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Report on potential emerging innovative monitoring approaches, identifying potential reductions in monitoring costs and evaluation of existing long-term datasets

Deliverable 4.3

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BOX 1. DEFINITION OF TERMS

AIS – *Automatic Identification System;*

AMBI – *AZTI Marine Biota Index;*

BACI – *Before-After-Control-Impact;*

BIMEP – *Biscay Marine Energy Platform;*

CODA – *Cetacean Offshore Distribution and Abundance in the European Atlantic;*

CTD – *Conductivity, Temperature and Depth profilers;*

CV – *Coefficient of Variation;*

DSLR – *Digital Single Lens Reflex;*

EIA – *Environmental Impact Assessment;*

EIS – *Environmental Impact Statement;*

EU – *European Union;*

HD – *High-Definition;*

ICES – *International Council for the Exploration of the Sea;*

MarEI – *Marine Renewable Energy Ireland;*

MRE – *Marine Renewable Energy;*

MS – *Member States;*

ORE – *Offshore Renewable Energy;*

ROV – *Remotely Operated Vehicle;*

SCANS – *Small Cetaceans in the European Atlantic and North Sea;*

SDM – *Survey, Deploy, Monitor;*

SMRU – *Sea Mammal Research Unit;*

UCC – *University College Cork;*

VMS – *Vessel Monitoring System;*

WGMME – *Working Group for Marine Mammal Ecology;*

WP – *Work Package;*

WGMME – *Working Group for Marine Mammal Ecology.*



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RiCORE Project Synopsis

The aim of the Risk-based Consenting for Offshore Renewable Energy (RiCORE) project is to establish a risk-based approach to consenting where the level of survey requirement is based on the environmental sensitivity of the site, the risk profile of the technology and the scale of the proposed project. The RiCORE project, which has received funding from the EU's Horizon 2020 research and innovation programme, will run between January 1st 2015 and June 30th 2016.

The consenting of offshore renewable energy is often cited as one of the main non-technical barriers to the development of this sector. A significant aspect of this is the uncertainty inherent in the potential environmental impacts of novel technology. To ensure consents are compliant with EU and national legislation, such as the EIA and Habitats Directive, costly and time-consuming surveys are required even for perceived lower risk technologies in sites that may not be of highest environmental sensitivity.

The RiCORE project will study the legal framework in place in the partner Member States (MS) to ensure the framework developed will be applicable for roll out across these MS and further afield. The next stage of the RiCORE project is to consider the practices, methodologies and implementation of pre-consent surveys, post-consent and post-deployment monitoring. This will allow a feedback loop to inform the development of the risk-based framework for the environmental aspects of consent and provide best practice. The project will achieve these aims by engaging with the relevant stakeholders including the regulators, industry and EIA practitioners, through a series of expert workshops and developing their outcomes into guidance.

The impact of the project will be to improve, in line with the requirements of the Renewable Energy Directive specifically Article 13 (1), consenting processes to ensure cost efficient delivery of the necessary surveys, clear and transparent reasoning for work undertaken, improving knowledge sharing and reducing the non-technical barriers to the development of the ORE sector so it can deliver clean, secure energy.

1. INTRODUCTION

In order to ensure the timely exploitation of our oceans and future sustainable development of marine renewable energy (MRE), the way must be paved for efficient streamlined cost-reducing EIA procedures in all MS. The main aim of the RiCORE project is to ensure the successful development of the sector in EU MS by reducing the cost and time taken to consent projects of low environmental risk, through the development of a risk-based approach during projects' consenting. This type of approach has already been developed in Scotland (Survey Deploy and Monitor Approach) and its application across Europe (with appropriate adaptations to each MS) may be a way of standardising the assessment of key components of environmental risk from MRE deployment.

In order to implement a risk-based approach through utilising the SDM approach, the existing requirements for pre-consent surveys in the EU MS were first assessed (Simas & Henrichs 2015). Generally such pre-consent surveys may be part of a preliminary site characterisation exercise or scoping as part of the EIA process. Different approaches are followed by EU MS during this licensing phase, which were reviewed in order to assess how well existing methods can be optimised across the EU, taking into account the potential positive implications for project timescales and costs (Simas et al. 2015). The principal objectives of the current deliverable are outlined in Section 1.1, and primarily focus on the potential application of state-of-the-art novel technology to monitoring programmes and broadly reviewing the financial cost of monitoring programmes. The overarching key outcome of the work developed under WP4 is to develop guidance for pre-consent surveys considering the spectrum of survey requirements for projects under SDM and existing project experience.

1.1 Objectives

The objectives of the present deliverable are to highlight the potential for using emerging and innovative technologies for pre-consent surveys of key receptor groups at proposed MRE sites and to identify potential reductions in cost through comparison



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of survey methods currently utilised. This deliverable will also review and examine patterns and trends in data from long-term studies to investigate how interpretation of data changes over time and what the implication of these findings has on defining a suitable survey duration for gathering baseline data, where required.

2. POTENTIAL FOR USING EMERGING AND INNOVATIVE MONITORING TECHNOLOGIES

Deliverable 4.2 summarised the typical approaches and methods used for collecting data across seven key receptors: 1) physical environment, 2) marine mammals, 3) fish and shellfish, 4) benthos and seabed habitats, 5) seabirds, 6) bats and 7) other users (socio-economy) (Simas et al. 2015). Within the corresponding sections of D4.2, emerging and innovative monitoring technologies that are beginning to be applied to offshore surveys, or may be feasible in the near future, were considered, such as high definition digital photography and/or video for seabird surveys. Continuing on from D4.2, Section 2 of this report will detail some of the emerging and innovative monitoring technologies that are showing promise for the monitoring of some of the aforementioned receptor groups during the pre-consent phase of offshore MRE developments.

2.1 High-Definition photography and video

Considerable advances in HD photography and video technology in recent years has led to their relatively successful application to seabird surveys (Mellor et al. 2007, Hexter 2009, Thaxter & Burton 2009, Buckland et al. 2012). Indeed, a comparison of HD video and stills (photography) with real-time visual surveys has shown that the former produced appreciably higher abundance estimates (Buckland et al. 2012). Conversely, for marine mammals, a preliminary study comparing marine mammal sightings from visual aerial surveys with images from HD video and a DSLR camera collected concurrently, found that fewer animals were identified in the HD video than by the observers, whilst the results were generally comparable between DSLR and visual observers (Koski et al. 2013). However, the authors issued a note of caution that more data were required, particularly across varying survey conditions. As such, the application of HD photography and/or video to marine mammal surveys have not been as successful, with the principal concerns relating to the influence of environmental conditions on sightings and species identification (Koski et al. 2013). Nevertheless,

recent improvements since these studies were undertaken, including an improved ability to identify individuals to species level (both marine mammals and seabirds) and increased strip width of the cameras, giving greater coverage of the development area as compared to visual aerial or boat-based surveys (Mackenzie et al. 2013) does suggest that HD photography and/or video will supersede visual aerial and boat-based offshore surveys for seabirds and marine mammals in the near future (where circumstances and logistics allow).

With respect to underwater HD video footage (often obtained using ROVs or diver surveys), efforts have been made to develop software tools that enhance image quality and eliminate (as much as possible) particle irradiation. These tools must be robust to certain external factors, such as variable light conditions and turbidity, which are common in a non-structured environment such as the marine environment. In parallel, robust video imagery tools have also been developed in MatLab environment for the automatic identification, detection and quantification of marine species of interest, such as mussels or commercially valuable fish species, therefore reducing the necessary time for image processing and analysis. This work is currently being developed under the Demowfloat project (<http://www.demowfloat.eu/>) (WavEC 2015) and follows previous work described by Marques (2011) and Rao & Chen (2012).

2.2 Unmanned Aerial Systems

Of the innovative monitoring technologies that are being progressively advanced, unmanned aerial systems (UAS) are probably one of the more likely to be applied to offshore surveys in the not too distant future. The attraction of UAS for replacing traditional manned aerial surveys comes from the potential to provide an improved method for monitoring, particularly for seabirds and marine mammal populations through: reduced cost, reduced human risk, increased accuracy of detection, location and identification of species and/or obtaining a permanent record of the survey (Hodgson et al. 2010). However, earlier reviews of available UAS deemed the equipment as too expensive and/or did not meet basic requirements for offshore biological surveys (Koski, Abgrall, et al. 2009, Koski, Allen, et al. 2009, Hodgson et al.

2010). More recent studies have shown that these platforms have great potential for near-shore environments on a relatively inactive species, *Dugong dugon*, close to the water surface (Hodgson et al. 2013). A review of over 600 UAS, published in 2010, considered several criteria, including size, cost, payload capacity, flight duration, speed, sensor capabilities and video resolution to assess each UAS potential for real-time survey platforms for marine mammals in offshore areas (Koski et al. 2010). Of these 600, 8 were deemed to be suitable; however, the authors cautioned that none of the UAS had been tested in the field to establish their efficiency for detection of marine mammals (or seabirds) and that some of these UAS would likely need improvements before they could be used for offshore surveys.

For coastal and terrestrial regions, UAS have shown great potential for monitoring seabird colonies and nests as a preferred approach to the often disruptive and time-consuming ground surveys (Chabot et al. 2015, Weissensteiner et al. 2015) and for obtaining abundance estimates of pinnipeds whilst on land during the breeding season (Perryman et al. 2010, Goebel et al. 2015). However, at present and as far as we are aware, there are no examples of UAS being used offshore for monitoring seabirds or marine mammals. As technology continues to advance in both HD photography/video and in UAS, these options are likely to become more feasible both financially and with respect to their capabilities. However, at present, beyond the potential to use UAS at coastal/terrestrial seabird and pinniped breeding colonies/haul-outs that may be of concern during offshore MRE installation (e.g. close to where cables make landfall), UAS are not currently a viable replacement for manned aerial- or boat-based surveys.

2.3 Remotely Operated Vehicles

In recent years, Remotely Operated Vehicles (ROVs) have been widely adopted as alternatives or additions to seabed surveys carried out using more traditional methods such as divers or towed or drop-down platforms. As noted in D4.2, ROVs are often used to obtain imagery for seabed mapping, habitat distribution and species composition and abundance (Simas et al. 2015). Although manned submersibles do exist and can transport humans to these depths, ROVs are a more compact, portable

and practical alternative, without the element of human risk. They are often deployed to extend diver only surveys into deeper water, to survey difficult areas (Sheehan et al. 2010) and/or to survey larger areas in shorter periods of time. For example, Galparsoro et al. (2015) used ROV surveys to improve the knowledge of benthic habitats in deep water (>100m) on the Basque continental shelf, for which there was previously little information. Similarly, Bald et al. (2015) used a ROV in areas inaccessible to divers to film specific areas and activities of interest, including the installation of the submarine cable at the BIMEP.

Despite their well-established use in offshore surveys, advances in ROV technology, coupled with advances in HD photography, video and storage capacity are continuing to improve the quality and quantity of data that can be obtained by ROVs. For example, in Portugal during 2013 and 2014 an ROV has been used to monitor the potential impact on existing *Sabellaria* biogenic reefs (Almagreira beach, Peniche) within the deployment site of a wave energy device (Machado et al. 2014). To further improve the data obtained, a side scan sonar and a multibeam sonar could be attached to the ROV to provide HD imaging and relatively accurate 3D images of the physical environment features of the MRE site. As the multibeam imaging sonar results are not affected by water clarity it works whether it is stationary or moving at speed, this may be a powerful ROV tool, which has been developed by several manufacturers (e.g. <http://www.teledyne-reson.com> and <http://www.seabotix.com>).

In another example, the Underwater Time Of Flight Image Acquisition system (UTOFIA, <http://www.utofia.eu/>) a recently launched H2020 project, aims to develop a compact and cost-effective underwater imaging system for turbid environments. By using range-gated imaging, the system will extend the imaging range by a factor of 2 to 3 over conventional video systems and, at the same time, the system aims to provide video-rate 3D information (Figure 1). This would fill the current gap between short-range, high-resolution conventional video and long-range low-resolution sonar systems with the potential to extract additional parameters, such as the volume of

objects from the images obtained. Consequently, UTOFIA could offer a new and efficient *modus operandi* for ocean ecosystem monitoring.

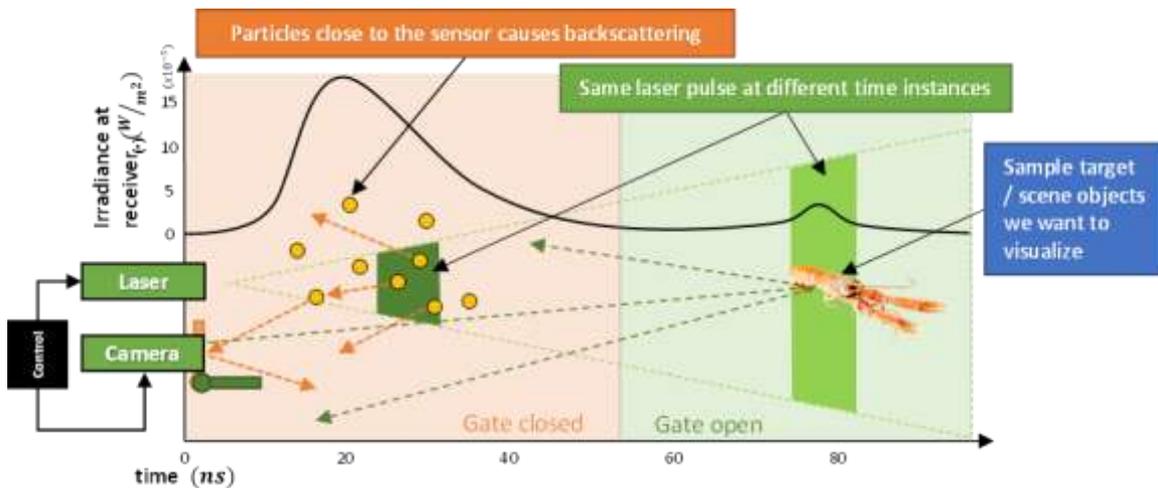


Figure 1. Range-gating reduces the effect of backscattering. In this figure an underwater object at a distance of ca. 9m is imaged. The graph shows the reflected signal from a laser pulse as a function of time. The first peak of the curve corresponds to backscattering from particles in the water. The second, attenuated peak corresponds to the reflection from the object that we are interested in (e.g., a lobster). The camera shutter is kept closed for approximately 50ns before it opens. Since the image is created from an integration of all light received, when the first 50ns is gated out, most of the backscattering contribution to the fundamental noise is removed.

2.4 High-frequency SONAR

For monitoring fish species, hydroacoustics, which is a non-invasive technique, is a method currently used in pre-consent surveys (Simas et al. 2015). Common hydroacoustic sampling techniques are based upon the use of split-beam scientific fishing echo sounders, such as SIMRAD EK60. Scientific sounders are made up of a power source, a transmitter-receiver, a laptop computer and one or several transducers operating at a different frequency, for example: 38, 120 and 200 kHz. The most commonly used frequency in hydroacoustic assessment of fishing resources is 38 kHz (Simmonds & Maclennan 2005), but information obtained at other frequencies can provide additional information that can assist in identifying other organisms (Fernandes et al. 2006). However, the major limitation of hydroacoustics is species

identification; to identify species, fishing hauls are used, typically done through pelagic gears, although other approaches such as purse seiners for detection ranges less than 50 metres (Boyra et al. 2013), can be used. The advantage of pelagic trawls is the possibility to sample at different depths; alternatively, purse seining can obtain a relatively small sample, with the benefit that the bulk of the animals captured can be released. Without this addition to the use of hydroacoustics, this method can only provide relative abundance and horizontal and vertical spatial distribution of biomass split into broad groupings (e.g. fish, plankton and krill; see Lezama-Ochoa et al. 2011). The new generation of split beam echo-sounders will move from narrow band to wide band, which will likely improve the species identification capacity of these systems in the near future (Stanton et al. 2010).

Of the innovative technologies that are being progressively advanced for better understanding of fish ecology, aspects of the dual-frequency identification sonar (DIDSON) developed by Sound Metrics (<http://www.soundmetrics.com/>) do provide great promise. The DIDSON has been used in shallow waters (particularly estuarine environments) to assist in environmental management for over a decade (Martignac et al. 2014). The acoustic camera uses higher frequencies and more sub-beams than more conventional hydroacoustic tools and, as such, provides near video quality images and allows observation of fish morphology and swimming behaviour. It is possible to measure fish length, which can also assist in species identification. However, its low detection distance, with associated decreased accuracy, has been cited as a limiting factor (Martignac et al. 2014), which could make its successful application in deeper offshore areas difficult. Since Martignac et al.'s (2014) review of the DIDSON, more recent advances in this technology have improved the DIDSONs ability to obtain video-quality images down to 300 metres, and Sound Metrics have since developed a range of Adaptive Resolution Imaging Sonars (ARISs) which have improved image clarity even in turbid waters, with a maximum range of 80 metres and a depth rating of 300 metres (<http://www.soundmetrics.com/>). Consequently, these devices do offer the potential to monitor, non-invasively, fish movement and

abundance, and provide species identification at the site of a proposed MRE development.

2.5 The FLOW, Water column and Benthic ECology 4-D (FLOWBEC-4D)

The FLOWBEC-4D is a device recently trialled at the European Marine Energy Centre (EMEC), Orkney, UK (Williamson et al. 2015). This device is a sonar platform that combines several instruments, including below-the-water instruments like sonars and above-the-water sensors like radar to record a range of information. Data are collected continuously for a period of 2 weeks, capturing an entire spring-neap tidal cycle. The data collected are over a wide range of both physical and multi-trophic levels (e.g. phytoplankton, zooplankton, fish seabirds, mammals). It is possible to identify fish species, and there is potential to identify seabird and mammal species, whilst all three groups of receptors can be tracked (above and below the water, where relevant for seabirds and mammals). Currently, techniques for analysing the raw data and statistical modelling are being refined. As such, this technology holds much promise for an integrated approach for monitoring several receptor groups; for example, detailed information on depth preference and interactions of birds, fish schools and marine mammals at proposed sites of MRE devices could be obtained, and individuals could be tracked to assess the likelihood of collision risks with turbines (Williamson et al. 2015).

2.6 Telemetry and other remote transmitters

Telemetry devices are well established in the study of marine mammals, and in particular, pinnipeds. As such, the majority of telemetry devices in Europe are designed and applied to pinnipeds (as licences for tagging cetaceans are unattainable or rarely applied for or issued, depending on the EU MS). Depending on the manufacturer and the specifications, there are a broad range of devices available from more basic models that provide location, samples of dive records, depth, temperature and speed to more sophisticated devices that can also provide information on oceanographic quality (e.g. temperature and salinity profiles). The longevity of devices

varies between a few months to several years; however, in practice, for pinnipeds, these devices are limited to a maximum of one year, as the seal will shed the tag during the annual moult (for example, in the UK the annual moult occurs between December and April for grey seals and in August for harbour seals). The volume of data, the interval and the lag in data retrieval also varies between devices. As technology advances, these tags are likely to become more sophisticated and further assist in pre-consent (and post-consent) monitoring of pinniped (and cetaceans in MS where licences are attainable e.g. Denmark; Sveegaard 2011, Sveegaard et al. 2011) habitat use, behaviour and movement patterns. For cetaceans, the principle logistical difficulties and welfare concerns pertain to potentially having to capture the animal and using invasive procedures to affix the tag to the animal (Sveegaard 2011, Sveegaard et al. 2011) or remotely fixing the tag at sea (e.g. tagging poles, cross-bows, firearms or air guns; see review by McIntyre 2014). In contrast, pinnipeds haul-out on land (capture opportunity) and have fur (tag attachment is not to skin and comes off when the animal moults). Non-invasive options for cetaceans do exist, such as suction cup telemetry tags, which may be suitable for shorter-term deployments (McIntyre 2014).

The disturbance effects on harbour porpoise population in the North Sea (DEPONS) project (<http://depons.au.dk/>) has tagged harbour porpoises in Danish waters to monitor the potential impact of noise generated by the construction phases of MRE developments. The project used Fastloc GPS tags, which were set to provide accurate positions approximately every 1.5 minutes (dependant on how often the animal surfaced). These tags provide detailed movement data for a period of up to 10 days. The tags need to be recovered in order to obtain the data; therefore, these tags are combined with Argos tags and VHF transmitters, which remain on the animal for up to 1.5 years. The tags were affixed using pin attachment (requiring two holes to be drilled through the dorsal fin; Teilmann et al. 2007). These data are intended to provide information on the movement and dispersal of these individuals before, during and after animals are exposed to noise associated with the construction-phase of a MRE development. For this project, the animals tagged were mostly accidentally caught in

pound nets (i.e. bycatch) but some were also actively caught in pound nets. To better inform movement and dispersal models, DEPONS aims to capture and tag porpoise in other regions of Danish waters and in Scottish waters. Should the DEPONS project provide valuable, unparalleled data (as compared to other methods herein, see also D4.2, Simas et al. 2015) for monitoring harbour porpoises near MRE devices, then this approach of invasive tagging of small cetaceans could be considered by other MS.

With respect to seabirds, there are a broad range of telemetry tags available; these include geolocators, radio tags, satellite transmitters, GPS, accelerometers and temperature depth recorders, all of which are reviewed in detail by Masden (2015). However, in her review of telemetry technologies in relation to the MRE sector and seabirds, Masden (2015) acknowledged that whilst devices continue to become smaller, they are not yet suitable for all purposes for all species of seabird. The main constraints highlighted in the review were the size and weight of tags, which limit the amount of data that can be collected simultaneously, and the longevity of the operational duration of the tags. Furthermore, Masden (2015) highlighted that no tags have the ability to remotely download Temperature Depth Recorder data.

Acoustic transmitters, which can give location, temperature and depth readings with no need to recapture the animal can be surgically implanted into fish; however, in an effort to increase battery longevity, the intervals between data collection tend to be coarser for these devices (as compared to telemetry tags for marine mammals) (Martins et al. 2014, Masden 2015). These devices have been used, with some success, in a recent bull trout monitoring programme, which assessed the risk of fish displacement from reservoirs to downstream waters through turbine intakes (Martins et al. 2014). The authors did report several issues with the transmitters, which included systematic and random errors associated with the number of receivers used and variability of detection efficiency (caused by noise from boat traffic, turbines and rain). Using a similar approach, Sims & Cotterell (2013) have developed a novel acoustic array-based fish tracking and monitoring programme, trialled at the Wave Hub, Cornwall. These unique 'seabed landers' house data-logging receivers that

monitor the movement of commercially valuable fish species tagged with acoustic transmitters. This array-based approach has the potential to investigate the movement of fish species in the area of a proposed MRE development to better understand impacts, such as collision risk on fish species. As such, acoustic transmitters may well prove to be a valuable tool for monitoring fish populations to better understand migration routes, habitat use and the potential for collision risks.

In summary, as telemetry devices and acoustic tags continue to evolve, they will allow for more data types to be collected from smaller devices on shorter temporal scales over longer time periods. Consequently, these will become increasingly valuable tools for monitoring animals across several receptor groups (e.g. fish, seabirds and marine mammals) and the characteristics of the fine-scale physical environment used by these groups.

2.7 PAM devices

Wilson et al. (2013) have trialled the use of C-PODs (<http://www.chelonia.co.uk/>; see Section 3.1.2 for more information on this device) as drifting PAM devices in tidal areas. Wilson et al. (2013) adapted these PAM devices by affixing a GPS unit and attaching the device to a drifting drogue and surface float that are deployed upstream and recovered for redeployment once the current has carried them beyond the site. This system allows for the mapping of odontocete vocal detections within tidal areas and can be used to investigate temporal variation across low speeds and tidal phases, for example. As the survey effort is effectively uncontrolled, the metric of effort is perhaps best based on time spent within cells of a spatial grid, rather than linear travelling distance; as such, these data are not capable of informing on absolute abundance (Wilson et al. 2013). Nevertheless, it does provide relative densities of vocalising odontocetes in a tidally active area and it can provide other environmental data, such as flow speed and background noise, that are likely useful in other applications, such as environmental modelling (Wilson et al. 2013).

As highlighted by Sparling et al. (2015), PAM systems will continue to improve. The areas they highlighted for improvement were: 1) an increased storage capacity 2) for electronic packages to get smaller, more reliable and cheaper and 3) for devices to be more streamlined/hydrodynamic so that they would be better suited for tidal current sites. The combination of these features should make for easier and more reliable data collection on presence/absence data for odontocetes. Sparling et al. (2015) also discuss PAM developments in progress at the SMRU, which include small bottom mounted arrays that should allow for the calculation of bearings to sound sources and, with two or more arrays, cross bearings could be obtained, which can give the location of vocalising animals. At present, locating animals is typically a limitation of PAM devices; therefore, this has clear potential for better understanding the potential of collision risk, as odontocetes (so long as they are vocalising), can be tracked throughout the water column. An alternative approach to obtaining these data are drifting vertical hydrophone arrays; Gordon et al. (2011) trialled a simple 4 element vertical array to test the feasibility of obtaining data on underwater movements and dive behaviour of porpoise at tidal sites. This work has progressed to 10 element, vertically orientated arrays that can track vocalising animals in 3D, which can assist in predicting collision risk (Macaulay 2010). The principal drawbacks of this technique is that it requires a technically competent and experienced team to operate it, coupled with the relevant costs required for a suitable vessel for deployment, could make this an unfeasibly expensive approach. At present, an effort is being made to develop a more affordable system that can yield the same data (Sparling et al. 2015). With respect to drifting arrays, the data collected are limited in some respects (i.e. uncontrolled, unplanned and uneven effort) but they do have the potential to provide valuable data on underwater movements and dive behaviour, which are important for better informing collision risk models (Sparling et al. 2015). With further development, data obtained from multiple arrays may also be able to provide information on density, which is yet another common limitation to the PAM devices typically used at present. As Sparling et al. (2015) note, additional development is required to obtain

these functions; however, the components and the software for the most part already exist.

2.8 VMS to monitor vessel traffic and fishing activity

Capture fisheries are major users of the seas, which cover a diverse range of commercial fisheries using both smaller (<12 m) and bigger vessels (≥12 m). The latter are covered by the satellite-based Vessel Monitoring System (VMS), and those over ≥300 Gross Tonnes are additionally covered by the Automatic Identification System (AIS) (prior to 2012, the size criteria for smaller and bigger vessels was <15m and ≥15 m, respectively). The latter of these systems is a maritime navigation safety communications system used to provide vessel information, primarily for the purposes of maritime safety. AIS data provides a source of information that can be used to spatially represent vessel movements within the receiving range of transmissions, with signals broadly classified as ‘Class A’ and ‘Class B’. AIS-A is carried by international voyaging ships of ≥300 gross tonnage (GT) and all passenger ships regardless of size, whereas AIS-B is a non-mandatory form of AIS typically used by small commercial craft, fishing vessels and recreational vessels; as such a very small proportion of the fishing fleet are fitted with these devices. Data from AIS is routinely used in a pre-consent desk-based review of vessel traffic in the area of a proposed MRE development (see D4.2, Simas et al. 2015).

In addition to the application of AIS for understanding spatial and temporal use of the seas, there is also VMS, which is a fisheries compliance tool offering bi-hourly location data that can be linked to landings information (European Commission 1997). Complementing VMS data with catch data can provide information on the spatial and temporal distribution of fishing activity and landings (Eastwood et al. 2007, Bastardie et al. 2010); this information could be used to assess the potential impact of MRE developments on commercial fisheries, for example. However, smaller vessels (<12 m) do not carry VMS and most are not fitted with AIS devices either. As such, the availability of spatial data is often limited to coarse sea areas (ICES rectangles) coming from logbook data. However, in Scotland, a novel approach to baseline

characterisation of inshore fishing activity for smaller vessels and commercial fisheries used participatory data collection (face-to-face interviews with fishermen) as well as data from AIS devices to map activity (Kafas et al. 2014, MMO 2014). Therefore, to provide a complete picture of vessel traffic and fisheries activity, all forms of information should be considered (AIS-A, AIS-B, VMS, radar, visual observations and interviews with fishermen) (Kafas et al. 2014, MMO 2014) and used, where required, as part of a desk-based study (and a field study, if required). Given that these data are now more readily accessible (e.g. AIS) and/or are available via the relevant authorities (e.g. VMS), this all-encompassing approach is becoming more viable.

2.9 RADAR

Radar systems for tracking birds are progressively becoming more sophisticated. For example, the Merlin radar system (DeTect Inc., Panama City, Florida, USA), was used to monitor seabirds, post-consent, at the Egmond aan Zee Offshore Wind Farm (OWEZ) (the Netherlands) (Hartman et al. 2012). The system consisted of two radars and dedicated software designed to record bird activity. The first radar rotated horizontally and recorded the spatial patterns, flight routes, migration routes and avoidance of the wind farm and turbines. The second radar rotated vertically and recorded information on flight heights and intensities of birds. The radars scanned an area up to 5.6 km around it and up to 1.4 km above it (Hartman et al. 2012).

The system is operational 24 hours a day, during poor weather, and can be accessed and controlled remotely from offices on the mainland. Bird echoes are automatically logged into a database as the signal is taken directly from the radar and is filtered using algorithms developed specifically for recording bird flight activity (Krijgsveld et al. 2011). With each recorded echo, the Merlin system can record a large number of parameters, including exact location, direction, speed and altitude. However, one limitation is the ability to identify species (Krijgsveld et al. 2011); this may be an important limitation if a particular species is of concern. Other companies, for example Robin Radar Systems (<http://www.robinradar.com/>), have developed radar systems that can also track wing-beat frequencies, which in combination with flight

characteristics (behaviour, speed, movement) and echo characteristics (size and shape) could potentially give an insight into species composition, solely using radar. The radar systems developed by Robin Radar Systems have recently been applied to pre- and post-consent monitoring of birds at offshore MRE sites in Norway and Estonia, for example (<http://www.robinradar.com/environmental-references/>). As radar systems continue to advance both technologically and in their application to offshore sites, their use will likely become common practice, given the distinct advantage of being able to gather data both during poor weather and night-time. The latter of which would be beneficial for better monitoring of nocturnal as well as the diurnal occurrence and behaviour of birds at offshore MRE sites.

3. IDENTIFY POTENTIAL REDUCTIONS IN COST THROUGH COMPARISON OF METHODS

As identified in the 1st RiCORE expert Workshop (Simas & Henrichs 2015), seabirds and marine mammals are often the most challenging and controversial of the several receptors to overcome. The principal issues identified are the typical requirement to gather data over multiple years, with surveys covering all seasons and/or important life-history events (e.g. breeding season, moulting season). As such, these surveys are often the most costly and logistically difficult due to the nature of the study species (e.g. highly mobile, covering large areas). Furthermore, there is a wide range of approaches for gathering data on these species, which will be dependent on the requirements of the monitoring programme, the energy harnessed (i.e. wind, wave or tidal, see Simas et al. 2015) and the location of the site. Consequently, this section will first focus on potential cost reductions in monitoring seabirds and marine mammals as a function of cost per unit effort, followed by an overview of potential reductions in cost for surveying two other receptor groups: 1) the physical environment and 2) fish. All costs will be presented in € for consistency (relevant exchange rates were calculated as £1 = €1.30 and \$1 = €0.88 using <http://www.xe.com/> on 12/Oct/2015).

3.1 Marine mammals and seabirds

3.1.1 Aerial and boat-based survey approaches

In a 2010 report, MacLeod et al. (2010) conducted a comprehensive review of cost per unit effort for marine mammal surveys (Table 1). Their costings were based on the cost of charter and observers only, and the hours of effort were based on data obtained during the SCANS-II survey, except for aerial surveys where MacLeod et al. (2010) based the calculations on hourly charter rates and assumed the ratio of transit/survey time. The cost per hour and per km of effort were not given in monetary value, rather they expressed these relative to the cheapest method, which was a towed hydrophone array on a platform of opportunity. Therefore, when interpreting Table 1, ship-based double platform line transects are 51 times more expensive than a towed hydrophone

array on a platform of opportunity and both cost per hour and cost per km of effort is 205 times more expensive. This illustrates the point that the charter costs are the biggest outlay for ship-based surveys. Aerial surveys benefit from the ability of covering more track line in a relatively shorter period of time, as compared to ship-based surveys, which therefore reduces the charter costs. In turn, towed acoustic arrays have the benefit of being able to gather data during night-time and in worse sea conditions, so can yield more data at relatively lower cost. Ultimately, the method, or the combination of methods used will depend on the nature and requirements of the pre-consent monitoring. Specifically, the methods do provide different data per unit effort, where some are better at detecting certain species than others and some are more suitable for particular logistic constraints, for example. Therefore, whilst effort is standardised in Table 1, the data obtained are not the same and may not be suitable for the purposes of a particular monitoring programme, as such selecting an option on a financial basis without first considering the requirements of the monitoring project is not advised.

Table 1. Standardised costs of visual and acoustic cetacean survey methods. Daily costs and Cost Per Unit Effort (CPUE) figures are expressed relative to the cheapest method (PoOP towed array); DP = Double Platform, SP = Single Platform, LT = Line Transect, PoOP = Platform of Opportunity. Table replicated from McLeod et al. (2010).

Method	Hours on effort	Daily field costs	Cost per hour of effort	Cost per km of effort
Ship-based DP LT	5.5	51	205	205
Aerial DP LT	4	29	158	16
Ship SP LT	5.5	26	103	103
Aerial SP LT	4	27	147	15
Towed hydrophone array	22	6	6	6
PoOP visual survey	5.5	4	16	16
PoOP towed survey	22	1	1	1

Evans & Thomas (2011) provided costings for a dedicated cetacean monitoring programme in UK waters. For inshore and nearshore regions they identified vessel-based double platform line transects as the most suitable option; based on a vessel of

10-15m at a charter rate of €95 per hour and a survey team of six persons (@ €34 per hour) covering 200km over 10 hours they estimated the cost to be €2,990. For the offshore areas, they opted for aerial surveys and costed 25 survey days with 28 overnight stays as €163,132, with 3 persons as €22,169, totalling €185,301. Aerial surveys are typically conducted between 165-205 km per hour; therefore, in this rather specific example, where the vessel surveys cover 200km in 10 hours, the aerial surveys can cover 200km in one hour and require a substantially reduced staff, once again highlighting the financial benefit of aerial surveys for cetacean and seabird surveys, particularly for offshore surveys (i.e. with respect to transit time to line transect). Since the time of publishing, inflation has likely increased the costs reported in Evans & Thomas (2011) by approximately 10% to 15%.

Where seabird surveys are required, aerial surveys with dedicated marine mammal and seabird observers, either identifying both groups concurrently, or taking observations of seabirds and marine mammals independently, would be the cheaper option per unit effort. This is particularly true for large sites as aerial surveys can cover more area than ship-based surveys (as noted above). However, important considerations would include development type, as surveys should be continued post-consent to allow for direct comparisons in the metric of interest (e.g. species' abundance) between the phases of the development. Therefore, for offshore wind farms no-fly zones could be an issue, which may then favour ship-based surveys. For marine mammal and seabird aerial surveys currently underway in large offshore areas off the west-coast of Ireland, MaREI noted that the time and cost saved, as compared to vessel-based surveys, was considerable. Furthermore, the short time required to complete aerial surveys also enabled year-round survey coverage in winter months, where available daylight becomes limiting. Alternatively, ship-based surveys would need to be conducted over multiple days to cover required visual line transect lengths in daylight hours, with associated extra personnel and accommodation costs. Additionally, the short times needed to complete line transects using aerial surveys enable surveyors to take advantage of short weather windows, particularly in high

energy environments such as the Atlantic where sea states suitable for surveying (Beaufort <4) can be limited.

The preferred method of aerial survey be it visual and/or HD photography/video will depend on several factors; for example, if particular species are of greater concern, then identification to species level will be important, therefore, with HD photography and/or video there are data to evaluate post-survey, making species identification more likely, given that observers only having a couple of seconds in real-time to see the animal. So there are trade-offs between the higher costs of HD and the reliability of detection rates and species identification obtained from visual aerial surveys.

3.1.2 Static Passive Acoustic Monitoring

Static passive acoustic monitoring (PAM) for marine mammals is becoming a more common tool in pre-consent surveys, the most common of which in Europe is the C-POD (<http://www.chelonia.co.uk/>), which detects odontocete vocalisations in the range of 20 – 160kHz. These are priced at approximately €4,000, with additional costs between €250 and €500 for moorings (concrete blocks, chain, rope) per C-POD, depending on depth and turbidity of the area. In addition, 10 D-Cell batteries are required to run C-PODs (approximately €20 per deployment, per C-POD). Additional costs not included here are staff time for setting up, retrieving and processing the data, or the cost of deployment and retrieval every 3-5 months. Wilson et al. (2013) provide a promising method for using C-PODs to drift in tidally active areas to obtain information on presence/absence of porpoise (see Section 2.7); the additional costs to those outlined above are likely to be minimal.

There are other static PAM devices on the market, such as microMARS (<http://desertstar.com/product/micromars/>) that operate within a broader frequency range than C-PODs, such as 25 - 250kHz, in this case. Depending on the specifications, the cost of these devices range from €1,750 to €3,100. These are small devices (c. 20cm x 6.5cm) that can operate for 10-12 days on one D-Cell battery. The device can mark data segments of potential interest and can be optimised for high or low

frequency sampling, operating at maximum depths ranging from 300m to 4,000m, depending on the model. The sensitivity and frequency range of microMARS is set by the hydrophone end-cap based on the characteristics of the sound source (marine mammal, industrial noise, etc.) to be studied, and the expected ambient or background noise level. The end-caps, which cost around €440, are easily replaceable. The smaller size makes deployment simpler with less floatation (small hard float c. 1.5-4kg) and anchor weight (c. 7-10kg) required, which could be done from small vessels. The device can be retrieved using an acoustic release mechanism that costs approximately €1,750 per device and a software charge of €2,600 that synchronises with all acoustic releases.

Wildlife acoustics (<http://www.wildlifeacoustics.com/>) produce the Song Meter SM3M Deep water, which is a long-deployment bioacoustics recorder and noise logger, recording between 2Hz - 192KHz, which can be equipped with different hydrophones depending on the users' requirements. This device can be deployed at a maximum depth of 800m. The standard unit price is c. €9,250 with additional hydrophones ranging from €1,700 - €2,600. The same company produce a model, Song Meter SM3M Submersible, for shallower water (up to 150m) that operates within the same frequencies at a unit price of €5,930 and additional hydrophones at the costs stipulated above. Depending on the duty cycle and the frequencies recorded within, both of these devices can record from 26 to 1,236 days.

Despite the common use of PAM devices in cetacean monitoring programmes, many have limitations with respect to detecting particular species (typically due to the range of frequencies the device is operating within) or cannot distinguish between species, which is particularly true for delphinids as they often produce highly variable calls that overlap to a large degree with other species. As such, caution should be exercised when using automated processes to identify species of interest (Caillat et al. 2013), as misclassification may result in data that are not fit for purpose, and as such provide no benefit to the species' monitored. Yet, some species, such as large whales, have distinctive acoustic calls that can be identified by experienced PAM operators and/or

have reasonably efficient automated call classifiers. Therefore, this may require further consideration if there is a requirement to monitor a specific cetacean species, other than the harbour porpoise (which is more readily identified due to vocalisations in higher frequencies), for example. This brief review of a small number of available devices currently used in the field to monitor cetaceans gives a general introduction to the variation in several key parameters when discussing static PAM, including operating frequency range (i.e. which species can be detected), duration of deployment, ease of deployment, overall cost. For a more comprehensive comparison of static PAM devices (including the C-POD and the predecessor to the SM3M, the SM2M), see Sousa-Lima et al. (2013), for an in-depth review of capabilities, costs and ease of deployment for over 30 PAM devices.

3.1.3 Telemetry tags

The cost of telemetry tags for both marine mammals and seabirds vary substantially depending on a number of factors. For marine mammals, the majority of telemetry devices in Europe are designed for pinnipeds. Satellite tags are attached externally to the animal and transmit a signal to the Argos satellite system or GPS satellite system. Depending on the manufacturer and the specification of the tags, prices can range from €4,000 for the more basic Argos telemetry tags to €7,000 for the more sophisticated tags with the ability to record oceanographic data (e.g. temperature, salinity and fluorescence profiles). With respect to seabirds, basic GPS tags can range between €40-€500 depending on manufacturer, amount of waterproofing and deployment duration. Argos satellite tags range between €1,000-€1,500 depending on configuration, but do enable the user to obtain the data without needing to recapture the animal. Basic Time Depth Recorders for obtaining dive depths range between €400-€500, whilst more sophisticated GPS, Time Depth Recorder accelerometer tags can be upwards of €1,000 depending on configuration. More information on specific tags for seabirds can be found in Masden (2015) (which does not provide costings).

The prices provided here are only approximations of cost from a small variety of manufacturers. Given that this field is rapidly evolving, new and innovative telemetry tags are regularly being developed, with many developers each with expertise in different species, data acquisition and duration of recording, for example. Consequently, more specific specifications and costings of telemetry tags would be study specific, as such, for more specific information, these are just some of the companies currently manufacturing telemetry devices for fish, sea birds and marine mammals:

- Cefas (<http://www.cefastechnology.co.uk/>),
- Lotek (<http://www.lotek.com/>),
- Sirtrack (<http://www.sirtrack.com/>),
- SMRU instrumentation (<http://www.smru.st-andrews.ac.uk/Instrumentation/>),
- Wildlife Computers (<http://wildlifecomputers.com/>)

3.2 Physical environment and benthos

Different methods can be used to assess benthic communities such as divers, drop-down cameras and ROVs. *In situ* sampling by divers presents the poorest cost efficiency as the area covered by divers is limited and costs are comparably higher. The use of an ROV is normally an expensive alternative to systems such as drop-frames; yet for surveys of large seabed areas ROVs are often the better option. In calm, nearshore conditions, a small ROV can be operated from vessels as small as 6m with a minimum of equipment and crew. In contrast, conducting safe, quantitative surveys with a small ROV in more extreme marine environments increases the complexity of the operation and requires additional equipment and personnel to ensure success. ROVs can be equipped with additional sampling gear (e.g. claw-and-suction samplers depth sensor, compass, and two parallel laser beams) to obtain more detailed data; however, the size of the ROV will determine the payload, manoeuvrability and uses of the vehicle (Rees 2009). It is important to note, that in areas with relatively high current speeds, the effect of drag on the cable may cause problems and, in current speeds greater than 1.5 knots, smaller ROVs may struggle to operate effectively (Rees 2009).

ROVs are particularly useful when more detailed information on abundance, size, and morphology of large organisms is needed. However, limitations with respect to image quality typically mean that identification of sessile epifauna smaller than 2cm is not possible (Mitchell & Coggan 2007). Nevertheless, the use of still images (photographs) obtained during the survey may be able to assist in species identification of smaller taxa such as gastropods and stone crabs, but only if the camera is close enough to the seabed at the time the image is taken (Coggan et al. 2009). Data processing is a desk-based task, analysing photographs and video imagery, which requires less time and is less resource consuming as compared to *in situ* sampling, which, in comparison requires long laboratory screenings and equates to more effort in terms of both human resources and consumables (Mitchell & Coggan 2007).

The comparably higher cost of divers over ROVs is highlighted in a case study from Portugal in 2014 and 2015, where professional divers were subcontracted at a cost of €2,500 per day to collect benthos samples at artificial reefs in order to assess the potential impact of a floating offshore wind turbine. Conversely, to perform similar work, the rental of a ROV Seabotix LBV200 (with laser scaling, positioning system and sonar), including the required personnel and boat rental, was €1,900 per day. The equipment on board the ROV included two cameras, one for navigation, which can be moved remotely from the surface and a HD GoPro with a resolution of 1080p for capturing video footage of the site. These costs are similar to those incurred for the seabed and benthos community characterisation at BIMEP (at depths ranging between 50m and 90m) where an underwater video camera attached to a Seaeye Falcon ROV was rented for €2,000 per day. For seafloor mapping, ROVs with multi-beam echosounders (MBE) are commonly used, ranging in cost depending on the depth. For example, seafloor mapping of the Basque continental shelf (SE Bay of Biscay) used a high-resolution SeaBat (ca. 100 m water depth) and EM3002D (ca. 200 m water depth) MBEs at a cost of €4,000 and €8,000 per day (excluding the cost of the ROV), respectively (Galparsoro et al. 2015).

3.3 Fish

Broadly speaking, there are two overarching techniques that can be used for monitoring fish: (i) capture methods (traps, seine nets and purse seines, selective fishing: trawling and dredging gears and angling and line fishing) and (ii) observation methods (visual census with SCUBA divers, underwater video cameras and hydroacoustics). The cost of these monitoring approaches depends on several factors, such as cost of personnel, equipment, shipping and laboratory analyses. Other factors, such as quality of the information provided by the methodology, the level of maturity of the methodology, the required level of expertise to undertake the sampling, analysis and data interpretation will also influence the cost and the decision about which methodologies and techniques are most suitable. In Table 2, each of the aforementioned monitoring approaches are assessed according to the following criteria:

- 1) **Level of maturity:** the assigned value varies between High (H), Medium (M) and Low (L) according to how widely the methodology is used.
- 2) **Technical costs:** costs of technical equipment. The value assigned varies between Low (L, €1,000 – 10,000), Medium (M, €10,000 – 50,000), and High (H, >€50,000).
- 3) **Personnel Expertise:** level of expertise required for sampling, analysis and data interpretation, the value assigned varies between High (H, high expertise and specialist skills required), Medium (M, trained personnel with specific professional skillset) and Low (L, trained personnel without specific professional skillset).
- 4) **Total Cost:** personnel, shipping, travel costs, etc. The value assigned varies between Low (L, €1,000 – 10,000), Medium (M, €10,000 – 50,000) and High (H, >€50,000).
- 5) **Quality of information:** the value assigned varies between High (H), Medium (M) and Low (L) according to accuracy and how detailed the information generated is.

Table 2. Comparison of fish sampling techniques according to their Level of Maturity (LM), Technical Cost (ThC), Expertise of personnel required (E), Total Cost (TC) and the Quality of information provided (QI), '-' indicates that the information is unknown or has not been evaluated.

	Technique	Typologies	LM	ThC	E	TC	QI
Capture Methods	Explosives/Ichthyocides	Rotenone	M	L	M	L	H
		Clove oil (anaesthetic)	H	L	L	L	M
	Traps	Barriers	L	L	L/M	L	M
		Pots	L	L	L/M	L	M
	Encircling and Vertical Seines	Seine nets	L	L	L/M	-	L
		Purse seine	L	L	L/M	-	L
		Drift nets	L	L	L/M	-	M/H
	Trawling	Semi pelagic trawling	H	L	L/M	H	H
		Bottom trawling	H	L/M	L/M	H	H
	Angling and line fishing	Vertical logline fishing	-	L	M	M	L
		Bottom-logline fishing	-	L	M	M	L
		Hand-lining	-	L	M	M	L
Observation Methods	Underwater Visual Census (UVC)	Divers	M	M	M	M	M
		Divers + camera	M	M/H	M/H	M/H	M
		Underwater video camera	L	M	H	M	M
		ROV	M	H	H	H	M
	Hydroacoustics	Split-beam scientific	M	H	H	H	L
		Fishing echo sounder	M	H	H	H	L
		DIDSON	M	H	H	H	M/H
		Buoys	M	L	M	M	L

3.3.1 Capture methods

a) Traps

Traps are a fixed fishing approach, where trap characteristics vary depending on the target species. This approach is used in capture and recapture studies, age studies, reproduction surveys and circadian activity studies, for example. The main advantage is that traps can be used at depths where divers cannot reach and the captured individuals can be freed alive and without serious damage once data collection has been carried out. The main disadvantages are: (i) Species' capture depends on the mesh size; (ii) predation on captured individuals may occur; (iii) some species can go in and out of the trap; (v) it is difficult to estimate the number of individuals per unit area; (vi) many repeat samples are needed to better understand differences between trap variance and low capture rates. Costs in general are low and the expertise needed and the quality of information provided is low/medium (Table 2).

b) Seine nets and Purse seines

Seine nets and purse seines obtain a larger number of captures in less time than traps or bating hooks. However, studies developed in reefs have shown abundance estimates of species with no commercial interest are not accurate and captures may depend on density and fish movement (foraging, migration, etc.). Consequently, these techniques are recommended for use in tandem with suitable sampling approaches for species without commercial interest (Acosta 1997). Costs in general are low and the expertise needed and the quality of information provided is low/medium (Table 2).

c) Selective fishing with trawling and dredging gears

Trawling is used to determine the species, size, age, reproductive status and biomass of schooling fish. It is also used to validate results from hydroacoustics and visual surveys made by divers, for example (Watson 2008). This capture method does not discriminate between organisms and its bottom modality can damage the benthic habitat (Kulbicki 1998, Bailey et al. 2007). Selecting a suitable gear depends on the species, type of seabed (sand, mud or rock) and the environment (demersal, pelagic) and must be complemented by other sampling techniques (diving with or without video cameras, ROVs, etc.) in areas of bedrock, for example, where dredging is not possible. The quality of the information obtained from these techniques is high, but so are the general costs (Table 2).

d) Selective fishing with angling and line fishing

The use of this approach, such as baited hooks, have the following advantages: they are affordable, replicate samples can be obtained quickly, they have a high survival rate of individuals captured and samples can be collected from depths much deeper than divers can reach (Willis et al. 2000). However, this approach also has its disadvantages, primarily due to biases in the factors affecting the fish captured (selectivity in species caught and their size, for example); consequently, community structure or abundance estimates are unlikely to be accurate. Moreover, logistics and personnel involved in this kind of sampling (e.g. vessel, fishermen and technicians) are

also rather demanding and Captures Per Unit of Effort (CPUE) will depend on fishermen's skill, which will vary. Last but not least, hooks may cause damage to the fish (e.g. natatory bladders, body or gill) and/or predation of fish on hooks may occur, which will increase mortality rate (Willis et al. 2000). General costs are considered to be medium and the quality of information as low (Table 2).

3.3.2 Observation Methods

a) Visual Census with SCUBA divers

This is a selective technique focused on size, appearance and behaviour of the target species and community. The main advantage lies in the fact that these are non-invasive techniques that can be repeated. As such large databases can be generated quickly allowing for information to be obtained on species abundance and diversity, for example. Methods for data collection include: line transects, strip transects, stationary method, random method, visual census and video recorded visual census. Data need to be gathered in optimal conditions of luminosity, turbidity and sea state; therefore, sampling should be carried out during the beginning of summer (more hours of daylight and less turbidity), in the morning (more light) and in good visibility conditions (low turbidity). However, there is a linear relationship between species detectability and the number of replicates; therefore, the greater number of replicates, the more species detected (MacNeil et al. 2008). The costs, expertise and the quality of the information obtained have been assessed as medium (Table 2).

b) Underwater video cameras

Thanks to technological advances, traditional visual census can be complemented or substituted by underwater video cameras via a variety of options: (i) fixed on a structure anchored to the bottom; (ii) operated by a diver (iii) ROV, human operated underwater vehicles (HOV) or autonomous underwater vehicles (AUV). At present, line transects are the most commonly used sampling method (Shortis et al. 2007); however, strip transects or a combination of both is also used.

The advantage of video cameras is that they are not restricted by immersion time or diver constraints (i.e. they can be used at greater depths), time of day (i.e. can be used during night-time) or selectivity of species, they are non-invasive techniques, provide a permanent record of the survey and data can be gathered following a standardised methodology (Watson & Quinn Li 1997, Cappo et al. 2006, Costa et al. 2006, Morrison & Carbines 2006, Heagney et al. 2007, Shortis et al. 2007, Stobart et al. 2007, Stoner et al. 2008, Watson 2008, Yoklavich & O'Connell 2008). Some disadvantages of this method are a consequence of: (i) cryptic and/or small species that are more likely to be missed; (ii) visibility limitations; (iii) repeated entries into the field-of-vision by the same individual that cannot be distinguished (Watson 2008); (iv) the density estimates generated are usually relative (e.g. maximum number of fish of a same species represented in the camera's field-of-vision at a given time).

Other considerations are that underwater video cameras can be baited to attract a greater number of individuals and species, which could be advantageous for detecting cryptic, less common species, for example. However, some potential biases may occur in the ability to identify fish species and their behaviour (i.e. fish may remain in front of bait and individuals may be obscured by other fish milling around the bait and/or as a consequence of current direction and turbidity) (Cappo et al. 2006, Heagney et al. 2007, Watson 2008). There are some potential direct and indirect effects (attraction, repulsion or indifference to survey equipment e.g. ROV with underwater camera) that may be caused by artificial light (intensity and wave length), sound (intensity and frequency) and speed and size of the ROV, for example, which will vary according to the environmental conditions and the way in which the ROV is operated (Trenkel et al. 2004, Stoner et al. 2008). In conclusion, general costs and expertise required are expected to be medium to high and the quality of information obtained is likely to be medium (Table 2).

c) Hydroacoustics

As stated in Section 2.4 common hydroacoustic sampling techniques are based upon the use of split-beam scientific fishing echo-sounders, with the dual-frequency

identification sonar (DIDSON) and other sonar technologies (Martignac et al. 2014) showing promise in their ability to identify species. Given the high costs associated with these techniques (Table 2), an alternate approach, depending on the requirements of the monitoring project, is hydroacoustic buoys, which are a comparably lower-cost method that has been successfully used to obtain data on relative biomass (Table 2). For example, as part of the environmental monitoring on the BIMEP project, five M3i hydroacoustic buoys were deployed on 6th June 2012, one in each of the four future mooring areas of wave energy converters and one far enough from BIMEP to act as control site. Table 3 shows the sampling periods of the five M3i buoys between 2012 and 2014.

Table 3. Sampling periods of the five M3i buoys in BIMEP between 2012 and 2014

	2012												2013												2014					
	J	J	A	S	O	N	D	E	F	M	A	M	J	J	A	S	O	N	D	E	F									
Buoy 1																														
Buoy 2																														
Buoy 3																														
Buoy 4																														
Control																														

M3i buoys, developed by Marine Instruments (www.marineinstruments.es), are specially designed for tuna fishing with fish aggregating devices (FADs). The M3i buoys are equipped with a GPS and echo-sounder (50kHz and 500W) and solar electric panels as an energy source. Whilst the GPS allows tracking of the buoy itself, the echo-sounder provides measurements of the relative biomass below the FAD. Once activated, M3i buoys transmit messages via satellite with echo-sounder information, GPS position, sea water temperature and battery level. Sounder information shows fish presence at 6-150m depth, with a 3m resolution. It records data every two hours during day and night (12 measurements per 24 hours). Data can be viewed as shown in Figure 2 and they can be downloaded to an Excel worksheet in order to extract information and carry out appropriate statistical analysis.

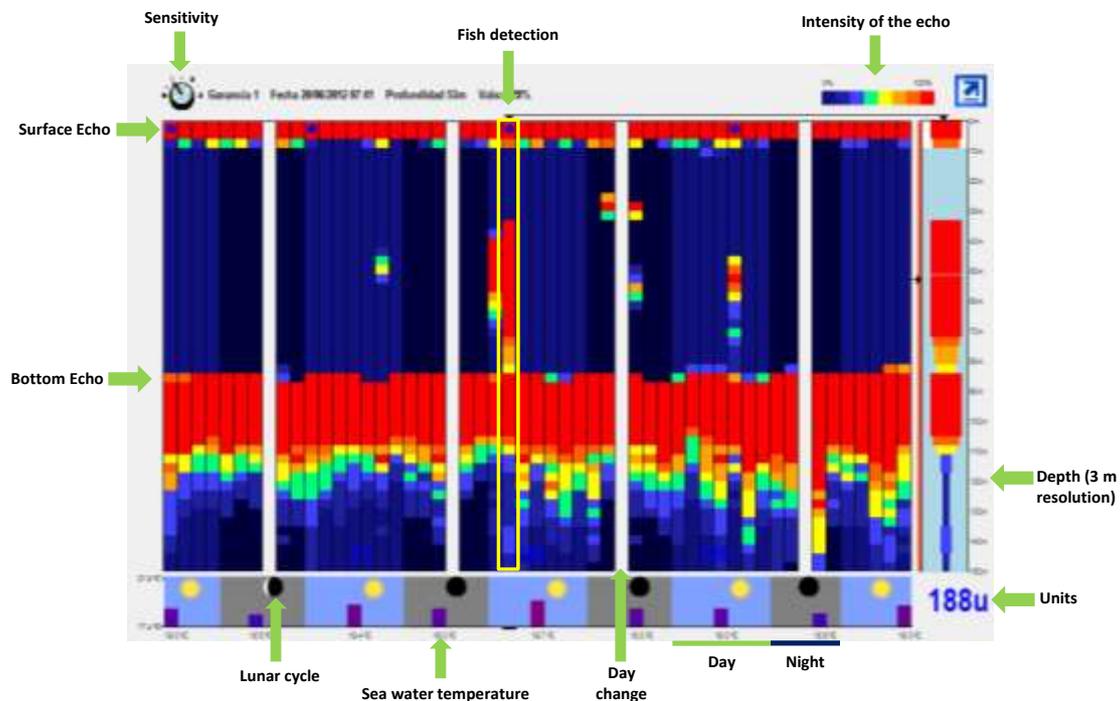


Figure 2. Data display format provided by the control M3i buoy between 18/06/2012 at 11:54 (GMT) and 22/06/2012 at 12:33 (GMT).

Figure 3 shows the deployment of an M3i buoy with in BIMEP. Each M3i buoy was secured to a signalling buoy (for maritime safety purposes) and the system was moored to the seafloor. The approximate cost for one of these systems (M3i buoy, signalling buoys and moorings) was approximately €2,500-€3,000, thus, €12.500-€15.000 for the five monitoring systems. In addition, there are costs associated with communication between the M3i buoys and a computer on the mainland (c. €22/buoy/month) and personnel costs for the deployment and periodic maintenance of the buoys. Even if total costs of this monitoring methodology are deemed to be medium (Table 2), one of the principal disadvantages of the M3i buoy is that differentiation between species is not possible.

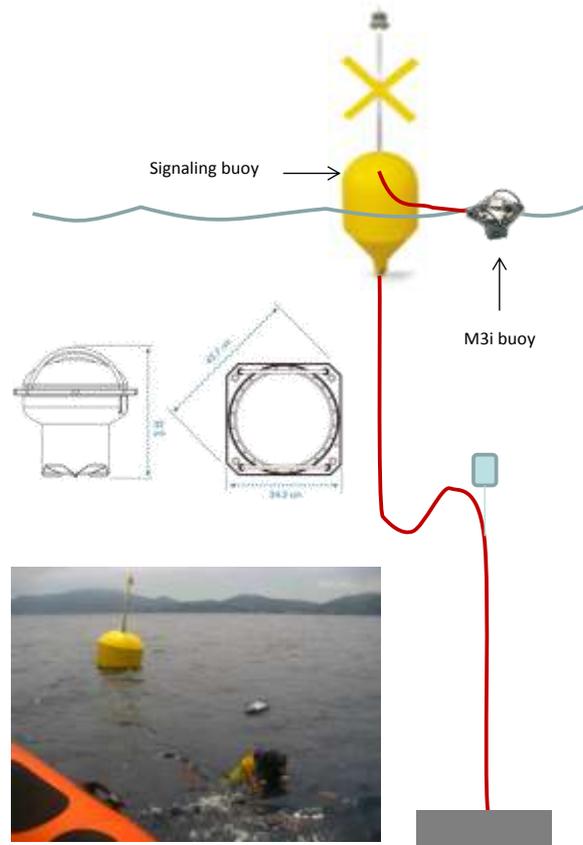


Figure 3. M3i buoy, as designed by Marine Instruments and deployment in the field.

4. EXPLORING LONG-TERM DATASETS AND CASE STUDIES

The aim of pre-consenting (and post-consent) monitoring for MRE developments should be to ensure that regulatory requirements (as determined by relevant legislation) are met (these requirements, across several EU MS, have been reviewed and discussed in Deliverable 2.1, O’Hagan et al. 2015). Furthermore, it is important that monitoring programmes use methodologies that are cost effective for the effort involved (see Section 3), and that the overall level of effort provides data that meaningfully informs the analyses and is comparable to other relevant situations. Particularly since regulators are likely to seek to manage the risks in a consistent fashion across a range of human activities, and may wish to avoid requesting less or more data collection without a justification that considers the risks to the receptors of interest alongside the monitoring costs.

4.1 Statistical concepts relevant to monitoring programmes

The conventional approach to identifying a change in the receptor of interest over time is to apply a suitable statistical test to the data and if the P value is >0.05 , we conclude that there is no statistically significant difference (i.e. we accept the null hypothesis). In other words, there is no significant change in the metric (e.g. abundance) for the receptor of interest over time. Where this outcome is perceived as positive (e.g. comparing baseline data to post-consent data we would conclude that there is no significant impact on the receptor as a result of construction activity), there is growing concern that in some cases these analyses may be failing to detect an effect that is present; which is referred to as a Type II error. In statistical terms, a Type II error occurs when there is failure to reject a false null hypothesis (false-negative). The converse is also possible, where an incorrect rejection of a true null hypothesis (false-positive) occurs, i.e. the analysis detects an effect that is not present; this is referred to as a Type I error. The likelihood of a Type I or Type II error occurring can, in part, be

addressed by using a statistical power analysis of simulated or existing data. This can then be used to better inform the survey design and data collection process during the pre-consent phase of the project. From the perspective of a monitoring programme, ensuring that a Type II error does not occur, particularly from a regulators perspective, is arguably more important. The following section considers the important attributes of power analysis and will put these into context with respect to pre-consent monitoring at proposed MRE sites.

4.2 Statistical power of long-term datasets

The most pertinent questions that need to be addressed prior to commencing a monitoring programme is ‘what change in population size needs to be detected?’ and ‘how confident do we need to be in detecting this trend?’ The latter can be addressed using power analysis, which is a statistical approach that can be used to design an effective monitoring programme and thus minimise the potential for wasting resources on a programme that is unlikely to yield useful results. As such, the value of power analysis to monitoring programmes is now widely acknowledged (Paxton & Thomas 2010, Mackenzie et al. 2013) given that, if a monitoring programme is unable to detect trends within the scope of the regulators requirements, then it will not be able to meaningfully inform judgements associated with the risks of impact by the project to the receptor. Consequently, power analysis has become more sophisticated, particularly as more advanced statistical techniques become available; in turn, this allows researchers to consider both realistic features of the data (e.g. autocorrelation and overdispersion) and the natural environment (e.g. Beaufort sea state and water depth) (e.g. Guillera-Arroita & Lahoz-Monfort 2012, Mackenzie et al. 2013, Embling et al. 2015).

The ability to increase statistical power is dependent on a number of factors, which include sample size, rate of change in the quantity being measured and the measure of precision (often referred to as the Coefficient of Variation, CV). More specifically, high statistical power (i.e. the ability to detect trends if they are occurring and thus avoid a Type II error) is a result of precise studies where the size effect is large and, as survey

effort increases, the precision of the estimate decreases (i.e. CV decreases). Inevitably greater effort through increasing the number of surveys or the duration of time spent on survey will incur greater costs, and thus initiates the widely accepted trade-off between statistical power and the cost of monitoring (Thomas 2009). Therefore, generally speaking, to attain both high statistical power and a high level of precision means that the financial costs will be high. For some key receptors, like seabirds and marine mammals, the emphasis is often on detecting trends in abundance over time, where baseline requirements for some MS may be a minimum of 2 years of data to account for seasonal and inter-annual variation (see D4.2, Simas et al. 2015). Therefore, it is essential that potentially time consuming and costly monitoring programmes are designed in a way that meaningfully informs the detection of trends at relevant spatial and temporal scales.

One approach to identifying statistical power is to run multiple iterations (e.g. 1,000) of a subset of the data through the suitable statistical test of choice and then calculate the proportion of these tests that obtain a significant difference using a given P value (e.g. ≤ 0.05). This can be done so as to investigate the influence of effort, where we would expect statistical power to increase as more surveys are undertaken. As Figure 4 shows, for 100 surveys if we use a P value of ≤ 0.05 we have a statistical power of 0.35 (i.e. 350 of the 1,000 iterations were significant at $P \leq 0.05$); however, if a P value of ≤ 0.2 is used, we have a statistical power of 0.73 (i.e. 730 of the 1,000 iterations were significant at $P \leq 0.2$).

As Figure 4 shows, to increase the number of surveys to 160, for example, would increase the statistical power across both P values to 0.52 and 0.87, respectively. This illustrates the issues of Type I and Type II errors; specifically, for higher P values (e.g. ≤ 0.2) there is an increased risk of a Type I error, i.e. incorrectly identifying a trend that does not exist despite having high statistical power to identify such a trend. Conversely, for low P values (e.g. ≤ 0.05) there is an increased risk of a Type II error, i.e. failing to detect an effect that is present because the statistical power may be substantially reduced, as is the case in this example. This highlights an important

trade-off with respect to the considerations that need to be given to the statistical power of a monitoring programme.

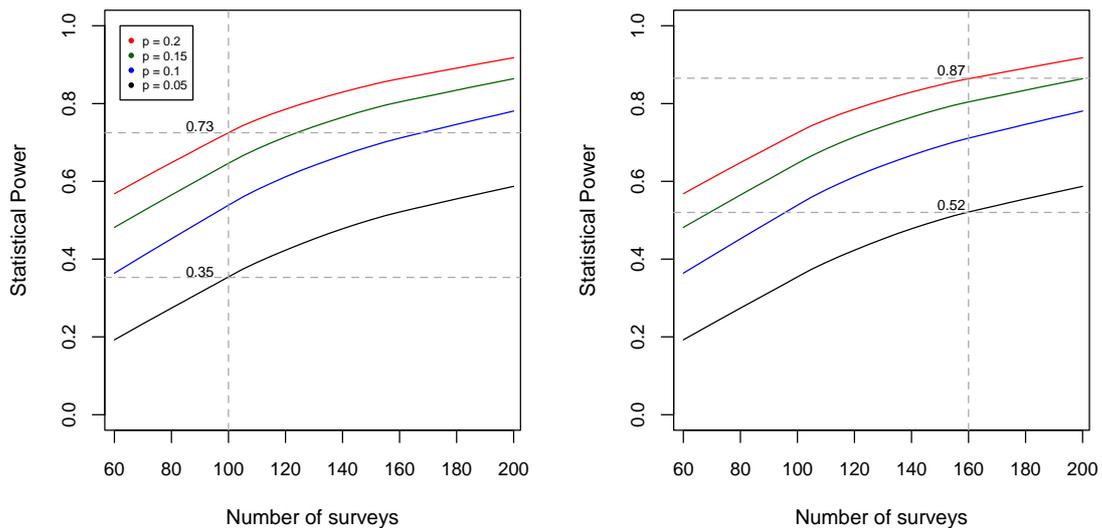


Figure 4. A simulation study illustrating how statistical power varies depending on the threshold applied to P values (i.e. what is considered as a significant difference) and how statistical power improves with additional survey effort (NB: both plots are the same). For example, if regulators deemed $P \leq 0.2$ to be a reasonable threshold with a requirement of 0.8 power the power analysis indicates that approximately 120-130 surveys would be required to attain this.

To highlight the importance of statistical power for detecting trends, Taylor et al. (2007) used several marine mammal datasets to examine their ability to identify a precipitous decline, defined as a 50% decrease in abundance over 15 years. They found, based on the conventionally used level of significance ($P \leq 0.05$), that the percentage of precipitous declines that would not have been detected as declines (i.e. Type II errors) was 72% for large whales, 90% for beaked whales, 75% for dolphins and porpoise and 5% for pinnipeds on land. Similarly, MacLeod et al. (2010) found that in most cases, using the SCANS-II data (SCANS-II 2008), statistical power to detect a 50% change between two surveys was very poor, despite the high expenditure of the SCANS-II project. For example, they found that double platform ship-based surveys of harbour porpoise (with a budget of c. €945,000) achieved a power between 0.17 and 0.6, depending on porpoise density (MacLeod et al. 2010). Conversely, at Strangford

Lough (the site for SeaGen, a tidal turbine), survey effort of approximately 25-30 hours per month was calculated to be enough to identify a 50% change in harbour seal abundance with a statistical power of 0.88 after just one month of monitoring. However, for grey seals, a 50% change in abundance would only have a 0.12 chance of detection (i.e. statistical power) after 6 months of monitoring (Sparling et al. 2011).

Using a BACI design, Vanermen et al. (2013) set out to investigate whether or not the windfarm in Thorntonbank, Belgium displaced seabirds. They investigated how survey length, monitoring intensity and data characteristics influenced statistical power. Conducting monthly surveys of 10km² in both the control and impact areas, they found, for 12 species of seabird, that a change in density of 25% with a power of more than 0.55 was not possible, not even after 15 years of monitoring. A change in 50% was detectable within 10 years for two of the 12 species, with a power >0.9. Under these conditions (within 10 years and a statistical power of >0.9) Vanermen et al. (2013) calculated that they would be able to detect a change of 75% in all but one of the 12 species.

In a similar study, MacLean et al. (2013) conducted power analyses based on real data obtained from aerial seabird surveys that covered areas of 'Round 2' offshore wind farm developments in UK waters. They investigated the power of being able to identify several thresholds for decline (50%, 33%, 25%, 15% and 10%) and how these could be influenced by survey duration and frequency, spatial scale and variability in bird numbers. They concluded that the standardised survey design protocols used did not provide adequate means of detecting changes in numbers, even when declines are >50% and assumptions regarding certainty are relaxed to $P < 0.2$. Although extending duration, frequency and spatial extent of the survey area did provide an increase in the probability of detecting a trend; this was only possible when certainty was relaxed to $P < 0.2$. For example, for four taxa, MacLean et al. (2013) varied the spatial scale of the survey area whilst looking at the statistical power to identify a 50% decline over 4 years with 4 surveys per year accepting a level of significance of $P = 0.2$ (Figure 5). They showed that, on average, the statistical power could be as low as ca. 0.1 and no higher

than 0.65, depending on the taxa and the spatial scale. Ultimately, MacLean et al. (2013) concluded that despite the substantial survey effort the statistical power remained low, which they suggested was most likely due to seabird numbers being highly variable over space and time making it difficult to distinguish an overall trend from fluctuations in numbers.

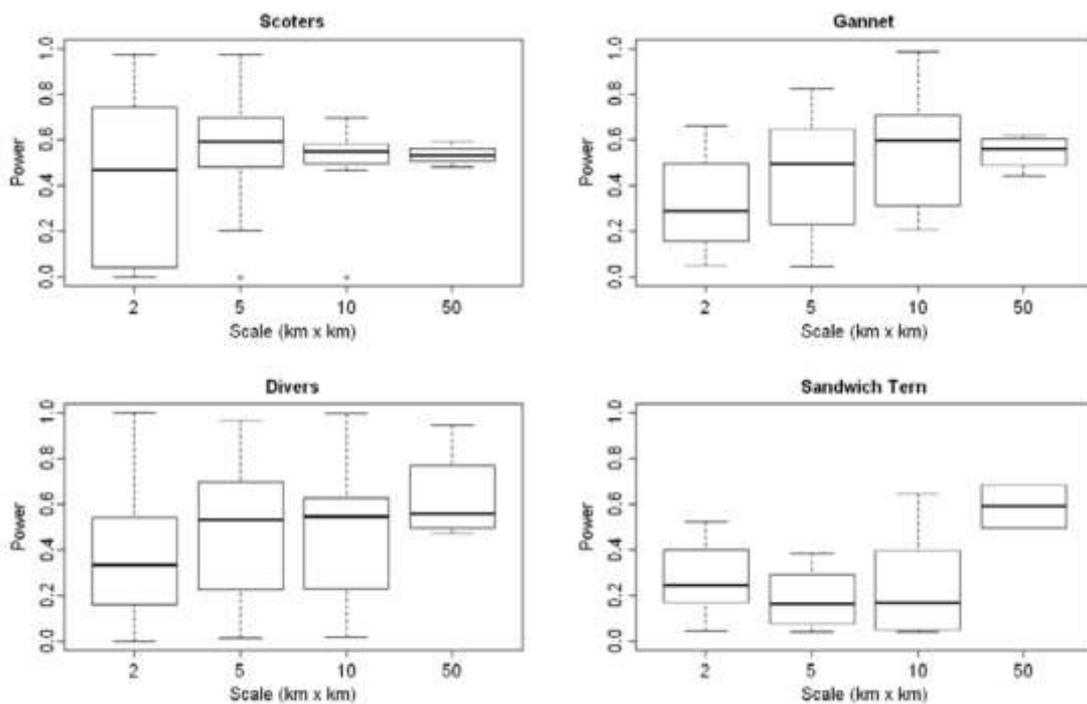


Figure 5. Box plots of variation in statistical power across sites for each of the taxa and each of the spatial scales for analyses (duration: 4 years, frequency: four surveys per year, $P = 0.2$, decline = 50%). The solid black line represents the 50th percentile and the box the 25th and 75th percentiles. The whiskers extend to the most extreme data point that is no more than 1.5 times the interquartile range of the box. Taken from MacLean et al. (2013).

In another example from the Thornbank windfarm in Belgium, Coates et al. (2013) investigated the statistical power of the BACI design for quantifying macrobenthos abundance, species number and species composition. They used a Van Veen grab to sample within 5 zones in the autumn of 2005 through to 2012. Each zone had multiple stations (ranging from 4 to 20), where one to three replicates were taken per station. The in-depth study looked at how control data from different locations and time periods influenced the results (i.e. whether or not and to what extent the abundance,

species number and species composition changed in the impact areas). Coates et al. (2013) did find cases where the power of the analysis was too low (c. 0.4). This was primarily due to low amounts of impact samples and/or control samples. Although some comparisons did yield a power >0.7 , Coates et al. (2013) used their findings to emphasise the need for a well-balanced survey design, with similar and adequate numbers of samples being collected in both the control and impact area to ensure that there is sufficient power in the data to allow for a meaningful assessment.

In many cases, studies that have applied the conventional level of significance ($P \leq 0.05$) have shown that it is either not possible to detect changes in abundance, or it is only possible once a substantial decline has occurred. Furthermore, where monitoring is only over a shorter period or small spatial scale, the power to detect change will be lower unless the magnitude of change per annum is high and the annual CV is low. The ICES Working Group for Marine Mammal Ecology (WGMME) (ICES 2008, 2010, 2014) have proposed that, for marine mammals, monitoring should achieve $\geq 80\%$ power and consideration should be given to the use of a significance level of $P \leq 0.2$, rather than $P \leq 0.05$. These suggestions have been made based on datasets with a large spatio-temporal scale (e.g. SCANS and CODA), as such, it is important to consider that distinguishing the signal from noise is harder at smaller spatial and temporal scales. Nevertheless, the ICES WGMME advice is a pragmatic approach that allows more lenient standards for detecting change compared to established conventions that were based on datasets with far smaller CVs than can realistically be achieved when taking measurements from the marine environment (particularly at smaller management units). As such, the following section uses data collected on a marine mammal monitoring programme by UCC to investigate how varying significance levels influences statistical power and the subsequent interpretation of the data.

4.2.1 Case study: Land-based Marine Mammal Monitoring at Broadhaven Bay, County Mayo, Ireland

Since 2009, there has been a year-round marine mammal monitoring programme in place at Broadhaven Bay, County Mayo, Ireland (Culloch et al. 2014). One of the

principal aims of the project was to gather data year-round in suitable conditions for sighting marine mammals (Beaufort sea state <4, Visibility > 7km); as such, the project maximised survey effort. Land-based surveys were conducted from a cliff top, each survey lasting approximately 60 mins followed by a 60 min break to prevent observer fatigue. Within the 60 mins the entire bay was scanned for marine mammals, by either one or two observers using a combination of the naked eye, binoculars and a telescope. Of the several species sighted within Broadhaven Bay, the grey seal was one of the more frequently sighted species. Using these data a *post hoc* power analysis was conducted for this species, with the aim of investigating the variation between pairwise years, to ascertain at what level of significance we could detect an increase or decrease in the sightings rate (defined as whether or not the species was sighted in a survey). The lowest sightings rate occurred in 2009, followed by 2012, with all six years having relatively high standard errors (Figure 6).

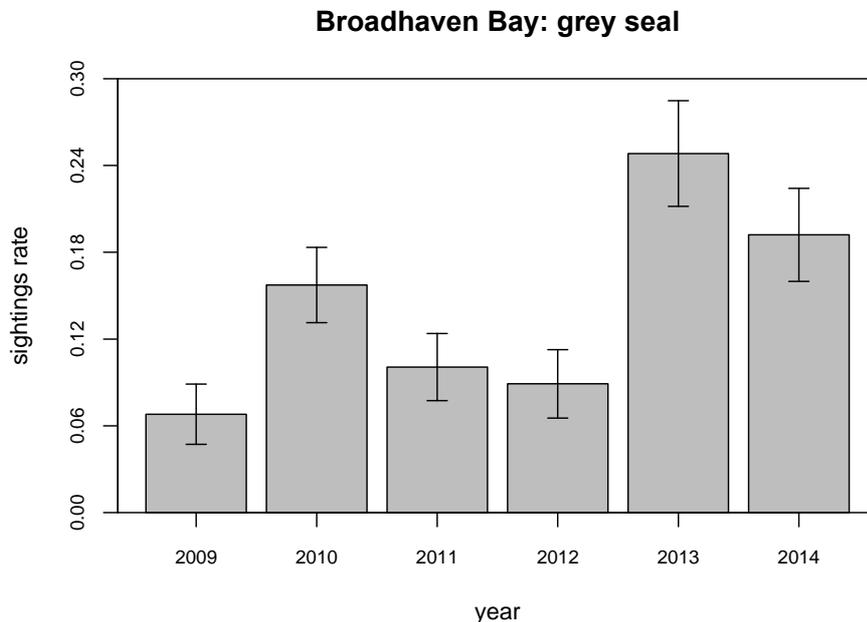


Figure 6. The average number of grey seals sighted per survey, for each year; the error bars show the standard error.

The power analysis was conducted following the methodology in Embling et al. (2015). Briefly, the data were presence/absence of a sighting during a survey. Generalised Estimating Equations (GEE) were employed so as to account for autocorrelation

between surveys within a given day and the model also took into account the Beaufort sea state during surveys, which is a variable that does significantly influence observers' ability to detect marine mammals (Evans & Hammond 2004). The maximum number of surveys was set to 200, which is a realistic number to achieve within a calendar year. The baseline year was taken to be 2009, which was compared to the other five years. From the original dataset, data were resampled, with replacement, 1,000 times. This was done for 60 to 200 surveys at intervals of five. A GEE was run on each block of 1,000 iterations and the proportion of P values that were within 0.05, 0.01, 0.015 and 0.2 were extracted from the models. For each block of 1,000 iterations the CV of the sightings rate was also calculated (standard deviation / mean).

There was an observed increase of 31% between 2009 and 2012, which was not significant, even at $P \leq 0.2$ with a power of 0.8 after 200 surveys (Figure 6 and 7). Similarly, for 2009 and 2011, an observed increase of 48% was not significant under the same conditions. However, for the 2009 and 2010 comparison, where there was a 131% observed increase in sightings rate, there was a significant difference at $P \leq 0.15$ with a power of 0.8 after 200 surveys. The comparison between 2009 and 2013 saw an observed difference in sightings rate of 265%, which, with a power = 0.8, was significant at $P \leq 0.05$ after approximately 140 surveys. The simulations showed that the CV for these data was high, and this was especially true for 2009 (Figure 8). For all six years of data the mean CV does gradually decrease as the number of surveys increase (and the 95% CIs also become more narrow).

Reducing the CV will increase the statistical power (Paxton & Thomas 2010), and this is an important consideration when designing monitoring programmes. In the case of Broadhaven Bay, previous analyses of this long-term data set identified a significant seasonal pattern in many of the marine mammal species recorded. For some species, such as common dolphins (data not presented here) sightings occurred more during the autumn and winter months during which point effort (due to shorter days and poorer weather conditions), was generally lower (Culloch et al. 2014). These attributes (i.e. higher effort during periods with a lower likelihood of sightings) will increase the CV and thus decrease statistical power. Therefore, one consideration may be to

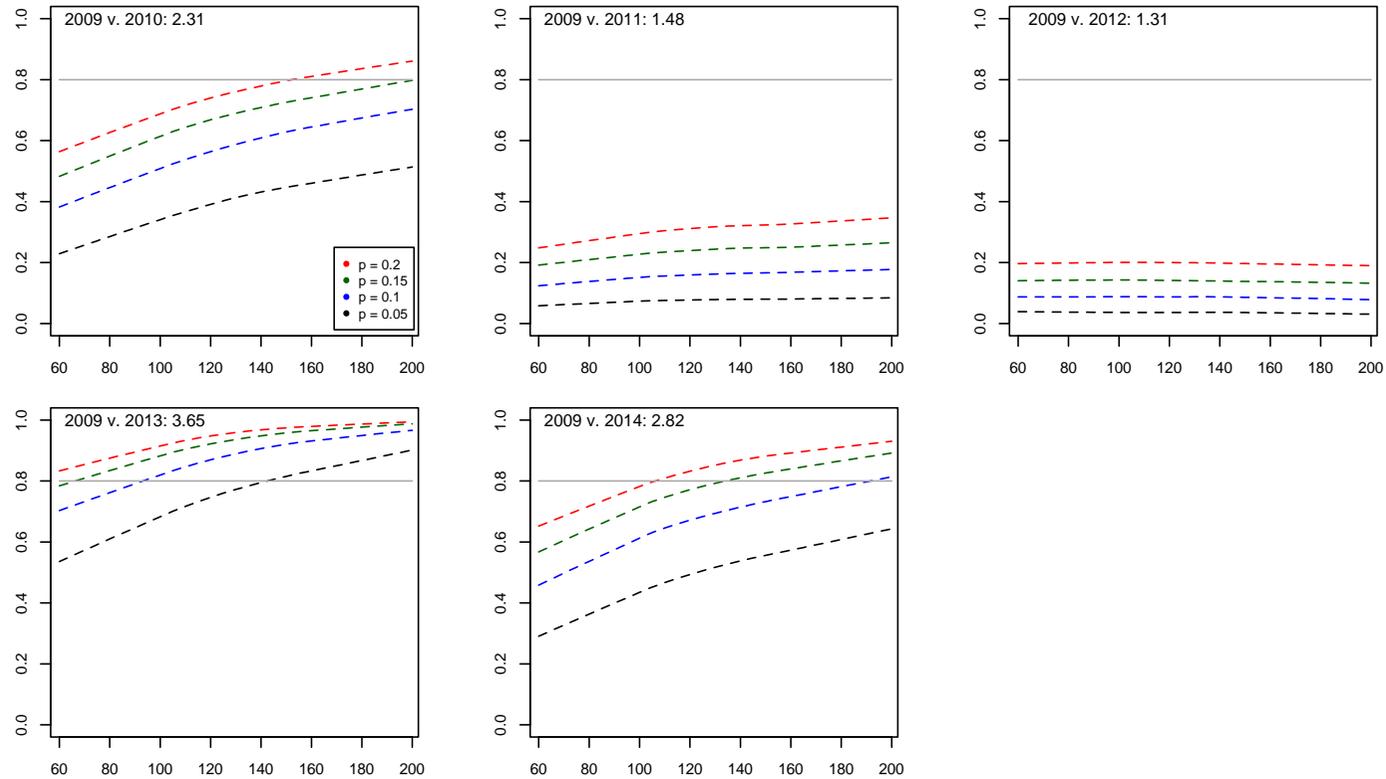


Figure 7. The power analysis for the pairwise years for grey seals, with statistical power on the y-axis and number of surveys on the x-axis. Each coloured dashed line is a mean of the power for the blocks of 1,000 iterations, which pertains to a given P value (see legend), the grey line shows the cut-off for a power of 0.8, as suggested by the ICES WGMME, the number after the pairwise year indicates the observed change in sightings rate, where 2.31 indicates a change of 131%.

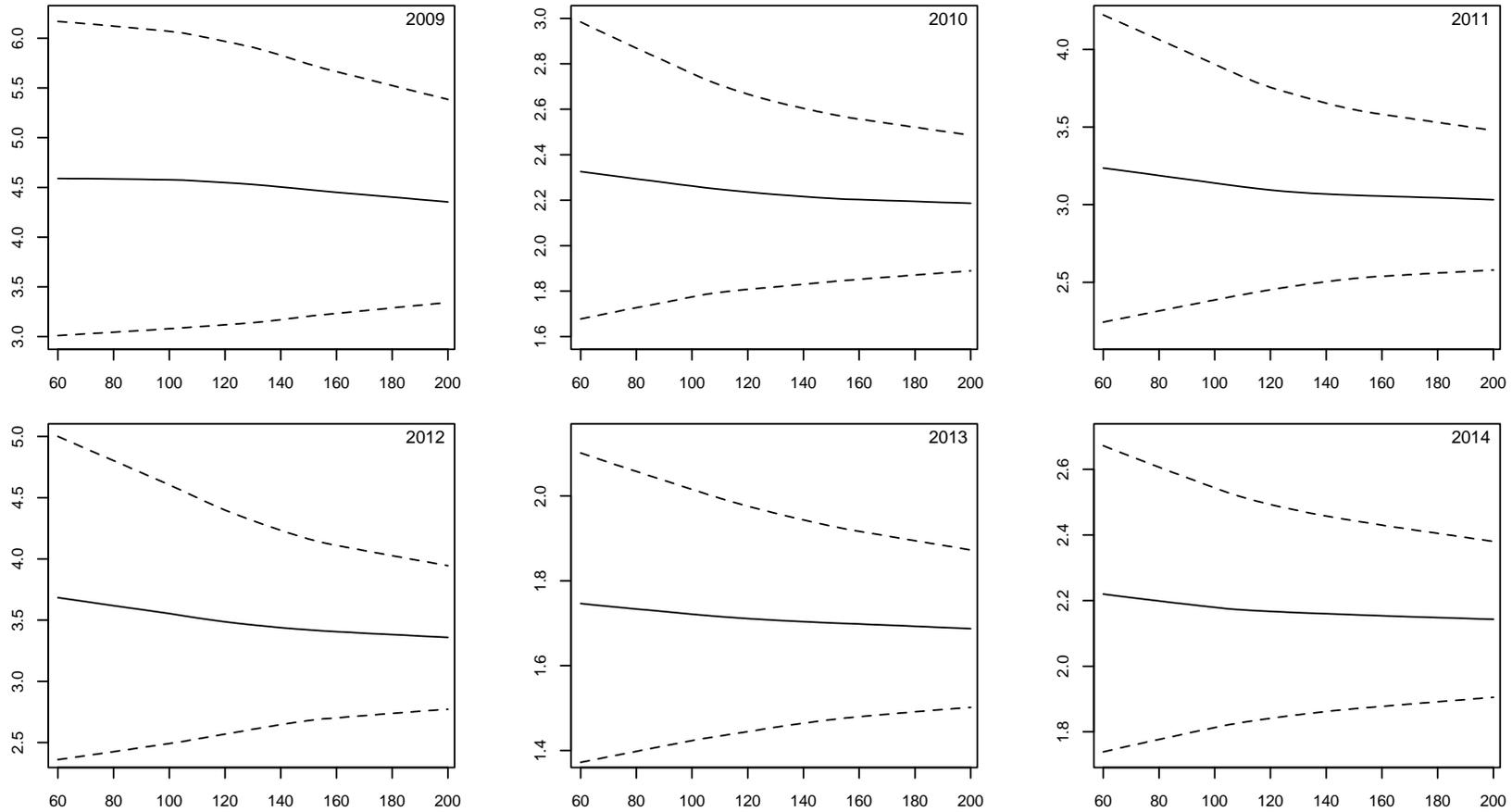


Figure 8. The Coefficient of Variation (y-axis) for the blocks of 1,000 iterations for the grey seal power analysis, depending on the number of surveys (x-axis) for each of the six years. The dashed lines show the 95% Confidence Intervals.

conduct surveys during the period where the species of greatest concern is more common. This illustrates the value of developing an understanding of the specific circumstances that are contributing to the variation in the data and how these can be managed on a case-by-case basis depending on these potentially unique circumstances.

4.3 Statistical power: further considerations and applications

Most notably, where population size is low, the power of the available data to detect a decline in abundance can become effectively meaningless. This scenario may be normal for a large number of protected populations/species, particularly if the regulator wishes to manage small magnitudes of change. When coupled with variable sightings rates and infrequent surveys (e.g. one survey per month, which is a recommend approach in some MS; see D4.2, Simas et al. 2015) the outcome will often be the provision of data that are likely to be not fit for purpose, as has been shown in the several case studies in Section 4.2. Consequently, these data provide no benefit to the species' monitored and can only serve to add cost and potentially delay the consenting process if regulators request more data.

Given how informative power analysis can be, it is undoubtedly a statistical tool that should be employed when considering which survey method to use and how to design the spatial and temporal nature of the surveys. As such, this approach will likely become commonplace in the near future; for example, power analysis has been used to identify the level of survey effort required to detect a 50% decline in Atlantic puffin (*Fratercula arctica*) within the boundaries of a consented offshore wind farm in East Scotland (Jared Wilson, Marine Scotland, pers. comm.). The target was to achieve a probability of 0.80 (i.e. $P = 0.2$). The power analyses addressed this particular issue because displacement effects on Atlantic puffin were identified as a key potential impact from the wind farm, and in the assessment of the application a displacement rate of 50% was assumed. The power analyses were based on existing, pre-construction boat based surveys from the area, and a single digital aerial survey. The results indicated that 6 surveys during the breeding season would have a 0.95

probability ($P = 0.05$) of detecting a 50% decline, whilst 3 surveys during the same period would have a 0.69 probability ($P = 0.31$) of detecting a 50% decline in abundance within the wind farm. It was concluded that a minimum of 5 surveys (with an associated probability of 0.86, i.e. $P = 0.14$) should be undertaken during the first year of pre-construction monitoring, with the power analyses to be repeated when the first year of aerial data became available to ensure that the simulated boat-based data were representative of actual data.

5. CONCLUDING REMARKS

This deliverable has highlighted a number of innovative technologies that are currently being developed specifically for monitoring aspects of the marine environment, or could be adapted for this purpose. Those documented in Section 2 covered several of the key receptors including seabirds, marine mammals, fish and the seabed and benthic environment. This is unlikely to be a comprehensive list, as other devices and approaches are likely being trialled at sites prior to them being outlined in technical reports, after which time it may be a period of months to years before this information is published in peer-reviewed academic journals. Ultimately, this is a clear indication of how rapidly this field is evolving in an attempt to improve all aspects of pre-consent monitoring (e.g. cost, data quantity, data quality, health and safety).

In Section 3, the deliverable also considered the cost of many of the approaches currently used for pre-consent monitoring of several receptor groups. In many cases these costs varied substantially within receptor groups, with some approaches more suitable for a particular data type or information (e.g. abundance estimates or informing collision risk modelling) or were more suitable given certain logistic constraints (e.g. offshore vs. nearshore, shallow waters vs. deeper waters). This highlighted the fact that, although cost is an important consideration of survey design, the initial stage of the process should be to consider the logistic constraints of the site coupled with the requirements requested by regulators to ensure that these can be met by selecting a suitable survey method or combination of survey methods.

Section 4 considered other aspects of survey design, including power analysis, which can be used to confirm that the data gathered can identify a change in abundance if one does occur; therefore ensuring that the data collected are fit for purpose. This is likely to become a commonly used approach in pre-consent survey design, as it can identify how much data is required to address the requests made by regulators. In using this approach, developers can obtain a better understanding of the financial costs likely to be involved during this phase of the monitoring programme, and, if

suitable data for the area already exist, then it may be possible to do this without having to conduct initial surveys at the proposed MRE site.

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