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Assessing the environmental status of the short-beaked common dolphin (*Delphinus delphis*) in North-western Spanish waters using abundance trends and safe removal limits

Camilo Saavedra[a,b,*], Tim Gerrodette[c], Maite Louzao[d], Julio Valeiras[a], Salvador García[e], Santiago Cerviño[a], Graham J. Pierce[f,g,h], M. Begoña Santos[a]

[a] IEO Instituto Español de Oceanografía, CO Vigo, Subida a Radio Faro 50, 36390 Vigo, Pontevedra, Spain
[b] Campus Do Mar, Universidad de Vigo, Plaza Miralles, A-7, 36310 Vigo, Pontevedra, Spain
[c] Southwest Fisheries Science Center, NOAA, 8901 La Jolla Shores Drive, La Jolla, CA 92037, USA
[d] AZTI-Tecnalia, Herrera Kaia, Portualdea, Pasaia, 20110 Gipuzkoa, Spain
[e] IEO Instituto Español de Oceanografía, CO Málaga, Puerto Pesquero, 29640 Fuengirola, Málaga, Spain
[f] CSIC Consejo Superior de Investigaciones Científicas, Eduardo Cabello, 6, 36208 Vigo, Pontevedra, Spain
[g] Oceanlab, University of Aberdeen, Main Street, Newwburgh, Aberdeenshire AB41 6AA, UK
[h] CESAM & Departamento de Biologia, Universidade de Aveiro, 3810-193 Aveiro, Portugal

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ABSTRACT

Monitoring and assessment of the status of marine mammal populations is a requirement of the European Marine Strategy Framework Directive (MSFD). Due to the difficulty of collecting data in the marine environment and because many populations of these highly mobile species inhabit waters of several Member States, monitoring of marine mammals is particularly challenging. In the present work we have used a 10-year time-series of data collected from multidisciplinary research surveys to estimate common dolphin (*Delphinus delphis*) abundance and trends in continental shelf waters of the northwest Spanish sub-region. We argue that this approach provides a valuable addition to large-scale dedicated surveys, offering a shorter interval between surveys and hence offering the possibility to track abundance changes at a regional scale. Trends in the number of dolphins present in the study area over the last 10 years show a mean increase of about 9.6% per year, which results in an evaluation of Good Environmental Status for the species in the area using the abundance indicator adopted in the framework of the MSFD. Data obtained from dedicated dual-platform surveys have been used to correct the detection bias in our data collected using single-platforms (attraction toward the observation platform and animals missed on the track-line), to obtain absolute abundance estimates for calculating bycatch limits. The average abundance over the study period was 12,831 dolphins [CI 95%; 9025, 18,242] calculated with the conventional distance sampling methodology, 4747 [3307, 6816] corrected for attraction and missed animals on the track-line, and 22,510 [15,776, 32,120] corrected only for missed animals on the track-line. The estimated safe bycatch limit for this area calculated from these abundance values were 218 [153, 310], 81 [56, 115] and 383 [268, 546] per year, respectively. Comparing these figures with estimates based on different sources, the percentage of dolphins that die in this study area is higher than the maximum limit allowable under the OSPAR criteria for population mortality adopted as an indicator for the MSFD.

1. Introduction

The common dolphin (*Delphinus delphis*) is one of the most abundant species of small cetaceans in the Northeast Atlantic Ocean and the most abundant in the Atlantic shelf waters of the Iberian Peninsula (Hammond et al., 2013; Méndez-Fernandez et al., 2012). However, the abundance, threats, population structure and population trends of this species are poorly known, even if this information is needed to assess its conservation status as required by national, European and other international legislation.

Abbreviations: MSFD, Marine Strategy Framework Directive; MS, Member States; GES, Good Environmental Status; AU, Assessment Units; ER, Encounter Rate; DF, Detection Function

* Corresponding author at: IEO Instituto Español de Oceanografía, CO Vigo, Subida a Radio Faro 50, 36390 Vigo, Pontevedra, Spain.
E-mail addresses: camilo.saavedra.penas@gmail.com (C. Saavedra), tim.gerrodette@noaa.gov (T. Gerrodette), maite.louzao@gmail.com (M. Louzao), xulio.valeiras@vi.ieo.es (J. Valeiras), salvador.garcia@ma.ieo.es (S. García), santiago.cervino@vi.ieo.es (S. Cerviño), g.j.pierce@iim.csic.es (G.J. Pierce), m.b.santos@vi.ieo.es (M.B. Santos).

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The Marine Strategy Framework Directive (MSFD; Directive 2008/56/EC), requires Member States (MS) to monitor and manage their marine environment and species, to achieve (or maintain) Good Environmental Status (GES) of their seas by 2020. The Directive further requires MS to implement management measures when necessary to achieve GES. Due to the varying environmental characteristics of Spanish marine waters, they have been divided into 5 sub-regions for management purposes (Santos and Pierce, 2015). The evaluation of GES is based on 11 Descriptors. Within Descriptor 1, Biodiversity, marine mammals are considered as a specific functional group on which MS must report separately. In Spain, to aid reporting, various Assessment Units (AU) for marine mammals have been identified (i.e., species, populations or part of them that permanently or seasonally inhabit a certain sub-region). In the Northwest Spanish sub-region, which includes the Spanish waters delimited by the marine boundaries of France and Portugal, the common dolphin was reported as a single AU (Santos and Pierce, 2015). High rates of fishery bycatch have been reported for common dolphins in Atlantic Spanish waters (Goetz et al., 2014; Pierce et al., 2010), with levels that could be well above the maximum limits that this species is able to endure, compromising its viability (López et al., 2003). However, in most cases, it is difficult to estimate cetacean mortality accurately, and therefore, trends in abundance become a key descriptor of the status of cetacean populations, necessary to assess the potential effect of human perturbations (Barlow, 2010; Taylor et al., 2007). Thus, two indicators were proposed to assess the environmental status of this AU following the OSPAR recommendations (see ICG COBAM, 2012), a state indicator (population abundance trends), and a pressure indicator (mortality due to bycatch).

The criterion subsequently proposed to determine whether a cetacean AU is at GES was based on criteria proposed by IUCN: to maintain populations size at or above baseline levels, with no decrease of $\geq 30\%$ over a three-generation period (ICES, 2014). In the absence of good historical data, the baseline is considered to be the population size when modern day monitoring (of a given AU) commenced. The recommended maximum rate of decline recognizes the fact that small rates of decline are statistically difficult to detect. However, this rule does not specify an absolute drop in populations' size that should be considered critical and the precise criterion to be used is still being debated (ICES, 2017).

Thus far, the only absolute estimate of common dolphin abundance in the shelf Atlantic waters of the Iberian Peninsula was obtained during the SCANS-II survey in 2005 (Hammond et al., 2013). This large-scale survey used mark-recapture distance sampling methodology with two observation platforms (Buckland et al., 2004). Large-scale surveys such as SCANS allow the entire distribution area to be covered for most of the studied species (and therefore, their populations) over a relatively short period of time, thus avoiding under- or overestimating abundance due to shifts in distribution within the range over the surveyed period. In addition, the double-platform methodology used during these surveys permits correction for animals missed on the track-line, thus generating absolute rather than relative estimates of abundance, and also for responsive movements by cetaceans to the observation platform (i.e. attraction or avoidance), which avoids over- or underestimation of abundance (see Buckland et al., 2004).

However, the high costs involved, the complex logistics of organization and the analysis of the large amount of data are the main reasons why the frequency of these surveys has so far been approximately decadal. This makes it difficult to detect negative trends in populations over appropriate time periods to decide (and implement) management decisions (Hammond et al., 2013). For this reason, the study of cetacean abundance trends at shorter time scales has been encouraged on several occasions (Hammond et al., 2013; ICES, 2014), especially in regions where cetacean populations are subject to specific threats, as is the case of our study region (due to fishery bycatch; Goetz et al., 2014).

A viable and cost-effective approach for use as a complement of large scale surveys can be the use of oceanographic vessels that carry out periodic (generally annual, sometimes seasonal) monitoring and sampling of the marine environment. For instance, surveys designed to provide fishery-independent abundance indices for commercial fish stocks. Currently, acoustic surveys for the assessment of the living resources of the pelagic ecosystem usually perform linear transects perpendicular to the coast, with an annual frequency and a level of effort higher than that of the dedicated large-scale cetacean surveys. We argue these characteristics make them suitable platforms for applying distance sampling methods to estimate cetacean abundance. However, challenges include the adaptation of the methodology to the characteristics of the ship, as well as the tradeoffs with the other objectives and activities of the survey. One of the limitations, for example, tends to be the space for marine mammal observers, and therefore the number of personnel available to carry out these observations is limited. This sometimes makes the use of two observation platforms unfeasible.

The absence of a double platform prevents corrections for attraction or avoidance movements against the observation platform (e.g., common dolphin and harbour porpoise *Phocoena phocoena* respectively) and for animals missed on the track-line, which can be particularly problematic for inconspicuous species or those that perform long dives (e.g., sperm whale *Physeter macrocephalus* or beaked whales). As is the case for dedicated cetacean surveys, if cetaceans are not sighted before they respond to the ship, in cases of attraction to the ship, abundance will be overestimated (Mullin and Fulling, 2003). These attraction movements can be important in the case of common dolphins, and have been reported on several occasions (e.g., Cañadas et al., 2004). All these factors mean that accurate estimates of absolute abundance cannot be obtained in the study area from single-platform surveys alone. However, assessing trends in population size can use an appropriate index of relative abundance (Hammond et al., 2013). We presume that attraction or evasion movements that may be exhibited by certain species of cetaceans are maintained over time if the same platform and method of sampling is maintained, so that the positive or negative bias of abundance estimates would not be altered over time. In addition, the number of individuals of a given species missed along the track-line would also remain constant if the conditions and observer experience are also constant. Therefore, although the absolute estimates may be biased, the trends we detect will still be informative.

The bycatch indicator was designed to be applied to populations as a whole, and therefore, absolute abundance estimates are required to assess it (Hammond et al., 2013). Moreover, it is necessary to know the percentage of the population that can be safely removed, according to its biological parameters and population dynamics, without compromising its viability. This index is not currently available for the common dolphin and a common short-term target calculated for harbour porpoises has been proposed for all cetaceans (i.e., a total annual bycatch level in all fisheries of 1.7% of the maximum likelihood estimate of abundance; ASCOBANS, 2000; IWC, 2000). Finally, to make the bycatch indicator operational, the bycatch level to which the population is subjected must be known. The MSFD requires MS to design and implement monitoring programs and programs of measures to ensure reliable estimates are obtained and, if necessary, that bycatches are reduced to sustainable levels. MS are encouraged to work at regional or sub-regional scale by co-operating with neighbouring MS (in particular for highly mobile species like the common dolphin). However, MS are in the position to decide if the maximum percentage of mortality caused by bycatch should be measured only at population level (meaning that some areas are allowed to suffer high levels of bycatch as long as the total bycatch does not exceed the established limits), or if each MS has to ensure that this percentage is not exceeded in its waters (regardless of what may happen in waters of the other MS). To do this, the abundance of the fraction of the population that inhabit their waters has to be known. To obtain unbiased estimates of abundance from single-platform surveys, data from dedicated double-platform cetacean surveys (i.e., SCANS-II) can be used to correct abundance estimates from response movements and/or missed animals in the track-line.

Within this context, our main objective was to assess the GES of the common dolphin in North-western Spanish waters using abundance trends and safe removal limits. We have estimated the abundance of common dolphins over a 10-year time-series using data collected during multidisciplinary oceanographic surveys carried out in the north and northwest coasts of the Iberian Peninsula. During these surveys the methodology of distance sampling in line-transects was applied, based on a single observation platform. Relative annual abundance of common dolphins was estimated for the shelf waters of the Northwest Spanish sub-region using conventional multiple covariates distance sampling methodology (Buckland et al., 2001) and trends were analyzed to evaluate, following the criterion of the abundance indicator, if the population could be considered at GES. We used data from the dedicated double-platform cetacean SCANS-II survey, which covered our study area in 2005, to derive correction factors for the abundance estimates. Different abundance scenarios were used to define thresholds for the bycatch indicator.

2. Material and methods

2.1. Study area and data collection

Over the last two decades, the Spanish Institute of Oceanography (IEO) has carried out annual acoustic surveys (PELACUS) to estimate pelagic fish biomass along the north and northwest coasts of the Iberian Peninsula (ICES subareas IXaN and VIIIc). Since 2007, an observer program for top predators has been integrated into the surveys, collecting sightings on cetaceans and seabirds using distance sampling methodology in line-transect with a single platform configuration. Surveys took place during March and April, each lasting approximately one month. The study area comprises the continental shelf waters belonging to the Northwest Spanish sub-region defined under the MSFD, with a total surface of $\approx 37{,}000\ km^2$ (Fig. 1). The acoustic survey design consists of a systematic grid with equally spaced (8 nMi apart) transect lines perpendicular to the coastline which are conducted at 10 knots. Transects are also performed inside the *rías* (i.e., coastal inlets) but this part of the study area was removed from our analyses because of the different environmental characteristics of these inlets. Although common dolphins have been found to visit the *rías* occasionally, *rías* do not represent the usual habitat of this species. From 2007 to 2012, surveys were performed on-board the French R/V *Thalassa*, of 70 m in length, whereas and the Spanish R/V *Miguel Oliver*, of 67 m in length, was used from 2013 to 2016. Observations platforms were located at approximately 16 m and 12 m height above the sea level in the R/V *Thalassa* and R/V *Miguel Oliver*, respectively. Differences in height between observation platforms on the two ships were not taken into account because in preliminary analyses the effect was not significant. In some years, transects were extended into Portuguese and French adjacent waters. Sightings performed on these additional transects were not used to estimate the Encounter Rates (ER) and abundances but were used to fit the Detection Function (DF) (see below). In addition, the navigation routes between predefined transects (namely non-predefined transects) followed a fixed course and at a constant 10-knots speed and were also used to make observations. Common dolphin observations performed during the navigation routes were only used to fit the mean DF of the entire period and study area.

Sampling was performed using a standard distance sampling methodology (see Buckland et al., 2001). A team of three observers collected data on sightings during each survey, with two observers on duty during daylight hours at the same time. The observers searched, by naked eye, a sector of 90° from the track-line to 90° port and starboard respectively. Binoculars were used only to identify the species and/or estimate the group size. Because observers search during the acoustic prospection, transects were covered steadily and a passing mode methodology was applied (i.e., without stopping and approaching the boat when a sighting is performed). Transects were only disrupted for fishing, which is carried out to help ground truth the acoustic signal and to collect fish samples and length measurements. Fishing was performed with a pelagic trawl. During fishing, searching effort was stopped, resuming when the vessel had returned to the same point of the track-line after completing the haul and acoustic prospection resumed. Every period of observation was called a *leg* and a new *leg* started when searching conditions varied, composition of the observer team changed or transects finished or were interrupted. For every *leg*, environmental conditions were recorded (i.e., Beaufort, wind speed, wave direction, wave height, visibility, cloudiness, glare intensity and degrees and general searching conditions), as well as ship course direction, composition of the observer team, starting time and platform.

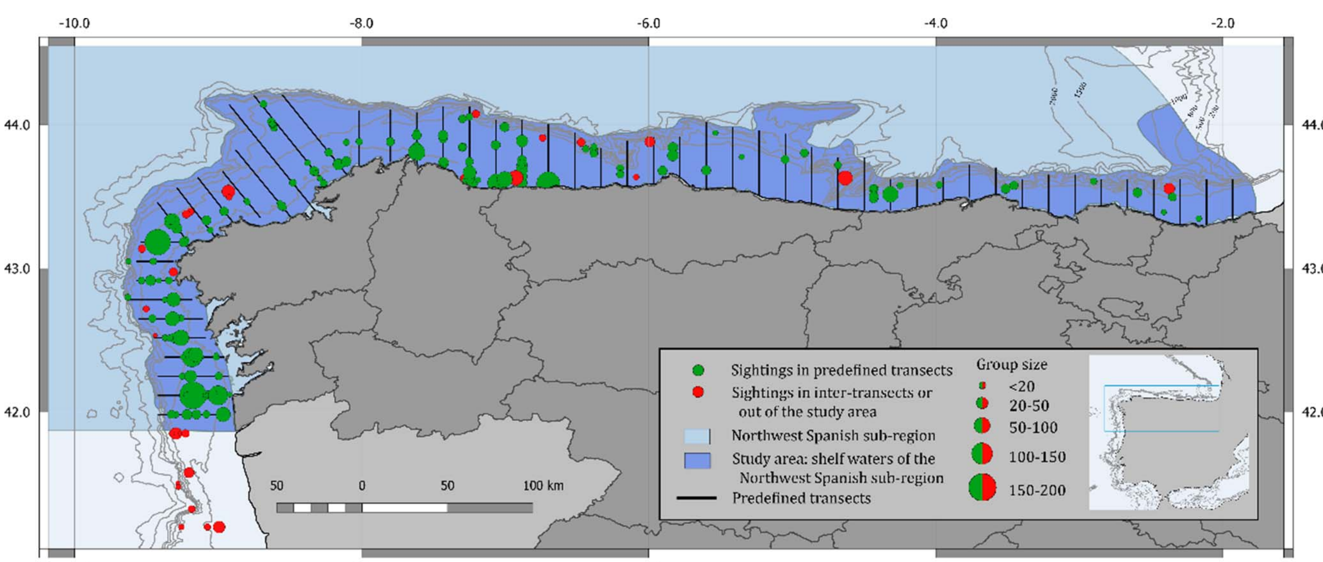

Fig. 1. Map of the study area. Northwest Spanish sub-region as defined under the MSFD (light and dark blue). Shelf waters of the Northwest sub-region and study area (dark blue). Predefined transects within Spanish waters (dark lines). Sightings outside the Spanish study area or belonging to non-predefined transects (navigation routes) are represented by red bubbles. Sightings in the Spanish study area and belonging to predefined transects are shown as green bubbles. The size of the bubbles is proportional to the estimated group size of sightings (see map legend). The blue rectangle on the map of Spain, in the lower right corner, indicates the location of the study area. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

During the searching effort, observers filled a form with the observer code, date, number of the *leg* and platform description. Each time a sighting occurred, the observer recorded the time, species, group size, angle with respect to the ship's course, radial distance, behaviour and reaction to the ship. The angle was calculated with an angle meter and the distance with a measuring stick based on the Heinemann (1981) method.

2.2. Analytical methods

2.2.1. Distance sampling theory and abundance estimates

Common dolphin sightings from 2007 to 2016 were analysed using a conventional design-based distance sampling methodology (Buckland et al., 2001) with multiple covariates (Marques and Buckland, 2003) using the standard software Distance version 6.2 release 1 (Thomas et al., 2010) and the R package *Distance* version 0.9.6 (Lawrence Miller, 2016) under the R software version 3.3.2 (R Core Team, 2016).

The method assumes that dolphins presence varies spatially but transects randomly placed through the study area allow mean Density (D) to be estimated. If dolphins are recorded within a given perpendicular or truncation distance (w) along the length of transects or *legs* (L) the mean D within the surveyed area (a) will depend on the expected number of clusters of dolphins ($E_{(n)}$) within distance w, multiplied by the mean expected cluster size ($E_{(s)}$) divided by a times the probability of detection in that area (P_a).

$$D = \frac{E_{(n)}E_{(s)}}{aP_a}$$

For estimating aP_a, a form must be specified for the DF, denoted as $g_{(y)}$, which represents the probability of detection of an object at a given perpendicular distance (y) from the track-line. The density function of distances $f_{(y)}$ is identical in shape to the detection function $g_{(y)}$ but rescaled so that it integrates to unity. The surveyed area can be expressed as $a = 2Lw$. So, if we define the effective strip half-width as $\mu = wP_a$ then $aP_a = 2\mu L$. Since $\mu = g_{(x)}/f_{(x)}$ and $g_0 = 1$ after rescaling, then $\mu = 1/f_0$ and the D equation becomes (see Buckland et al., 2001 for detailed explanation).

$$D = \frac{E_{(n)}f_{(0)}E_{(s)}}{2Lg_0}$$

In surveys of inconspicuous animals, the probability of detecting an animal at zero distance g_0 would be expected to be lower than 1 and the unconditional probability of detection of an animal in the surveyed area can be factorized as the product of g_0 by P_a. However, g_0 was assumed to be 1: the bias that can occur when not taking this fact into account was not considered in this study because we expected that it will remain relatively constant throughout the time series and therefore would not impact our estimated trends.

The final objective of the method is to estimate an index of the number of dolphins (N) in the whole studied area (A). Replacing the parameters in the previous equation by their estimators, the final equation will be as follows:

$$\widehat{N} = A\widehat{D} = A\frac{\widehat{n}\widehat{f}_{(0)}\widehat{E}_{(s)}}{2L}$$

For the analysis, only *legs* and sightings performed with Beaufort Sea state lower or equal to 5 were used. Due to the season in which the surveys take place, the observation conditions usually exceeded Beaufort 4 (commonly used as the maximum limit for observation), and for that reason it was decided to also use sightings performed with Beaufort ≤ 5 (Fig. 2).

All sightings and years were pooled to fit a common DF (green and red bubbles in Fig. 1). The w was selected following the recommendations of Buckland et al. (1993). Half-normal and hazard-rate functions were tested as DF, with covariates Beaufort, wind speed, wave

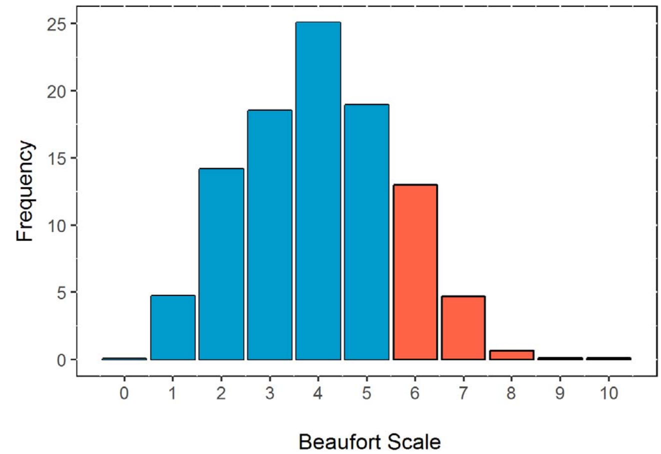

Fig. 2. Beaufort Sea state. Percentage of kilometres surveyed from 2007 to 2016 under different sea conditions, measured with the Beaufort scale (0–10). In blue, percentages of kilometres included in the analysis, in red kilometres removed. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

height, visibility, cloud cover, glare intensity, group size and log of the group size as proposed by Marques and Buckland (2003). The best DF was selected based on the Akaike's Information Criterion (AIC).

D and N were estimated with sightings made inside the study area and along predefined transects (i.e., green bubbles in Fig. 1). The analysis was post-stratified by year and the global DF and the estimated average cluster size over the whole time period were applied to the annual ER to derive annual estimates.

2.2.2. Trends and the abundance GES indicator

ICES Working Group on Marine Mammal Ecology (WGMME) suggested that a suitable indicator target to assess the status of each AU of cetaceans could be based on the criterion developed by the IUCN of 'maintaining population size at or above baseline levels, with no decrease of ≥30% over a three-generation time' (ICES, 2014). Although the baseline ideally should be derived from historical data obtained prior to major human impacts, ICES (2016) acknowledged that these are not usually available and that in many cases, even if the historical abundance and distribution was known it cannot realistically be restored as today's marine environment is very different. For these reason, for our assessment of the status of the common dolphin population we used the approach proposed by ICES (2014) namely to use the start of the data time-series as the baseline, with indicator assessment thresholds set as a deviation from that baseline value. Moreover, the ICES WGMME proposed to base the assessment on a time-series covering at least the last 10 years, with a minimum of four counts during that period (ICES, 2016).

The generation time of common dolphins reported in the literature varies depending on the parameters used and the rate of growth assumed, ranging between 12.8 (Danil and Chivers, 2007) and 14.8 years (Taylor et al., 2007), a range that includes the value of 12.94 estimated by Murphy et al. (2007). Based on these values, three generations would represent a period of between 38.4 and 44.4 years. Therefore, a decrease of ≥30% over a three-generation period is equivalent to a decrease of 0.86% (0.80–0.92) per year or approximately 8% (7.72, 8.87) over a 10-year time-series.

In our analyses we fitted a linear regression with 95% confidence limits to the 10-year time-series of annual estimates of abundance obtained in the Northwest Spanish sub-region in order to evaluate if the common dolphins present in the continental shelf of that region meet the proposed criteria for GES. However, the uncertainty around each estimate is not taken into account when fitting a regression in this way. Therefore, using the mean and coefficient of variation (CV) of abundance estimates for each year and assuming a log-normal distribution of errors, 1000 simulated datasets were generated from these data and each was tested for the existence of a trend.

2.2.3. Corrected abundance estimation

The estimates of abundance obtained with the conventional distance sampling methodology were corrected to take into account the animals missed on the track-line and possible movements of attraction/avoidance of common dolphins towards the observation platform.

Existing data from the SCANS-II survey (SCANS-II, 2006) were used for this purpose. The available abundance estimates were derived from data collected in the same area (using a 44.5 m vessel with a tracker platform of 8.8 m and primary platform of about 5.6 m in height). We assume that the attraction to the vessel, as well as the proportion of animals missed on the track-line should be very similar, and confirmed that the difference in platform height did not affect the results. In SCANS-II, two estimates of abundance were obtained for common dolphins in an area (i.e., block W) that includes our study region, one using the same methodology as the one used in our analysis (i.e., conventional distance sampling) and another using the mark and recapture methodology (i.e., with a double-platform configuration) that corrects for these biases (i.e., attraction/avoidance and $g_{(0)}$). The estimated abundance and CV with these two methodologies were 48,743 (CV 0.25) and 17,916 (CV 0.22) common dolphins, respectively, the latter representing about 37% of the abundance estimated with the conventional distance sampling (SCANS-II, 2006) (note that block W is almost 4 times larger than our study area). The decimal form of this value (0.37) with a CV of 0.25 (standard deviation; SD 0.09) was, therefore, the correction factor (CF) used for correcting our initial estimate, by including a new parameter in the abundance equation:

$$\widehat{N} = CF\cdot A\frac{n\hat{f}_{(0)}\widehat{E}_{(s)}}{2L} \quad \text{where } CF \sim \boldsymbol{N}(0.37{,}0.09)$$

An alternative method of correction was also applied using only the $g_{(0)}$. The $g_{(0)}$ calculated in the SCANS-II survey for the common dolphin is 0.57 (CV 0.16) which means that by applying this correction our estimate of abundance should increase by approximately 175% in relation to the estimates made with conventional distance sampling methodology. However, when applying the correction on the effects of attraction this has an opposite effect indicating that the abundance was being overestimated and the corrected value should be approximately only 37% of the previous one. These data indicate that the effect of the estimated attraction is very high and suggest that it is possibly being overestimated, as recently discussed in ICES WGMME (ICES, 2017). Therefore, abundance was also corrected by using only the SCANS-II $g_{(0)}$ and not taking into account the attraction.

$$\widehat{N} = A\frac{n\hat{f}_{(0)}\widehat{E}_{(s)}}{2Lg_{(0)}} \quad \text{where } g_{(0)} \sim \boldsymbol{N}(0.57{,}0.08)$$

These two corrections of abundance, although subjected to certain limitations and assumptions, give a range of possibilities over the absolute abundance of common dolphins that inhabit the waters of the northern Spanish shelf and allow us to use this abundance to calculate tentative values of safe limits of bycatch for the fleet that operates in that area as we explain below.

2.2.4. Safe mortality limits and the bycatch MSFD indicator

We use some approximations to calculate the necessary values described above in order to assess common dolphin environmental status considering that the maximum percentage of bycatch calculated for the whole population should not be exceeded in waters of any MS. Three approaches were used to calculate the number of animals present in the waters of our study area and therefore susceptible to be bycaught by the fleet that operates there. Two corrected mean common dolphin abundances over the 10-year period (i.e., one assuming attraction and animals missed in the track-line and another corrected only by missed animals in the track-line) and the estimated average abundance using the conventional distance sampling methodology without bias correction were used to calculate safe bycatch limits. In absence of a specific figure for common dolphin, the percentage of bycatch considered safe for harbour porpoises (i.e., 1.7% following IWC, 2000) was used as a proxy in this work for common dolphins and the number of dolphins that it would represent was calculated. Bycatch limits estimated were compared with bycatches reported in the literature. For such purpose, we performed a literature review to obtain bycatch information of the study area indicating the type of survey used, affected species, bycaught individuals, and fishing gear involved.

3. Results

3.1. Distance sampling analysis and raw abundance estimates

Ten years of surveys were analysed. A total of 19517.2 linear km was surveyed (as explained before, this figure does not include transects within the *rías*). 81.5% of the surveyed linear kilometres (15904.2 km) were performed under Beaufort ≤ 5 and used for the DF selection, including sightings along predefined transects within the study area, inter-transect travel and transects outside the study area (Table 1).

Of the 193 sightings of common dolphins registered over the ten years, 166 took place in Beaufort ≤ 5 and 150 (90.4%) were within the truncation distance of 650 m (Table 1). The truncation distance was established according to the distribution of the perpendicular distances to locations at which the dolphins were spotted (Fig. 3).

The best model for the DF (with the lowest AIC) had a half-normal key with Beaufort and log of the cluster size as covariates (Fig. 4) with an effective strip half width of 230.35 m (CV 0.07). The resulting half-normal DF followed the expression described below:

$$g_{(y)} = \exp(-y^2/2(a\cdot\exp(bf + lgs))^2)$$

where y is the perpendicular distance and the scale parameter σ is defined by an exponential function where a is the intercept with a mean value and SD of 167.1 and 8.566 respectively, bf (−0.1135 SD 0.06) the coefficient of the covariate Beaufort and lgs (0.2319 SD 0.05) the coefficient of the covariate log of the group size.

Table 1
All linear kilometres surveyed (km) and common dolphin sightings (N) by year. Only under Beaufort ≤ 5 (B ≤ 5), under Beaufort ≤ 5 and truncation distance of 650 m (B ≤ 5 T650 m) and under Beaufort ≤ 5 but only predefined transects inside the study area (B ≤ 5 SA) and with truncation distance of 650 m (B ≤ 5 SA T650m).

	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	Total
km	1834.5	2588.6	2282.3	2375.1	1336.6	2220.3	1754.7	1754.8	1676.1	1694.2	19517.2
km (B ≤ 5)	1698.2	1836.6	2125.6	1578.9	1248.8	1756.9	1262.5	1541.1	1424.5	1431.1	15904.2
km (B ≤ 5 SA)	1555.1	952.0	1442.1	1046.0	1185.8	1680.5	1024.6	1318.8	1398.5	1431.1	13034.5
N	12	18	12	27	17	20	21	24	13	29	193
N (B ≤ 5)	11	12	12	20	17	17	17	20	12	28	166
N (B ≤ 5 T650 m)	11	12	11	20	15	17	15	16	9	24	150
N (B ≤ 5 SA)	7	4	9	18	15	17	16	17	12	28	143
N (B ≤ 5 SA 650 m)	7	4	8	18	13	17	14	13	9	24	127

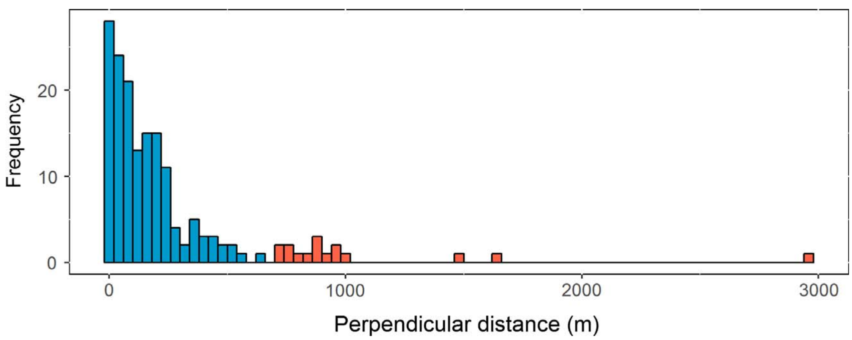

Fig. 3. Perpendicular distance of sightings. In blue, sightings included in the analysis, in red sightings removed. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

The DF estimated was applied to the 13034.5 linear km belonging to the predefined transects inside the Spanish study area and to the 127 sightings of common dolphins within the truncation distance (Table 1). The estimated mean group size was 16.6 (CV 0.12), the mean abundance over the whole period was 12,831 (CV 0.18) and the mean density 0.35 animals/km^2 (CV 0.18). Annual abundance and density estimates are shown in Table 2.

3.2. Trends and the abundance MSFD indicator

The linear regression fitted to the annual abundance estimated versus a 10-year time-series was: $N = 7096.8 + 1086.4 \cdot Yr$ with a R^2 of 0.269, where N is the annual abundance and Yr the position of the year in the time series denoted as 1 to 10 (Fig. 5).

The predicted fitted value of the first year (baseline) was 8183.2 dolphins, and the predicted abundance of the tenth year was 17961.0 [10171.6, 25750.4]. The fitted trend showed an increase of 1086.4 dolphins per year, suppose a mean annual increase of 9.6% and a 119.5% [24.3%, 214.7%] increase in the 10-year time-series.

Therefore, taking the fitted abundance of the first year of the time-series as the baseline value, the abundance of common dolphins in the study area is not showing a 8% decrease in 10 years (resulting from applying the established threshold of a decrease of ≥30% over a three-generation time) or indeed any decrease at all, and therefore the common dolphin AU could be considered to be at GES for this indicator.

The alternative approach using the variation in the abundance estimates yielded negative trends in 1.7% of cases (none of which were individually significant), while 98.3% yielded a positive trend of which 10.4% were individually significant. The fact that over 95% of simulations showed a positive trend could be viewed as indicating a statistically significant upward trend in abundance.

Table 2
Estimated abundance and density of common dolphins in shelf waters of the Northwest Spanish sub-region. Mean abundance (N) and density (D) over the whole period. Annual abundance (N_{YEAR}) and density (D_{YEAR}). Coefficients of Variation (CV) and 95% Confidence Intervals (95% CI low and high) of the estimates.

	Mean	CV	CI low	CI high
N_{mean}	**12,831**	**0.18**	**9025**	**18,242**
N_{2007}	5928	0.43	2613	13,449
N_{2008}	5533	0.62	1785	17,150
N_{2009}	7305	0.41	3347	15,946
N_{2010}	22,662	0.36	11,410	45,009
N_{2011}	14,437	0.33	7584	27,484
N_{2012}	13,322	0.34	6924	25,631
N_{2013}	17,994	0.53	6643	48,745
N_{2014}	12,981	0.46	5421	31,084
N_{2015}	8475	0.40	3946	18,202
N_{2016}	22,084	0.30	12,335	39,540
D_{mean}	**0.35**	**0.18**	**0.24**	**0.49**
D_{2007}	0.16	0.43	0.07	0.36
D_{2008}	0.15	0.62	0.05	0.46
D_{2009}	0.20	0.41	0.09	0.43
D_{2010}	0.61	0.36	0.31	1.22
D_{2011}	0.39	0.33	0.20	0.74
D_{2012}	0.36	0.34	0.19	0.69
D_{2013}	0.49	0.53	0.18	1.32
D_{2014}	0.35	0.46	0.15	0.84
D_{2015}	0.23	0.40	0.11	0.49
D_{2016}	0.60	0.30	0.33	1.07

3.3. Corrected abundance estimation

The corrected mean abundance using the 0.37 (CV 0.25) correction factor was 4747 (CV 0.19) common dolphins in the study area. This amount corresponds to a density of 0.128 dolphins/km^2 very similar to that obtained during the SCANS-II survey in 2005 (i.e., 0.129 dolphins/

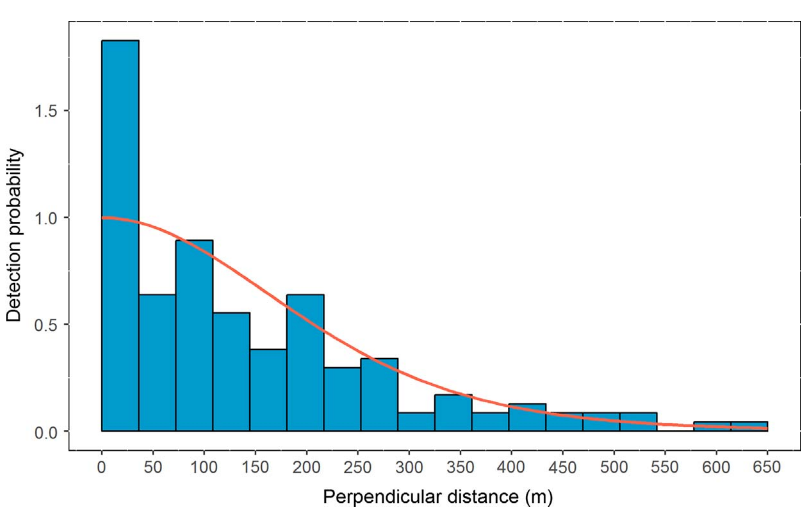

Fig. 4. Half Normal Detection Function for common dolphin sightings truncated at 650 m with Beaufort and log of the group size as covariates.

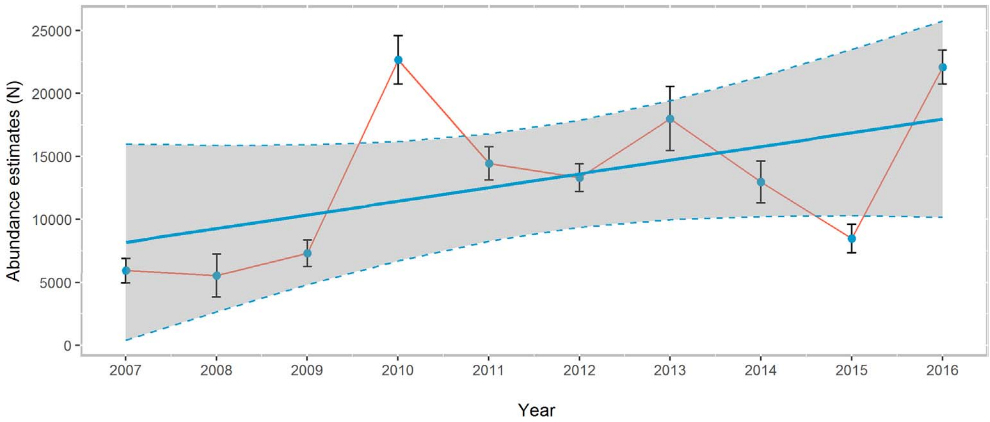

Fig. 5. Time series of common dolphin estimated abundance (blue dots) in the Northwest Spanish sub-region shelf waters, with standard errors (SE bars). Linear regression fitted trend (blue line), with 95% CI (dashed lines and grey area). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 3
Abundance of common dolphins in shelf waters of the Northwest Spanish sub-region corrected for attraction and missed animals on the track-line. Mean abundance (N) over the whole period. Annual abundance (N_{YEAR}). Coefficients of Variation (CV) and 95% Confidence Intervals (95% CI low and high) of the estimates.

	Mean	CV	CI Low	CI High
N_{mean}	**4747**	**0.19**	**3307**	**6816**
N_{2007}	2193	0.43	963	4994
N_{2008}	2047	0.62	659	6360
N_{2009}	2703	0.41	1233	5923
N_{2010}	8385	0.36	4203	16,730
N_{2011}	5342	0.34	2792	10,219
N_{2012}	4929	0.34	2549	9530
N_{2013}	6658	0.53	2451	18,084
N_{2014}	4803	0.47	1999	11,540
N_{2015}	3136	0.40	1454	6762
N_{2016}	8171	0.30	4538	14,712

km^2). The annual abundances, CV and confidence intervals (CI) are shown in Table 3.

The mean abundance corrected using only the proportion of animals missed on the track-line was 22,510 (CV 0.18) common dolphins in the study area with a mean density of 0.608 dolphins/km^2. The annual abundances, CV and confidence intervals (CI) are shown in Table 4.

3.4. Safe mortality limits and the bycatch GES indicator

Following the criteria described (i.e., 1.7% of the best abundance estimate), 81 [95% CI 56, 115] common dolphins could safely be removed from the corrected estimate of 4747 [3307, 6816] individuals in the shelf waters of the Northwest Spanish sub-region, 218 [153, 310] from the 12,831 [9025, 18,242] common dolphins estimated with the conventional distance sampling without bias correction, and 383 [268, 546] from the 22,510 [15,776, 32,120] common dolphins estimated corrected by the animals missed on the track-line.

Table 4
Abundance of common dolphins in shelf waters of the Northwest Spanish sub-region corrected for missed animals on the track-line. Annual abundance (N_{YEAR}). Coefficients of Variation (CV) and 95% Confidence Intervals (95% CI low and high) of the estimates.

	Mean	CV	CI low	CI high
N_{mean}	**22,510**	**0.18**	**15,776**	**32,120**
N_{2007}	10,400	0.43	4578	23,626
N_{2008}	9707	0.62	3129	30,113
N_{2009}	12,816	0.41	5863	28,016
N_{2010}	39,758	0.36	19,985	79,095
N_{2011}	25,328	0.33	13,280	48,305
N_{2012}	23,371	0.34	12,126	45,046
N_{2013}	31,569	0.53	11,642	85,601
N_{2014}	22,774	0.46	9499	54,600
N_{2015}	14,868	0.40	6912	31,980
N_{2016}	38,745	0.30	21,595	69,512

Bycatch information of the study area has been summarised in Table 5. All estimates reported refer to waters within the study area. The Galician fleet, to which the data refer, involves a high number of vessels and a range of different fishing gears (e.g., trawls, purse-seines, trammel-nets, driftnets, longlines, pots). The only data coming from fleet monitoring was evaluated only in pair-trawlers and reported a mean annual bycatch of 394 common dolphins during the period 2000–2001 (Fernández-Contreras et al., 2010). Due to the lack of an adequate monitoring, in other fleets, the best data come from interviews with fishermen. López et al. (2003) calculated an accidental catch in inshore waters of Galicia (encompasses mainly semi-confined areas with bottom depth < 50 m) of about 200 cetaceans and more than 1518 in offshore waters (until continental shelf, 200 m depth approx.), mainly common dolphins. Some years later, Goetz et al. (2014) reported 159 common dolphins bycaught by Galician fisheries (operating in national waters), plus almost 1300 unidentified cetaceans, probably most of them common dolphins. Available data from on-board observation represent a fraction of the fleet (i.e., only pair-trawlers) while those from interviews are believed to underestimate bycatch mortality (López et al., 2003). Therefore, the total bycatch of common dolphins is expected to be higher in our study area, far exceeding the maximum allowed bycatch estimation under any of the abundance scenarios. This indicates that this common dolphin AU would not be at GES for the bycatch indicator, even considering the last scenario where the number of animals that can be safely removed is highest.

4. Discussion

Monitoring the conservation status of marine mammals involves long and costly monitoring schemes. Although monitoring population abundance has been considered the most suitable mechanism for assessing cetacean conservation status (Barlow, 2010), effective conservation also needs knowledge on the threats that affect populations, and their impacts, to ensure that appropriate management measures are put in place.

In this study we have emphasized the importance of obtaining abundance estimates more frequently, reducing the large time gaps between surveys. It allows us to provide the temporal resolution needed to follow populations trends with the temporal resolution needed for the MSFD and to follow changes that may in turn allow us to identify factors modulating the distribution and/or abundance of populations (i.e., shifts in prey availability, fisheries bycatch, etc.; Hammond et al., 2013). We propose to use the platforms provided by the annual acoustic surveys, the objectives of which include the estimation of the pelagic living resources biomass, as a good and cost-effective complement (but not an alternative) to dedicated large-scale cetacean surveys. The use of these existing annual surveys can greatly reduce monitoring costs,

Table 5
Bycatch information available for the study area and indication of the survey method, area/fleet surveyed, affected species, number of reported/estimated bycaught individuals, study period and reference.

Method	Area/fleet	Species	Number	Study period	Reference
On-board monitoring	Galician pair-trawlers	Common dolphin	394	2000–2001	Fernández-Contreras et al. (2010)
Interviews	Galician inshore fleet	Mainly common dolphins	210	1998–2000	López et al. (2003)
Interviews	Galician offshore fleet	Mainly common dolphins	1518	1998–2000	López et al. (2003)
Interviews	Galician fleet	Common dolphin	159	2008–2010	Goetz et al. (2014)
Interviews	Galician fleet	Unidentified (mostly common dolphins)	1300	2008–2010	Goetz et al. (2014)

resulting in a realistic supplement for the assessment of population trends of several cetacean species. The temporal frequency of these surveys and the high survey effort ensure a wide spatiotemporal coverage as is the case in our study area. In addition to estimating the biomass of pelagic fish stocks, which are the main prey of small cetaceans (Santos et al., 2014), in these multidisciplinary surveys samples of the water column are also taken to characterise the plankton community and the oceanography of the region. Therefore, in addition to the application given to the data in this study, all this information provide an opportunity to be used to calibrate ecosystem models which will allow a more effective management of cetacean populations, by evaluating the implication at population level of bycatch mortality caused by the fishing fleet that is also targeting their prey species, and how fluctuations in prey abundance, cetaceans abundance and fishing effort impact each other (e.g., Lassalle et al., 2012; Saavedra et al., 2015). Moreover, this information can help build habitat models with *in situ* predictive variables (e.g., prey concentration, physicochemical parameters) at much finer temporal and spatial scale than those normally available (e.g., Becker et al., 2008; Torres et al., 2008).

We have shown that to obtain reliable abundance estimates when performing a single-platform survey, a number of assumptions must be met and/or violations corrected for. One major assumption is that all animals on the track-line are detected, which is unlikely to be fully met. Species-specific traits condition the probability of being detected (e.g., size, time spent under the surface, surface behaviour), but we argue that these do not change over time. The methodology used, the type of vessel (e.g., height of the observer platform) and observer experience can also affect abundance estimators, but if the methodology and vessel are kept constant, and, as far as possible, the level of experience of the observers, the probability of detection on the track-line, g_0, will be relatively stable over the time-series. Another requirement for unbiased line-transect estimates of abundance is that the cetacean group should not move in response to the vessel before it is sighted. If these assumptions are indeed violated, and no corrections are made, the estimated density and abundance will be biased to some degree (Buckland et al., 2001; Mullin and Fulling, 2003). As with the probability of detection, if the methodology and platform of observation are maintained, there is no reason to think that the reaction that dolphins may show would vary over time. All these conditions, except the height of the sampling platforms, have remained constant in our study. However, preliminary analyses did not detect significant differences between the two vessels used. Therefore, we again argue that we are justified to consider that the bias committed by not applying these corrections will remain relatively constant over time and will not alter the trends in the abundance estimates obtained with a conventional distance sampling analysis of data collected from single-platform surveys.

Although we are aware that the assessment is not performed at the level of the whole common dolphin population (a single population across the NE Atlantic; Natoli et al., 2006), at the level of the Spanish AU we can conclude, using the population abundance indicator, that the common dolphins present in the shelf waters of the Northwest Spanish sub-region can be considered to be at GES. This does not mean that the population as a whole fulfils this same criterion and, therefore, our evaluation cannot be extrapolated to other areas or to the entire population. However, it is important to note that the increases in abundance observed in some years in our study would be demographically impossible in an isolated population given expected maximum birth rate (Reilly and Barlow, 1986) so we may also corroborate that this common dolphin UA is not a closed population and exchange with areas outside the study area is likely to have happened. Thus the observed abundance increase in this sub-region may be a reflection of a trend at population level or simply be the result of distribution changes due to short-term movements in response to changes in prey abundance, environmental conditions, oceanographic events, or threats operating in other areas where the population is distributed (Certain et al., 2008; Hammond et al., 2013; Johnston et al., 2005; MacLeod et al., 2009).

Our results indicate that common dolphins in the study area seem to meet the requirements for GES but the assessment of the GES at the level of the NE Atlantic population is beyond the scope of this study. Although the MSFD specifically requests that MS use the regional structures already in place (e.g., Regional Sea Conventions) to coordinate the implementation of the MSFD at regional level, MS are ultimately responsible for the assessment of the status of their waters (Santos and Pierce, 2015). MS can therefore assess if cetaceans present in their waters show positive or negative trends. If through these MS assessments a generalised decrease in abundance is detected, it can be inferred that the population as a whole not at GES; if there is a generalised increase, GES may be inferred (at least in relation to abundance). However, if different and indeed opposite trends are observed in the assessments by different MS, it will be more difficult to reach a conclusion about GES. The existence of different trends in different subregions could be due to other factors not detectable with this type of sampling, justifying further research.

Ecological tracer analyses (stable isotope, contaminants, etc.) suggest that common dolphins in shelf habitats might be distinguished from those of slope and oceanic habitats (Chouvelon et al., 2012; Das et al., 2003; Méndez-Fernandez et al., 2012). However, there is consensus that a full assessment of both abundance trends and the impact of bycatch should be carried out at population level (i.e., at the NE Atlantic level in the case of the common dolphin). Therefore, the use of small-scale surveys that we propose does not imply that larger dedicated surveys such as SCANS should not take place. On the contrary, these large dedicated surveys, following a standard methodology, still represent the best way to study global trends in abundance and population distribution. Ideally, the frequency should be increased over time to make it coincide, for example, with the cycles of the MSFD as suggested by the ICES WGMME (ICES, 2016). Moreover, these large-scale and double-platform surveys not only provide relative trends but also absolute abundances. Absolute abundances are needed for a variety of reasons, as for example to improve the accuracy of multispecies or ecosystem models that include cetaceans (e.g., Lassalle et al., 2012; Saavedra et al., 2015; Tjelmeland and Lindstrøm, 2005; Torres et al., 2013) and/or to calculate the number of cetaceans that could be safely removed from a population due to bycatch (or any other anthropogenic threat) as we have done in the current paper.

In the present study we have taken advantage of the existence of a dedicated double-platform survey (SCANS-II) carried out in the same study area, with a methodology very similar to that used in our surveys, to correct the bias of our estimates. Ideally, we should not use correction indices coming from different surveys (Buckland et al., 2004; Cañadas et al., 2004) but we have considered that the boats used are similar, as well as the methodology and the experience of the observers. A possible source of error due to this assumption may occur because of the lower height of the observation platforms used in SCANS-II. The relative low g_0 of SCANS-II may produce an overestimation of abundance when applied to the PELACUS data, with higher platforms and probably with higher probability of detection. On the other hand, g_0 may decline strongly with Beaufort (Barlow, 2015). In our analysis we have included data collected under Beaufort up to 5. In addition, due to the time of the year in which the surveys are carried out, the average Beaufort has been higher in PELACUS than in SCANS-II. Contrary to the previous case, this may cause an underestimation in the abundance when applying the SCANS-II g_0 to the PELACUS data. With regard to attraction/evasion movements, both the species and the study area are the same and therefore behaviour towards the observation platform should be very similar. Although it is true that some authors have considered that the acoustic signal used for the estimation of the pelagic biomass can affect the behaviour of cetaceans, causing avoidance, after years of experience and several experiments (i.e., long time series of surveys in the Eastern Pacific and Northeast Atlantic oceans) some of the authors of the present study have noticed that these emissions could affect the detection of certain species but not common dolphins. In addition, common dolphin is not an evasive species and has been identified on several occasions (direct observation or underwater cameras) playing a few centimetres from the acoustic transducers, which indicates that common dolphins experience no discomfort or minimal discomfort, and in no case sufficient to cause evasion at big distances since the acoustic spectrum is directional to the bottom and not horizontally dislodged (P. Carrera, 2016, pers. comm.). Although the possible errors introduced when applying this methodology should be further study in the future, we assumed that the data and knowledge arising from SCANS-II can be used to make corrections in the estimates obtained from our surveys.

However, SCANS-II data for the common dolphin seem to overestimate the attraction toward the observation platform (ICES, 2017) and therefore we used three different scenarios for calculating the maximum bycaught level without putting at risk the viability of the population. Obtaining reliable estimates of the population still remains difficult with the mark and recapture methodology using dual platform, and by extension, also using a single platform. Another source of uncertainty associated with this indicator is the lack of a specific threshold for the common dolphin (the figure calculated for porpoise is tentatively used for all cetaceans, but a specie-specific threshold should be calculated; ASCOBANS, 2015). In relation to data available for the assessment of this indicator, these are both limited and flawed, derived from on-board observation for one fleet component (Fernández-Contreras et al., 2010) or from self-selecting respondents to interview surveys (Goetz et al., 2014; López et al., 2003). However, the estimates are more likely to be underestimates than overestimates, and the high proportion of stranded common dolphins showing signs of bycatch (e.g., López et al., 2002; Read et al., 2009; Silva and Sequeira, 2003) is consistent with this inference. Due to the uncertainty of the parameters used to calculate threshold values in the present study, we do not know by how much the catches exceed the proposed limits, but the data seem to indicate that bycaught numbers exceed by far these thresholds. In addition, other threats that affect the common dolphin population may aggravate the bycatch effect (see Murphy et al., 2013), so adjustment of bycatch thresholds maybe needed to account for this. Consequently, while further, more comprehensive and more systematic monitoring of cetacean bycatch in NW Spanish fisheries is urgently needed, our tentative conclusion is that, according to the bycatch indicator, common dolphins in this sub-region are not at GES.

The assessment of whether the common dolphins in the Northwest Spanish sub-region are at GES gives contradictory results depending on the indicator used. Clearly this could be explained because we are working with an open population and migration movements from north to south could be, and are probably, taking place. This may be due to short-term movements because the study area may represent a more favourable habitat, or because the NE Atlantic short-beaked common dolphin population is increasing and as a consequence its presence in Spanish waters is also increasing. If this was the case, it could indicate that mortality at the population level is low, even if mortality is above threshold values at a local level in Galician waters. Even if this is the case, applying the precautionary principle and following the requirement of the MSFD that MS must individually assess the trends of abundance in their waters and ensure that the mortality (or other threats) does not exceed the limits established, the part of the population that inhabits in northern Spanish waters cannot be considered at GES in relation to the mortality indicator.

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References

ASCOBANS, 2015. Workshop on the Further Development of Management Procedures for Defining the Threshold of "Unacceptable Interactions" Part I: Developing a Shared Understanding on the Use of Thresholds/Environmental Limits. London, United Kingdom.

ASCOBANS, 2000. Resolution No. 3 Incidental Take of Small Cetaceans. 3rd Session of the Meeting of Parties. Bristol.

Barlow, J., 2015. Inferring trackline detection probabilities, g(0), for cetaceans from apparent densities in different survey conditions. Mar. Mammal Sci. 31, 923–943. http://dx.doi.org/10.1111/mms.12205.

Barlow, J., 2010. Cetacean Abundance in the California Current Estimated from a 2008 Ship-based Line-transect Survey, NOAA Technical Memorandum. NOAA Southwest Fisheries Science Center, La Jolla, CA.

Becker, E.A., Forney, K.A., Ferguson, M.C., Barlow, J., Redfern, J.V., 2008. Predicting Modeling of Cetacean Density in the California Current ecosystem Based on Summer/Fall Ship Survey in 1991–2008.

Buckland, S.T., Anderson, D.R., Burnham, Kenneth Paul, Laake, J.L., Borchers, D.L., Leonard, T., 2001. Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford Univ. Press, Oxford, England.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., 1993. Distance Sampling: Estimating Abundance of Biological Populations. Chapman and Hall, London.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L., 2004. Advanced Distance Sampling. Oxford University Press, Oxford.

Cañadas, A., Desportes, G., Borchers, D.L., 2004. The estimation of the detection function and g(0) for short-beaked common dolphins (*Delphinus delphis*), using double-platform data collected during the NASS-95 Faroese survey. J. Cetacean Res. Manage. 6, 191–198.

Certain, G., Ridoux, V., van Canneyt, O., Bretagnolle, V., 2008. Delphinid spatial distribution and abundance estimates over the shelf of the Bay of Biscay. ICES J. Mar. Sci. 65, 656–666.

Chouvelon, T., Spitz, J., Caurant, F., Mèndez-Fernandez, P., Chappuis, A., Laugier, F., Le Goff, E., Bustamante, P., 2012. Revisiting the use of δ 15N in meso-scale studies of marine food webs by considering spatio-temporal variations in stable isotopic signatures – the case of an open ecosystem: the Bay of Biscay (North-East Atlantic). Prog. Oceanogr. 101, 92–105. http://dx.doi.org/10.1016/j.pocean.2012.01.004.

Danil, K., Chivers, S.J., 2007. Growth and reproduction of female short-beaked common dolphins, *Delphinus delphis*, in the eastern tropical Pacific. Can. J. Zool. 85, 108–121. http://dx.doi.org/10.1139/z06-188.

Das, K., Beans, C., Holsbeek, L., Mauger, G., Berrow, S.D., Rogan, E., Bouquegneau, J.M., 2003. Marine mammals from Northeast Atlantic: relationship between their trophic status as determined by δ13C and δ15N measurements and their trace metal

concentrations. Mar. Environ. Res. 56, 349–365. http://dx.doi.org/10.1016/S0141-1136(02)00308-2.
Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 Establishing a Framework for Community Action in the Field of Marine Environmental Policy (Marine Strategy Framework Directive). OJ L 164/19.
Fernández-Contreras, M.M., Cardona, L., Lockyer, C.H., Aguilar, A., 2010. Incidental bycatch of short-beaked common dolphins (*Delphinus delphis*) by pairtrawlers off northwestern Spain. ICES J. Mar. Sci. 67, 1732–1738. http://dx.doi.org/10.1093/icesjms/fsq077.
Goetz, S., Read, F.L., Santos, M.B., Pita, C., Pierce, G.J., 2014. Cetacean–fishery interactions in Galicia (NW Spain): results and management implications of a face-to-face interview survey of local fishers. ICES J. Mar. Sci. 71, 604–617. http://dx.doi.org/10.1093/icesjms/fst149.
Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., Burt, L., Cañadas, A., Desportes, G., Donovan, G.P., Gilles, A., Gillespie, D., et al., 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. Biol. Conserv. 164, 107–122.
Heinemann, D., 1981. A rangefinder for pelagic bird censusing. J. Wildl. Manage. 45, 489–493.
ICES, 2017. Report of the Working Group on Marine Mammal Ecology (WGMME). 6–9 February, 2017. Aberdeen, Scothland, UK.
ICES, 2016. Report of the Working Group on Marine Mammal Ecology (WGMME). 8–11 February 2016, Madrid, Spain. ICES CM 2016/ACOM:26. 117 pp.
ICES, 2014. Report of the Working Group on Marine Mammal Ecology (WGMME). 10–13 March, 2014. Woods Hole, Massachusetts, USA. ICES CM 2014/ACOM:27. 234 pp.
ICG COBAM, 2012. MSFD Advice Manual and Background Document on Biodiversity. A Living Document - Version 3.2 of 5 March 2012. Approaches to Determining Good Environmental Status, Setting of Environmental Targets and Selecting Indicators for Marine Strategy Framework Direc.
IWC, 2000. Report of the IWC-ASCOBANS working group on harbour porpoises. J. Cetacean Res. Manage. 2, 297–305.
Johnston, D.W., Westgate, A.J., Read, A.J., 2005. Effects of fine-scale oceanographic features on the distribution and movements of harbour porpoises *Phocoena phocoena* in Bay of Fundy. Mar. Ecol. Prog. Ser. 295, 279–293.
Lassalle, G., Gascuel, D., Le Loc'h, F., Lobry, J., Pierce, G.J., Ridoux, V., Santos, M.B., Spitz, J., Niquil, N., 2012. An ecosystem approach for the assessment of fisheries impacts on marine top predators: the Bay of Biscay case study. ICES J. Mar. Sci. 69, 380–388. http://dx.doi.org/10.1093/icesjms/fst048.
Lawrence Miller, D., 2016. Distance: distance sampling detection function and abundance estimation. R Package Version (9), 6.
López, A., Pierce, G.J., Santos, M.B., Gracia, J., Guerra, A., 2003. Fishery by-catches of marine mammals in Galician waters: results from on-board observations and an interview survey of fishermen. Biol. Conserv. 111, 25–40. http://dx.doi.org/10.1016/S0006-3207(02)00244-6.
López, A., Santos, M.B., Pierce, G.J., González, A.F., Valeiras, X., Guerra, A., 2002. Trends in strandings and by-catch of marine mammals in north-west Spain during the 1990s. J. Mar. Biol. Assoc. United Kingdom 82, 513–521.
MacLeod, C.D., Brereton, T., Martin, C., 2009. Changes in the occurrence of common dolphins, striped dolphins and harbour porpoises in the English Channel and Bay of Biscay. J. Mar. Biol. Assoc. United Kingdom 89, 1059–1065.
Marques, F.F.C., Buckland, S.T., 2003. Incorporating covariates into standard line transect analyses. Biometrics 59, 924–935. http://dx.doi.org/10.1111/j.0006-341X.2003.00107.x.
Méndez-Fernandez, P., Bustamante, P., Bode, A., Chouvelon, T., Ferreira, M., López, A., Pierce, G.J., Santos, M.B., Spitz, J., Vingada, J.V., Caurant, F., 2012. Foraging ecology of five toothed whale species in the Northwest Iberian Peninsula, inferred using carbon and nitrogen isotope ratios. J. Exp. Mar. Bio. Ecol. 413, 150–158. http://dx.doi.org/10.1016/j.jembe.2011.12.007.
Mullin, K.D., Fulling, G.L., 2003. Abundance of cetaceans in the southern U. S. North Atlantic Ocean during summer 1998. Fish. Bull. 101, 603–613.
Murphy, S., Dabin, W., Ridoux, V., Morizur, Y., Larsen, F., Rogan, E., 2007. Estimation of Rmax for the common dolphin in the Northeast Atlantic, Report to the European Commission, NECESSITY Contract 501605 Periodic Activity Report No. 2–Annex 8.4. Ijmuiden, The Netherlands: IMARES Wageningen UR, Institute for Marine Resources & Ecosystem Studies.
Murphy, S., Pinn, E.H., Jepson, P.D., 2013. The short-beaked common dolphin (*Delphinus delphis*) in the North-East Atlantic: distribution, ecology, management and conservation status. Oceanogr. Mar. Biol. An Annu. Rev. 51, 193–280.
Natoli, A., Cañadas, A., Peddemors, V., Aguilar, A., Vaquero, C., Fernández-Piqueras, P., Hoelzel, A.R., 2006. Phylogeography and alpha taxonomy of the common dolphin (*Delphinus* sp.). J. Evol. Biol. 19, 943–954. http://dx.doi.org/10.1111/j.1420-9101.2005.01033.x.
Pierce, G.J., Caldas, M., Cedeira, J., Santos, M.B., Llavona, Á., Covelo, P., Martínez, G., Torres, J., Sacau, M., López, A., 2010. Trends in cetacean sights along the Galician coast, north-west Spain, 2003–2007, and inferences about cetacean habitat preferences. J. Mar. Biol. Assoc. United Kingdom 90, 1547–1560.
R Core Team, 2016. R: A Language and Environment for Statistical Computing.
Read, F.L., Santos, B., González, Á.F., Martínez-Cedeira, J., López, A., Pierce, G.J., 2009. Common Dolphin (*Delphinus delphis*) in Galicia, NW Spain: Distribution, Abundance, Life History and Conservation. Reports Int. Whal. Comm.
Reilly, S.B., Barlow, J., 1986. Rates of increase in dolphin population size. Fish. Bull. 84, 527–533.
Saavedra, C., Cerviño, S., Northridge, S., 2015. Improving Stock Assessment and Managing Bycatch Rates using a Multispecies Approach. A Case of Study of the European Hake, Common and Bottlenose Dolphins in Atlantic Waters of the Iberian Peninsula. Copenhagen, Denmark.
Santos, M.B., Pierce, G.J., 2015. Marine mammals and good environmental status: science, policy and society; challenges and opportunities. Hydrobiologia 750, 13–41. http://dx.doi.org/10.1007/s10750-014-2164-2.
Santos, M.B., Saavedra, C., Pierce, G.J., 2014. Quantifying the predation on sardine and hake by cetaceans in the Atlantic waters of the Iberian peninsula. Deep Sea Res. Part II Top. Stud. Oceanogr. 106, 232–244. http://dx.doi.org/10.1016/j.dsr2.2013.09.040.
SCANS-II, 2006. Design-based abundance estimates from SCANS-II. In: Burt, M., Borchers, D., Paxton, C. (Eds.), Small Cetaceans in the European Atlantic and North Sea. Final Report to the European Commission under project LIFE04NAT/GB/000245, St. Andrews, p. 31.
Silva, M.a., Sequeira, M., 2003. Patterns in the mortality of common dolphins (*Delphinus delphis*) on the Portuguese coast, using stranding records, 1975–1998. Aquat. Mamm. 29, 88–98. http://dx.doi.org/10.1578/016754203101023924.
Taylor, B.L., Martinez, M., Gerrodette, T., Barlow, J., Hrovat, Y.N., 2007. Lessons from monitoring trends in abundance of marine mammals. Mar. Mammal Sci. 23, 157–175. http://dx.doi.org/10.1111/j.1748-7692.2006.00092.x.
Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R.B., Marques, T.A., Burnham, K.P., 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. J. Appl. Ecol. 47, 5–14.
Tjelmeland, S., Lindstrøm, U., 2005. An ecosystem element added to the assessment of Norwegian spring-spawning herring: implementing predation by minke whales. ICES J. Mar. Sci. 62, 285–294. http://dx.doi.org/10.1016/j.icesjms.2004.12.011.
Torres, L.G., Read, A.J., Halpin, P., 2008. Fine-scale habitat modeling of a top marine predator: do prey data improve predictive capacity. Ecol. Appl. 18, 1702–1717.
Torres, M.Á., Coll, M., Heymans, J.J., Christensen, V., Sobrino, I., 2013. Food-web structure of and fishing impacts on the Gulf of Cadiz ecosystem (South-western Spain). Ecol. Modell. 265, 26–44. http://dx.doi.org/10.1016/j.ecolmodel.2013.05.019.