

Agenda Item 4.1

Review of New Information on Other Matters
Relevant for Small Cetacean Conservation

Population Size, Distribution, Structure and
Causes of Any Changes

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**Marine megavertebrates adrift: a
framework for the interpretation of
stranding data in a monitoring
perspective with applications for
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area**

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Marine megavertebrates adrift: a framework for the interpretation of stranding data in a monitoring perspective with applications for small cetaceans in the ASCOBANS area.

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Summary:

For many marine species, stranding events remain the most important source of biological samples. Because of insufficient sampling strategies however, stranding data have long been under- or misused in the assessment of population conservation status. Even if many national and international regulations encourage the use of stranding data series in monitoring strategies, their interpretation must be improved. The aim of this paper is to provide a context for interpreting marine megafauna stranding data sets. Appendices provide examples of application in the ASCOBANS area. The first step is to construct a prior distribution under the null hypothesis H_0 . The prior distribution of theoretical dead animals can be set either uniform or according to available knowledge on abundance of marine vertebrates. The drift prediction of these theoretical carcasses will provide time series of stranding events expected under the null hypothesis. Back-calculation the reverse drift of observed stranding events highlights mortality areas of stranded animals. The correction of these areas by the probability to get stranded as determined by drift conditions would provide the distribution of dead animals inferred from strandings. The differences between observed and expected stranding data and the difference between inferred and prior distributions of dead animals are defined as the stranding and the distribution anomalies respectively. This work proposes a context for interpreting stranding data sets to be incorporated into monitoring strategies for all large marine vertebrates that are likely to float after death and get stranded. It is in line with priorities set up by the EU Marine Strategy Framework Directive and more specifically by ASCOBANS for small cetaceans.

Introduction

Top predator diversity can impact lower trophic levels (Heithaus et al., 2008; Wallach et al., 2009) and their conservation has to be considered for both marine ecosystem good health and biodiversity conservation. Many national and international regulations or agreements are aimed to assess, restore or maintain the conservation status of marine top predators. In most cases collecting data from dedicated methodologies is encouraged as a primary monitoring strategy (Evans and Hammond, 2004), in particular when absolute abundance estimates are required. However, collecting data at sea by using a standardized protocol generally remains quite expensive, because one has to incur the cost of the platform itself, which hinders the capacity to repeat this type of survey at sufficiently short intervals to detect trends in populations of pelagic megafauna. To minimise the cost of data acquisition, visual observations can be conducted from platform of opportunity, but the lack of dedicated sampling scheme can be a major disadvantage. For vocal species (mostly cetaceans), acoustic detections can be recorded from fixed devices or from moving platforms. For seabirds or pinnipeds, population estimates can be obtained from visual or photographic counts in breeding colonies or at haul-out sites. Nevertheless, these aggregations are usually segregated, notably by either sex, reproductive status or age, creating important biases that have to be estimated if population size is to be extrapolated from visual counts. Similarly, sea-turtles are often monitored by adult counts on nesting beaches where only the reproductive female segment of the population is present. Finally, stranded carcasses of marine megavertebrates also have long been a major source of biological information. The main benefit of this approach is to give access to tissue and organ samples without lethal issue for animals (Hall et al., 2010); therefore the scientific use of stranded animals has been encouraged for centuries (Hunter and Banks, 1787). For protected and vulnerable species, stranded carcasses are the main source of information on their biology, physiology, and health condition (Kirkwood et al. 1997; Geraci et al. 1999; Wilkinson & Worthy 1999; Jepson et al. 1999; Santos et al. 2001; Lahaye et al. 2005; McFee et al. 2006; Spitz et al. 2006; Hall et al. 2010; Tollit et al. 2010; Norman et al. 2011). Because of insufficient sampling strategies however, strandings data have long been under- or misused in the assessment of population conservation status (Wiese and Elmsie, 2006). Patterns in stranding frequency are often considered to be masked by drift-generated noise in data sets and their composition biased in favour of the weaker segments of the populations (new-born individuals, pregnant females, pathologic and older animals...). Nevertheless, strandings provide relevant information on species richness (Maldini et al.,

2005, Pyenson, 2011) and for rare and pelagic species can be the only source of evidences for their presence. Moreover, collecting stranding material and data is the best method available to estimate a variety of vital rates and identify natural and anthropogenic causes of death in top predators. For many marine species death in fishing gears is the most important cause of mortality (Reeves et al., 2013; Hamel et al., 2009; Prado et al., 2013, Kirkwood et al., 1997; Rogan and Mackey, 2007; Leeney et al., 2008; Murphy et al., 2009; Robins 1995; Lewison & Crowder 2003; Lewison et al. 2004; Read et al., 2006; Zeeberg et al., 2006; Tomas et al., 2008). As much as 75% of Odontocete, 66% of Pinniped, 66% of Mysticete, and all Sirenian species have been impacted by fishery activities over the past 20 years (Reeves et al., 2013).

Most regional agreements and regulations promote the assessment and reduction of by-catch. It is commonly admitted that mortality induced by interactions with fisheries should be estimated by dedicated observer programs. Nevertheless the implementation of such programs proves to be quite difficult because of insufficient regulation, administrative complexity or practical implementation difficulties. Important gaps in marine mammal by-catch data have been identified around the world (Reeves et al. 2006). By-catch estimates could also be approached by analyzing stranding data sets as this cause of death is often easily diagnosed on beached carcasses. Estimating the proportion of by-catch related death among stranded animals as well as its temporal and spatial patterns by analyzing stranding datasets would neither be hindered by heavy administrative or logistic constraints nor by regulation disparities between countries and fisheries. However previous studies concluded that strandings were not efficient in seabirds by-catch estimates (Zydelis et al., 2006), but when stranding representativeness 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 issues appeared in interpreting oiled marine mammals or seabirds during acute oil spill events (Degange et al., 1994, Hlady and Burger, 1993, Hope Jones, 1978, Piatt et al., 1990, Williams et al., 2011). Interpretation of oiled carcasses recorded during beach surveys was also improved by drift experiments (Lloyd et al. 1974; Piatt et al. 1990; Hlady & Burger 1993; Flint & Fowler 1997) or by incorporating proxy of oil spill risk. Finally, many natural and anthropogenic causes of death are detected through the examination of stranded carcasses.

The relevance of stranding records is substantially improved by the joint analysis of physical parameters (wind, tides, currents, aspects of the coast line), biological components (abundance, natural or anthropogenic mortality) and societal aspects (discovery, reporting)

involved in the complete stranding process. Spatiotemporal patterns in stranding follow a complex function of biological, physical and societal factors (equation 1).

Equation 1

$$N_{\text{stranding}} \sim \text{Abundance} \cdot \text{mortality} \cdot \text{buoyancy} \cdot \text{drift} \cdot \text{reporting}$$

Relative 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 (daily changes in direction and speed) and tidal currents (6-hourly changes in direction, lunar oscillations in strength) which greatly vary in the short to very short term. By comparison, carcass buoyancy and reporting efficiency, although difficult to estimate, are not supposed to vary greatly in the short term. The probability that a stranded carcass would be discovered and reported ($P_{\text{reporting}}$) is governed by physical and societal aspects. Indeed, access to the coast line can be made difficult due to the nature of the coast and public awareness is needed so that lay users of the coast know that their observations are worth reporting to the authorities. Public awareness would vary in time and space (Evans et al. 2005), mostly at large scales (decades, hundreds of kilometers). Buoyancy is another key contributing factor, as it determines the proportion of carcasses that float, and how high above the surface they float. This can vary greatly between taxa: cetaceans are close to neutrally buoyant whereas seabirds are positively buoyant; but short term variation is unknown.

The aim of this work is to provide a conceptual context of stranding interpretation which can be applied in all oceans and for all large marine vertebrates (seabirds, marine mammals, sea turtles...) based on the understanding of drift conditions in the stranding process. Specific examples for small cetaceans in the ASCOBANS area (harbor porpoise - Peltier *et al.*, 2013; common dolphin - Peltier *et al.*, *in review*) are given as appendices.

General design

The general principle is to determine a prior distribution of marine megafauna (either theoretical, or from actual data) to determine its stranding probability at any location of the study area and to create a predicted stranding data sets by using a drift model over a given period of time and across a given study area (Figure 1). Conversely, the observed stranding data set for the same area and period can be used to generate an inferred distribution of dead animals at sea, when corrected by stranding probability. Finally, stranding anomalies are defined as the difference between observed and predicted strandings, whereas anomalies in distribution/mortality are defined as the difference between the inferred and prior distributions of dead megafauna (Figure 1).

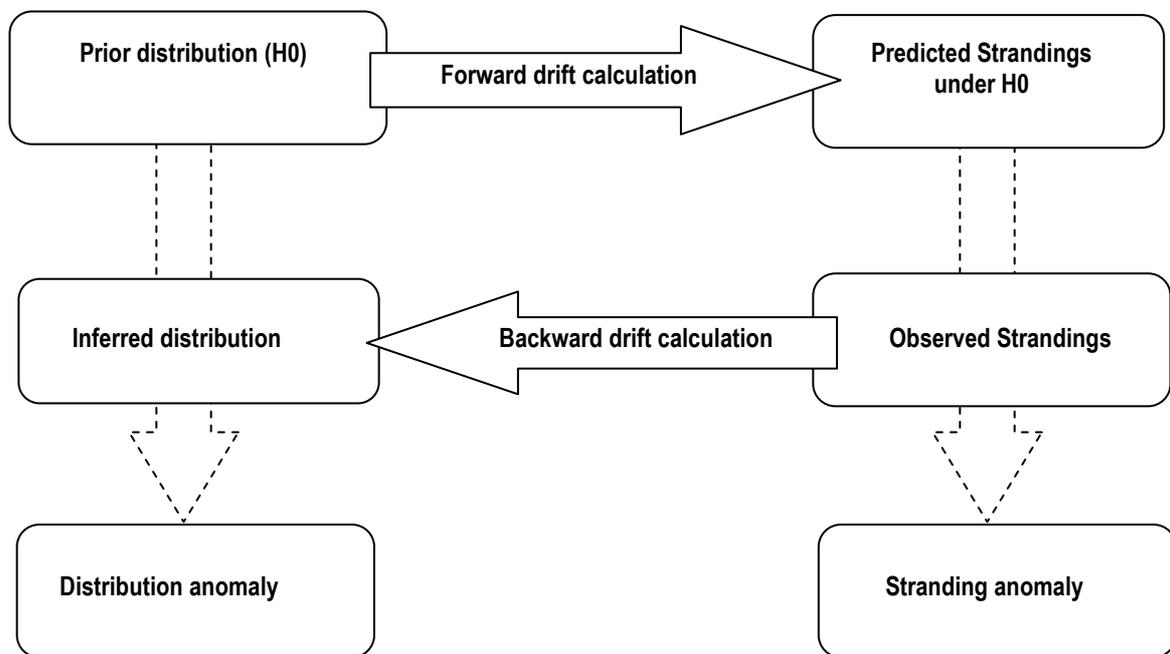


Figure 1: Conceptual framework for the interpretation of marine megavertebrate stranding data series.

This general strategy can be divided up into four main steps (Figures 1) and twelve detailed steps (Figure 2). Firstly, a prior distribution of dead animals at sea has to be defined according to the scientific or monitoring objectives and available knowledge. The prior distribution is analogous to a null hypothesis (H0) and will allow predictions to be made. In the absence of information over the study area, the prior distribution can be set uniform; otherwise actual distribution data or outputs of distribution models can be used. Secondly the drifts of these theoretical animals are modeled over the study period and area to estimate if they would strand as a result of day to day or even hourly drift conditions (wind, tides, currents...)

encountered during 30 days after their time of death. The theoretical carcasses predicted to strand constitutes the expected stranding data set, as predicted under the null hypothesis. It can be described both spatially and temporally. At each location (*e.g.* each cell of a gridded map) across study area, the proportion of dead animals predicted to get stranding is defined as the stranding probability ($P_{\text{stranding}}$), that can be averaged at different time scale as needed (yearly, seasonal, monthly climatologies). Thirdly, the actual stranding data set collected by local stranding schemes constitutes the observed strandings that can be described both spatially (maps) and temporally (time series). Finally, by using the same drift model backwards, the reverse drifts of all observed strandings (calculated from stranding site to death location at sea) provide a map of stranding origin. When weighed by $1/P_{\text{stranding}}$, it would represent the distribution of dead cetaceans at sea inferred from observed strandings. Differences between observed and expected strandings and differences between inferred and prior distributions are the stranding and distribution anomalies respectively (Figure 1). Peltier et al. (2013), given in appendix 1, exemplifies these first three steps as well as the stranding anomaly on the basis of a collaborative analysis of harbor porpoise stranding in the ASCOBANS area from data sets collected in Denmark, Germany, the Netherlands, Belgium, the United Kingdom and France. Peltier et al. (in review), given in appendix 2, exemplifies all steps as well as the stranding and distribution anomalies on the basis of a collaborative analysis of common dolphin stranding data in the ASCOBANS area from data sets collected in the United Kingdom and France.

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. The model MOTHY, initially designed by MétéoFrance for calculating the drift of oil slicks and of solid objects of interest to maritime safety (Daniel et al., 2002; 2004), was adapted to small cetaceans (Peltier et al., 2012; 2013). Several parameters are to be considered: carcass thickness, buoyancy rate, drift duration, drift conditions (wind and current), as well as date and location of drift start (*i.e.* date and location of death in forward drift or date and location of stranding in backward drift modeling) are considered.

Thickness can be measured on fresh carcasses and averaged for all stranded animals. Buoyancy rate is one of a key parameter in drift prediction as it determines how the carcass is exposed to air *versus* water movements. An experiment conducted on two small cetacean carcasses kept in a floating pontoon during 45 days allowed buoyancy rate of small cetaceans (delphinids and porpoises) to be estimated at about 90% immersion, which means that 90 % of body volume is under the surface (Peltier et al. 2012). The duration of drift is also critical. Visual criteria on skin degradation were used to estimate post-mortem drift duration over 30 days from pictures (Peltier et al. 2012). Another study examined changes in seabird carcass buoyancy and estimated floating duration at no more than 8 days (Wiese 2003). Before predicting drift of any marine top predators, it is required to examine how buoyancy changes as decomposition progresses.

Prior distributions and predictions

Prior distribution under the null hypothesis and predicted strandings

The prior distribution is a theoretical distribution of dead animals at sea and constitutes the null hypothesis. The number of theoretical dead animals in the study area depends on calculation capacities in drift prediction modeling and of the expected spatiotemporal resolution. It can be set either uniform or based on current knowledge about megavertebate abundance and distribution in the study area. For a flat prior distribution, mortality and abundance are supposed constant in time and uniform in space. In this case, patterns in predicted strandings would reflect the effects on stranding data of patterns in drift conditions only. Using actual distribution and abundance data as a prior distribution implies a comprehensive knowledge of top predator distributions in the area, including changes in time. In this case, patterns in predicted strandings would reflect patterns in drift conditions together with spatiotemporal heterogeneity in distribution.

Stranding probability and origin of predicted strandings

As explained above, various distributions of theoretical dead animals can be constructed. The drift of these animals from death locations onwards is calculated at a regular time interval by using a drift prediction model. The duration of the interval and the temporal span must be defined in relation with the expected spatiotemporal resolution and the temporal span of the wind archives to be used as entries to the model. For each theoretical animal at each selected date, a value of 1 is attributed to the cell of origin (death location) if stranding is predicted to

occur and 0 if 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 is defined as stranding probability $P_{\text{stranding}}$. When scaled up to the total number of observed stranding, this map represents the origin of expected strandings. The map of stranding probabilities is a requirement in stranding analysis as it highlights the areas of likely origin of stranded animals and is key to the interpretation of stranding patterns. $P_{\text{stranding}}$ is heavily dependent on distance to coastline and orientation of the coast relative to prevailing winds. At some distance $P_{\text{stranding}} = 0$, indicating that dead megavertebrates originating from offshore areas cannot be found stranded.

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. These first steps are illustrated in appendix 1 for the harbor porpoise in the North Sea and Atlantic parts of the ASCOBANS area (Peltier et al., 2013).

Observations and inferences

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 for a monitoring perspective. Stranding data are typically composed of the following fields: species, date, location, decomposition code, sex, and measurements; and can be associated to a variety of biological samples or recordings (notably necropsies, tissues samples and pictures) which in turn can be analyzed to produce individual-specific information such as age, reproductive state, general health status, cause of death, diet, contaminant load, population genetics,... In the present context, species, locations and dates are the primarily required data, together with decomposition code or any other proxy of postmortem drift duration such as pictures (Peltier et al., 2012). 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.

Origin of observed strandings

The use of a drift prediction model allows back-calculating drift trajectories (from stranding place and date 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 or according to its reported decomposition status (Peltier et al. 2012). The inferred death locations for all observed strandings can be displayed on maps of the actual stranding likely origins.

Inferred distribution

To take into account spatial patterns in stranding probability $P_{\text{stranding}}$, the origin of observed strandings is multiplied by $1/P_{\text{stranding}}$ to infer the distribution of dead animals at sea from stranding records. At greater distance from the coastline, where $P_{\text{stranding}} = 0$, no inference can be made. In order to limit uncertainty at the fringe of the area where $P_{\text{stranding}} > 0$, it is advised to truncate cells where $P_{\text{stranding}}$ is very low (e.g. < 0.05 or < 0.10). This final step provides a gridded map of numbers of dead animals found stranded corrected by $P_{\text{stranding}}$, which is the number of dead animals drifting at sea. The number of dead drifting 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 calculation area, it is crucial to carefully estimate the proportion of carcasses that float after death. This proportion varies greatly among marine megavertebrates, in particular in diving taxa whose buoyancy is generally close to neutral, and therefore limited changes in condition can make it float or sink. As an example, it was recently estimated that about 13 to 18% small delphinids would float at death in the eastern North Atlantic (Peltier et al. 2012).

Anomalies

Stranding anomaly

The stranding anomaly is defined as the difference between observed strandings and strandings expected under H_0 (illustrated for harbour porpoise in the ASCOBANS area in appendix 1; Peltier et al. 2013). To do this, expected strandings must be scaled to the total number of observed stranding so that overall stranding anomaly is 0 over the whole study period and area. Local positive (vs negative) anomalies suggest that more (vs less) strandings were observed than expected under H_0 . When H_0 is set uniform, stranding anomaly shows

changes in abundance and/or mortality of stranded animals. When H0 is set as an actual distribution, stranding anomaly highlights changes in mortality rate of stranded animals.

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. Map of expected origin must be calibrated on the map of observed origin so that the overall anomaly in stranding origin summed over the study area and period is zero. This anomaly reflects changes in origin of stranded animals compared to H0: the anomaly is positive when drifting dead animals are more abundant than expected under H0 and the anomaly is negative when they are less abundant than under H0.

Distribution anomaly

The distribution anomaly is the difference between prior distribution and distribution inferred from strandings. The two distributions are scaled up to the same totals across study area and period and the distribution anomaly is calculated in each cell of the gridded map. When H0 is set uniform, positive (vs negative) anomalies highlights areas where abundance and/or mortality are higher (vs lower) than in the prior 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 H0 is set as an actual distribution, positive (vs negative) anomalies would highlight areas where mortality is higher (vs lower) than in the prior distribution. Hence the distribution anomaly would highlight spatial patterns in mortality alone.

The complete process is illustrated in appendix 2 for the common dolphin in the Bay of Biscay and Western Channel of the ASCOBANS area (Peltier et al., in review).

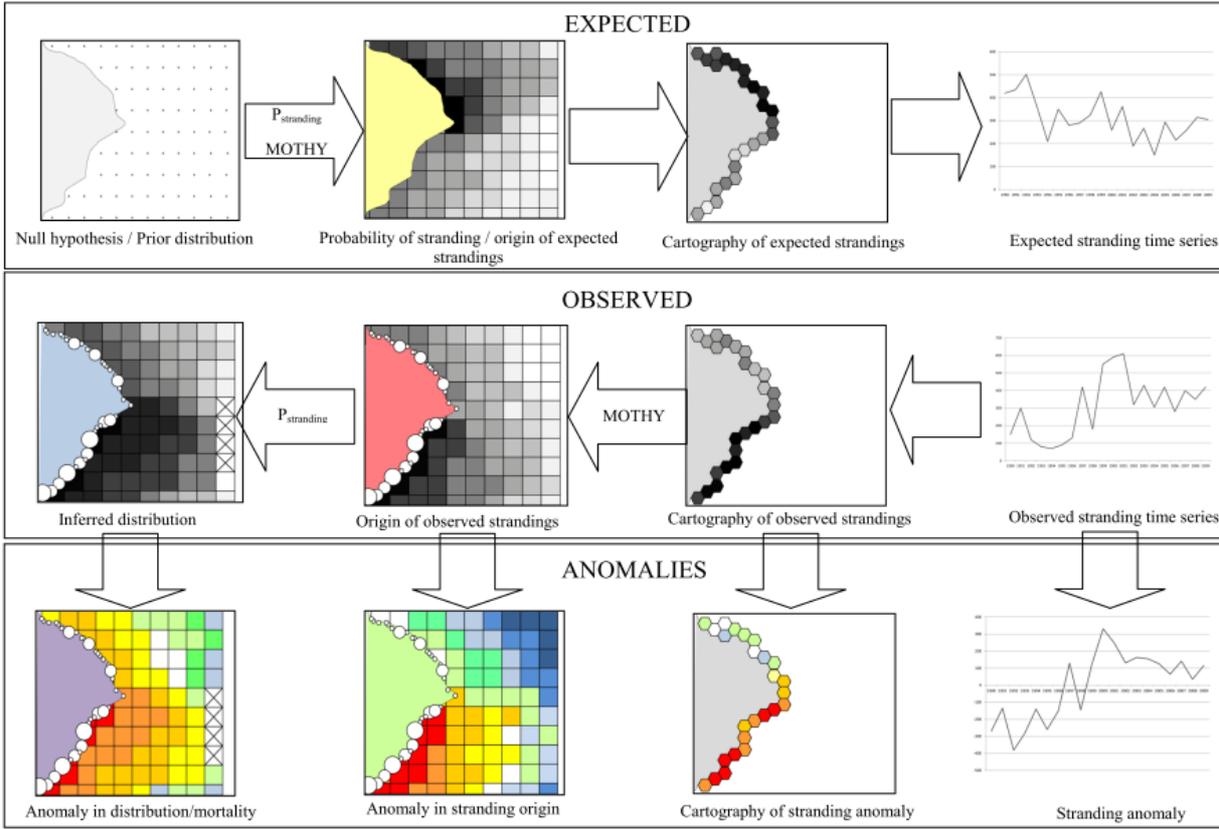


Figure 2: Step-by-step framework for interpreting megavertebrate stranding data sets: prior/expected and observed/inferred patterns in at-sea distributions and strandings and the corresponding anomalies.

Discussion

General

Because it is scientist role to provide interpretation tools of 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 be beached ashore and for a variety of marine species, including cetaceans, seabirds, sea-turtles. After setting a hypothesis on the distribution of the marine megafauna of interest, a probability to get stranded can be estimated and an expected stranding data set can be constructed to be compared to actual stranding records. Conversely, actual stranding data can be converted into an inferred distribution of dead animals. The differences between observed and expected strandings are defined as the stranding anomalies. High stranding anomalies reveal conditions of particular conservation concerns, either spatially or temporally, and are thus obtained by disentangling variations in the biological components of strandings (the parameters to be monitored) from possible variation due to drift conditions, the physical component of strandings and major confounding. This can be done under the assumption that carcass buoyancy and stranding reporting do not change at the spatiotemporal scale that is relevant for monitoring.

One of the most important steps is the choice of the drift prediction model. The size and buoyancy of marine megafauna suggest that model predicting drift of large dimension objects are more suitable for carcass modeling than models designed for calculating the movements of water masses or of small objects such as fish or invertebrate eggs or larvae. The adaptation of the model to top predator carcasses can be based on in situ experiments. Several parameters can affect the sinking probability of dead animals at sea (causes of death, health status, body condition...) and it can be considered that carcasses sunk on the seafloor are quickly consumed by scavengers and most often lost for the stranding process (Jones et al. 1998; Baco & Smith 2003; Smith & Baco 2003). Hence, carcass buoyancy is one of the most sensitive parameter in drift simulations (Peltier et al., 2012). The spatial and temporal range of the model is also crucial in using strandings for conservation purposes. Moreover, the integration in drift simulations of atmospheric models, tide models and currents could improve the efficiency of trajectory prediction.

Producing spatial indicators

Various steps of the present interpretation framework for stranding data sets could be used as indicators, generally represented as maps.

In the context of increasing utilization of marine habitats (marine renewables, dredging and marine aggregate extraction, gas and oil exploration and extraction, marine traffic, fisheries...), human pressures on cetaceans and other 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 these activities on marine species. This would provide indicators of the impact of these activities on top predators.

The likely origin of observed stranding can be used to spatialize samples collected on stranded animals as well as the biological information derived from their analyses. For many protected species like cetaceans and other large marine vertebrates, strandings remain the main source of biological samples. Back-calculating sampled carcasses drift would allow mapping biological parameters at sea. It would become possible to *inter alia* map dietary information by locating carcasses according to their stomach contents. Maps of carcasses according to traits of life could be considered in order to detect potential short term segregation in the population. Maps of stranding origin can also be used to locate tissue samples used in population structure analyses (M. Louis, unpublished, for bottlenose dolphins in the ASCOBANS extension area).

The distribution anomaly would probably be one of the most relevant indicators in the context of conservation strategies. It detects anomalies in the number and distribution of dead animals at sea compared to a prior distribution. The prior distribution can be constructed either as a hypothetical situation set uniform or according to existing knowledge on the distribution of marine megafauna. The interpretation of anomalies requires complementary information, notably on man-induced mortality. In the context of marine megafauna conservation, the detection of critical areas with high mortality level or abundance is a major criteria when marine protected areas are to be designated.

Monitoring and conservation strategies

The present work has developed a context for interpreting stranding data sets and proposed different population indicators based on beached carcasses. Providing cost-efficient indicators based on strandings is an increasing need in monitoring strategies endorsed by a number of regional, EU or national policies and agreements.

The OSPAR convention aims to protect and conserve the North-East Atlantic Ocean and its resources and was adopted by fifteen European countries. Many indicators are monitored to provide information on 12 parameters (e.g. climate change, radioactive substances, eutrophication...). Among them, the number of oiled common guillemots (*Uriaa algae*) informs 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. Their stranding origin could be identified by using the present framework, which could provide additional relevant information on mortality areas of oiled seabirds. Another of the OSPAR indicators aimed to ensure that the by-catch rate of harbour porpoises is maintained under 1.7% of recent population abundance estimates. This indicator relates to actions proposed in the harbour porpoise conservation plan of the Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas (ASCOBANS). Among twelve indicators dedicated to harbour porpoise conservation, four of them aimed to evaluate the effect of fishery activities on porpoise populations based on data collected on boats and from strandings as well. These indicators could be further improved by estimating the origin of stranded porpoises and the distribution inferred from stranded porpoises with by-catch evidences. Anomalies in distributions would highlight critical areas with high level of interactions with fisheries. This information would be crucial to assessing small cetacean conservation status and could help designing better adapted mitigation actions.

The Marine Strategy Framework Directive (MSFD) aims to restore and maintain a Good Environmental Status in European waters by 2020. Several descriptors are to be considered, including biological diversity and the impact of anthropogenic activities. In this context, the long term use of indicators dedicated to marine top predators and based *inter alia* on the interpretation of stranding data series could provide relevant information on distribution, causes of death and mortality fields, and detect anomalies of these parameters compared to the Good Environmental Status.

These 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.

Besides these applications, the following cases illustrate some other 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.

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

Early questioning about stranding representativeness appeared for interpreting strandings recorded during various oil spills in recent decades (e.g. oil spills of the *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 & 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 "Cetacean Unusual Mortality Events" procedure of investigation and monitoring during 13 months. This event engaged scientists to re-examine the issue of the stranding representativeness relative to actual mortality at sea (Williams et al. 2011). This work did not consider the importance of drift conditions in the stranding process. The use of distribution data as a null hypothesis would allow calculating inferred distribution and hence monitoring mortality rate of marine mammals, seabirds, turtles before, during and after the oil spill, in the fraction of the monitored area where $P_{\text{stranding}}$ would be > 0 .

Finally, interactions with fisheries remain one of the most important causes of death for many marine large vertebrates. In the European Union (EU), regulation CE812/2004 requires dedicated programmes of observers on 5% of >15 m vessels of a selection of métiers. In most EU countries this regulation is rather poorly implemented with important spatial and temporal heterogeneity in sampling effort that impaired by-catch estimation efficiency. The origin of stranded animal indicator would highlight major areas of interactions between fisheries and top predators. Inferred distribution of stranded small cetaceans diagnosed as by-catch would

allow numbers of animals caught in all fisheries combined (not only those identified by EU regulation and with an actual monitoring programme being implemented) to be estimated and mortality areas identified. If an actual distribution was used as the prior distribution, anomalies in distribution would inform on the spatial pattern of mortality in fisheries. Hence, the combination of data collected by observers on board, origin of stranded top predators with by-catch evidences and inferred distribution of these animals would improve the comprehension of by-catch process and help designing relevant mitigation actions.

References

- Baco, A. R., and C. R. Smith. 2003. High species richness in deep-sea hemoautotrophic whale skeleton communities. *Marine Ecology Progress Series* **260**:109–114.
- Daniel, P., Josse, P., Dandin, P., Lefevre, J., Lery, G., Cabioc'h, F., Gouriou, V., 2004. *Forecasting the Prestige oil spills*. Interspill Conference, 14–17 June 2004, Trondheim, Norway.
- Daniel, P., Jan, G., Cabioc'h, F., Landau, Y., Loiseau, E., 2002. Drift modelling of cargo containers. *Spill Science Technology Bulletin* 7 (5-6), 279–288.
- Degange, A. R., A. Doroff M., and D. Monson H. 1994. Experimental recovery of sea otter carcasses at Kodiak Island, Alaska, following the Exxon Valdez oil spill. *Marine mammal science* **10**:492–496.
- Epperly, S. P., J. Braun, A. J. Chester, F. A. Cross, J. V. Merriner, P. . Tester, and J. H. Churchill. 1996. Beach Strandings as an indicator of at-sea mortality of sea turtles. *Bulletin of Marine Science* **59**:289–297.
- Evans, P. G. H., and P. S. Hammond. 2004. Monitoring cetaceans in European waters. *Mammal review* **34**:131–156.
- Flint, P., L., and A. Fowler C. 1997. A Drift Experiment to Assess the Influence of Wind on Recovery of Oiled Seabirds on St Paul Island, Alaska. *Marine Pollution Bulletin* **36**:165–166.
- Frantzis, A. 1998. Does acoustic testing strand whales? *Nature* **392**.
- Garshelis, D. L. 1997. Sea Otter Mortality Estimated from Carcasses Collected after the Exxon Valdez Oil Spill. *Conservation Biology* **11**:905–916.
- Geraci, J. R., J. Harwood, and V. J. Lounsbury. 1999. Marine Mammal Die-Offs. Pages 367–395 In Twiss, J.R. and Reeves, R.R. (Eds) *Conservation and Management of Marine Mammals*. Smithsonian Institution Press, Washington.
- Gordon, J., D. Gillespie, J. Potter, A. Frantzis, R. J. Swift, and D. Thompson. 2004. A review of the Effects of Seismic Surveys on Marine Mammals. *Marine Technology Society Journal* **37**:16–34.
- Hall, A. J., F. M. D. Gulland, J. A. Hammond, and L. H. Schwacke. 2010. Epidemiology, disease, and health assessment. Pages 144–164. In Ian L. Boyd, W. Don Bowen, Sara J. Iverson, (Eds). *Marine Mammal Ecology and Conservation: A Handbook of Techniques*. Oxford University Press, Oxford.
- Hamel, N. J., A. E. Burger, K. Charleton, P. Davidson, S. Lee, D. F. Bertram, and J. K. Parrish. 2009. Bycatch and beached birds: assessing mortality impact in coastal net fisheries using marine bird strandings. *Marine Ornithology* **37**:41–60.

- Hart, K. M., P. Mooreside, and L. B. Crowder. 2006. Interpreting the spatio-temporal patterns of sea turtle strandings: going with the flow. *Biological Conservation* **129**:283–290.
- Heithaus, M. R., A. Frid, A. J. Wirsing, and B. Worm. 2008. Predicting ecological consequences of marine top predator declines. *Trends in Ecology and Evolution* **23**:202–210.
- Hlady, D., A., and A. Burger E. 1993. Drift-Block Experiments to analyze the mortality of oiled seabirds off Vancouver Island, British Columbia. *Marine Pollution Bulletin* **26**:495–501.
- Hope-Jones, P., J.-Y. Monnat, C. J. Cadbury, and T. J. Stowe. 1978. Birds oiled during the *Amoco Cadiz* incident - An Interim Report. *Marine Pollution Bulletin* **9**:307–310.
- Hunter, J., and J. Banks. 1787. Observations on the structure and oeconomy of Whales. By John Hunter, Esq. F. R. S.; Communicated by Sir Joseph Banks, Bart. P. R. S. *Philisophical Transactions of the Royal Society B* **77**.
- Jepson, P. D. et al. 2003. Gas-bubble lesions in stranded cetaceans. *Nature* **425**:575–579.
- Jepson, P. D., P. M. Bennet, C. R. Allchin, R. J. Law, T. Kuiken, J. R. Baker, E. Rogan, and J. K. Kirkwood. 1999. Investigating potential associations between chronic exposure to polychlorinated biphenyls and infectious disease mortality in harbour porpoises from England and Wales. *Science of the Total Environment* **243/244**:339–348.
- Jones, E. G., M. Collins, P. M. Bagley, S. Addison, and I. G. Priede. 1998. The fate of cetacean carcasses in the deep sea: observations on consumption rates and succession of scavenging species in the abyssal north-east Atlantic Ocean. *Proceedings of the Royal Society B* **265**:1119–1127.
- Kirkwood, J. K., P. M. Bennet, P. D. Jepson, T. Kuiken, V. R. Simpson, and J. R. Baker. 1997. Entanglement in fishing gear and other causes of death in cetaceans stranded on the coasts of England and Wales. *Veterinary record* **141**:94–98.
- Koch, V., H. Peckham, A. Mancini, and T. Eguchi. 2013. Estimating At-Sea Mortality of Marine Turtles from Stranding Frequencies and Drifter Experiments. *PLoS ONE* **8**:e56776.
- Lahaye, V., P. Bustamante, J. Spitz, W. Dabin, K. Das, G. J. Pierce, and F. Caurant. 2005. Long-term dietary segregation of common dolphins *Delphinus delphis* in the Bay of Biscay, determined using cadmium as an ecological tracer. *Marine Ecology Progress Series* **305**:275–285.
- Leeney, R. H., R. Amies, A. C. Broderick, M. J. Witt, J. Loveridge, J. Doyle, and B. J. Godley. 2008. Spatio-temporal analysis of cetacean strandings and bycatch in a UK fisheries hotspot. *Biodiversity and Conservation* **17**:2323–2338.
- Lewis, R. L., and L. B. Crowder. 2003. Estimating fishery bycatch and effects on a vulnerable seabird population. *Ecological Applications* **13**:743–753.
- Lewis, R. L., L. B. Crowder, A. J. Read, and S. A. Freeman. 2004. Understanding impacts of fisheries bycatch on marine megafauna. *Trends in Ecology and Evolution* **19**:598–604.
- Lloyd, C., S., J. A. Bogan, and W. R. P. Bourne. 1974. Seabird mortality in the North Irish Sea and Firth of Clyde early in 1974. *Marine Pollution Bulletin* **5**:136–140.
- Maldini, D., L. Mazzuca, and S. Atkinson. 2005. Odontocete stranding patterns in the main Hawaiian Islands (1937-2002): How do they compare with live animal surveys? *Pacific Science* **59**:55–67.
- McFee, W. E., S. R. Hopkins-Murphy, and L. H. Schwacke. 2006. Trends in bottlenose dolphin (*Tursiops truncatus*) strandings in South Carolina, USA, 1997-2003: implications for the Southern North Carolina and South Carolina Management Units. *Journal of Cetacean Research Management* **8**:195–201.

- Murphy, S., Winship, A., Dabin, W., Jepson, P.D., Deaville, R., Reid, R.J., Spurrier, C., Rogan, E., López, A., González, A.F., Read, F.L., Addink, M., Silva, M., Ridoux, V., Learmonth, J.A., Pierce, G.J., Northridge, S.P. 2009. Importance of biological parameters in assessing the status of *Delphinus delphis*. *Marine Ecology Progress Series* **388**: 273-291.
- Peltier, H., Baagøe, H.J., Camphuysen, K.C.J., Czeck, R., Dabin, W., Daniel, P., Deaville, R., Haelters, J., Jauniaux, T., Jensen, L.F., Jepson P.D., Keijl, G. O., Siebert, U., Van Canneyt, O., Ridoux, V. 2013. The stranding anomaly as population indicator: the case of harbour porpoise *Phocoena phocoena* in North-Western Europe. *PLoS ONE* 8(4): e62180. doi:10.1371/journal.pone.0062180
- Peltier, H., W. Dabin, P. Daniel, O. Van Canneyt, G. Dorémus, M. Huon, and V. Ridoux. 2012. The significance of stranding data as indicators of cetacean populations at sea: modelling the drift of cetacean carcasses. *Ecological Indicators* **18**:278–290.
- Piatt, J. F., C. J. Lensink, W. Butler, M. Kendziorek, and D. Nysewander. 1990. Immediate impact of the *Exxon Valdez* oil spill on marine birds. *Auk* **107**:387–397.
- Prado, J. H. F., E. R. Secchi, and P. G. Kinas. 2013. Mark-recapture of the endangered franciscana dolphin (*Pontoporia blainvillei*) killed in gillnet fisheries to estimate past bycatch from time series of stranded carcasses in southern Brazil. *Ecological Indicators* **32**:35–41.
- Pyenson, N. D. 2011. The high fidelity of the cetacean stranding record: insights into measuring diversity by integrating taphonomy and macroecology. *Proceedings of the Royal Society B*.
- Read, A. J., P. Drinker, and S. P. Northridge. 2006. Bycatch of Marine Mammals in U.S. and Global Fisheries. *Conservation Biology* **20**:163–169.
- Reeves, R. R., K. McClellan, and T. B. Werner. 2013. Marine mammal bycatch in gillnet and other entangling net fisheries, 1990 to 2011. *Endangered Species Research* **20**:71–97.
- Robins, J. B. 1995. Estimated catch and mortality of sea turtles from the east coast otter trawl fishery of Queensland, Australia. *Biological Conservation* **74**:157–167.
- Rogan, E., and M. Mackey. 2007. Megafauna bycatch in drift nets for albacore tuna (*Thunnus alalunga*) in the NE Atlantic. *Fisheries Research* **86**:6–14.
- Santos, M. B., G. J. Pierce, J. S. Herman, A. Lopez, A. Guerra, E. Mente, and M. R. Clarke. 2001. Feeding ecology of Cuvier's beaked whale (*Ziphius cavirostris*): a review with new information on the diet of this species. *Journal of the Marine Biological Association of the United Kingdom* **81**:687–694.
- Smith, C. R., and A. R. Baco. 2003. Ecology of whale falls at the deep-sea floor. *Oceanography and Marine Biology: an Annual Review* **41**:311–354.
- Southall, B. L., R. C. Braun, F. M. D. Gulland, A. D. Heard, R. W. Baird, S. M. Wilkin, and T. K. Rowles. 2006. Hawaiian melon-headed whale (*Peponacephala electra*) mass stranding event of July 3-4, 2004. Page 78. *NOAA Tech. Memo.*, NOAA.
- Spitz, J., Y. Rousseau, and V. Ridoux. 2006. Diet overlap between harbour porpoise and bottlenose dolphin: An argument in favour of interference competition for food? *Estuarine Coastal and Shelf Science* **70**:259–270.
- Tollit, D. J., G. J. Pierce, K. A. Hobson, W. Don Bowen, and S. J. Iverson. 2010. Diet. Pages 191–221. In Ian L. Boyd, W. Don Bowen, Sara J. Iverson (Eds), *Marine Mammal Ecology and Conservation: A Handbook of Techniques*. Oxford University Press, Oxford.
- Tomas, J., P. Gozalbes, J. A. Raga, and B. J. Godley. 2008. Bycatch of loggerhead sea turtles: insights from 14 years of stranding data. *Endangered Species Research* **5**:161–169.
- Wallach, A. D., B. R. Murray, and A. J. O'Neil. 2009. Can threatened species survive where the top predator is absent? *Biological Conservation* **142**:43–52.

- Wiese, F. K. 2003. Sinking rates of dead birds: improving estimates of seabird mortality due to oiling. *Marine Ornithology* **31**:65–70.
- Wiese, F. K., and K. Elmslie. 2006. Underuse and misuse of data from beached bird surveys. *Marine Ornithology* **34**:157–159.
- Wilkinson, D., and G. A. J. Worthy. 1999. Marine mammal strandings networks. Pages 396–411. In Twiss, J.R. and Reeves, R.R. (Eds), *Conservation and management of marine mammals*. Smithsonian Institution Press, Washington.
- Williams, R., S. Gero, L. Bejder, J. Calambokidis, S. D. Kraus, D. Lusseau, A. J. Read, and J. Robbins. 2011. Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conservation Letters* **4**:228–233.
- Zeeberg, J., A. Corten, and E. de Graaf. 2006. Bycatch and release of pelagic megafauna in industrial trawler fisheries off Northwest Africa. *Fisheries Research* **78**:186–195.
- Zydelis, R., M. Dagys, and G. Vaitkus. 2006. Beached bird surveys in Lithuania reflect oil pollution and bird mortality in fishing nets. *Marine Ornithology* **34**:161–166.