Small cetacean bycatch as estimated from stranding schemes: the common dolphin case in the northeast Atlantic

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Abstract

Death in fishing gear of non-target species (called 'bycatch') is a major concern for marine wildlife, and mostly worrying for long-lived species like cetaceans, considering their demographic characteristics (slow population growth rates and low fecundity). In European waters, cetaceans are highly impacted by this phenomenon. Under the Common Fishery Policy, the EC 812/2004 regulation constitutes a legal frame for bycatch monitoring on 5 to 10% of fishing vessels > 15 m. The aim of this work was to compare parameters and bycatch estimates of common dolphins (*Delphinus delphis*) provided by observer programmes in France and UK national reports and those inferred from stranding data, through two approaches. Bycatch was estimated from stranding data, first by correcting effectives from drift conditions (using a drift prediction model) and then by estimating the probability of being buoyant. Observer programmes on fishing vessels allowed us to identify the specificity of the interaction between common dolphins and fishing gear, and provided low estimates of annual bycaught animals (around 550 animals.year^{-1}). However, observer programmes are hindered by logistical and administrative constraints, and the sampling scheme seems to be poorly designed for the detection of marine mammal bycatches. The analyses of strandings by considering drift conditions highlighted areas with high levels of interactions between common dolphins and fisheries. Since 1997, the highest densities of bycaught dolphins at sea were located in the southern part of the continental shelf and slope of the Bay of Biscay. Bycatch numbers inferred from strandings suggested very high levels, ranging from 3,650 dolphins.year^{-1} [2,250-7,000] to 4,700 [3,850-5,750] dolphins.year^{-1}, depending on methodological choices. The main advantage of stranding data is its large spatial scale, cutting across administrative boundaries. Diverging estimates between observer programmes and stranding interpretation can set very different management consequences: observer programmes suggest a sustainable situation for common dolphins, whereas estimates based on strandings highlight a very worrying and unsustainable process.

Keywords: bycatch, drift modelling, common dolphins, observer programmes, CE 812/2004 regulation, Marine Strategy Framework Directive.

1. INTRODUCTION

The catch of non-target or non-commercial species in fishing gear, or bycatch, affects most marine species (Davies et al., 2009; Hall, 1996; Hall et al., 2000; Lewison et al., 2004; Peckham et al., 2008; Read, 2008; Reeves et al., 2013; Soykan et al., 2008; Thompson et al., 2013). Hall (1996) defined bycatch as: 'the portion of the capture that is discarded at sea dead (or injured to an extent that death is the most likely outcome) because it has little or no economic value or because its retention is prohibited by law' (Hall, 1996). The impact of bycatch on marine mega-vertebrates can be direct, such as additional mortality at unsustainable levels for populations, or indirect including depletion of prey, habitat destruction, disturbance of physical and chemical processes (Hall et al., 2000; Kumar and Deepthi, 2006; Read, 2008). Bycatch is a potent threat for long-lived species with slow population growth rates, low fecundity or low survival to adulthood such as seabirds, sharks, sea turtles and marine mammals (hereafter defined as mega-vertebrates) (Cox et al., 2007; Hall et al., 2000; Lewison et al., 2004; Mannocci et al., 2012; Peckham et al., 2008; Read, 2008; Soykan et al., 2008). Uncertainties around the true magnitude of bycatch delays management decision-making and their reduction is therefore a challenge for the effective conservation of mega-vertebrate populations (Lewison et al., 2004; Thompson et al., 2013). Recent studies on the effects of interactions between fisheries and mega-vertebrate demography or population genetics revealed pessimistic conservation scenarios (Mannocci et al., 2012; Mendez et al., 2010). In fact, most fishing gear, such as pelagic or bottom trawl nets, bottom-set gillnets or longlines, contribute to this worldwide threat to large marine vertebrates (Adimey et al., 2014; Davies et al., 2009; Gilman et al., 2005; Lewison et al., 2004; Lewison and Crowder, 2003; Read et al., 2006). Bycatch has been identified as a conservation issue since the 1970s; although it is probably one of the most important man-induced threats to marine mega-vertebrates, it still remains largely unresolved (Cox et al., 2007; Davies et al., 2009; Hall et al., 2000; Hamel et al., 2009; Lewison and Crowder, 2003; Peckham et al., 2008; Read et al., 2006).

Bycatch issues have long been ignored or under-documented, mostly because the process remains barely visible as it takes place far from ports and fish markets (Hall et al., 2000). Fisheries management has focused for decades on commercial species only. Historically, rising awareness of the detrimental effects of bycatch on species persistence and ecosystems functioning has occurred through charismatic species (marine mammals, sea turtles, etc.).

Because bycatch occurs far from the public eye and affects species for which public concerns can quickly become salient, obtaining reliable estimates of its magnitude at a population scale is a difficult endeavour (Read, 2008).

Implemented in 1983 in European waters, the Common Fishery Policy (CFP) is the marine translation of the Common Agricultural Policy (CAP). Its goals are manifold including (i) setting total allowable catches (TACs) for commercial squid, fish and shellfish species; (ii) regulating the market in order to ensure its stability, sustainable prices for fishermen and regular supply to consumers; and (iii) estimating and reducing the total incidence of non-target species bycatch. Since 1992, the Habitats Directive required bycatch monitoring by European Union (EU) Member States. In line with commitments towards individual protected species, incidental catches are addressed under Article 12(4) which establishes an obligation to address, inter alia, by-catches: 'Member States shall establish a system to monitor the incidental capture and killing of the animal species listed in Annex IV(a). In light of the information gathered, Member States shall take further research or conservation measures as required to ensure that incidental capture and killing does not have a significant impact on the species concerned'.

The latter goal is now specifically implemented by European Council (EC) Regulation n°812/2004. The two main actions of EC 812/2004 are the coordinated monitoring of cetacean bycatch through compulsory on-board observer programmes for selected fisheries and the mandatory use of acoustic deterrent devices ('pingers') in other fisheries. Member States are required to design and implement monitoring schemes for incidental catches of cetaceans. Programmes of observers on fishing vessels with an overall length of at least 15 meters or over constitutes a legal frame for bycatch monitoring (Table 1).

Two main biases were identified in these observer programmes: (i) the deployment effect, or non-random assignment of observers to vessels and ports due to the fact that accepting an observer on board is at the vessel master's discretion, and (ii) the observer effect, i.e. a change in fishing practices when an observer is present (Amandè et al., 2012; Benoît and Allard, 2009; Faunce and Barbeaux, 2011; Stratoudakis et al., 1998). Additionally, EC 812/2004 regulation, by selecting focus fisheries for the implementation of on-board monitoring programmes and excluding others, precludes any possibility of providing a synoptic view of cetacean bycatch in EU fisheries. The growing awareness of insufficient spatial, temporal and métiers coverage by EC 812/2004 observer surveys, and the incidence of the deployment and observer effects, has encouraged the development of alternative bycatch estimates from data sources that would be independent of the industry and of the regulation and could document the total extent of bycatch in fisheries.

Stranding records are an important source of information on marine mega-vertebrates, and can provide critical information to estimate a minimum level of bycatch across fisheries (Adimey et al., 2014; Leeney et al., 2008; Lopez et al., 2003; Silva and Sequeira, 2003). Because of a lack of control over the stranding process, strandings have long been underused as a source of quantitative indicators (Wiese and Elmslie, 2006). However, through the understanding of the small cetacean carcass drifting and stranding processes (eq. 1), the relationships between stranding records and cetacean relative abundance and mortality can be elucidated (Peltier et al., 2014):

$$N_{stranding} = f(\,Abundance, mortality, buoyancy, drift, discovery) \text{ (eq. 1)}$$

where $N_{stranding}$ is the observed number of stranded dead cetaceans; $Abundance$ is the total population size, $mortality$ is the mortality rate (including both natural and anthropogenic sources); $buoyancy$ is the probability of a dead animal to float; $drift$ is the probability of a floating dead animal to drift to a coast and get stranded; and $discovery$ is the probability of a stranded carcass to be discovered and reported.

Recent studies have aimed at improving the representativeness of strandings, by accounting for drift conditions and observation pressure (Authier et al., 2014; Epperly et al., 1996; Hart et al., 2006; Koch et al., 2013; Peltier et al., 2014, 2013, 2012), and provided relevant indicators on mega-vertebrate populations. The proportion of animals dying at sea found stranded was recently investigated by different studies and estimated at 0.02 (range: 0-0.06) in the Gulf of Mexico (Williams et al., 2011), 0.105 (CI 95%[0.05;0.18]) in Brazilian fisheries targeting white croakers (Prado et al., 2013) and 0.129 (CI 95% [0.047; 0.206]) along the French coast of the Bay of Biscay (Peltier et al., 2012). Here, we propose to estimate levels of dolphin bycatch in the northeast Atlantic from stranding records. In the northeast Atlantic, the short-beaked common dolphin (*Delphinus delphis*) is one of the most abundant species (Certain et al., 2011; Hammond et al., 2013, 2002; Kiszka et al., 2007; McLeod et al., 2003; Murphy et al., 2013), yet also one of the most exposed to being bycaught in fisheries (De Boer et al., 2008; Fernández-Contreras et al., 2010; Kirkwood et al., 1997; Leeney et al., 2008; de Boer, 2012; Peltier et al., 2014; Silva and Sequeira, 2003). In the Bay of Biscay and the English Channel, common dolphin bycatch are mostly reported in pelagic fisheries targeting sea-bass (*Dicentrarchus labrax*) or albacore tuna (*Thunnus alalunga*), as shown by compulsory observer programmes conducted under EC 812/2004 (Morizur et al., 1999; Rogan and Mackey, 2007; Spitz et al., 2013).

The aims of this work were: (1) to develop and adapt cartographic indicators inferred from strandings to inform common dolphin mortality in fisheries of the Bay of Biscay and the western Channel, (2) to estimate overall bycatch mortality of common dolphins from stranding recorded along French and British coasts of the Bay of Biscay and western Channel using two different approaches, and (3) to compare these estimates with figures obtained by on-board observer monitoring programmes conducted by France and United-Kingdom under regulation EC 812/2004.

2. MATERIAL AND METHODS

2.1- General considerations

Stranding data were selected from the French and UK stranding databases for the period 1990-2009. Only common dolphins found with lesions diagnostic of bycatch in fishing gear were considered (Kuiken and Hartmann, 1993) as well as those stranded during multiple stranding events, or 'unusual mortality events' related to bycatches in fisheries. Multiple stranding events were defined as high numbers of strandings occurring in restricted area with a common cause of death. The threshold was defined at 30 cetaceans over 10 consecutive days recorded along a maximal distance of 200 km in the Bay of Biscay, and 10 individuals.10 $days^{-1}$.200 km^{-1} along the coast of the western Channel (Peltier et al., 2014). Along the UK and French coasts, these events are related to bycatch in pair-trawl fisheries, with a high proportion of carcasses showing typical bycatch marks (Leeney et al., 2008; Morizur et al., 1999).

The study area was located in the northeast Atlantic from 43.3-51.3°N, encompassing neritic and oceanic waters of the Bay of Biscay (south of 48°N) and the western Channel and Celtic Seas (north of 48°N) bordering the coasts of France and southern Great Britain (Figure 1). Previous studies on the development of cartographic indicators of common dolphin mortality in the Bay of Biscay and western Channel showed that the same stranding profiles were recorded in these areas (Wiese and Elmslie, 2006). The eastern Channel was excluded from the study area as very low numbers of common dolphin strandings were reported from this area.

Both deterministic and stochastic approaches were developed to estimate bycatch levels from stranding data (Figure 2). In both approaches, drift conditions that led to stranding events were explicitly considered.

The first approach is geographically explicit and is based on drift back-calculations (thereafter named ‘reverse drift modelling’) in order to reconstruct the trajectory of every stranded common dolphin from its stranding location to its likely area of death at sea. The number of dead stranded animals in each cell is then corrected by the cell-specific probability of being stranded (Peltier et al., 2013). The left-hand panel of Figure 2 illustrates the reverse drift approach. The study area is sub-divided in 89 cells of size 0.75° x 0.75°. The drift trajectory of a dolphin dying in each cell centroid is simulated with a physical drift model (MOTHY, developed by *Météo-France*; (Daniel, 2004)). After 30 days, whether the carcass was predicted to reach a coast within the study area was recorded. Cells in which a dead dolphin would strand (as predicted by MOTHY) are highlighted in green (Figure 2, lower left panel). The probability of a dolphin dying in a given cell to strand is the long-term frequency over the study period with which it was predicted to strand.

The second approach used probabilistic modelling to quantify different sources of uncertainties intrinsic to the stranding process (eq. 1) and to obtain uncertainty measures associated with bycatch levels. This approach is not geographically explicit and therefore does not allow at-sea mortality maps to be inferred. It relies on direct drift modelling, and is thereafter referred to as ‘direct drift modelling’ (Figure 2). The right panel of Figure 2 illustrates the direct drift approach: the study area is sub-divided into 0.75° x 0.75° cells. The drift trajectory of a dolphin dying in each cell centroid (black dots on the upper right panel of Figure 2) was simulated with the drift model MOTHY. After 30 days, the total number of dolphins predicted to strand over the study area was recorded irrespective of the cell where they originated (green triangles on the lower right panel). This number was used as data to model p_{jt}, the stranding probability of a floating dead dolphin in the study area to strand in month j and year t on the coastline of the study area. The probability is different from the previous one as it is not spatially explicit.

In the reverse drift approach, stranding probability is a long-term frequency calculated over the study period at the cell level. In the direct drift approach, stranding probability is related to the current total number of predicted strandings over the whole study area and does not take into account where a predicted-to-strand dolphin came from within the study area.

2.2- Cartographic indicators of common dolphin bycatch

Cartographic indicators were constructed following previously described methods (Peltier and Ridoux, 2015), but for the present analyses, only data on multiple stranding events and carcasses found outside these events but showing bycatch marks were used. Relative density maps of dead common dolphins were inferred from stranded animals using MOTHY, which predicts the drift of floating objects under the influence of tides and wind. Through reverse drift modelling, observed stranded dolphins were mapped back to their likely location of death. The probability of stranding for an animal bycaught in each cell $p_{stranding}$, was estimated during computer experiments with MOTHY for every period of ten days between 1990 and 2009 (Peltier et al., 2013). The drift of uniformly distributed theoretical small cetaceans was predicted for 30 days in order to estimate $p_{stranding}$ for each cell at sea. The number of observed dolphins in each cell was corrected (divided) by $p_{stranding}$ in order to estimate the total number of bycaught dolphins (Authier et al., 2014), irrespective of drift conditions. In order to avoid major uncertainty around extrapolations made from rare events, cells with stranding probability $p_{stranding} < 0.1$ were removed.

Bathymetric maps were plotted with the R package *marmap* (Pante and Simon-Bouhet, 2013).

2.3- Estimating bycatch numbers based on strandings

2.3.1- Estimations based on reverse drift modelling

Maps of bycaught common dolphins inferred from strandings show the spatial distribution of bycaught animals across the study area. The sum of dead dolphins in each cell provides an estimate of dolphin mortality in fishing gear every year, uncorrected for the proportion of dead animals that sink to the sea floor and are therefore lost to the stranding process.

To estimate the proportion of floating and sinking bycaught dolphins, an experiment was carried out between 2004 and 2009 with tagged carcasses (Peltier et al., 2012). A total number of 100 dolphins that were caught in fishing vessels were marked and their carcasses released back into the Bay of Biscay at a known time and place. Their stranding location was then predicted by using MOTHY. Of the 100 dead dolphins dropped at sea, 62 were predicted to strand, and among those only 8 carcasses were subsequently reported. The number found can be viewed as the result of a binomial process:

$$n_{found} \sim Binomial\ (N_{predicted}, p_{buoyant} \times p_{discovery})\ \text{(eq.2)}$$

where $p_{discovery}$ is the probability to discover a stranded dolphin in the study area and $p_{buoyant}$ is the probability that a dead bycaught dolphin floats rather than sinks to the seabed. An informative prior was elicited for $p_{discovery}$: given the stability of the French National Stranding Network since 1990 (Authier et al., 2014), $p_{discovery}$was elicited to have a 95% credible interval of 0.800-0.975 with 0.95 probability using the software Parameter Solver v3.0 (Cook et al., 2013). The resulting beta distribution is $p_{discovery} \sim Beta(36{,}3.71)$.

To improve the estimation of $P_{buoyant}$, we used the 'add two successes and two failures' rule (Agresti and Coull, 1998) and implemented the following model in WinBUGS v1.4.3 (Lunn et al., 2000):

$$\begin{cases} (n_{found} + 2) \sim Binomial(N_{predicted} + 4, p_{buoyant}\ p_{discovery}) \\ p_{discovery} \sim Beta(36{,}3.71) \\ p_{buoyant} \sim Beta(1{,}1) \end{cases}$$

(eq.3)

The *cut()* function was used on the parameter $p^{discovery}$ to ensure that the estimate of $P_{buoyant}$ is conditional on $P_{discovery}$. Four chains were run for 20,000 iterations. The first 10,000 were discarded as burn-in, and 1 iteration out of 10 was kept for posterior inference. The final posterior sample was thus 1,000 iterations per chain. The Gelman-Rubin diagnostic suggested model convergence (Cowles and Carlin, 1996).

Time series at the year level were then constructed of estimated bycaught common dolphins corrected by both drift conditions and the proportion of buoyant animals.

2.3.2- Estimations based on direct drift modelling

Let p_{jt} denote the probability of a floating dead dolphin in the study area to strand in month j and year t on the coastline of the study area. p_{jt} is different from $p_{strandings}$: the former refers to the whole study area while the latter is cell-specific. Let y_{ijt} denote the number of bycaught dolphins to strand during the i^{th} period of ten days in month j and year t on the coastline of the study area. Similarly, let z_{ijt} denote number of bycaught dolphins that did not strand over the same period. Finally, let B_{ijt} denote the total number of bycaught dolphins (conditional on them being afloat) over the same period. While y_{ijt} is observed, z_{ijt} is not and their sum $B_{ijt} = y_{ijt} + z_{ijt}$ is thus unknown.

If B_{ijt} were known, y_{ijt} could be modelled as the result of a binomial process with success probability p_{jt}. However, with a random B_{ijt}, joint modelling of both y_{ijt} and z_{ijt} are required (Comulada and Weiss, 2007). We chose to model y_{ijt} and z_{ijt} with negative binomial processes to account for overdispersion (Authier et al., 2014) (Appendix I). Under this model, there is a simple relationship between z_{ijt}, the number of floating dead dolphins that did not strand, and y_{ijt} and p_{jt} (Appendix I). We could thus estimate B_{ijt} and correct these estimates by the probability of being buoyant previously estimated (see 2.3.1). The full methodology is described in detail in Appendix I. A sensitivity analysis is described in Appendix II. Code and data to replicate the analyses are available at https://github.com/mauthier/bycatch.

3. RESULTS

3.1- Cartographic indicators

A total number of 3714 common dolphins found stranded related to fishery activities were collected between 1990 and 2009.

Spatial distributions of bycaught common dolphins inferred from strandings recorded from 1990 to 2009 along French and English coasts of the eastern North Atlantic showed an expansion of mortality in fishing gear over this period (Figure 3). Before 1997, densities of bycaught dolphins at sea were the lowest (max. 1 ind.1000 km^{-2}) and their distribution was inferred from only a few individual trajectories. From 1997 onwards, densities were higher (19 ind.1000 km^{-2}) and mostly located on the continental shelf of the Bay of Biscay and in the western Channel. Bycatch mortality is mostly observed over the continental shelf and slope of the southern Bay of Biscay, from the Loire estuary to the Spanish border. A secondary area of recurrently high bycatch mortality is also found south and southwest of Cornwall. The strongest mortality events occurred between 1997 (massive mortality mapped over the slope of the Bay of Biscay) and 2002 (mortality recorded in shallow waters of southern Bay of Biscay). However, events occurring beyond the continental slope were poorly informed by stranding records.

3.2- Estimating bycatch numbers based on strandings

3.2.1- Estimations from reverse drift modelling

The numbers of dead common dolphins in each cell were corrected by the proportion of buoyant animals that was estimated at 17.9% [9.3%; 28.8%]. This correction provided minimal and maximal estimates of common dolphins dying in fishing gear across the study area in all cells where $p_{stranding} > 0.1$. The average mortality of common dolphins from 1990 to 2009 was 3650 [2250; 7000] dolphins per year (Figure 4), mostly from shelf and slope cells (Figure 3). Before 1997, bycatch estimates were the lowest (below 720 individuals). From 1997 onwards, the average mortality was 4950 [3100; 4950] animals per year. 2001 was estimated as the peak year with 10300 common dolphins [6400; 19850] dying in fishing gear in the Bay of Biscay and western Channel. From 2001 onwards, estimated bycatches decreased.

3.2.2- Estimates from direct drift modelling

Estimates provided by the direct drift modelling approach were on average 4700 [3850; 5750] common dolphins dying in fishing gear every year between 1990 and 2009 (Figure 4). These numbers are approximately 30% higher than those provided by reverse drift modelling. Between 1990 and 1996, estimations were quite high but on average fewer than 2000 individuals (1850 [650; 5200] animals) died in fishing gear. From 1997 onwards, mortality estimates were very high, and averaged 6250 [1250; 8800] dead common dolphins per year. Standard error was on average 1700 over the study period.

4- DISCUSSION

4.1- General

Developing mortality indicators based on strandings ensure a broad spatial and temporal continuity to bycatch monitoring, irrespective of the administrative boundaries within which observer programmes are implemented. The use of strandings is strengthened when coupled with modelling techniques that can provide spatial and temporal indicators in order to come up with areas of interactions with fisheries and bycatch estimates. The interpretation of common dolphin strandings through the use of these indicators highlighted that carcasses found along the coasts constituted a small proportion of mortality at sea. Cartographic indicators allowed mortality areas to be identified on the shelf and continental slope of the central and southern Bay of Biscay, and to a lesser extent south and southwest of Cornwall. Correcting stranding numbers by drift conditions and probability of being buoyant by two different approaches provided estimates of common dolphin mortality in fishing gear. These estimates were between 3600 and 4700 dolphins per year on average over the study period. Peak years were 2001 and 2003 with more than 8500 animals estimated from both approaches bycaught yearly in fishing gear.

4.2- Bycatch estimated from strandings

We developed two different approaches based on the same data. In both cases, estimates are corrected by the proportion of buoyant animals, based on an *in situ* experiment (Peltier et al., 2012), which estimated the probability for a bycaught dolphin to float. This correction factor has a major effect on final estimates and could be further improved by increasing the number of experimentally released carcasses and by refining estimates of discovery rates along the French and UK coasts.

Reverse drift modelling provided minimal numbers of dead animals, and allowed cartographic indicators of mortality areas to be constructed for by-caught cetaceans. This method does not consider offshore cells where $p_{stranding} < 0.1$, thus omitting bycatch from oceanic waters (Figure 5). Furthermore, for a few individual cases of stranded dolphins, the MOTHY model failed to provide a reverse drift trajectory. These few cases had to be removed from the analysis. Thus, estimates from the reverse modelling approach are under-estimates. Another shortcoming of this approach is that it cannot generate proper confidence intervals around estimates. The only source of uncertainty (as shown on Figure 5) stems from uncertainty around buoyancy probability.

The direct drift modelling generated higher estimates overall. This can be mostly explained because the model takes into account the whole study area, including cells with low stranding probabilities. An interesting feature of this approach is how it deals with 0 observed strandings during a time period. Here, 0 either means no bycatch mortality occurred during that period, or that p_{ijt}, the probability of a floating dead dolphin in the study area to strand was very low. Unlike the reverse drift method, the direct modelling approach distinguishes between these two situations. Moreover, it provides uncertainties associated with estimates.

4.3- Comparison with observer programmes

Since 2007, the UK and France have presented bycatch estimates to the European Council based on their observer programmes implemented under regulation EC 812/2004. Available reports suggest a yearly average of 546 common dolphins by-caught in all fishing gear of relevance to the regulation for the period 2007-2011 and in the area of interest of the present work (Table 2). Estimates vary between countries, fisheries and years (Table 3). These figures are approximately one degree of magnitude lower than the reconstructions made from stranding records in the same study area. The comparison was made with the end of the stranding time series (Figure 4) that coincides with the implementation of national observer programmes under regulation EC 812/2004. This marked discrepancy indicates that observer programmes are far from exhaustive and reveal only about 10% of the total small cetacean bycatch in the area.

Several explanations can be considered. Firstly, regulation EC 812/2004 is not aimed at monitoring all fisheries, but only the most relevant ones for small cetacean bycatch. Either the fisheries of interest were misidentified at the time of drafting and negotiating the regulation or the contributions of specific fisheries to total cetacean bycatch have varied greatly over time, making the regulation gradually maladapted. Indeed, an extensive part of the pelagic pair trawl fleets switched to other gear in the early 2000s as a result of anchovy quotas being set to zero for several years (Vermard et al., 2008). Some large-scale fisheries, like fish-meal fisheries, are not considered by this regulation. Nevertheless, they represent a major fishing pressure in the area and target small pelagic fishes known to be prey species for the common dolphin, a situation that would be favourable to high bycatch rates. Secondly, for practical reasons, only vessels over 15 m in length are considered in regulation EC 812/2004, and the coverage of their fishing effort depending on the fleet size and the type of gear (Table 1). Neglecting smaller and artisanal fishing boats can have serious management consequences (Peckham et al., 2008), as vessels under this size limit constitute the major component of many national fishing fleets in the EU. This is notably the case in France where almost 80% of vessels are less than 15 m long (FranceAgriMer, 2014). Artisanal fisheries have long been overlooked, although it is more and more admitted that even recreational and subsistence fisheries can jeopardize marine mammal populations (Lewison et al., 2004; Mangel et al., 2010; Peckham et al., 2008; Zappes et al., 2013). It can be considered that observer programmes in general tend to be biased unless they have a 100% observer coverage. Thirdly, several EU Member States provided uneven bycatch estimations, including Spain and Denmark that operate several major fisheries in the Bay of Biscay and Celtic Sea (ICES, 2014). For instance, during the 2011/2012 fishing season, Spain landed 20% of the catch selling value in Europe (against 12% for the UK and Denmark and 11% for France) (FranceAgriMer, 2014) for around 750,000 t of fishery products (around 255,000 t for France and 464,000 t for the UK) (European Commission, 2014). The lack of reports on cetacean bycatch using observer programmes for several major fishing countries can greatly affect the assessment and proper mitigation of the bycatch issue. Fourthly, even in Member States that have implemented observer programmes, the implementation of EC 812/2004 is not homogeneously distributed among fishing harbours. This observer programme appeared in the context of historically deteriorated relationships between fishermen, scientists and policy-makers. The final decision of accepting an observer on-board is that of the vessel master only and makes it difficult to implement any statistically meaningful sampling protocol (Stratoudakis et al., 1998). This spatial and temporal heterogeneity hinders the power of observer programmes to detect changes in catch, bycatch and discard estimations (Benoît and Allard, 2009).

However, observer programmes have specific value in responding to questions that stranding data can barely address. They can even be conducted out of the EC 812/2004 regulatory context, therefore improving the sampling scheme and the interpretation of bycatch numbers. Some of the most relevant information recorded by observer programmes has highlighted the specificity of interactions between cetaceans and fisheries (Brown et al., 2014; Fernández-Contreras et al., 2010; Marçalo et al., 2015; Rogan and Mackey, 2007). The type of fishing gear and several parameters can be tested as explanatory variables of cetacean mortality. Detecting the specificity of different fisheries in terms of bycatch is essential to determine efficient conservation mitigation measures. Moreover, observer programmes can be associated with biological sampling from bycaught cetaceans (Meynier et al., 2008; Pusineri et al., 2007), which is needed to document cetacean biological traits and to understand the ecological specificity of their interactions with fishing gear (Spitz et al., 2013) (Table 2).

Even if strandings generally cannot inform on the type of fishing gear involved in a majority of bycatch events, strandings collected along European coasts are an important source of information collected at a spatiotemporal scale that matches the cetacean population scale, irrespective of the size and flag of the fishing vessels involved, and independent of the industry's actual willingness to contribute. Stranding schemes can provide minimal numbers of total by-caught small cetaceans and their implementation is independent of the fishing industry. However, stranding only reflect processes affecting cetacean populations within a given distance from the coast; this distance varies regionally with current and wind regimes (Peltier et al., 2013).

4.4- Implications for conservation

We suggest the application of these results to cetacean mortality estimates. They must be carefully interpreted and considered as perspectives.

Considering the bycatch estimated from either stranding data or from dedicated observer programmes has key conservation implications. The current knowledge on common dolphin management areas (MA) in the NE Atlantic is still debated. According to the low genetic differentiation of this species in the north Atlantic, it is commonly admitted that common dolphins can be managed as a single MA (Murphy et al., 2013), but according to ecological tracers (stable isotopes, fatty acids, metal tracers, stomach contains), two MA could be considered for common dolphin management in the NE Atlantic (Caurant et al., 2011; Lahaye et al., 2005; Pusineri et al., 2007). In order to highlight the importance of the conservation consequences associated with the different estimations, the mortality rates of common dolphins were calculated following eq. 2:

$$\text{Mortality rate} = (\text{Number of dead animals (n)})/(\text{Absolute abundance estimation (n)})$$

Estimates of absolute abundances of common dolphins in NE Atlantic were provided by the SCANS II and CODA dedicated surveys (CODA final report, 2009; Hammond et al., 2013). Under the assumption of a single MA, the sum of the SCANS-II and CODA estimates was used, whereas the SCANS II estimate alone was selected to represent the coastal MA under the assumption of two distinct MAs (Table 4).

For bycatch estimates issued from EC 812/2004 reports, the 2007-2011 mean was used. These reports do not refer to spatialized bycatch estimations, and can therefore be used only in the case of one MA. Estimations inferred from strandings for the year 2005 were considered for coastal MA, to be compared with SCANS-II abundance estimations during the same year. Only bycatch predicted to originate from the continental shelf and slope was selected. In the case of one MA in the NE Atlantic, SCANS-II and CODA population estimations were summed and mortality rates were calculated using average bycatch estimates from 2005 to 2007, covering the years of the two dedicated surveys.

Under the hypothesis that common dolphins can be managed under one MA in the NE Atlantic, mortality rates differ according to both sources of bycatch estimations. Numbers proposed by national reports suggest very low mortality rates (less than 0.6%). Estimations inferred from strandings provided mortality rates from 0.9 to 5.7% according to the type of modelling considered. In the case of two MA, only the reverse drift modelling assesses the spatial distinction of the MA. This provides a high and unsustainable mortality rate for the common dolphin population in the costal MA (2.3 to 5.8%).

Assessing the importance of an anthropogenic pressure that generates additional mortality is generally performed following one of the three following approaches. Additional mortality can be kept below a fixed fraction of total population size; this threshold has been determined to be 1.7% for the harbour porpoise in the Gulf of Maine and widely used as a proxy for other species and regions. Note that the calving interval in this particular harbour porpoise population is almost annual, whereas this value is 3.8 years for common dolphins in the NE Atlantic (Murphy et al., 2013), suggesting a higher sensitivity of NE Atlantic common dolphins to additional mortality.

Additional mortality can be kept below the Potential Biological Removal (PBR), which is the maximum number of anthropogenic mortalities that allows the population to remain above its optimum sustainable level (Wade, 1998). The removal limit can be calculated as a function of population parameter estimates that are derived by fitting a population model to a time-series of absolute abundance estimates (Cooke, 1999).

In all three frameworks, bycatch limits are expressed as simple functions of cetacean abundance, with correcting factors related to the maximum growth rate of the population of interest and to its conservation status. The ratio between total bycatch and absolute abundance for a given population is therefore of paramount importance in all cases. Observer programmes are designed as to provide fishery specific estimates of bycatch rates, but fail to give an estimate of total bycatch incurred by a given population of small cetacean. This is particularly an issue in regions where fisheries are made of multiple fleets, gear types and métiers, as opposed to region where a single practice would represent most of the total fishing effort, hence considerably simplifying any monitoring strategy.

Some changes to the EC 812/2004 observer programme could greatly improve its representativeness. The systematic random sampling of fisheries could reduce the deployment effect (Benoît and Allard, 2009). The administrative and logistical complexity of taking an observer on board could be reduced in order to encourage the involvement of captains and crews in the programme. Nevertheless, bycatch estimations on small vessels and artisanal fisheries remain always challenging, and mounting electronic cameras on the vessels to replace on-board observers could increase observer coverage. The use of different complementary sources of data constitutes the most efficient way to estimate marine mega-fauna bycatch: observer programmes on large vessels to understand the specificity of interactions between mega-vertebrates and fisheries; questionnaire surveys (called a 'Rapid Bycatch Assessment') carried out for evaluating cetacean and seabird bycatch and specific interactions on smaller vessels and artisanal fisheries (Goetz et al., 2015; Moore et al., 2010; Oliveira et al., 2015; Poonian et al., 2008), and finally the interpretation of stranding data, in order to evaluate the impact of fisheries on mega-vertebrates at the population scale.

The European Commission has requested the appropriateness and effectiveness of the provisions of Regulation 812/2004 for protecting cetaceans to be reviewed by the end of 2015. The recent revision of the CFP (Regulation 508/2014) re-affirmed the need to mitigate, prevent and monitor the marine mammal bycatch (articles 38 & 77). Given the continuing saliency of marine mammal bycatch, it is critical to ensure continuous data acquisition and monitoring of the issue.

The recent European Marine Strategy Framework Directive (2008/25/EC, hereafter MSFD) was adopted in 2008 and aim to restore and maintain the 'Good Environmental Status' of Member State marine ecosystems by 2020. The MSFD embodies both an ecosystem-based approach and the precautionary principle applied to marine conservation (Dotinga and Trouwborst, 2011). It represents an improvement over preceding legal instruments in Europe (Dotinga and Trouwborst, 2011), by promoting a pro-active approach, setting deadlines to ensure progress toward 'Good Environmental Status' and requiring regional cooperation between Member States. Regional cooperation is leveraged via the Regional Seas Convention, in particular the OSPAR Convention. The latter includes a bycatch indicator for marine mammals. We propose that this indicator be also informed by stranding data collected by Member State national stranding networks. The present work, along with previous studies (Peltier et al., 2014, 2013, 2012), demonstrates the relevance and credibility of these data for estimating the marine mammal bycatch within the MSFD.

5- CONCLUSION

Cartographic parameters inferred from strandings were adapted to highlight the areas at sea with high vulnerability of common dolphins to fisheries. The highest densities of by-caught common dolphins at sea were predicted on the continental shelf to the slope of the Bay of Biscay. Then two complementary approaches were developed in order to provide new estimations of by-caught common dolphins in the Bay of Biscay and western Channel. We demonstrated that both approaches provided complementary estimates, which provided low and high bounds of an interval encompassing real estimate. Finally, the comparison of this interval with on-board observer monitoring programmes conducted under regulation EC 812/2004 demonstrates the complementarity of these tools, as both provide relevant and consistent estimates for cetacean conservation.

This work demonstrated the interest of including and associating other sources of indicators with observer programmes, in order to provide bycatch estimates at population scales rather than administrative boundaries. Whatever the method used to develop indicators based on strandings, these estimates were about 10 times higher than estimates produced by observer programmes conducted under EC 812/2004 regulation. According to the nature of the different estimates and the origin of the data, it can be concluded that observer programmes carried out under the EC 812/2004 regulation provided relevant information on the specificity of the interaction between small cetaceans and fishing activities, essential for relevant decision making. Nevertheless because of administrative and practical restrictions, this approach cannot be used for quantitative estimations of these interactions. The use of stranding data sets as a source of indicators for common dolphin mortality seemed more convincing. This suggested potentially unsustainable level of bycatch for common dolphin in the NE Atlantic.

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Tables

Table 1 Fisheries to be monitored and minimum level of fishing effort subject to on-board observers according to EC 812/2004 in ICES areas VII and VIII.

Gear	Coverage by on-board observers
Pelagic trawls (single and paired)	Fleets > 60 vessels: 10% observer coverage of fishing effort Fleets < 60 vessels:10%, at least three different vessels
Bottom-set gillnets or entangling nets (mesh ≥ 80 mm)	Fleets > 400 vessels: fishing effort of 20 vessels 400 > Fleets > 60 vessels: 5% observer coverage of fishing effort Fleets < 60 vessels: 5% observer coverage, at least three different vessels
Driftnets	Fleets > 400 vessels: fishing effort of 20 vessels 400 > Fleets > 60 vessels: 5% observer coverage of fishing effort Fleets < 60 vessels: 5% observer coverage, at least three different vessels
High-opening trawls	Fleets > 400 vessels: fishing effort of 20 vessels 400 > Fleets > 60 vessels: 5% observer coverage of fishing effort Fleets < 60 vessels: 5% observer coverage, at least three different vessels

Table 2: Comparison between common dolphin bycatch indicators based on observer programmes or inferred from strandings.

Parameter	Observer Programmes	Strandings
Specificity of the interaction	Yes	No
Spatial scale	Administrative	Population
Reproducibility	Difficult	Yes
Time series	Since 2005	Since 1990
Sampling strategy	Difficult	In progress
Biological samples	Yes	Yes
Mean estimated bycatches	≈550.year^{-1} (2007-2011)	≈3,600 to 4,700.year^{-1} (1990-2009)

Table 3: Information available on cetacean bycatch in EU Member Country reports under the EC 812/2004 regulation for the year 2012 (based on ICES report (ICES, 2014)). ('no monitoring obligation' means that countries have no monitoring obligation under EC 812/2004; 'no dedicated monitoring' means that countries do not implement the EC 812/2004 regulation observer programmes).

EU members	EC 812/2004 dedicated observer programme	Bycatch reported (all observer programmes)	Observer coverage
Belgium	No monitoring obligation	0	-
Denmark	No dedicated monitoring	17 cetaceans observed	752 days
Estonia	No dedicated monitoring	0	198 days (22 of 101 pelagic vessels)
Finland	Reported until 2008	?	?
France	Dedicated monitoring	207 common dolphins estimated (in 2011)	796 days
Germany	Dedicated monitoring	0	1225 hours on pelagic trawlers & 833 hours on static netters
Ireland	Dedicated monitoring	1 cetacean observed	227 days on pelagic trawlers
Italy	Dedicated monitoring	1 cetacean observed	518 days on pelagic/midwater trawlers
Latvia	Dedicated monitoring	0	1096 days on 9 pelagic trawlers
Lithuania	Dedicated monitoring	0	9 days on 2 pelagic trawlers
Netherlands	Dedicated monitoring	1 cetacean observed	123 days on pelagic fleet
Poland	Dedicated monitoring	0	70 days on pelagic trawlers & 59 days on set gillnetters
Portugal	Dedicated monitoring	5 cetaceans observed	71 days on gillnet/trammelnet fleet
Slovenia	Dedicated monitoring	0	?
Spain	Reported until 2008	?	?
Sweden	No report provided	?	?
United Kingdom	Dedicated monitoring	257 common dolphins estimated	100 days on pelagic trawlers and 299 on gill- and tanglenet vessels

Table 4: Abundance estimations and mortality rates due to bycatch for common dolphins in the case of one or two management areas in the NE Atlantic.

MA	One management area	Two management areas (coastal)
Absolute abundance estimations (n)	SCANS II + CODA [92,663 - 334,659] Years 2005 and 2007	SCANS II [35,748 - 88,419] Year 2005
Mortality rate estimations		
Observer programmes	[0.2% - 0.6%] Years 2007 to 2011	Not Available
Reverse drift modelling	[0.9% - 3.5%] Years 2005 to 2007	[2.3% - 5.8%] Year 2005
Direct drift modelling	[1.6% - 5.7%] Years 2005 to 2007	Not Applicable

FIGURE CAPTIONS

Figure 1: Study area and sub-regions. WC: western Channel, BB: Bay of Biscay.

Figure 2: Two approaches to estimating bycatch rates from stranding data. The latter were recorded on the coastline in the light blue area, which includes the following French *départements* from south to north: *Pyrénées Atlantiques*, *Les Landes*, *Gironde*, *Charente Maritime*, *Vendée*, *Loire Atlantique*, *Morbihan* and *Finistère*; and the following English counties from west to east: Cornwall, Devon and Dorset. Bathymetric maps were plotted with the R package *marmap*, and are represented by blue shading (Pante and Simon-Bouhet, 2013). The cell size is 0.75° x 0.75°.

Figure 3: Distribution of bycaught common dolphins inferred from strandings from 1990 to 2009. These densities of dead dolphins were calculated following reverse drift modelling and based from strandings collected along the coasts of the Bay of Biscay and the western Channel.

Figure 4: Common dolphin bycatch estimations (n individuals) inferred from strandings using direct drift modelling (black points, associated with the confidence interval in grey bars), and using reverse drift modelling (grey polygon).

Figure 5: Seasonal maps of stranding probability in the study area. The darker the colour, the higher the probability that animals dying in the corresponding cell would reach the coast (from Peltier et al., 2013)

FIGURES

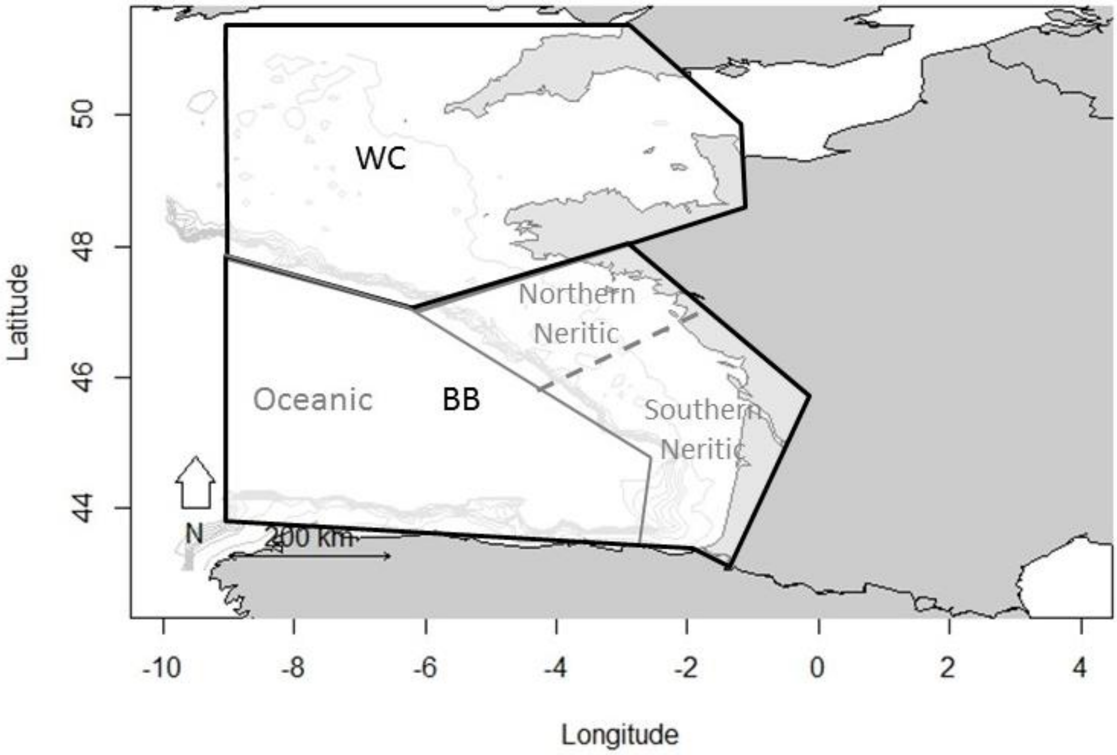
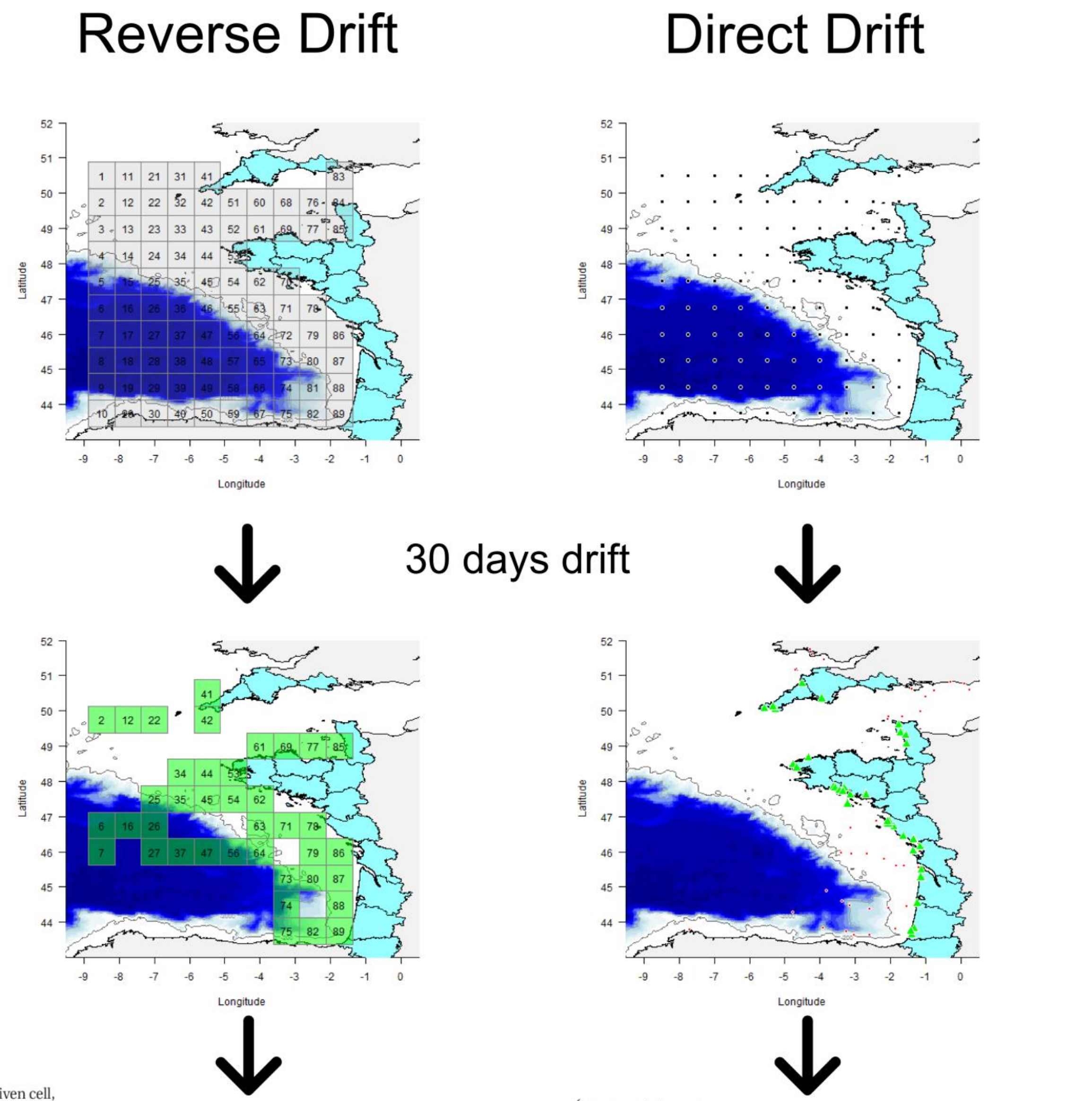


Figure 1



$$\begin{cases}\text{For a given cell,} & \\ p_{\text{strandings}} & = \dfrac{\sum_t^{year}\sum_j^{month}\sum_i^{decade}(\text{dolphin originating from cell predicted to strand})}{\sum_t^{year}\sum_j^{month}\sum_i^{decade}} \\ & = \text{long term frequency of predicted strandings from MOTHY}\end{cases}$$

$$\begin{cases}\text{For the whole study area,} & \\ p_{ijt} & = \dfrac{\sum \text{dolphins predicted to strand} + 2}{\sum \text{dolphins} + 4} \\ & = \text{proportion of predicted strandings from MOTHY}\end{cases}$$

Figure 2

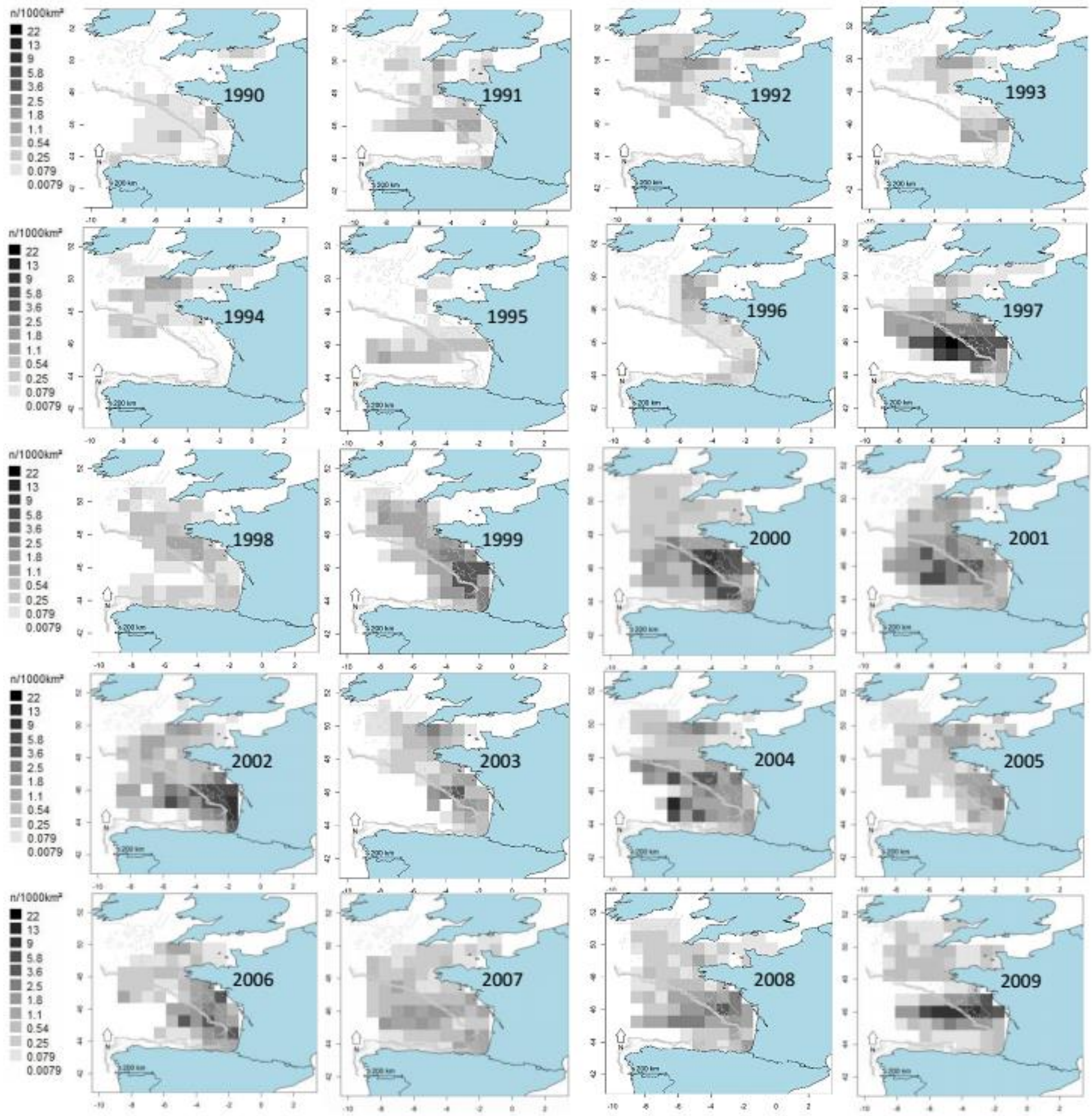

n/1000km²
22
13
9
5.8
3.6
2.5
1.8
1.1
0.54
0.25
0.079
0.0079
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
2009
200 km

Figure 3

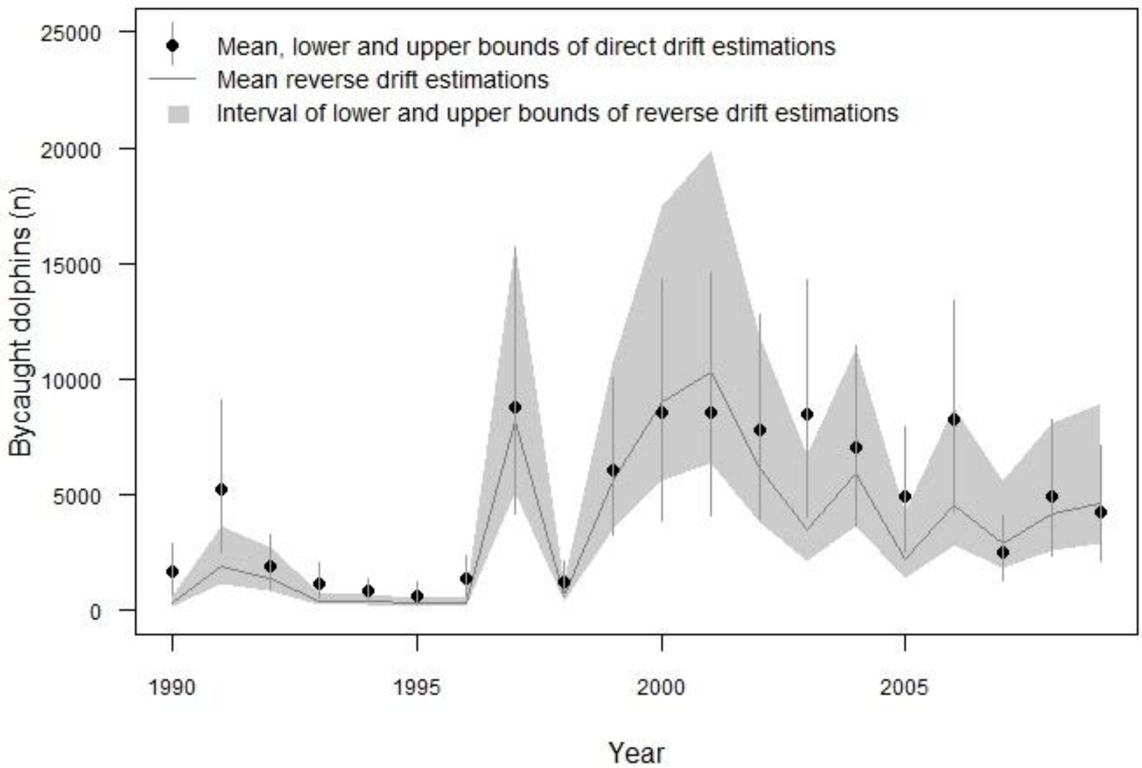

Mean, lower and upper bounds of direct drift estimations
Mean reverse drift estimations
Interval of lower and upper bounds of reverse drift estimations
Bycaught dolphins (n)
25000
20000
15000
10000
5000
0
1990
1995
2000
2005
Year

Figure 4

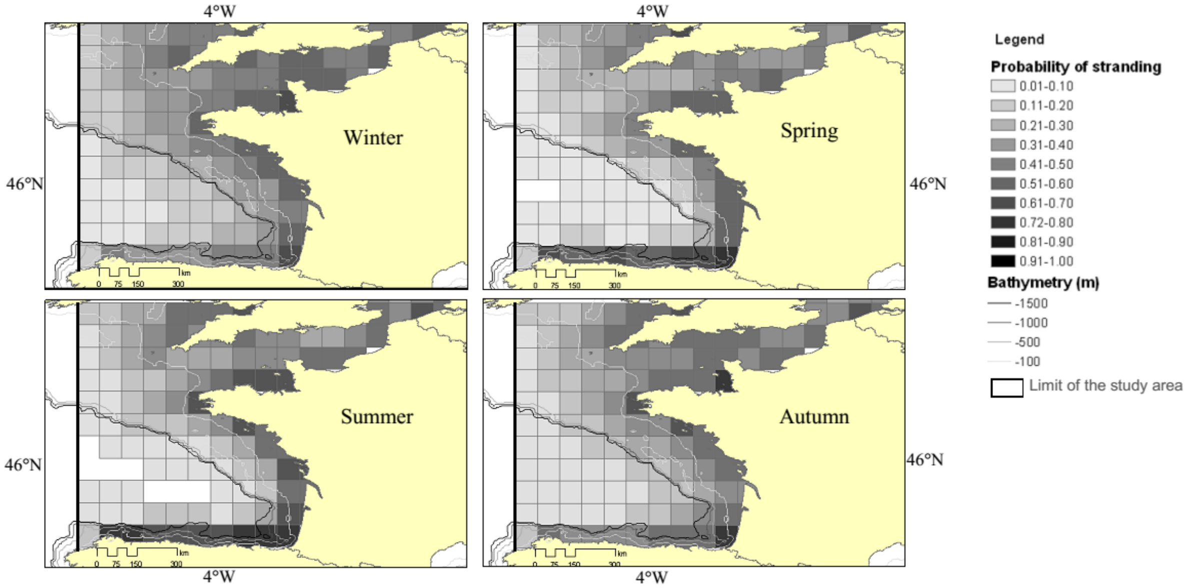

4°W
Winter
Spring
Summer
Autumn
46°N
Legend
Probability of stranding
0.01-0.10
0.11-0.20
0.21-0.30
0.31-0.40
0.41-0.50
0.51-0.60
0.61-0.70
0.72-0.80
0.81-0.90
0.91-1.00
Bathymetry (m)
-1500
-1000
-500
-100
Limit of the study area

Figure 5