



# Marine megavertebrates adrift: A framework for the interpretation of stranding data in perspective of the European Marine Strategy Framework Directive and other regional agreements



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## ABSTRACT

For many species of marine megafauna, strandings remain the most important source of biological samples. Because of their opportunistic nature however, strandings data have long been under- or misused in the assessment of population conservation status. Even if many national and international regulations promote the use of strandings in monitoring strategies, the interpretation of strandings remains controversial. The aim of this study is to provide a context for the interpretation of marine megafauna stranding data, in order to assess the achievement of specific objectives against Good Environmental Status criteria in the context of the EU Marine Strategy Framework Directive or other regional agreements. The first step is to construct an *a priori* spatial distribution under a null hypothesis  $H_0$ . The *a priori* spatial distribution of theoretical dead animals can either be set uniformly, consistent with current knowledge on abundance of marine vertebrates, or based on management objectives. The drift prediction of these theoretical carcasses would provide a time series of strandings expected under the null hypothesis. The reverse drift of observed strandings would highlight mortality areas of stranded animals. The correction of these areas by the probability of getting stranded according to drift conditions would provide an estimated distribution of dead animals inferred from strandings. The differences between expected and observed situations constitute anomalies and highlight cases where inferred distribution departs from the *a priori* spatial distribution. This work proposes several population indicators that can be used anywhere in the world and can be applied for all large marine vertebrates found stranded. The integration of these indicators in MSFD and various regional agreements could provide cost-effective and relevant information on protected species. In the context of impaired ecological situations, the complementary use of several population indicators could strengthen the diagnosis made regarding conservation status and hence conservation strategies.

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## 1. Introduction

In the context of many international regulations and agreements, the conservation status of marine megavertebrates is required. At the European Union (EU) level, the Marine Strategy Framework Directive (hereafter MSFD, 2008/56/EC) ratified in 2008 aims to achieve the “Good Environmental Status” (GES) by 2020. GES would be restored or maintained when 11 qualitative descriptors do not deviate from the undisturbed condition (Fletcher, 2007; Zampoukas et al., 2013; Freire-Gibb et al., 2014; Maier, 2014). Progress toward GES objectives is measured by ecological monitoring programs established in 2014. These

monitoring programs are encouraged to be maximized with existing regional agreements (Borja, 2006; Zampoukas et al., 2013). Agreements concerning top predators include among others OSPAR, HELCOM, ICES, ASCOBANS and ACCOBAMS. However, according to their protection status and habitat, indicators on top predator conservation status are difficult to collect.

The scientific use of stranded animals has been encouraged for centuries (Hunter and Banks, 1787). As part of many monitoring strategies, stranded carcasses of marine megavertebrates long have been a major source of biological information. The main benefit of this approach is to give access to tissue and organ samples with no lethal sampling (Hall et al., 2010). For protected and vulnerable species, stranded carcasses are the main source of information on their presence (Reyes et al., 1991; Findlay et al., 1992; Thompson et al., 2013), biology (Dabin et al., 2008; Murphy et al., 2009), physiology (Chaloupka and Zug, 1997; Rommel et al.,

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2006), cause of death (Geraci et al., 1989; Kirkwood et al., 1997; Sullivan et al., 2006; Tomas et al., 2008), specific richness (Maldini et al., 2005; Pyenson, 2010, 2011), health condition (Bjorndal et al., 1994; Krahn et al., 1997; Jepson et al., 1999; Lahaye et al., 2007) or diet (Plotkin et al., 1993; Santos et al., 2001; Spitz et al., 2006). Because of insufficient sampling strategies however, strandings data long have been under- or misused in the assessment of population conservation status (Wiese, 2003). Previous studies concluded that strandings were not efficient in seabird bycatch estimations (Zydalis et al., 2006), but when stranding representativity could be assessed, notably by drifter experiments in order to measure the effect of wind and currents, their interpretation became more relevant (Epperly et al., 1996; Hart et al., 2006; Peltier et al., 2012, 2013; Koch et al., 2013). Similar uncertainty appeared in interpreting oiled marine mammals or seabirds during acute oil spill pollutions (Hope Jones et al., 1978; Piatt et al., 1990; Hlady and Burger, 1993; Degange et al., 1994; Williams et al., 2011).

Stranding frequency is often considered to be biased by drift conditions and their composition biased in favour of the weaker segments of the population. Hence, spatio-temporal patterns in stranding follow a complex function of biological, physical and societal factors (Eq. (1)).

$$N_{\text{stranding}} \propto \text{Abundance} \times \text{Mortality} \times \text{Buoyancy} \times \text{Drif} \times \text{Reporting} \quad (1)$$

Abundance and mortality are the key biological parameters that a monitoring strategy should be aimed at documenting in the long-term; whereas carcass buoyancy, drift conditions and reporting rates are the main confounding factors that may mask variations in the two biological parameters of interest. Of these three confounding factors, drift is the one that by far would introduce more noise in the stranding data series as it is mostly driven by wind and tidal currents that greatly vary in the short to very short terms. By comparison, carcass buoyancy and reporting, although difficult to estimate, are not thought to vary greatly in the short term.

This paper will provide a context for the interpretation of stranding data that could be applied in all oceans and for all large marine vertebrates (seabirds, marine mammals, sea turtles, etc.) based on an understanding of drift conditions in the stranding process. The key to this approach is to possess or to encourage the development of an adequate physical drift model in order to disentangle the effects of confounding parameters from the effects of the biological component of the stranding signal. The use and benefits of these indicators in the EU MSFD and other international agreements will be discussed.

## 2. Context of stranding interpretation

### 2.1. General experimental design

During this work, the null hypothesis was used in its primary meaning defined by Strong (1980): “Null hypotheses entertain the possibility that nothing has happened, that a process has not occurred, or that change has not been produced by a cause of interest”. The terminology does not refer to inference statistics.

The general principle is to set *a priori* information on spatial distribution of marine megafauna, to determine the stranding probability at any location of the study area and generate predicted stranding data sets by using a drift model over a given period of time and across a given study area (Figs. 1 and 2). Conversely, the observed stranding data set for the same area and period can be used to infer the distribution of dead animals at sea, when corrected by stranding probability. Finally, stranding anomalies are defined as the difference between observed and predicted strandings under  $H_0$ , whereas anomalies in distribution and mortality are defined as the difference between the inferred and *a priori* spatial distributions of dead megafauna.

### 2.2. The drift prediction model and its parameterization

Using fully deterministic drift prediction models to compute the drift of dead top predators was recently developed to improve the representativeness of strandings as a source of indicators for cetacean populations. For example, the model MOTHY, initially designed by MétéoFrance for calculating the drift of oil slicks and of solid objects of interest to maritime safety, was adapted to small cetaceans in the Bay of Biscay, the English Channel and the North Sea (Peltier et al., 2012, 2013). Moreover ocean circulation models were used to simulate the dispersion of loggerhead turtle hatchlings across the Pacific Ocean (Okuyama et al., 2011) or the dispersion of Kemp’s ridley sea turtles across the Gulf of Mexico (Putman et al., 2013). In ideal situations, drift prediction models would be implemented with several parameters that can be obtained by existing databases or fieldwork (Table 1).

### 2.3. Predictions

#### 2.3.1. A priori spatial distribution under the null hypothesis and predicted strandings

The *a priori* spatial distribution is a theoretical distribution of dead animals at sea under the null hypothesis  $H_0$ . It can be set either to a uniform distribution, or based on current knowledge about megavertebrate abundance and distribution in the study

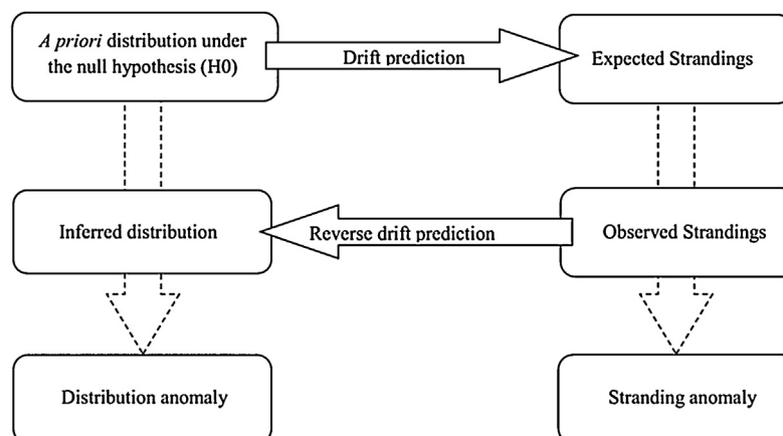


Fig. 1. General concepts of the experiment.

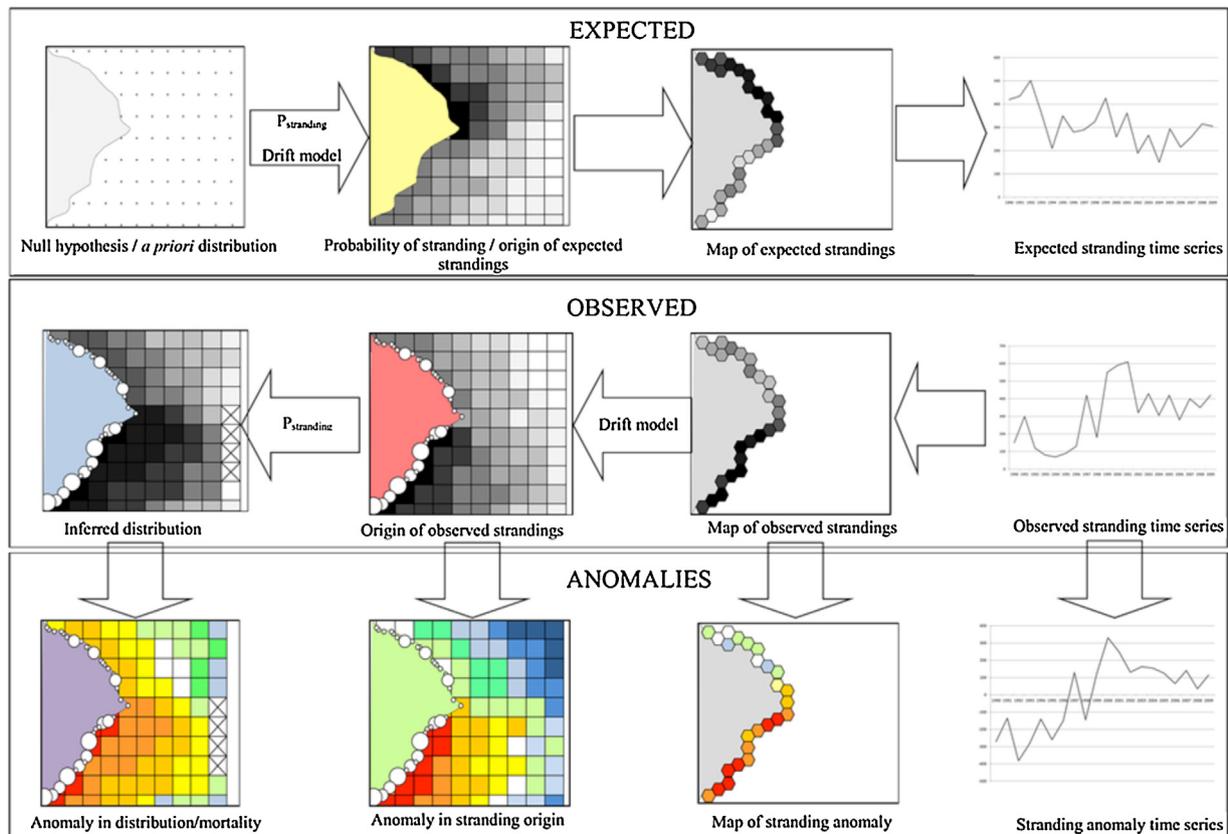


Fig. 2. Construction of theoretical context of stranding interpretation.

area. This distribution must be independent from stranding data (dedicated surveys, data collected on platform of opportunity). For a flat *a priori* spatial distribution, mortality and abundance are assumed constant in time and uniform in space. In this case, patterns in predicted strandings would reflect patterns in drift conditions only. The use of actual distribution and abundance data as the *a priori* distribution implies a comprehensive knowledge of top predator distributions in the area, including changes between seasons and among years. In this case, patterns in predicted strandings would reflect patterns in drift conditions together with spatiotemporal heterogeneity in distribution.

Table 1

Required parameters to implement drift prediction models in context of interpreting stranding data..

Parameters	Source	Examples
Atmospheric model	Regional or global databases	<ul style="list-style-type: none"> <li>•National Climatic Data Center (NOAA; world)</li> <li>•European Centre for Medium-Range Weather Forecasts</li> <li>•The Fifth Generation Penn State/NCAR Mesoscale Model (Mesoscale models)</li> </ul>
Tide and current models	Regional or global databases	<ul style="list-style-type: none"> <li>•High Resolution Limited Area Model</li> <li>•Center for Operational Oceanographic Products and Services (NOAA)</li> <li>•Danish National Space Institute</li> </ul>
Bathymetry	Regional or global databases	<ul style="list-style-type: none"> <li>•Service Hydrographique et Océanographique de la Marine</li> <li>•National Geophysical Data Center (NOAA)</li> <li>•Danish National Space Institute</li> <li>•General Bathymetric Chart of the Oceans</li> </ul>
Immersion rate	Based on experiment	<ul style="list-style-type: none"> <li>•Service Hydrographique et Océanographique de la Marine</li> <li>• Small cetaceans: 90% (Peltier et al., 2012)</li> <li>• Harbor porpoises (<i>Phocoena phocoena</i>) and bottlenose dolphins (<i>Tursiops truncatus</i>): positively buoyant (Kipps et al., 2002)</li> <li>• Manatees: negatively buoyant (Kipps et al., 2002)</li> </ul>
Drift duration	Based on experiment	<ul style="list-style-type: none"> <li>• Murres (<i>Aria</i> spp): positively buoyant during 10–14 days (Wiese, 2003)</li> <li>• Small cetaceans: 40 days (Peltier et al., 2012)</li> <li>• Murres (<i>Aria</i> spp): 10–14 days (Wiese, 2003)</li> <li>• Shearwaters (<i>Puffinus</i> spp): 7–10 days (Wood, 1996)</li> </ul>
Date and location of stranding	Collected on carcasses	–
Animal dimensions	Collected on carcasses	–

### 2.3.2. Stranding probability and origin of expected strandings

The drift of these animals from death locations onwards is calculated at regular time intervals using a drift prediction model. For each theoretical animal at each selected date, a value of 1 is attributed to the death location cell if stranding is predicted to occur and 0 if it is not. In each cell of the gridded map of the study area, the proportion of drift trajectories starting in this cell and predicted to reach a coast gives the local stranding probability  $P_{\text{stranding}}$ , which varies from 0 to 1. When scaled up to the total number of expected strandings, this map represents the origin of expected strandings.

### 2.3.3. Expected strandings

The number of theoretical animals reaching the coast and predicted to strand constitutes the stranding data set expected under the null hypothesis. These predicted strandings can be analyzed temporally or spatially. Expected strandings must be calibrated to match the total number of observed carcasses.

## 2.4. Observations and inferences

### 2.4.1. Observed strandings

Strandings schemes can provide long-term and geographically-extended data sets of observed strandings. The wider the spatiotemporal range, the more relevant the data set is from a monitoring perspective. In the present context, species, location and date are the primarily required data, together with decomposition code or any other proxy of postmortem drift duration, e.g. pictures (Peltier et al., 2012). However, subsets of stranding data can be defined and analyzed separately on the basis of individual-specific features, such as sex, age or cause of death.

### 2.4.2. Origin of observed strandings

The use of a drift prediction model allows back-calculating drift trajectories (from stranding place to likely death location at sea) of stranded animals, and hence identifying the origin of observed strandings. Drift duration is the key element in this process. For cetaceans, external criteria were developed to estimate drift duration of dead animals that could be estimated from pictures for each dead animal according to its decomposition status (Peltier et al., 2012).

### 2.4.3. Inferred distribution

To take into account spatial patterns of  $P_{\text{stranding}}$ , the origin of observed strandings can be multiplied by  $1/P_{\text{stranding}}$  to infer the distribution of dead animals at sea from stranding records. In order to limit uncertainty at the margin of the area where  $P_{\text{stranding}}$  gets close to 0, cells where  $P_{\text{stranding}}$  is very low may be truncated (e.g.  $P_{\text{stranding}} < 0.05$  or  $< 0.10$ ). These areas can generate great uncertainty in inferred dead animals at sea when rare events are multiplied by a high correction factor.

To summarize, this final step provides a gridded map of numbers of dead animals found stranded corrected by  $P_{\text{stranding}}$ , that is, the number of dead animals drifting at sea. The number of dead animals in each cell can be summed across the whole calculation area to provide an estimate of total dead animals floating at sea. Consequently, to derive total mortality across the calculation area, it is crucial to carefully estimate the proportion of carcasses that float after death. This proportion varies greatly among marine megavertebrates, particularly in diving taxa whose buoyancy is generally close to neutral and therefore limited changes in body composition, or amount of air in respiratory tracts can make it float or sink. This aspect is not developed further in the present work.

## 2.5. Anomalies

### 2.5.1. Stranding anomaly

The stranding anomaly is defined as the difference between observed strandings and strandings expected under  $H_0$  (Peltier et al., 2013). To do this, expected strandings must be scaled to the total number of observed strandings so that the overall stranding anomaly is 0 over the study period and area. Local positive (vs. negative) anomalies suggest that more (vs. less) strandings were observed than expected under  $H_0$ . Under a  $H_0$  set as a uniform and constant distribution, stranding anomaly would detect changes in abundance and/or mortality of stranded animals. If  $H_0$  is an actual distribution, stranding anomaly would highlight changes in mortality rate of stranded animals.

### 2.5.2. Stranding origin anomaly

In each cell of the gridded map, the difference between observed and expected stranding inferred origins constitutes the anomaly in stranding origin. Expected origin must be calibrated on the observed origin so that the overall anomaly in stranding origin summed over the study area and period is zero. This anomaly detects changes in origin of stranded animals compared to  $H_0$ : the anomaly is positive when dead drifting animals are more abundant than expected under  $H_0$  and the anomaly is negative when they are less abundant than under  $H_0$ .

### 2.5.3. Distribution anomaly

The distribution anomaly is the difference between the *a priori* spatial distribution and the distribution inferred from strandings. The two distributions are scaled up to the same total across the study area and period, and the distribution anomaly is calculated in each cell of the gridded map.

When  $H_0$  is set to a uniform distribution, positive (vs. negative) anomalies highlight areas where abundance and/or mortality of populations are higher (vs. lower) than in the *a priori* spatial distribution. Spatial patterns in the distribution anomaly would represent changes in the number of dead animals at sea, i.e. changes in either abundance or mortality.

If  $H_0$  is an actual distribution, positive (vs. negative) anomalies would highlight areas where mortality of populations is higher (vs. lower) than in the *a priori* spatial distribution. Hence the distribution anomaly would highlight spatial patterns in mortality alone.

## 2.6. Ethics statement

This work reports on new research that has never been and is not being submitted elsewhere. This constitutes a theoretical paper, no data or samples were used for this study.

## 3. Discussion

### 3.1. General

Because it is the role of scientists to provide interpretation tools for their results (Wiese and Elmslie, 2006), this work aimed to propose an interpretation context for marine megavertebrate stranding data sets. This context can be used everywhere in the world ocean where carcasses of dead megavertebrates are susceptible to become beached and for a variety of marine species, including cetaceans, seabirds, and sea turtles. It is particularly adapted for, and required in, many regional regulations and agreements in order to increase the cost-effectiveness of monitoring programs (Caughlan and Oakley, 2001; Heink and Kowarik, 2010; Bertram and Rehdanz, 2013; Zampoukas et al., 2013; Bertram et al., 2014).

In its primary meaning in conservation biology, the null hypothesis is a statement about a population of interest that can be theoretical or based on partial knowledge (Fewster, 2013). Null hypotheses are usually neutral because they can reflect our current understanding, acceptable situations or conservation objectives. These hypotheses are then compared with observations: anomalies between theoretical and observed situations provide evidence that the *a priori* distribution was not a reasonable description of the situation. The interpretation of these anomalies can provide relevant information and disentangle ecological from physical processes involved in stranding.

The use of the null hypothesis as a research hypothesis rather than a statistical hypothesis is very useful in exploratory studies in order to exclude random effects or disentangle process from known parameters (Clarke et al., 2008). The null hypothesis as

research hypothesis already has been used to study co-occurrence of fish communities (Costa de Azevedo et al., 2006) or tree diversity on tropical islands (Leigh et al., 1993).

One of the most important steps in the analysis is the choice of the drift model. The size and buoyancy of marine megafauna suggest that models predicting drift of large objects are more suitable for carcass modeling than models designed for calculating the movements of water masses or of small objects such as planktonic eggs or larvae. The parameterization of such models to top predator carcasses must be based on *in situ* experiments.

The choice of the null hypothesis is determinant and linked to scientific and conservation objectives. Therefore, the *a priori* spatial distribution can be set as uniform or be constructed from real data (either raw data, kriged data or habitat modeling predictions). It can be used as a reference distribution to highlight anomalies where dedicated management measures must be provided or as a hypothetical distribution where anomalies would confirm or contradict the *a priori* spatial distribution under  $H_0$ .

### 3.2. Producing spatial indicators

In the context of increasing utilization of marine habitats, human pressures on large vertebrates must be monitored. The maps of stranding probability and of expected strandings can be used to define stretches of coastline where strandings potentially linked to some specific and geographically defined pressures would be more likely to be reported. Particular vigilance would be needed in these areas to detect changes in stranded animals and evaluate the impact of new human activities on marine species like renewable energies before, during and after their construction, functioning and decommissioning. Such analysis would provide indicators of the impact of these activities on top predators.

The likely origin of an observed stranding can be used to locate the at-sea origin of samples collected on stranded animals as well as all derived biological information. For many protected species like marine mammals, seabirds and sea turtles, strandings remain the main source of biological samples. Back-calculating the drift of sampled carcasses provides a spatially explicit context for the interpretation of biological parameters at sea. It would become possible to locate carcass origins according to their stomach content composition (or any other biological information), in order to map the diet composition (or other biological traits of interest) of top marine predators found stranded. This opens new and potentially fertile grounds for investigating questions of long-standing interest such as detection of short-term segregation in population structure.

The distribution anomaly would probably be one of the most relevant indicators in the context of MSFD. It detects anomalies in the number and distribution of dead animals at sea compared to an *a priori* spatial distribution. The interpretation of anomalies requires complementary information, notably on the distribution of human-induced-mortality. In the context of marine megafauna conservation, the detection of critical areas with a high mortality level or abundance can become a major criterion for the designation of marine protected areas.

In addition to these applications, the following cases illustrate some situations where the present interpretation framework could improve the use of stranding data sets in the aim of better assessing the impact of already identified anthropogenic pressures.

Early speculation about stranding representativeness appeared for interpreting marine mammal, seabird, otter and turtle strandings recorded during various oil spills in recent decades (e.g. Amocco Cadiz, France, 1978; Exxon Valdez, USA, 1989; Erika, France, 1999; Prestige, Spain, 2002 ...) (Hope Jones et al., 1978; Piatt et al., 1990; Degange et al., 1994; Flint and Fowler, 1997; Garshelis, 1997). Recently, during the *Deepwater Horizon* oil spill in

the Gulf of Mexico the abnormally high marine vertebrate stranding numbers triggered the Unusual Mortality Event procedure during 13 months. These events prompted scientists to re-examine the issue of the stranding representativeness relative to actual mortality at sea (Dahlmann et al., 1994; Camphuysen, 1998; Williams et al., 2011). The use of distribution data as a null hypothesis would allow calculating inferred distribution and hence monitoring mortality rate of marine mammals, seabirds, and turtles before, during and after the oil spill.

Since the 1990s an increase of beaked whale mass strandings was recorded with severe injuries seemingly linked to decompression-related-mechanisms. In many cases these strandings were spatiotemporally associated with naval operation using low or mid-frequency sonar, or seismic prospection surveys that are localized in space and time (Frantzis, 1998; Jepson et al., 2003; Gordon et al., 2004; Southall et al., 2006). Nevertheless these data remained hard to collect, and back calculations of beaked whale strandings would detect critical areas of origin for these vulnerable species. This use of drift prediction would make the assumption that beaked whales died close to the disturbance source and did not strand alive.

For many marine species, death in fishing gears is the most important cause of mortality (Kirkwood et al., 1997; Lewison and Crowder, 2003; Gilman et al., 2005; Read et al., 2006; Sullivan et al., 2006; Cox et al., 2007; Rogan and Mackey, 2007; Leeney et al., 2008; Tomas et al., 2008; Wallace et al., 2008; Murphy et al., 2009; Reeves et al., 2013; Prado et al., 2013). Many regulations usually promote the assessment and reduction of bycatch, but very few provide protocols to assess mortality in fishing gears. It is commonly admitted that mortality induced by interactions with fisheries should be estimated by dedicated observer programs, however, important gaps in marine mammal bycatch data have been identified in many fisheries around the world (Reeves et al., 2013). The indicator describing the origin of stranded animals would highlight major areas of interactions between fisheries and top predators. Inferred distribution of stranded small cetaceans diagnosed as bycatch would allow numbers of animals caught in all fisheries to be estimated and areas of highest mortality to be identified. If an actual distribution was used as the *a priori* spatial distribution, anomalies in distribution would inform on the spatial pattern of mortality in fisheries. Hence, the combination of data collected by observers on board, of origins of stranded top predators with bycatch evidences and of the inferred distribution of these animals would improve the comprehension of the bycatch process and help in the design of relevant mitigation actions.

### 3.3. Monitoring in international and regional agreements

Providing cost-efficient indicators of top predator population status based on strandings is an increasing need in monitoring strategies. Through four cases, an overview of various uses of strandings as marine megafauna population indicators is provided below.

#### 3.3.1. CASE 1: The OSPAR commission

The OSPAR Commission aims to protect and conserve the North-East Atlantic Ocean and its resources through the involvement of 15 European countries. Many indicators were monitored to provide information on 12 parameters (e.g. climate change, radioactive substances, eutrophication ...). Among them, the number of oiled common guillemots (*Uriaa algae*) informed on offshore oil and gas industry activities and, as a conservation objective, this indicator must represent less than 10% of stranded guillemots across the OSPAR area. The identification of their stranding origin using the conceptual framework described here

could provide relevant information on mortality areas of oiled seabirds in addition to the proportion of oiled birds.

### 3.3.2. CASE 2: Agreement on the conservation of small cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS)

The OSPAR porpoise indicator relates to actions proposed in the harbor porpoise conservation plan developed under ASCOBANS. Among 12 indicators dedicated to harbor porpoise conservation, four of them aimed to evaluate the effect of fishery activities on porpoise populations based on data collected on boats and in strandings. These indicators could be further improved by the estimation of the origin of stranded porpoises and the distribution inferred from stranded porpoises with bycatch evidences. Anomalies in distributions would highlight critical areas with a high level of interaction with fisheries. This information is very important to assess small cetacean conservation status and could help in designing better adapted mitigation actions.

### 3.3.3. CASE 3: Indian Ocean–South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)

The IOSEA Marine Turtle Memorandum is an intergovernmental agreement that aims to protect, conserve, replenish and recover marine turtles and their habitat of the Indian Ocean and South-East Asian region. One of its objectives is to evaluate and reduce sea turtle bycatch in this vast area. In this case too, indicators based on strandings would inform on bycatch numbers by interpreting stranding data.

### 3.3.4. CASE 4: The Marine Strategy Framework Directive (MSFD)

The MSFD aims to restore and maintain GES in European waters by 2020. Eleven descriptors are to be considered, including biological diversity and the impact of anthropogenic activities. The use of long-term indicators dedicated to marine top predators could provide relevant information on abundance, distribution, causes of death and their impact on marine populations, and detect anomalies of these parameters compared to GES. These indicators can inform four of the 11 descriptors used for the restoration or maintenance of GES (the biological diversity, D1; elements of marine food webs, D4; marine litter, D10; introduction of energy, including underwater noise, D11).

With further methodological developments, the inferred distribution will allow calculation of the number of dead animals irrespective of drift conditions and the probability of being buoyant. Dividing population estimates provided by other monitoring programs by this mortality estimate could allow scientists to estimate mortality rate. As suggested in GES for cetaceans, mortality rate due to anthropogenic activity should not exceed 1.7%. The estimation of this indicator at cetacean population scale for different causes of death could monitor variations in mortality rates and prevent irreversible impacts on marine megafauna population. The fluctuation of mortality rates would inform on the restoration of GES at population scale.

This approach provides indicators at wide spatial and temporal range based on the tight cooperation between national stranding schemes in neighboring countries of the same marine region. Such cooperation is key to the implementation of MSFD (Borja, 2006; Bertram and Rehdanz, 2013; Zampoukas et al., 2013; Freire-Gibb et al., 2014).

These four examples related to existing regional regulations or agreements proposed different uses of indicators based on strandings but do not represent a comprehensive list of applications. Other applications for these indicators could be considered in many other contexts of national regulations (e.g. the Marine Mammal Protection Act in USA, MSFD in European Union), regional agreements (e.g. the Memorandum of Understanding concerning Conservation Measures for Marine Turtles of the Atlantic coasts of

Africa, the Agreement on the Conservation of Albatrosses and Petrels, ASCOBANS, IOSEA).

## 4. Conclusion

The aim of this work was to propose a context for the interpretation of marine megafauna stranding data sets. According to conservation objectives or current knowledge, the construction of a null hypothesis would allow for comparison of the observed situation with the expected one. This interpretation of stranding time series is based on the use of a drift prediction model.

The understanding of strandings would improve the spatial and temporal resolution of many monitoring strategies because stranding data are collected continuously in space and time, whereas other monitoring approaches are often dependent on periodic operations separated by extended gaps in knowledge. Their wide spatial and temporal range associated with high cost-effectiveness would highly improve the efficiency of many monitoring strategies in synergy with other available datasets. Improving the cost-effectiveness of frameworks, regulations or agreements enhance their longevity and efficiency.

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