to correct for perception bias.

The covariates considered for inclusion in the DS model were group size, glare severity (as a factor with 2 levels) and an assessment of survey conditions (with 2 factor levels). Group size was considered in two forms; as recorded group size and as a factor with four levels (1-5 animals, 6-15, 16-25 and >25). A minimum AIC and goodness of fit statistics were used to select the final model. Analyses were performed using Distance 7 Beta 1 (Thomas et al., 2010) and the *mrds* package version 2.1.12 (Laake et al., 2015).

For the MR detection function, logistic regression was used to model the probability of detection (given detection by the other team) with perpendicular distance and other covariates as possible explanatory variables. The covariates considered were group size, glare and survey conditions as for the DS model and also observer team. An interaction term between distance and observer was also investigated. Again AIC and goodness of fit statistics were used for model selection.

3. Results

3.1. Striped dolphins

Striped dolphins were sighted in all strata (Fig. 2), years and seasons. The sightings data (n=707 groups) and detections are summarised in Table 2a.

Table 2b provides all of the estimates for both design and model-based approaches. It is clear that the point estimates from both are very similar. Differences arise mainly in the confidence intervals. Both approaches are consistent and valid. In terms of the 'best' estimates, we have therefore chosen to use the estimates with the lower confidence intervals (highlighted in bold in the table). These are not corrected for availability or perception bias (see discussion under 4.1.3 below) and are therefore underestimates of true abundance.

With respect to the contiguous summer surveys in 2010, the highest densities were observed in the Pelagos Sanctuary (region A) with lower densities in regions B and C. However, as shown in Figs 4-6, there is considerable variation within the regions and the higher density areas are in the western part of the Pelagos Sanctuary and over into the northern part of region B. There does appear to be a band of lower densities between the eastern Pelagos and the northern Central Tyrrhenian regions in 2010.

3.1.1. Design-based estimates

For all datasets, a detection function with a hazard-rate key function and no covariates, apart from perpendicular distance, were selected according to AIC and other diagnostics. In all models, expected mean group size was estimated from a size bias regression since the regression was significant, at least in some sub-areas, for each model. The fitted detection functions are shown in Figure 3. Table 2b shows the estimated density and abundance for the different design-based season/year/region groupings for striped dolphins. The overall abundance estimate for striped dolphins in summer 2010, the year with the largest spatial coverage (regions A, B and C cover 236,271km^2) was 97,825 animals (CV=15%).

3.1.2. Model-based estimates

Table 2b shows the results in terms of density and abundance for the different spatial models run for striped dolphins.

The details of the resulting best models for abundance of groups for the seven datasets are shown in Table 3. All models retained depth as the main covariate, and in most cases latitude and longitude as an interaction. In all models over-dispersion with a Poisson error distribution ranged between 0.96 and 1.49, therefore the distributions chosen were considered appropriate and no over or under dispersion remained. Figs 4 to 9 show the predicted abundance of striped dolphins for the seven areas/periods considered. The overall abundance estimate for striped dolphins in summer in 2010 (Central Tyrrhenian, Pelagos and West Sardinia) was very similar to the design-based estimate: 95,013 animals but with a smaller CV (CV=11.6%). The deviance explained is low in all models, as is typical for "low density" and highly mobile species. This low density means that not every time a potentially good area (as defined by the significant covariates) is surveyed, animals are encountered. In addition, due to their high mobility, they may be found in "potentially poor areas" while transiting between two good areas. This means that animals are not always found in potentially good areas and they are also some times found in potentially poor areas. However, this is not in contradiction with obtaining good precision. Precision is telling us that after many resamplings, the results remain similar in each.

Table 2. Summary of the abundance estimates for available striped dolphins: a) Truncation distance (*w*; metres), number of detected groups (*n*), encounter rates of groups (ER, groups per km), expected group size, effective strip width (*esw; m*) and percentage coefficients of variation (CV) and b) design and model-based estimates of density (*D*; animals/km^2) and abundance of animals (*N*), and 95% confidence intervals of abundance. The regions are shown in Fig. 1.

a)

Season	**Year**	**Region**	***n***	**ER**		**Expected group size**		***esw***	
				Estimate	**%CV**	**Estimate**	**%CV**	**Estimate**	**%CV**
Summer	2010	A	132	0.0232	18.1	16.3	13.5	367	4.4
		B	59	0.0172	31.0	16.7	18.1		
		C	120	0.0196	13.2	12.4	14.1		
		Total	311	0.0204	11.3	14.1	8.5		
	2013	C	63	0.0103	22.1	19.6	16.4	376	3.2
Winter	2010	D	144	0.0252	16.0	7.8	9.2	275	6.6
	2014		91	0.0142	17.0	10.6	11.4	325	7.0
Spring	2010	E	98	0.0163	13.1	11.4	14.2	327	8.4

b)

Season	Year	Region	Design-based estimates					Model-based estimates					
			D	***N***	**%CV**	**95% CI of *N***		***D***	***N***	**%CV**	**95% CI of *N***		**Model**
Summer	2010	A	0.52	45598	23.0	29175	71264	**0.51**	**44557**	**15.8**	**37771**	**50702**	2
		B	0.39	21373	36.1	10365	44071	**0.39**	**21689**	**23.2**	**14312**	**28174**	
		C	0.33	30855	19.8	20974	45389	**0.31**	**28439**	**18.1**	**22479**	**33956**	
		Total	0.41	97825	15.1	72771	131500	0.40	95013	11.6	92893	120304	
	2013	C	**0.27**	**24861**	**28.7**	**14278**	**43288**	0.26	24339	36.4	15060	37272	3
Winter	2010	D	**0.45**	**37729**	**20.3**	**25382**	**56081**	0.30	32684	24.5	23134	48538	4
	2014		**0.23**	**25756**	**21.6**	**16898**	**39258**	0.25	27833	22.3	19147	36437	5
Spring	2010	E	**0.29**	**27813**	**21.0**	**18465**	**41893**	0.28	27214	29.4	19809	44087	1

Table 3. Summary of the spatial models for abundance of available striped dolphin groups: covariates and their respective estimated degrees of freedom (edf) and p-values, and the deviance explained by the model. '*' indicates an interaction term between the covariates.

Model	Region	Season	Year	Covariates	edf	p	Deviance explained
1	Ionian Sea	Spring	2010	Latitude*longitude Depth	27.6 4.4	<0.0001 <0.0001	13.8
2	All blocks	Summer	2010	Latitude*longitude Depth	17.0 8.6	<0.0001 <0.0001	11.2
3	C-Tyrrhenian	Summer	2013	Latitude*longitude Depth	17.0 5.0	<0.0001 <0.0001	22.4
4	S-Tyrrhenian	Winter	2010	Latitude*longitude Depth	27.1 4.4	<0.0001 <0.001	13.6
5	S-Tyrrhenian	Winter	2014	Latitude*longitude Depth	23.2 4.2	<0.0001 <0.0001	16.2

3.2. Fin whales

Fin whales were sighted (*n*=83 groups) only in regions A, B and C (Fig. 10) in summer; sightings information and detections are summarised in Table 4a. Data from the survey conducted in the Pelagos Sanctuary (region A) in summer 2009 (Panigada et al. 2011) were used to increase sample size for that block in order to estimate the detection function.

As shown in Table 4a, there were considerably fewer sightings of fin whales than for striped dolphins (Table 2a), with a total of only 66 sightings for all surveys. Fin whales were seen in the Central Tyrrhenian, Pelagos and West Sardinian regions. Insufficient sightings were made in the first of these for abundance estimates to be generated and even for the latter two, the number of sightings (less than 30 in each) is rather low. For this reason, the most robust estimate is for the total summer 2010 contiguous area, using a pooled detection function. Following the approach used for striped dolphins, we have chosen to use the estimates with the lowest CVs as our 'best' estimates, as highlighted in bold in Table 4b. No fin whale sightings were made during the spring (Ionian region) or winter surveys (South Tyrrhenian region).

3.2.1. Design-based estimates

The selected detection functions for both data groupings used a hazard-rate key function and no additional covariates were included (Figure 11). Table 4b shows the estimates of density and abundance for the two design-based models. The overall estimate of abundance in summer for all the areas was 665 animals (CV=33.1%).

3.2.2. Model-based estimates

The best model retained two covariates: depth (edf = 7.0) and mean summer sea surface temperature (edf = 4.8), both highly significant. The deviance explained was 21.8%. Table 4b shows the results in terms of density and abundance for the spatial model run for fin whales. Figure 12 shows the predicted abundance of fin whales. The point estimates are very similar to the design-based estimates, with an overall estimate of abundance of 663 animals but with a larger CV (CV=39.7%)

Table 4. Summary of the abundance estimates for available fin whales for summer: a) Truncation distance (*w*; metres), number of detected groups (*n*), encounter rates of groups (ER, groups per km), expected group size, effective strip width (*esw*) and coefficients of variation (CV) and b) design and model-based estimates of density (*D*; animals/km^2) and abundance of animals (*N*) and 95% confidence intervals of abundance. The regions are shown in Fig. 1.

a)

Region	**Year**	***n***	**ER**		**Expected group size**		***esw***	
			Estimate	**%CV**	**Estimate**	**%CV**	**Estimate**	**%CV**
A	2010	27	0.0048	25.9	1.3	8.0		
B	2010	26	0.0076	35.9	1.3	8.2		
	2010	6			1.5	33.3	750	25.1
C	2013	7			1.3	14.3		
	Total	13	0.0011	27.5	1.4	12.9		
Total		**66**	**0.0028**	**15.3**	**1.3**	**3.7**		

b)

Region	**Year**	**Design-based estimates**					**Model-based estimates**				
		D	***N***	**%CV**	**95% CI of *N***		***D***	***N***	**%CV**	**95% CI of *N***	
A	2010	**0.0037**	**330**	**33.9**	**172**	**633**					
B	2010	**0.0066**	**362**	**44.6**	**151**	**863**	0.0067	372	59.1	300	580
	2010										
C	2013										
	Total	**0.0010**	**91**	**41.1**	**42**	**199**	0.0008	71	38.9	40	110
Total		**0.0028**	**665**	**33.1**	**350**	**1,263**	0.0028	663	39.7	547	886

3.3. Estimates of g(0)

During the 2013 survey, 86 groups of striped dolphins were detected, of which 47 groups were duplicates. Note that this includes some groups which could not be included in the previous results (see Table 2) because perpendicular distance, group size etc. were missing but could be determined reliably from duplicate sighting information.

The longest perpendicular distance at which a group of striped dolphins was detected was 1,440m. To avoid a long tail in the detection function, 10 percent of sightings were truncated at a distance of 800m, consistent with the approach previously described to determine *esw*, and a reasonably good fit was obtained although the fit was poorer close to the trackline (Fig. 13a). For this reason a narrower truncation distance was also considered (300m) – this gave a better fit close to the trackline (Fig. 13b) but wider confidence intervals, largely due to the smaller sample size.

Model selection with a half-normal form of the DS detection function was performed for both truncation distances. With the 300 and 800m distances glare severity and survey condition were selected as additional covariates, respectively. The MR models for both truncation distances included terms for distance and recorded group size.

The estimates of detection probabilities and subsequent densities and abundance for the two truncation distances are provided in Tables 5 and 6. Given the uncertainty in the abundance estimates there is considerable overlap in the confidence intervals suggesting that these estimates are not substantially different from each other.

The number of sightings for fin whales was small (8 groups) and so we used the Petersen-type estimator to obtain a probability of detection for the front observing team (Table 5). This resulted in an estimate of 0.8 (%CV=22.4).

Table 5. Summary of detection probabilities from the MRDS analysis of the 2013 survey for the two truncation distances (meters) used: total number of detected groups (*n*), number of duplicates, estimated probability of detection on the trackline (i.e. at distance zero) by the front observing team; estimated probability of detection on the trackline by one, or both, observing teams; estimated probability of detection over all distances and teams; % CVs are provided in parentheses.

Parameter	**Striped dolphins**		**Fin whales**
	Truncation		
	300	**800**	**None**
N	56	78	8
Duplicates	37	44	4
Detection on the trackline by the front team	0.40 (37.2)	0.79 (7.8)	0.80 (22.4)
Detection on the trackline by at least one observing team	0.61 (29.5)	0.95 (2.9)	0.91 (10.4)
Overall probability of detection	0.46 (32.9)	0.41 (9.1)	0.43 (17.7)

Table 6. Summary of abundance estimates for striped dolphins for the 2013 survey corrected for perception bias for the two truncation distances: number of detected groups (*n*), encounter rates (ER, groups per km), expected group size, density (*D*; animals/km^2) and abundance of animals (*N*), coefficients of variation (CV; including uncertainty in the estimate of perception bias) and 95% confidence intervals (CI) for abundance.

Truncation	***n***	**ER**		**Expected group size**		***D***	***N***	**%CV**	**95% CI of *N***	
		Estimate	**%CV**	**Estimate**	**%CV**					
300	56	0.009	23.0	12.2	17.1	0.40	37050	32.1	19892	69009
800	78	0.012	21.7	17.8	12.4	0.33	31122	21.5	20261	47803

4. Discussion

This paper presents the results of a series of aerial surveys - based on line-transect distance sampling methodology - conducted between spring 2010 and winter 2014 in the waters around Italy. Data were collected for all large vertebrates seen but the numbers of sightings were only sufficient to estimate abundance for striped dolphins and fin whales. The resultant estimates represent important baseline data for future assessments of trends and to inform policy makers and stakeholders on how to manage human activities in order to minimize negative effects on Mediterranean cetaceans.

Comprehensive basin-wide estimates of density and abundance are lacking for all the species of cetaceans across the Mediterranean Region. Nonetheless, these parameters have been previously obtained for the striped dolphin and the fin whale over large portions of the Central and Western Mediterranean Basin, highlighting seasonal, annual and geographical patterns. Panigada et al. (2011) and Bauer et al. (2015) provide a synthesis of the available information on the species abundance, density and encounter rates in the Western portion of the Basin and present the most recent seasonal abundance and density estimates for the Pelagos Sanctuary, for both striped dolphins and fin whales uncorrected for perception and/or availability biases. Bauer et al. (2015) also provide estimates of density - corrected for the availability bias - for the same species in the Gulf of Lions. In this paper, we present a correction factor for the perception bias, obtained through a MRDS approach (Burt et al., 2014; Laake and Borchers, 2004) for the striped dolphin.

The overall higher density, and hence abundance, of both species observed in the North-Western portions of the surveyed areas, with values clearly decreasing during the winter months and towards the Southern and Eastern sectors, is consistent with previous studies on the ecology of these species, as well as, on the presence of suitable habitats (e.g. Azzellino et al., 2012; Gannier and Praca, 2007; Notarbartolo di Sciara et al., 2003, 1193; Panigada et al., 2008, 2005).

Differences in density and abundance, as well as in distribution, between successive surveys in the Central and South Tyrrhenian Sea were also observed and will be discussed.

4.1 Striped dolphins

These systematic surveys provide the first robust estimates for striped dolphins in the Tyrrhenian Sea and in the Ionian Sea and Gulf of Taranto. They also confirm that this species is the most abundant cetacean in the surveyed areas, as indeed it is in the whole Mediterranean (i.e. Aguilar, 2000; Aguilar and Raga, 1993; Forcada et al., 1995, 1994; Notarbartolo di Sciara et al., 1993; Reeves and Notarbartolo di Sciara, 2006). In terms of striped dolphins, it does appear that the present boundaries of the Sanctuary do not incorporate all of the high density striped dolphin areas (e.g. Cotté et al., 2010; Bauer et al., 2015; Lauriano et al., 2010; Panigada et al., 2011).

The density values estimated for striped dolphins in the Ionian Sea and the Gulf of Taranto are broadly consistent with those described for this species in other areas of the Mediterranean Sea, with the exception of the Pelagos Sanctuary, where the species occurs at higher densities (see Panigada et al., 2011). In the Ionian Sea, striped dolphins have been observed close to the coast, showing different habitat preferences from those exhibited in the Ligurian Sea and the Tyrrhenian Sea, where the species is observed mainly in the pelagic environment (Panigada et al., 2008).

Previous estimates of abundance and density for this species have been obtained for several other areas of the Mediterranean (see Appendix S1 in Panigada et al., 2011), with most of the research effort focused in the western sectors (Bauer et al., 2015; Forcada et al., 1994, 1995; Forcada and Hammond, 1998; Gannier, 1998; Gomez de Segura et al., 2006; Gannier, 2005; Laran et al., 2010;

Lauriano et al., 2010; Panigada et al., 2011). Fortuna et al. (2007) provided estimates of abundance relative to the summer months in 2002 and 2003 in the southern Tyrrhenian Sea. Our results are broadly consistent with previously published information (Azzellino et al., 2008; Cotté et al., 2010; Panigada et al., 2008) that the species is found primarily in deep pelagic waters and on the edge of the slope as depicted in Figures 4-9.

4.1.1 Changes over years

Table 2b shows that there are two regions for which data are available for the same season but different years: the Central Tyrrhenian Sea in summer (2010 and 2013) and the Southern Tyrrhenian Sea in winter (2010 and 2014).

With respect to the Central Tyrrhenian Sea, the abundance estimates are not significantly different, although the point estimates are lower in 2013 (24,900 *vs* 28,500). Examination of the spatial modelling results in Figs 6 and 8, however, does show considerable differences in distribution. In 2013, the highest density was in a narrow strip in the western part of the study area, off the continental shelf of eastern Sardinia. Densities decreased moving to the East of this area and the larger block that covers the Central and Eastern part of the study area had low densities throughout. By contrast, in 2010 the highest densities were along the northern edge of the study area and in deeper waters to the west of mainland Italy in the northern part of the study area. In broad summary, there was a sharp contraction of the range of higher density areas in 2013.

Interestingly, the mean group size in 2013 was somewhat larger (19.6, CV 16.4%) than in 2010 (12.4, CV 14.1%). In addition in 2010, the larger groups of striped dolphins are distributed all across the study area although with a slight tendency to be more common in the centre and west (Fig. 6), while in 2013, the larger schools mainly occurred in the western portion on the surveyed region (Fig. 8). A more general consideration of school size is given under 4.1.2 below.

In the Southern Tyrrhenian Sea in winter, again the estimates were not significantly different but the point estimates were rather lower in 2014 (25,800 vs 37,700). Examination of the spatial modelling predictions (Figs 7 and 9) again reveal a considerable change in distribution. In the latter period, there were only three main high density areas, a major one in the Southwest of the study area below eastern Sardinia, and two lesser ones to the north and south of the central portion of the study area. By contrast, in 2010 there were consistently higher densities across the northern part of the central and eastern sectors of the study area, particularly north of the Island of Sicily – an area where densities were particularly low in 2014. In broad summary, there was a contraction of the range of concentrations of striped dolphins into the southwest of the study area in 2014, compared to a more dispersed distribution in 2010. Mean school sizes were a little larger in 2014 (10.6, CV 11.4%) than in 2010 (7.8, CV 9.2%). As for the Central Tyrrhenian in summer, the contraction in range in this area also coincided with a change in the distribution of larger school sizes. In this case the broad distribution in 2010 contrasted with most of the larger schools being found to the central and west in 2014.

An unusually high mortality of cetaceans attributed to a morbillivirus epizootic, mostly striped dolphins (n=66, over 10 times the 25 yearly average), had been reported for these two regions in early 2013 (Casalone et al., 2014; Di Guardo et al., 2013; Di Guardo and Mazzariol, 2013). Cyclic DMV epidemics might be expected every 3–5 years that may severely impact the already endangered health and conservation status of Mediterranean striped dolphins (e.g. Panti et al., 2011). This emphasizes the need for regular monitoring and a better understanding of stock structure.

Our results reveal decreases of around 20% and 33% in the uncorrected point estimates for the Central and Southern Tyrrhenian Seas, respectively, but these are not statistically significant given the confidence intervals. Explaining the observed changes in distribution and habitat contraction and any potential conservation implications is not possible with only two seasons of data. As previously noted (e.g. Donovan, 2008), several years of data are required to determine the natural variability in distributions of highly mobile species, in addition to knowledge on the population structure, the temporal and spatial scales and patterns of movements and migrations. For both the study areas, the inter-annual and seasonal variations will be related to biological factors such as prey availability, which may well be related to many environmental, oceanographic and climatological conditions. It is clear for both regions that it will not be possible for trends in estimated abundance to be interpreted unless broader areas are surveyed simultaneously, better information on population structure is obtained and CVs are reduced. This is discussed further below.

4.1.2 School size

School size plays an important role in estimating abundance and in particular it is important to take into account the different detectability of larger schools (e.g. see section 2). However, aerial survey data on school size are also of interest from an ecological standpoint. A qualitative comparison of the school size data for the Central and Southern Tyrrhenian Seas across years revealed changes in school size and the distribution of large schools. In addition, a qualitative examination of the data in Table 2 suggests possible seasonal differences (e.g. winter school sizes are smaller than summer school sizes (c.f. Southern Tyrrhenian Sea in winter 2010 with summer surveys in all regions)) as well as regional differences within the same season (c.f. summer 2010 in the Central Tyrrhenian with Pelagos and West Sardinia).

Group size variability for the striped dolphin (and other social cetaceans) has been related to several factors, such as the physical environment, their diet, prey availability, and the life history (e.g. Gygax, 2002a). In particular, group size seems to be positively correlated with the openness of habitat for this species, with the size of striped dolphin groups increasing with more open habitat and deeper water, as found for other species of delphinids (e.g. Gygax 2002a, 2002b). Differences observed in the variation of the group size between the two study areas and study periods could be related to dynamic oceanographic and biological factors, that should be considered in future analysis if appropriate data are collected.

4.1.3 Perception bias

This paper provides the first attempt to estimate perception bias for aerial surveys of striped dolphins using a ‘double platform’ approach on a Partenavia P68. This is particularly important (in conjunction with obtaining estimates of availability bias) if absolute, rather than relative estimates of abundance, are to be obtained. The former are particularly important as part of any assessment of the effects of human activities such as bycatch. The experiment proved successful in showing that estimates can be obtained using the two-team approach tried. The MR model was fitted to obtain an estimate of detection by at least one observer on the trackline and the model truncated at 300m appears to be the best fit to the data close to the trackline, although the overall goodness of fit of the 800m truncated model is also reasonable, but poorer closer to the trackline. The confidence intervals are wider with the 300m truncation because some data are excluded.

In neither case the variable for observer was chosen, suggesting that there were no substantial differences between the observing teams and that application of a common correction factor to previous surveys is appropriate.

As shown in Table 6, corrected point estimates for data truncated at 800m are lower than those estimated by truncating the data at 300m (the uncorrected point estimate of around 24,500 increases to either around 37,000 for 300m or 31,000 for 800m). However, there is substantial overlap in the 95%

CI suggesting that these estimates are not significantly different. The lower detection on the trackline (for the 800m truncation) compared to further away may be due to observers sub-consciously searching a greater area and focusing further from the trackline, for example, to detect fin or sperm whales. As one would expect, the *esw* for fin whales is much larger than the ESWs for striped dolphins (see Table 2 and Table 4 of the paper). It is necessary to continue to collect double-platform data to investigate this issue further (e.g. by comparing estimates for regions with and without many large whale sightings).

In the meantime, it is clear that perception bias can result in uncorrected estimates being around 20-30% lower than the corrected estimates. In addition, data to obtain estimates of availability bias (e.g. through telemetry) are required.

4.2 Fin whales

Inspection of Figures 10 and 12 reveal a gradient in the occurrence from east to west within the northern Pelagos; highest predicted densities occur in the adjacent West Sardinia study region, primarily in deep offshore waters. Medium densities are found in the northwestern Central Tyrrhenian Sea along the southeastern border with the Pelagos Sanctuary. Fin whales were largely absent in a large northeast-southwest band of shelf waters in the Ligurian Sea, the east coast of Corsica and the island of Sardinia, as well as the southeastern part of the central Tyrrhenian Sea. This is broadly consistent with our recent understanding of fin whale distribution, including its occurrence well beyond the borders of the Pelagos Sanctuary (e.g. Arcangeli et al., 2013; Notarbartolo di Sciara et al., 2013).

Lower fin whale encounters in the Pelagos Sanctuary in the last few years, witnessed also by local whale watching operators, have been correlated with a decrease in the intensity of the spring phytoplankton bloom in the area since early 2000s, which may have altered food availability (Lauriano et al., 2010). This is in contrast with the cyclic bloom which regularly occurs in the Gulf of Lions i.e. to the west of the West Sardinia study region (B) (Finoia et al., 2007), where fin whales, cetaceans and other species of large vertebrates are hypothesized to concentrate in higher numbers as a consequences of more abundant prey.

Although the sample sizes are small (as are the ranges in school size from 1-4), there is a slight tendency for larger group sizes in the higher density areas (Figs 10 and 12). Forcada et al., (1996) suggested a correlation between size of groups and the patchiness of prey.

It should be noted that there have been no regular systematic studies of fin whale density and abundance in the Mediterranean Basin or in the Pelagos Sanctuary, even though this is part of the requirements of the Habitat and Marine Strategy Framework Directives of the European Union. A monitoring programme is among the priority actions in the Pelagos Sanctuary management plan, but given the information on distribution and density presented here, it is clear that from a management perspective, surveys must extend well beyond the Sanctuary boundaries; a small scale approach will clearly not provide a truthful picture of the status of Mediterranean fin whales. That being said, in terms of recent summer surveys in the Pelagos Sanctuary, there were insufficient data to obtain a reliable estimate in 2008 (Lauriano et al., 2010), an estimated abundance of 148 individuals (CV 27%) in 2009 (Panigada et al., 2011), and 330 (CV 34%) in the present study.

The range of density values for fin whales reported here (0.0028-0.0037 km^{-2}) partially overlap with the results recently reported by Bauer et al., (2015) in the Western Pelagos Sanctuary area and Gulf of Lions, and those available from previous surveys (Gannier 1997; Laran et al., 2010; Panigada et al., 2011). Of more concern is the fact that they are considerably lower than those reported by Forcada et al., (1995) for the Pelagos Sanctuary in 1992 (n= 901; CV= 21.77%). There are no earlier estimates of density or abundance to compare with the Central Tyrrhenian Sea estimates provided here.

In terms of abundance, estimates of fin whales in the Pelagos Sanctuary have decreased from nearly 1,100 individuals (CV 30%) (Forcada et al., 1995; Gannier, 2006) to our estimate of around 330 (CV 34%), over about two decades. Our estimate of 670 (CV 33%) of fin whales for the combined summer 2010 survey is closer to the estimates obtained by Forcada et al., (1995) for the Pelagos Sanctuary alone (n= 901; CV= 21.77%).

A changed distribution of fin whales in a wider area of the Central Mediterranean Sea, extending well beyond the Pelagos Sanctuary borders (e.g. Arcangeli et al., 2013; Notarbartolo di Sciara et al., 2013), potentially caused by a shift in food availability (Finoia et al., 2007), could explain the apparent decline in fin whales in the Sanctuary. However, the lack of previous knowledge on distribution and density at the Basin level make it impossible at this stage to determine the status (including trend) of the Mediterranean sub-population of fin whales and thus a true decline cannot be ruled out.

In this context, on a precautionary base, the actual and potential threats to fin whales in the Mediterranean Sea should also be re-evaluated in light of an actual decrease of fin whale numbers, as well as priority being given to a full-scale Mediterranean survey, as has been recommended frequently by ACCOBAMS.

4.3 General comparison between design-based and model-based estimates

In general, in the model-based approach the inclusion of environmental variables to predict abundance increases the precision of the estimate, and actual density heterogeneity along the trackline can be taken into consideration (Hedley et al., 1999; Forney 2000; Williams 2003). The abundance estimates generated in this study by the two methods are comparable for each species, supporting the assumption that model-based estimates are robust.

For striped dolphins, point estimates from both approaches are very similar, as are the CVs, although in some cases one approach yields better precision (Table 2b). Nonetheless, for the majority of estimates, the model-based approach yielded smaller CV, confirming that spatial distance sampling models are a good approach to estimate cetacean abundance (Hedley et al., 1999; Hedley & Buckland 2004). The same is true for fin whales (Table 4b), although the CVs are generally larger than for striped dolphins as a result of the smaller sample sizes.

Nonetheless, the relatively low levels of deviance explained by the variables considered (see Table 3) also reveal that further effort needs to be put into finding additional explanatory variables that are better proxies for the spatiotemporal patterns of density and abundance of the species concerned, as well as increasing the numbers and extent of surveys. This will improve sample sizes, increase the deviance explained, decrease the resultant CVs and improve the models' predictive abilities in terms of robust spatial mapping. This is particularly important in terms of development of mitigation measures to identified threats by providing reliable identification and delimitation of critical areas of species occurrence to be considered in management plans that take into account natural variability in abundance and distribution.

5. Conclusions

The information gathered during the systematic aerial survey monitoring scheme between 2009 and 2014 represent an important step towards obtaining essential baseline data to assess potential fluctuations at the population level over the short, medium and long term in the Mediterranean Sea. They also have confirmed the reliability of the aerial survey methodology to monitor cetacean populations and to obtain robust minimum abundance and density estimates (e.g. Panigada et al., 2011). The advantages of this approach - the possibility to cover large areas in relatively short time, maximising the effort in good weather conditions, and the high accuracy and precision in the collection of data - allow robust estimates with small coefficients of variation (CV) and confidence intervals (CI). Furthermore, alongside the scientific benefits, it is important to reiterate the fact that this approach is also very cost-efficient compared to other methods; in fact, given a similar spatial coverage and survey design, aerial surveys are significantly cheaper than ship-based surveys, allowing to survey the study area in a much shorter time, with less personnel, as well as to capitalise on very short time windows of good weather.

However, the results also support the view that to assess and evaluate possible population trends for a given species, and therefore to trigger adequate mitigation and conservation measures, knowledge on the population structure, as well as on the temporal and spatial scales and patterns of movements and migrations, is essential (Donovan et al., 2008). This knowledge can only be gained through a series of systematic and long-term monitoring programmes (including genetic studies) at the appropriate scale. This is the principle behind the ACCOBAMS Survey Initiative, which has for many years been deemed the highest priority research activity for the region and will be implemented in summer 2018. Without such studies, it will not be possible to determine whether changes in estimates of abundance for particular areas can be related to true population-level changes, as illustrated by the large-scale SCANS surveys that showed that despite major changes in distribution, the total abundance of harbor porpoises in the North Sea and adjacent waters had not declined (Hammond et al., 2002; 2013). Small scale surveys may have implied either a decline or an increase (or neither), without the ability to determine what actually the case was.

Results of the Italian programme so far have *inter alia* shown that the present boundaries of the Pelagos Sanctuary do not capture the complete primary summer range of a number of species including striped dolphins and fin whales, and thus surveys (and management actions) limited to that area alone cannot be considered adequate to ensure good conservation status.

Further collection of data from surveys such as those undertaken thus far, in addition to data collected at similar spatial and temporal scales on potential dynamic explanatory variables (including prey availability, changing oceanographic factors, etc.) should allow for more complex spatial and habitat models to better explain distribution and changes in distribution of striped dolphins and fin whales, reduce the CVs of estimates and therefore improve the ability to detect change, and to provide management advice on the most important areas to focus mitigation measures.

Finally, it should be noted that in addition to the clear conservation benefits of additional aerial surveys in the light of the shift in distribution described here, such work also fulfils requirements under a number of international agreements including the European Union (Habitats and Marine Strategy Framework Directives) and by the Protocol Concerning Specially Protected Areas and Biological Diversity Mediterranean (SPA/BD Protocol) in the framework of the Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean (Barcelona Convention).

Acknowledgments

The equipment for data collection was that used during the EU LIFE Nature programme, project LIFE04NAT/GB/000245 project ‘‘Small Cetaceans in the European Atlantic and North Sea (SCANS-II)’’, funded in 2006. Many thanks to Anita Gilles (Research and Technology Centre Westcoast, University of Kiel) for providing the data collection and analysis software. Roberto Gramolini kindly developed CR Tools, used for GIS analysis. We are grateful to Mach014 and to the pilots of the planes, Michele Albertario and Claudio Castagna, for their professionalism. Margherita Zanardelli, Leà David and Joan Gonzalvo participated to data collection as secondary observers. Jean-Noël Druon (Joint Research Centre of the European Commission) provided remote sensing data used for spatial modelling. Thanks for the contribution of the anonymous reviewers who improved the quality of the manuscript. This research was funded thanks to a series of grant from the Italian Ministry of the Environment.

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Fig. 1 – The study area subdivided into 5 sub-areas: (A) Pelagos Sanctuary; (B) Sea of Sardinia; (C) Central Tyrrhenian Sea; (D) Southern Tyrrhenian Sea and (E) Ionian Sea. The planned transects, spaced 15 km apart, for each sub-area are also represented.

Fig. 2 - Striped dolphins sightings in all sub-areas, scaled by group size.

Fig. 3 – Perpendicular distance distribution (histograms), and fitted detection functions (lines) for striped dolphins.

Fig. 4 - Predicted abundance of striped dolphins for the summer 2010 from spatial models.

Fig. 5 - Predicted abundance of striped dolphins for the spring 2010 from spatial models.

Fig. 6 - Predicted abundance of striped dolphins for the summer 2010 from spatial models.

Fig. 7 - Predicted abundance of striped dolphins for the winter 2010 from spatial models.

Fig. 8 - Predicted abundance of striped dolphins for the summer 2013 from spatial models.

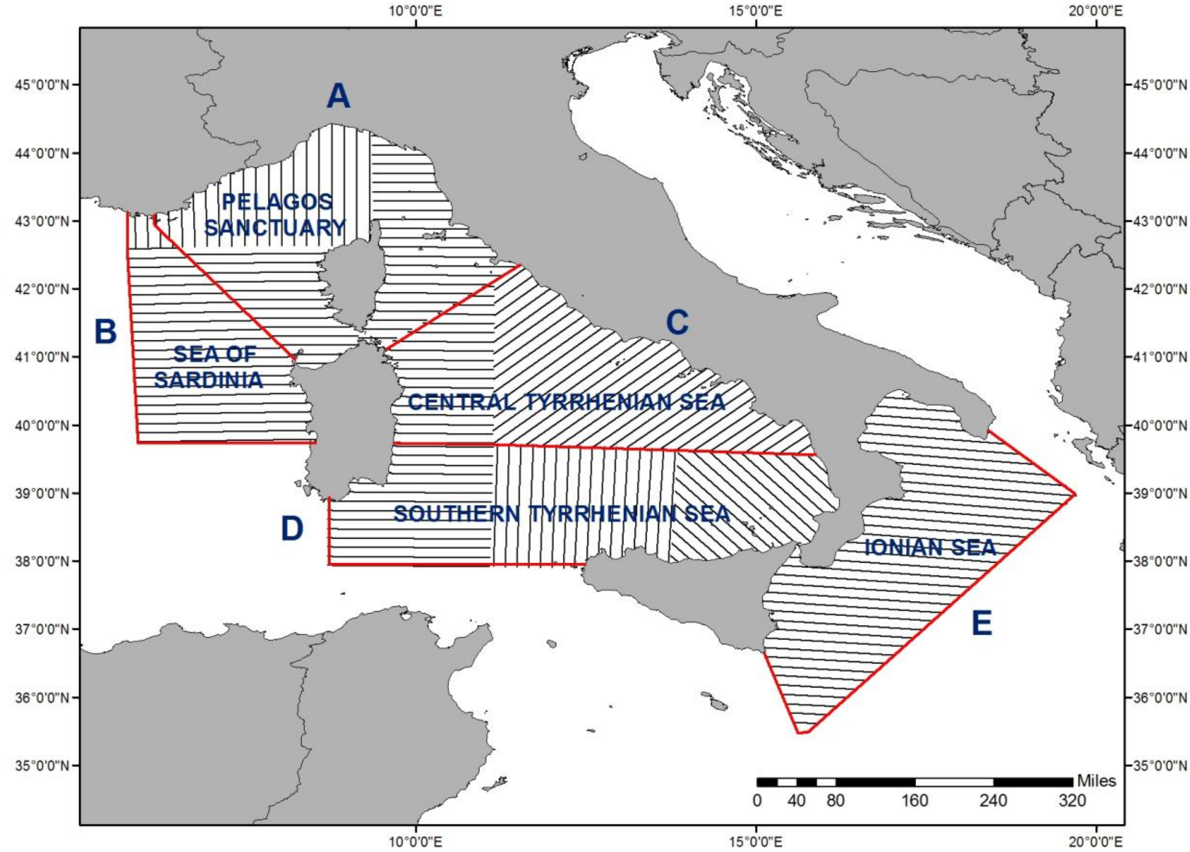
Fig. 9 - Predicted abundance of striped dolphins for the winter 2014 from spatial models.

Fig. 10 – Fin whales sightings in all sub-areas, scaled by group size.

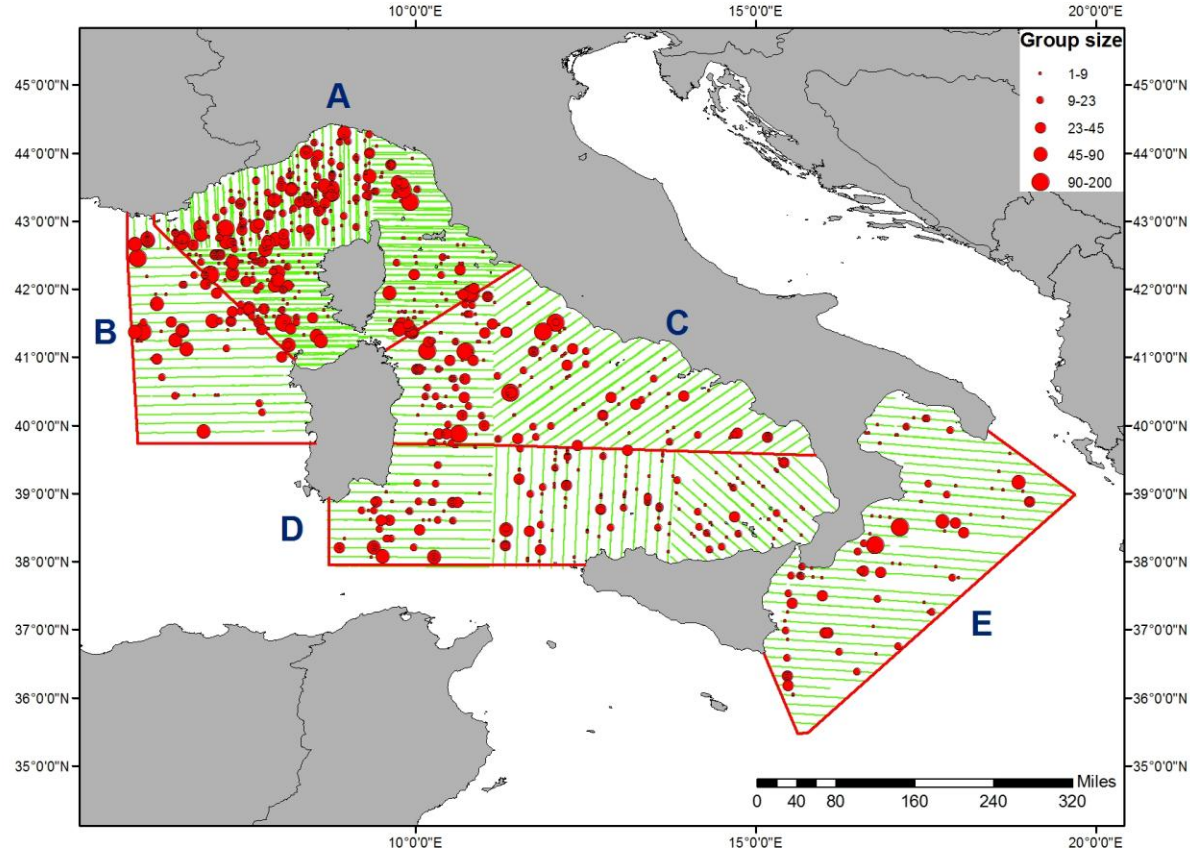
Fig. 11 – Perpendicular distance distribution (histograms), and fitted detection functions (lines) for fin whales.

Fig. 12 - Predicted abundance of fin whales for the summer 2010.

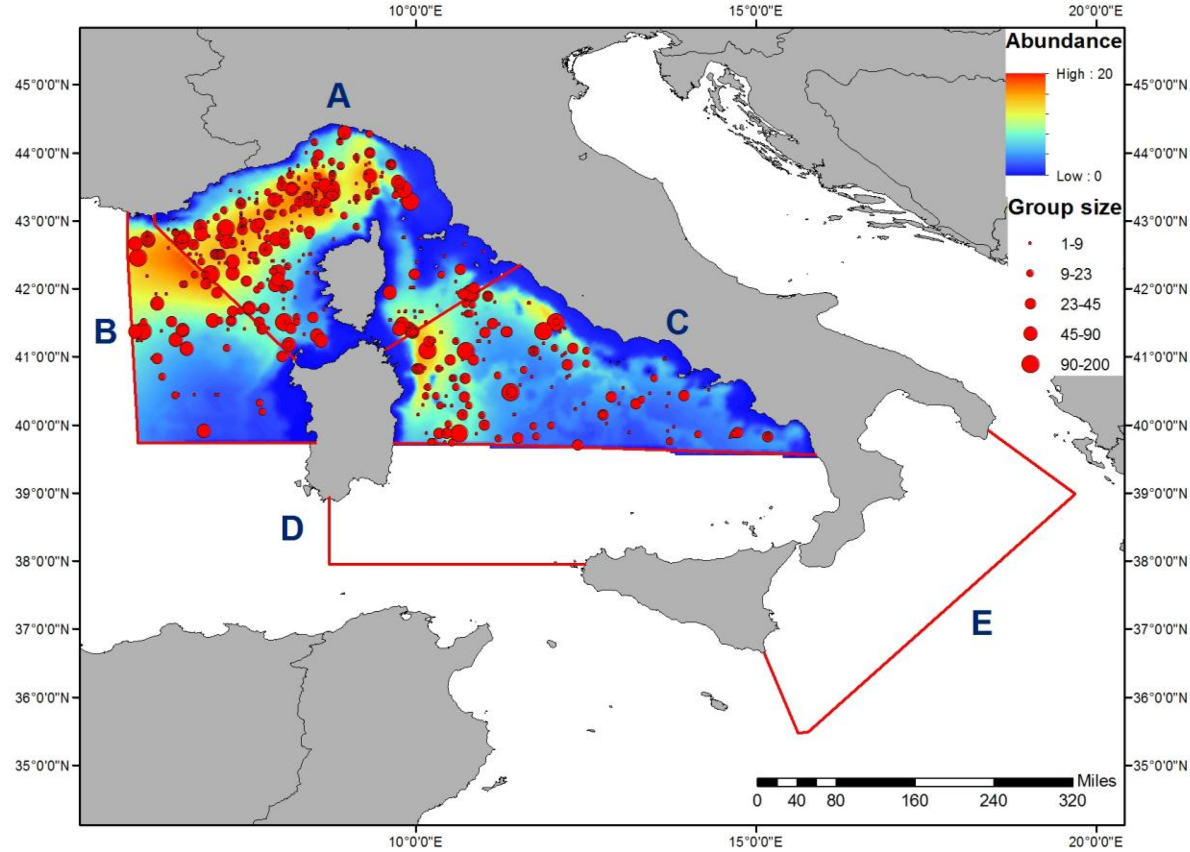
Fig. 13 - The selected MR model (i and ii) used to estimate perception bias and fitted detection function corrected for perception bias (iii); a) truncation at 800m and b) truncation at 300m. Plot (i) shows the estimated (line) and observed (histogram) probability of detection for team 1 given detection by team 2; plot (ii) shows the estimated (line) and observed (histogram) probability of detection for team 2 given detection by team 1.

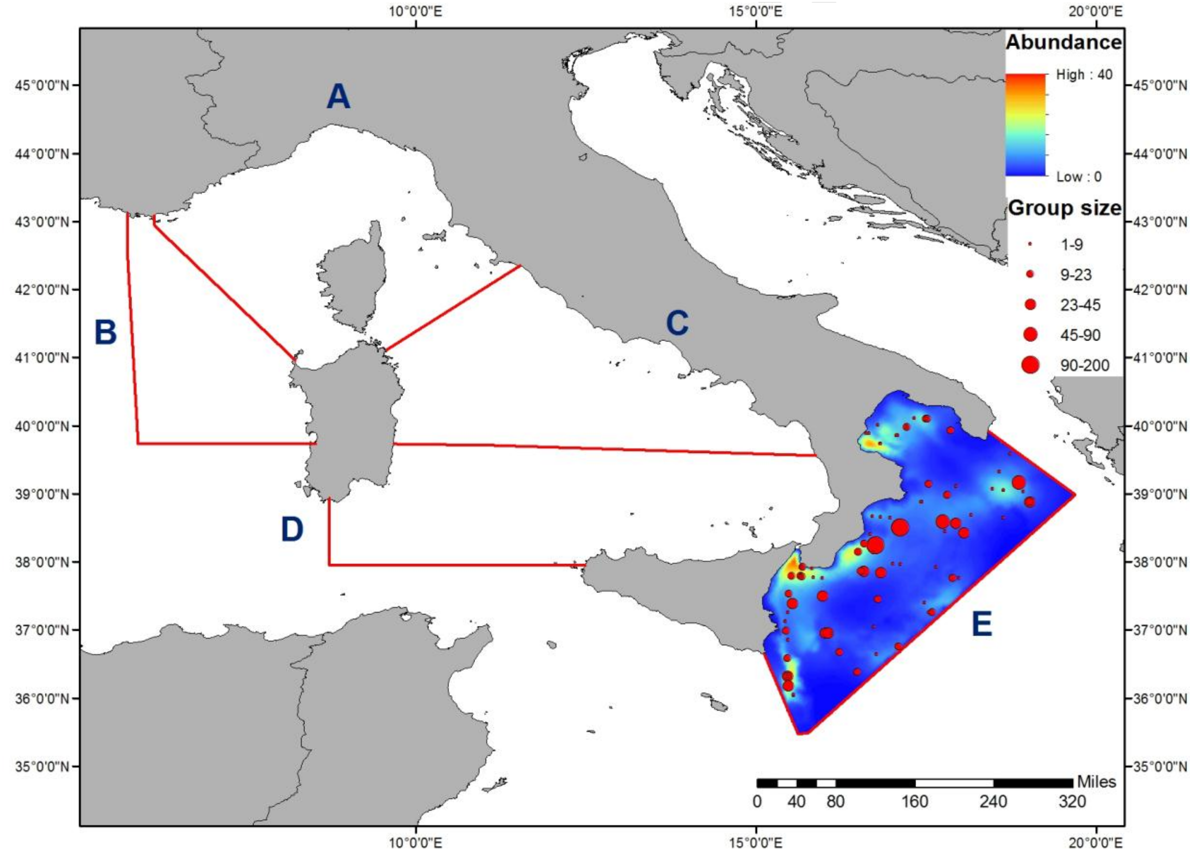


A
B
C
D
E
PELAGOS SANCTUARY
SEA OF SARDINIA
CENTRAL TYRRHENIAN SEA
SOUTHERN TYRRHENIAN SEA
IONIAN SEA
10°0'0"E
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
Miles
0 40 80 160 240 320



Group size
1-9
9-23
23-45
45-90
90-200
A
B
C
D
E
10°0'0"E
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
Miles
0 40 80 160 240 320


10°0'0"E
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
Abundance
High : 20
Low : 0
Group size
1-9
9-23
23-45
45-90
90-200
A
B
C
D
E
Miles
0 40 80 160 240 320


10°0'0"E
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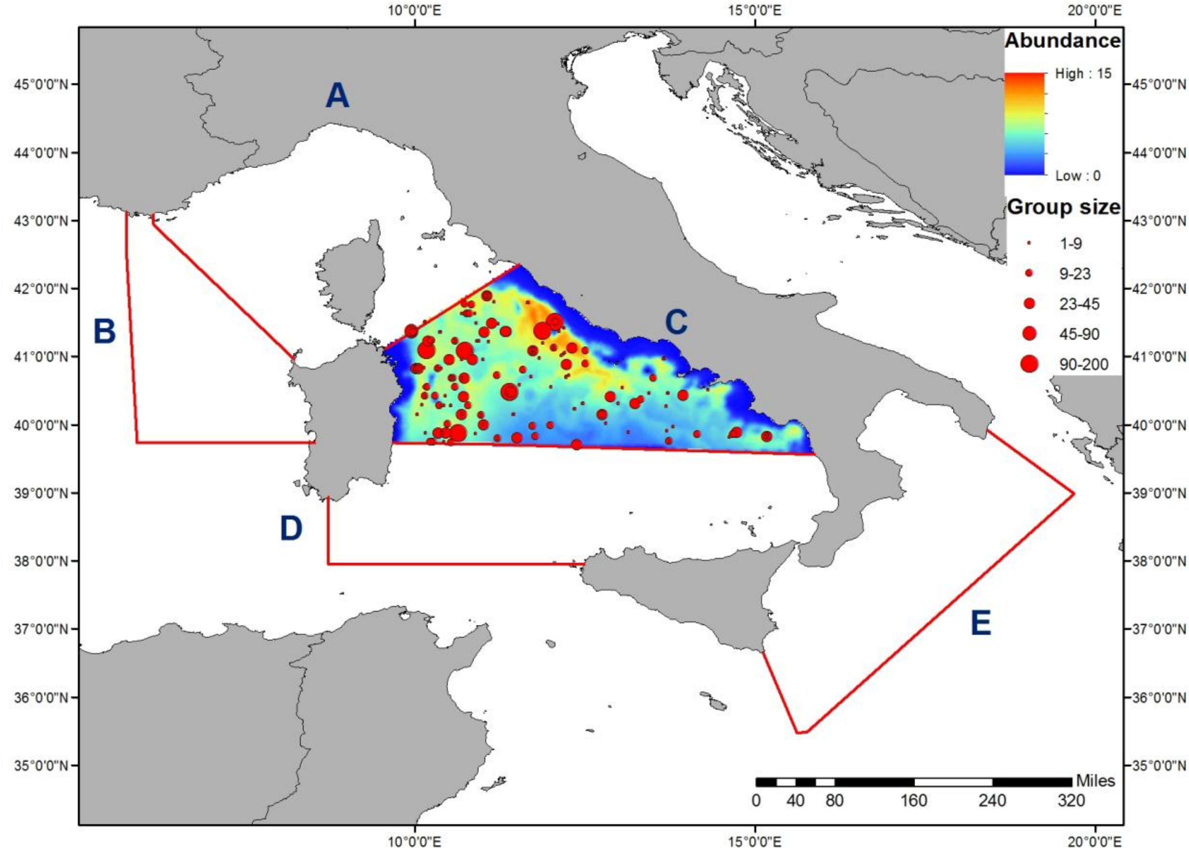
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
Abundance
High : 40
Low : 0
Group size
1-9
9-23
23-45
45-90
90-200
A
B
C
D
E
Miles
0 40 80 160 240 320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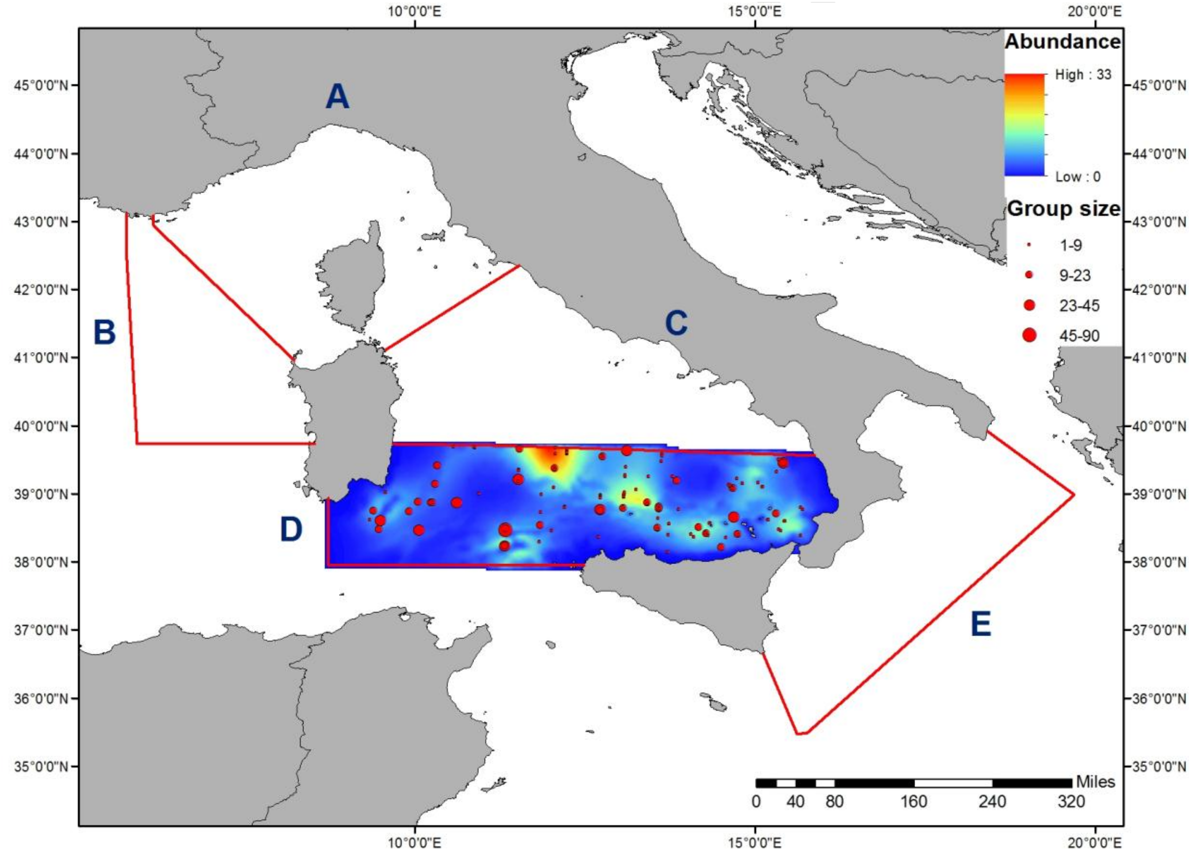
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
A
B
C
D
E
Abundance
High : 15
Low : 0
Group size
1-9
9-23
23-45
45-90
90-200
Miles
0 40 80 160 240 320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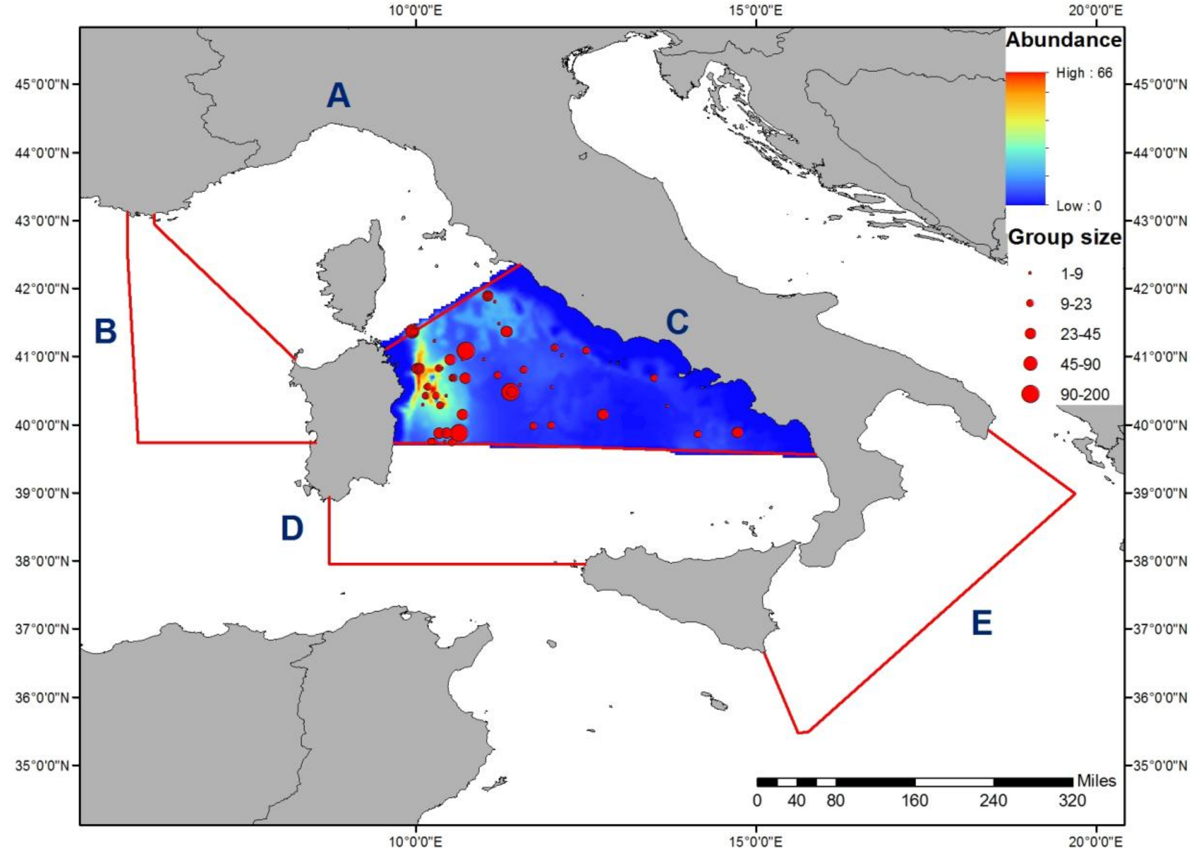
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
A
B
C
D
E
Abundance
High : 33
Low : 0
Group size
1-9
9-23
23-45
45-90
Miles
0 40 80 160 240 320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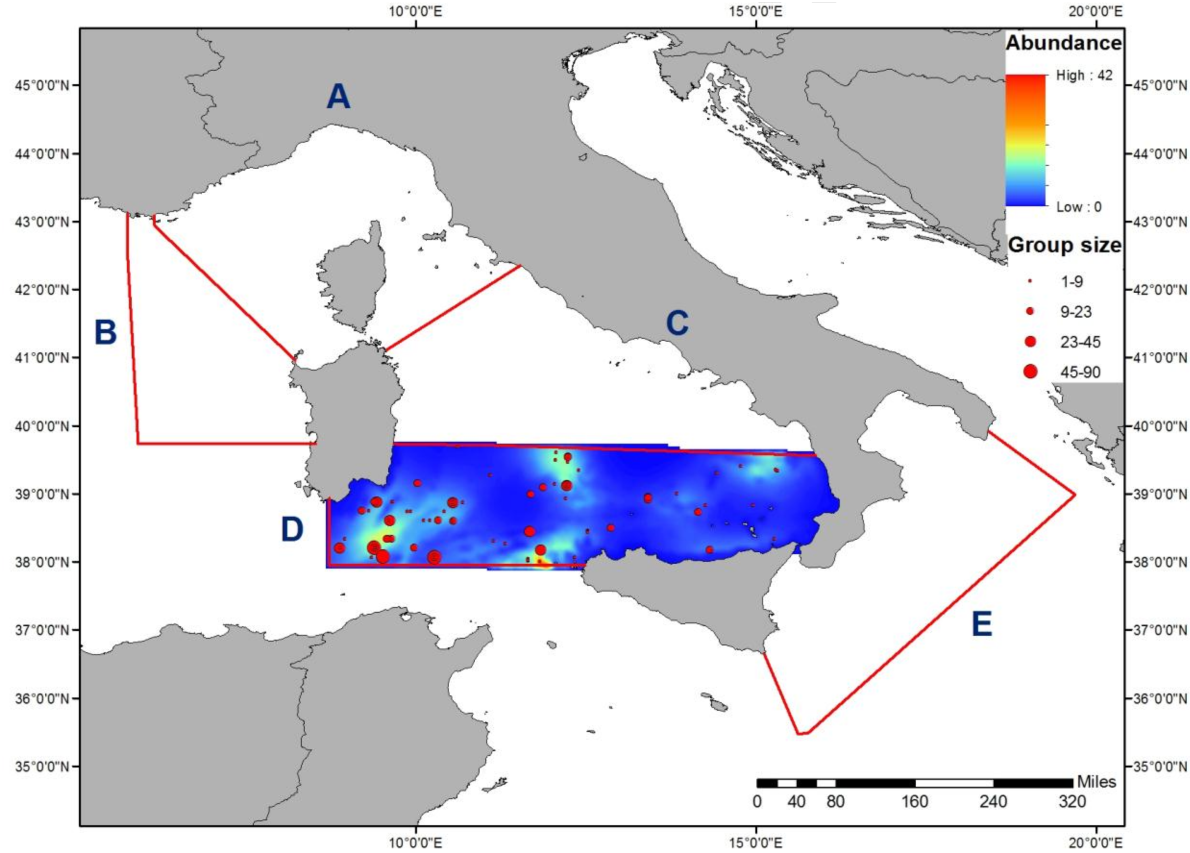
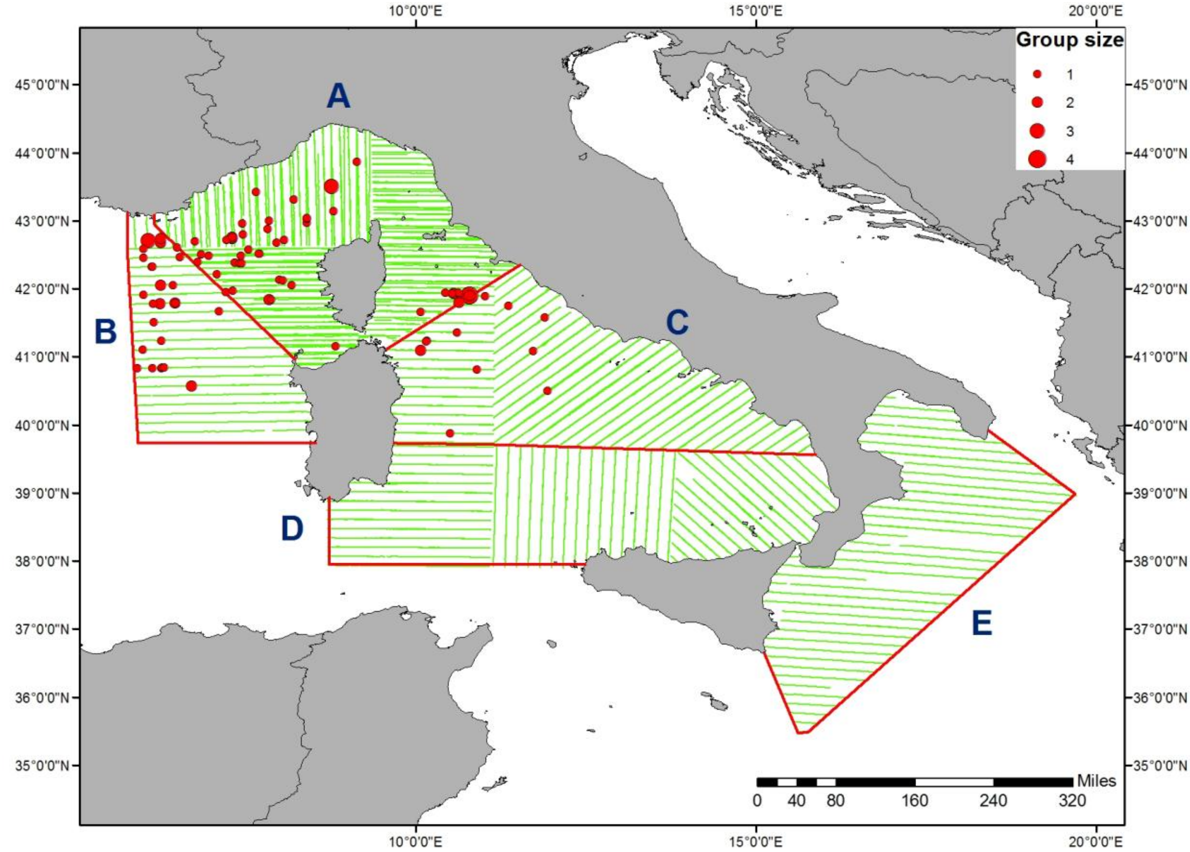
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
A
B
C
D
E
Abundance
High : 66
Low : 0
Group size
1-9
9-23
23-45
45-90
90-200
Miles
0 40 80 160 240 320


10°0'0"E
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
38°0'0"N
37°0'0"N
36°0'0"N
35°0'0"N
A
B
C
D
E
Abundance
High : 42
Low : 0
Group size
1-9
9-23
23-45
45-90
Miles
0 40 80 160 240 320


Group size
1
2
3
4
A
B
C
D
E
Miles
0 40 80 160 240 320
10°0'0"E
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
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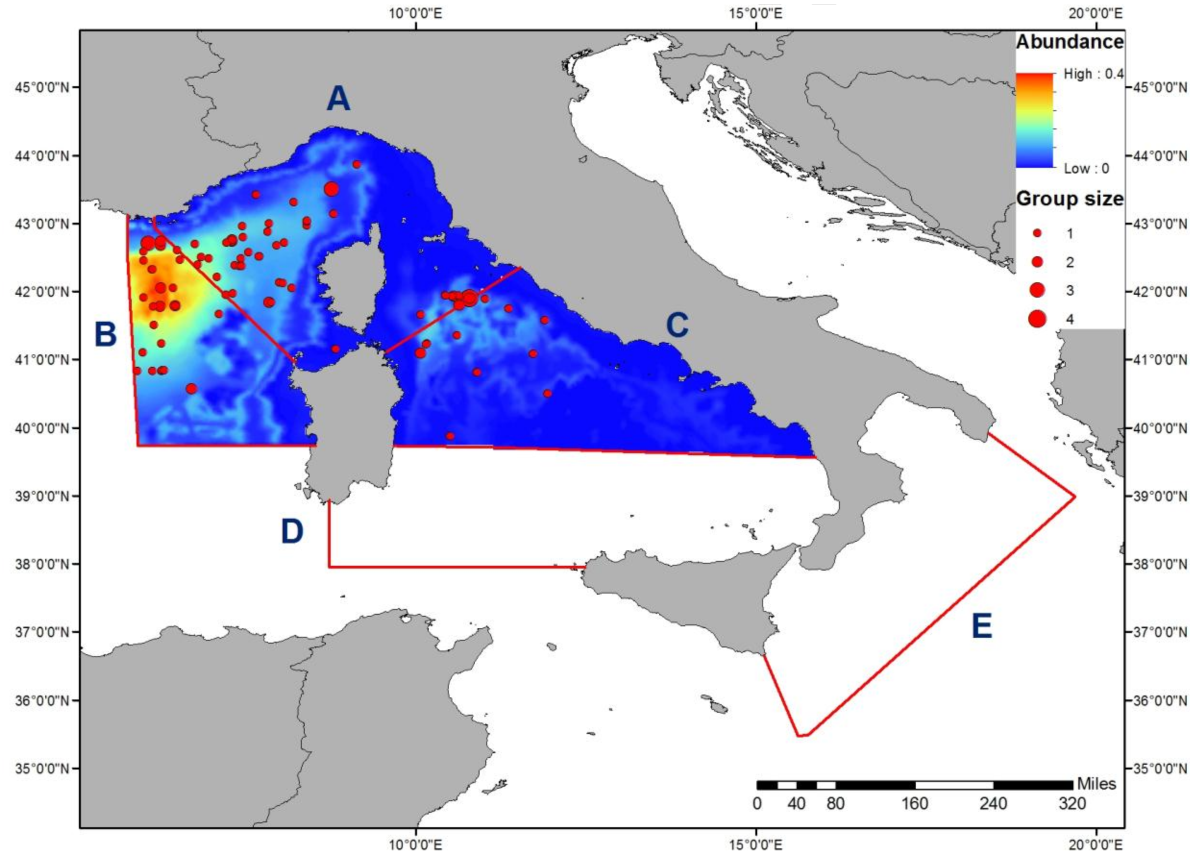
37°0'0"N
36°0'0"N
35°0'0"N


Abundance
High : 0.4
Low : 0
Group size
1
2
3
4
A
B
C
D
E
Miles
0 40 80 160 240 320
10°0'0"E
15°0'0"E
20°0'0"E
45°0'0"N
44°0'0"N
43°0'0"N
42°0'0"N
41°0'0"N
40°0'0"N
39°0'0"N
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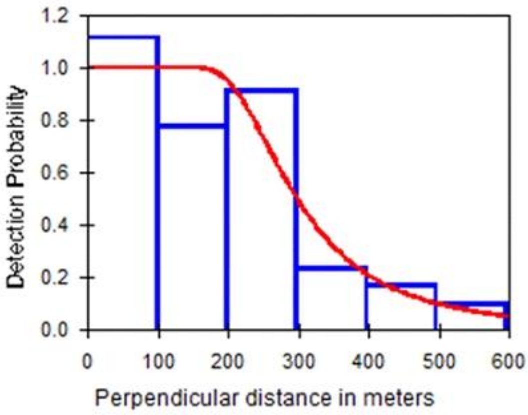
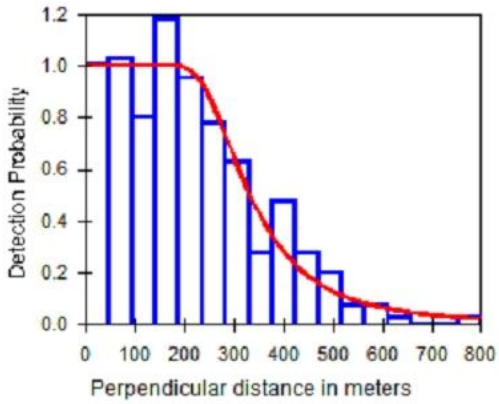
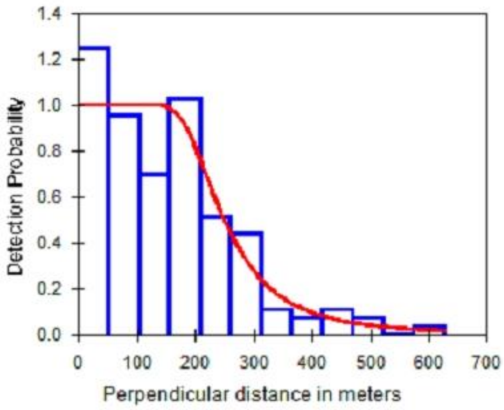
35°0'0"N



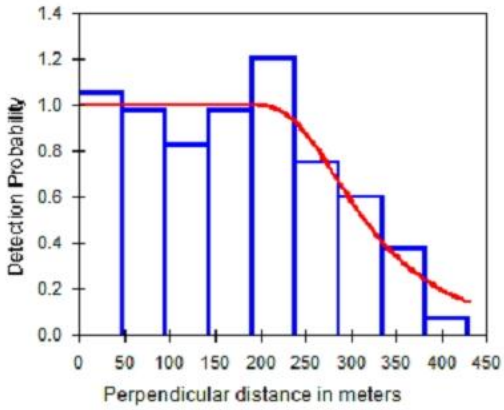
(a) Ionian in spring 2010



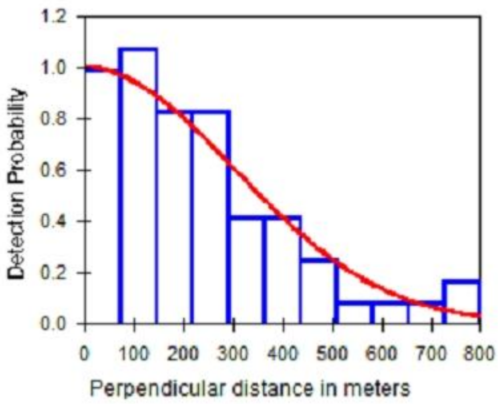
(b) summer 2010 stratified by blocks (West Sardinia, Pelagos and C-Tyrrhenian)



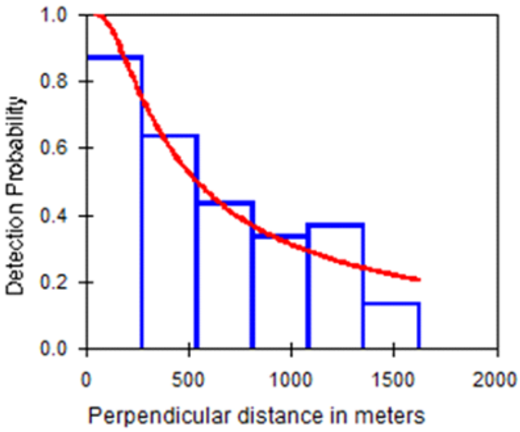
c) S-Tyrrhenian in winter 2010



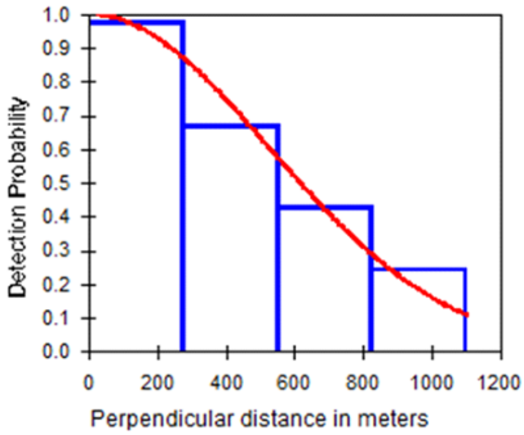
(d) S-Tyrrhenian in winter 2014



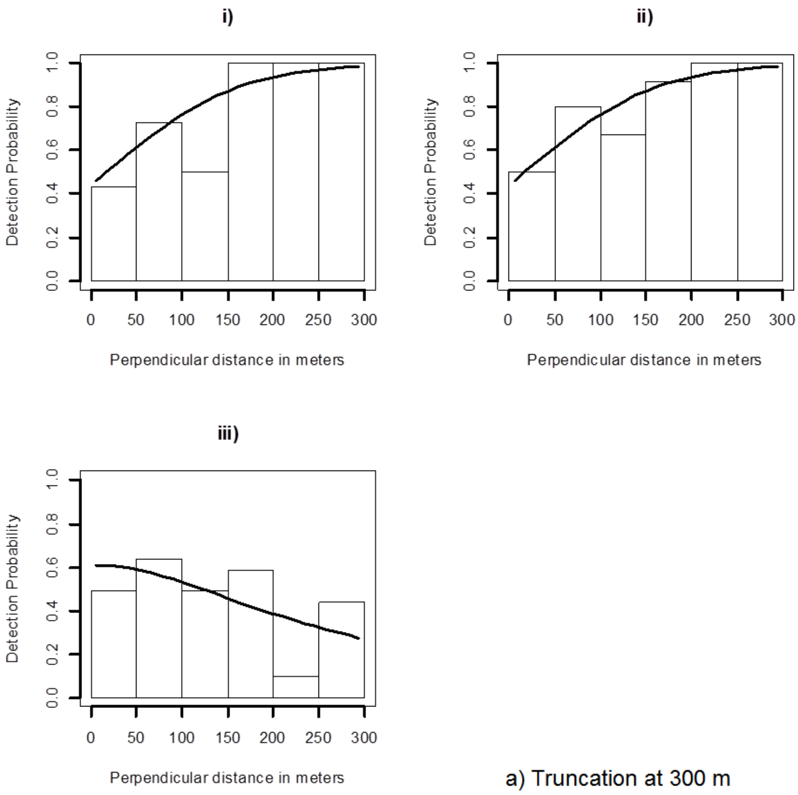
(e) C-Tyrrhenian in summer 2013



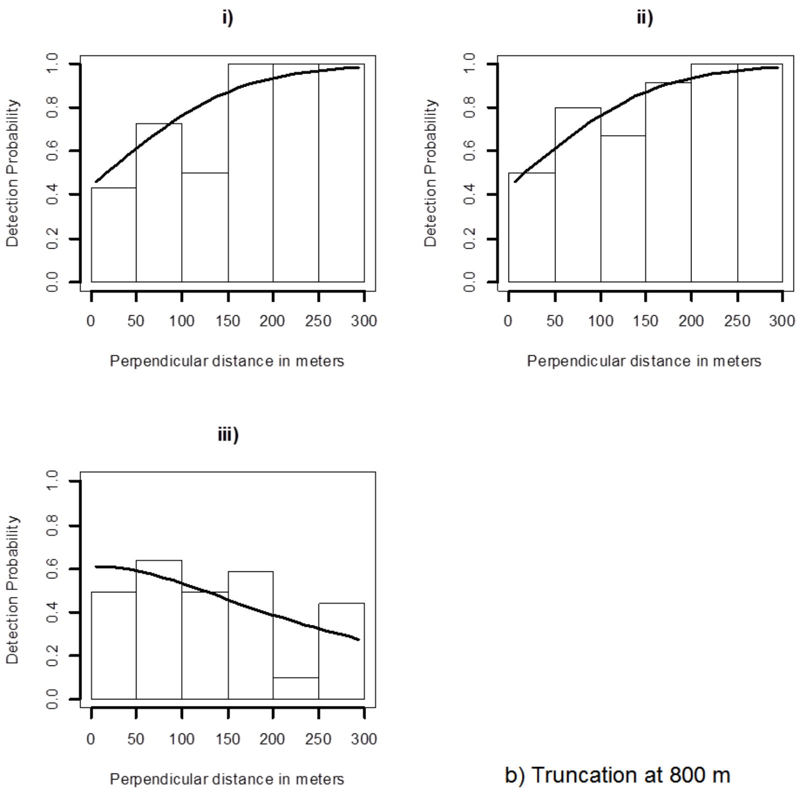
(b) summer all years stratified by blocks (West Sardinia, Pelagos and C-Tyrrhenian)



(d) Pelagos in summer



a) Truncation at 300 m



b) Truncation at 800 m