Guidance to inform marine mammal site characterisation requirements at wave and tidal stream energy sites in Wales

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NRW Evidence Report Number 82

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Report Code:	SMRUC-NRW-2015-012
Date:	Friday, 17 July 2015

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Carol Sparling was the principal author of the report and devised and developed the matrix based risk assessment approach.
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Tom Stringell and Ceri Morris provided input into the original scope for the report, provided input into the development of the matrix based risk assessment approach and into the text of the report.
Gordon Hastie, Paddy Pomeroy and Dave Thompson provided individual accounts of methodologies for the Appendix.

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1 Executive Summary

There is a growing consensus amongst regulators, statutory nature conservation advisors and developers and their environmental consultants that a 'one size fits all' approach to marine mammal site characterisation survey to inform consenting processes for wave and tidal stream projects is not fit for purpose. Furthermore, it is generally recognised that they may not always provide useful information for underpinning environmental assessments. There is a need to tailor pre-application surveys to a) provide specific information in relation to the particular types of impacts posed by the project, and b) to the likely degree of risk of significant impacts to marine mammals posed by the project. This report provides a mechanism for assessing b) and provides guidance on a) how to tailor survey effort to better provide information to inform specific impact assessment predictions. Section 2 of the report provides an introduction and section 3 provides a summary of the legislative background to the requirement for pre-application data gathering.

Section 4 provides an overview of the information requirements for wave and tidal stream energy projects across a number of identified main impact pathways. A number of general requirements for informing all potential impacts of projects were identified, such as a basic understanding of which species are present at a site and an understanding of the functional use of a site. A number of specific impact pathways were identified for wave and tidal energy projects, such as collision (mainly tidal) and disturbance (wave and tidal). For each impact pathway a set of individual information requirements were identified. For collision impacts the key information requirements were those which would enable a robust, quantitative prediction of collision rates. Specifically these are metrics which will help to predict the potential rate of encounter between marine mammals and the moving parts of devices such as information on animal flux through the project area and how this varies over time. It is accepted that there are important pieces of information that have a large bearing on the prediction of the magnitude and consequence of collision impacts, such as evasion and avoidance, which cannot be informed by pre-application surveys and the uncertainty in collision predictions as a result should be borne in mind.

For disturbance related impacts the important metrics are likely to be a measure of the density of animals at the site to allow an estimate of the number of animals likely to be affected and an understanding of why animals are present at a site, to allow a prediction of the nature and consequences of any disturbance. Disturbance can result in displacement of animals away from areas but it can also result in disruption to behaviours which may result in a reduction in breeding success or survival without any displacement.

A number of other impact pathways were also explored although it is noted that these impacts are unlikely to drive survey requirements in isolation from those considered above. The third impact pathway considered was indirect impacts mediated through a change in prey availability. The information requirements are similar to the impact of disturbance; how many animals are using the site and their behaviour and whether they are feeding (and what they are feeding on) in particular.

Entanglement with mooring lines was also considered and information requirements are similar to those for collision risk. An assessment of the risk of this impact occurring requires information on the

likely rate of encounter between marine mammals and the device which presents the risk. For entanglement with mooring lines, a prediction of the rate at which marine mammals are likely to come into contact with the hazard will be important but pre-application survey can inform very little about the likelihood of entanglement happening, other than highlighting whether species which may be particularly vulnerable to entanglement are present (e.g. larger baleen whales).

A number of other issues influencing pre-application data requirements were considered and discussed. This included consideration of how much reliance could be placed on existing data, how to assess whether survey methodologies are likely to be fit for purpose and issues involved in setting thresholds against which to assess potential population consequences of predicted impacts.

Section 5 then provides a detailed step by step process for an initial assessment of risk of wave and tidal stream energy projects to assist in determining pre-application survey requirements. The process is intended to be similar in principle to Marine Scotland's draft Survey, Deploy and Monitor policy but is adapted and developed more specifically for a more detailed consideration of marine mammals and extends to provide guidance on the type of survey that may be useful to inform the EIA and HRA in each case. This assessment should take account of a variety of features, e.g.:

- The type of device and its physical and mechanical features and the resulting likely impact pathways – for example a horizontal axis- rotor tidal stream turbine with large, fast moving parts that could cause injury and mortality to marine mammals poses a different level of risk of impact than a surface floating wave energy structure with no such apparent pathway for injury. This would involve a review of the evidence base, including research and knowledge of impacts for a given device or similar types of device. This assessment would need to be carried out for each specific impact pathway identified.
- The scale of the project – for example, a single demonstrator device is likely to pose a much smaller risk of impacts than a large array of devices;
- Site sensitivity – the extent to which the proposed project site is used by marine mammals, the importance of the site for those animals and the degree of legislative protection afforded those animals (either as EPS and/or as part of an SAC population) will affect the degree of risk of impacts.
- The duration of project - clearly the risk of significant impacts will be higher for projects of longer duration.

It is intended that this process would be followed by developers and their consultants at the scoping stage of projects, as part of the evidence plan development. This will allow developers to take responsibility for decisions on pre-application data gathering and to understand the rationale behind any proposed data collection and to understand the costs and benefits of survey work.

A staged, matrix based approach is proposed to classify these various predictors of risk and combine the assessment of biological features and technology related features. Each stage is as follows (note that further guidance on these classifications is given in the main report):

Stage 1: The sensitivity of marine mammal populations = classified as low, medium or high. A predetermined classification has been applied to each of the main Welsh marine mammal populations (harbour porpoise, grey seal, bottlenose dolphins and common dolphins).

Stage 2: The importance of project location to marine mammals based on existing knowledge = classified as low medium or high.

Stage 3: An overall sensitivity matrix combining the results from Stages 1 and 2 to assign a classification of low, medium or high.

Stage 4: Technology risk assessment, which is an assessment of the likelihood that a device associated with a project will result in impacts on marine mammals based on the current evidence base. An assignment of low, medium or high is given. Guiding principles for applying this assessment across the main impact pathways is provided.

Stage 5: Project risk assessment. This combines the technology risk from Stage 4 with the overall project duration to assign a risk of low, medium or high.

Stage 6: Overall risk assessment which combines in a matrix the outcome of Stage 3 (sensitivity) with the outcome of Stage 5 (project risk) to provide an overall classification of low, medium or high.

Stage 7: Determining survey requirements. The final stage in the process determines the survey needs based on the overall assignment of risk. For example, if for a given impact pathway, say for the risk of collision, a low risk is assigned, it is likely that a qualitative impact assessment detailing the justification of low risk will be sufficient based on existing information. If a high risk were assigned, this would require a quantitative collision risk assessment which may require the collection of site specific data to inform. The question of how much data are sufficient to inform a risk assessment is a difficult one and is best answered on a project and site specific basis. However the spatial and temporal resolution of previous surveys, the overall area of coverage, and the methods used to collect and analyse the data and how long ago the data were collected will be material considerations. General rules of thumb can be applied (e.g. data should be from within the last five years, though there may be exceptions to this).

A number of worked hypothetical examples are provided to illustrate this process.

Section 6 of the report details the range of survey methodologies likely to provide the information required to evaluate the main impact pathways as detailed in section 4. For collision risk the options for understanding animal flux rate through the swept area of devices which pose a collision risk are explored. These include site specific surveys measuring density and how it varies across the tidal cycle, or individual tagging studies for species where this is possible. The options for understanding animal depth distribution are also discussed. For disturbance related impacts, a quantification of animal abundance and surveys which are focused on understanding animal behaviour at a site are important. Across all impact pathways understanding the potential connectivity with protected sites will be important and techniques like telemetry and photo id may be useful. For all surveys it is important to consider the degree of precision that can be achieved. This may be particularly pertinent to small sites where even intensive effort would result in small sample sizes. Extended details of all survey methodologies discussed are provided in Appendix 1.

Section 7 provides a summary of the report and presents a series of recommendations. In summary, developers of wave and tidal stream energy projects should follow the initial risk assessment process outlined in this report to determine the survey needs for a project. This process will allow the developer to identify which impact pathways might result in potentially significant impacts and which impacts and species should be the focus of any pre-application survey. Alongside careful consideration of available data and the likelihood of site specific investigations providing data of sufficient quality, a specific and appropriate data gathering approach can then be devised and implemented. This process will ensure that developers take responsibility for making informed decisions about survey and the need for data to inform assessments, understanding the risks of potential operational restrictions that may result from poor or inadequate baseline data.

2 Introduction

2.1 A 'proportionate' requirement for survey to inform consenting process

Experience and learning have resulted in a general consensus that a 'one size fits all' approach to marine mammal 'site characterisation' surveys to inform the Environmental Impact Assessment (EIA) and Habitats Regulations Appraisal (HRA) processes for wave and tidal stream projects is not fit for purpose and may not always provide useful information for these environmental assessments. There is particular concern from the industry that a similar survey requirement is imposed for projects which are predicted to have a low impact as for those with the potential for more significant impacts, or for more uncertain projects (e.g. Scottish Renewables (2014))

Natural Resources Wales (NRW) wishes to understand how a more proportionate and appropriate approach to determining pre-application survey requirements might be developed in Wales. Options include drawing on elements of Marine Scotland's 'Survey, deploy and monitor' policy, which provides for a transparent risk-based approach to determining the level of survey required for wave and tidal stream projects, although it does not provide any guidance on the type of monitoring that is required.

The risk of impacts posed to marine mammals by projects might be assessed both at the outset, using existing information about mammals and the location and scale of the project and type of device and also through a continuing adaptive assessment process informed by data as it is gathered. Marine mammal pre-application characterisation surveys (or ongoing surveys) can then be tailored according to the perceived risk that projects might have unacceptable impacts. There must also be recognition in this process of the uncertainties inherent in predicting the potential impacts of wave and tidal stream projects and careful consideration of the information requirements of EIA and HRA. 'Traditional' approaches to survey may need to be adapted to provide the appropriate kind of information. In some cases even extensive pre-application monitoring may not provide the required information for a confident prediction of impact. A process for making an assessment of the most appropriate survey will be required. This report explores these issues and makes a number of suggestions as to how such an approach could be developed.

Box 1: Proportionality

Proportionality: A call for proportionality in survey and assessment requirement is a common occurrence, but is rarely defined. The typical interpretation is that survey and assessment requirements should be proportionate to the risk of significant impacts posed by the project – projects considered to be lower risk should have less onerous survey requirements which may involve lower effort, less detailed assessments, and entailing overall lower cost. Conversely, developments deemed to present a higher risk of impact should require a greater degree of survey and more detailed assessment to inform the consenting processes. This is how the term proportionality is intended in this report. In addition, the term proportionality is often used when comparing the requirements imposed on the wave and tidal stream industry relative to other, much larger scale industries which may have a corresponding greater risk of impact. In this report we follow the first definition.

2.2 Ensuring marine mammal characterisation surveys are fit for purpose

Pre-application surveys should be designed specifically with the likely requirements of HRA and EIA in mind, although consideration should be given at this early stage to data which might be useful baseline for future impact monitoring. Understanding marine mammal distribution over the impact footprint and gaining an estimate of abundance have been a common goals of pre-application survey to date. However, before any survey is implemented an assessment should be made of whether the proposed surveys are likely to provide sufficient data to provide particular information (such as densities) and how such data could be applied to assessing the potential scale and significance of impacts. For example, will the size of the survey area and the intensity of effort enable the collection of enough data to allow the calculation of a site specific density estimate with sufficient precision (e.g. for modelling collision risk)? If there is uncertainty around this then survey design should be revisited.

The traditional requirement for pre-application survey for wave and tidal stream (and other marine renewable energy) projects in Wales and elsewhere in the UK has been for a 'standard' 2 years baseline survey carried out either using shore based visual observers at vantage points (VP) for near shore sites (e.g. TEL's DeltaStream project in Ramsey Sound, MCT's Anglesey Skerries Tidal Array and ongoing wildlife monitoring at the Falls of Warness and Billia Croo at the European Marine Energy Centre (EMEC) in Orkney) or boat based line transect visual surveys where the site is too far from shore for VP surveys (e.g. DP Energy's West Islay site, MeyGen's Pentland Firth site). These surveys typically consist of counts of detections (typically visual but in some cases supplemented by acoustic) along with information on survey effort (kms trackline, observer effort, sighting conditions etc). Some behavioural information is also collected for each sighting but given that marine mammals spend the majority of their time submerged, this is usually not sufficient to inform understanding about functional use or importance of a site –information which is important for understanding the potential consequences of any impact. More detail is given on these methods and their benefits and limitations in Section 1 and Appendix One. The desired outcome is a site specific estimate of abundance and distribution and an understanding of the variation in these (seasonally, spatially and for tidal projects, over the tidal cycle). A combination of the limitations of the methodologies adopted and the amount of effort expended, the low sightings generated, coupled with the large inherent variability in both survey condition and animal densities generally means that absolute measures of abundance from these surveys are rare or come with large confidence intervals.

It is questionable in many cases to what extent surveys which are only designed to 'count' mammals can actually inform impact assessment processes and ultimately help to remove uncertainty around key consenting issues, or how much information these surveys provide to inform the development of mitigation or adaptive approaches – e.g. identifying risk factors for collisions. In addition there are questions as to whether the benefits of the information they provide justify their cost, in both financial and health and safety terms.

It may be better to shift emphasis towards gathering data which would increase the ability to predict whether, and the extent to which,

a) Marine mammals utilise areas earmarked for wave and tidal stream developments; and;
b) Marine mammals use these sites for particular life functions;
c) Marine mammals are likely to interact positively or negatively with devices within these areas?

As an extension of a risk-based approach to determining survey requirements, this report details some of the key information required for the assessment of the main marine mammal related impacts currently thought to be of importance for wave and tidal stream projects. We then review a range of available survey methodologies in the context of their ability to provide this information.

3 Legislative requirements

This section summarises the legislative framework which drives the need for survey and assessment for wave and tidal stream projects. This provides the context for understanding the regulators' task of ensuring that the legislative needs of the assessment and consenting process are met. The primary statutory processes driving the requirement for collection of baseline and monitoring data are Environmental Impact Assessment (EIA), European Protected Species (EPS) licencing and Habitats Regulation Assessment.

3.1 Environmental Impact Assessment

The EIA regulations require developers of marine renewable energy projects that are likely to have a significant effect on the environment to undertake an assessment of the positive and negative environmental impacts of developments from the construction stage through to decommissioning. This assessment must be documented within an Environmental Statement (ES). The ES should include sufficient information to enable the licencing authority(ies) to determine the extent of any environmental impacts arising from the proposed scheme and should cover direct, indirect, secondary, cumulative, short, medium and long-term, permanent and temporary effects.

3.2 European Protected Species licensing

All cetaceans found in Northern European waters are listed under Annex IV of the EU Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora (the Habitats Directive) as European Protected Species ('EPS') of Community Interest and in need of strict protection. The Habitats Regulations and the Offshore Marine Regulations make it an offence to deliberately kill, injure, capture or disturb any EPS. The SNCBs have published guidance on the interpretation of these regulations and the circumstances in which an EPS licence is required (JNCC et al., 2010).

Mitigation measures should be put in place if there is a significant risk of an offence. If there is a reasonable expectation that there is risk of deliberately killing, injuring (including auditory injury), capturing or disturbing an EPS as defined above, despite mitigation plans, a derogation licence is required.

Regulators will grant such a licence if the following three tests are met:

- The purpose of the work is for preserving public health or public safety or other imperative reasons of over-riding public interest including those of a social or economic nature and beneficial consequences of primary importance for the environment
- There is no satisfactory alternative to the activity
- The action authorised will not be detrimental to the maintenance of the population of the species concerned at a favourable conservation status (FCS) in their natural range

Given this legislative requirement, it is clear why a good understanding of the potential magnitude of any impacts which might cause injury or death, or affect the survival and fecundity of individuals and therefore consequences for cetacean populations is required and why information outside the narrow footprint of an activity is often required.

3.3 Habitats Regulations Appraisal (HRA)

The harbour porpoise, bottlenose dolphin, grey and harbour seal are protected under Annex II of the Habitats Directive as species of Community Interest whose conservation requires the designation of Special Areas of Conservation ('SACs'). The Habitats Directive is transposed into UK legislation through the 2010 Habitats Regulations, whereby European Sites (e.g. SACs) are given protection.

The 2010 Habitats Regulations require that the competent authority, before authorising a project likely to have a significant effect on a European site 'must make an appropriate assessment of the implications for that site in view of that site's conservation objectives'. Anyone applying for development consent must provide the competent authority with such information as may reasonably be required 'for the purposes of the assessment' or 'to enable them to determine whether an appropriate assessment is required'. This information is normally provided within the Environmental Statement, or in supplementary 'information to inform an HRA' report.

In practice this places a burden on the applicant to 'prove' there will not be a 'Likely Significant Effect' (LSE) on the European site(s), either alone or in combination with other plans and projects.

Where LSEs on a European site cannot be discounted, the competent authority needs to consider whether those effects will adversely affect the integrity of the site in view of its conservation objectives. The HRA should therefore include evidence about the projects impacts on the integrity of protected sites and a description of any mitigation measures proposed which avoid or reduce each impact, and any residual effect.

For highly mobile species, such as marine mammals, competent authorities typically consider the site to be affected if animals that are connected to the site (and can therefore be considered as animals from the site) are affected by an activity, even if that activity may be some distance from the SAC itself.

As such it is clear why a good understanding of the connectivity of a project site with any SAC with marine mammal features, and a good understanding of the magnitude of any potential impact that might affect survival and fecundity of individuals from that site is important to both the developer and the competent authority. Survival and fecundity (birth rate) are the two most important life history traits that contribute to the status and health of a population; the difference between mortality (the inverse of survival) and fecundity is the rate of change (decrease or increase) of a population. Since the majority of legislation protecting species and habitats is ultimately concerned with population level consequence, these two vital rates are often the focus of impact assessments.

4 Questions for EIA/HRA

Defining an appropriate pre-application survey approach involves consideration of potential impacts and a 'scoping' of the information required to predict and assess each impact. Therefore, it follows that survey requirements may be different depending on the impacts of concern. Here we review the key information requirements across a range of impacts.

4.1 General requirements for all potential impacts/key info for initial assessment

To characterise the sensitivity of a location there is a basic need to understand which species are present. The importance of the location for the relevant species/populations and how this may vary over time needs to be determined. Multiple visits may also be required to assess the presence of less common and/or migratory species. Much of this general characterisation can be achieved through visual and/or acoustic surveys repeated on many occasions over a number of years, or reliance on existing data. It can also be valuable to understand what the animals are *doing* at the location at different times of the year and therefore to what life functions the area contributes. This may include gathering information on animal distribution in three-dimensional space and time over small scales, as well as dietary and behavioural observations.

Abundance estimates for the relevant population units for all species of interest may also be a required; without reliable abundance estimates, it is difficult to put any potential impact into a broader context and to meet the regulatory requirements of assessing impacts on populations. In the UK, marine mammal reference populations have been defined by the UK Inter Agency Marine Mammal Working Group (IAMMWG, 2015), these represent ecologically relevant units for each marine mammal population for reporting of status to the EU and for the management of anthropogenic impacts. Furthermore, for marine mammals potentially associated with SACs the extent of 'connectivity' between the development site and SACs needs to be established.

The subsequent sections provide consideration of the primary impact pathways of wave and tidal stream projects. This is not exhaustive but is intended to cover the main issues of concern. It is not expected that other impacts would drive the requirements for data collection in isolation of those pathways considered here.

4.2 Specific impacts: Collision with the moving parts of devices

The potential for marine mammals to collide with the moving parts of devices, particularly the rotors of horizontal axis tidal stream turbines, is a primary concern for the consenting and licencing of projects. There is an absence of empirical data to determine the ability of animals to avoid coming into contact with devices, either through close-range evasion, where animals take last minute evasive action, or through avoidance, which may operate at a wider scale with animals avoiding the area the devices are located in. Predictions of the potential magnitude of collision risk rely on 'encounter rate' models that predict the potential rate of encounter between animals and the 'zone of risk'. Typical approaches to making these predictions are based on either the density estimate of animals, their movements and the area swept by the rotor blades (e.g. Wilson et al., 2007; 2014), or on passage rates of animals through the swept area (e.g. Davies and Thompson, 2011). Both approaches incorporate information on each species' vertical use of the water column (i.e. the proportion of time animals are spending at the depth of the devices). The outputs from these approaches allow an assessment of collision risks to be made before developments are consented. These assessments could be modified in light of empirical data on avoidance or evasion behaviour that might become available in the future. Adaptive responses such as evasion and avoidance have generally been based on assumptions (which are not yet validated). However, the possibility that animals will be attracted into areas after devices are installed, resulting in higher local densities and an increased collision risk, cannot be discounted, especially if devices serve to aggregate fish.

The temporal variability in these input parameters (e.g. site specific density and vertical use of the water column, direction of movement) across the tidal cycle is also important, since the risk posed by the devices will vary significantly over the tidal cycle as a result of variations in rotor speed, approach velocities etc. Understanding the temporal patterns in these two parameters (likelihood of animal encountering the device and the risk posed by the device) is crucial to an accurate prediction of risk.

In addition, understanding the degree of residency and the rate of individual turnover at a site is potentially important for the interpretation of the significance of current collision risk model outputs. To our knowledge, this has not been addressed in collision risk assessments thus far. A project at a site where animal turnover is high and is used by many transient (but presumably naive) animals is likely to pose a different risk than a project at a site where there is a small resident population that has the opportunity to gain experience of the devices. The number of animals affected as a proportion of total vulnerable animals will be lower for the former scenario, but the absolute number of animals affected may be much larger. Conversely the opportunities for learning and behavioural modification will be lower where there is a large number of transient animals passing through the site. Thus, the rate of movement past a focal point will fundamentally affect the number of animals at risk. Current models of collision risk (Wilson et al. 2014; Davies and Thompson 2011) typically do not differentiate between these two scenarios, but it is possible to develop individual based modelling approaches to collision risk which could take this factor into account if sufficient data were available to inform them.

At the EIA/HRA stage of projects, a quantitative prediction of collision risk is usually carried out using these collision risk modelling approaches and the resulting rate of potential encounters per year is assessed in the context of legislative requirements for that particular species and population. This is typically assessed against an assessment of the level of additional mortality that would be considered significant for that population. For assessment purposes, model outputs are interpreted in a precautionary manner, whereby encounters are assumed to represent collisions, which are assumed to represent mortalities[1].

It is clear that there are a number of key information requirements to allow robust quantitative predictions of collision risk:

1) Information on animal flux through the swept area (and how this varies across the tidal cycle), although in many cases this is not possible to collect;
2) In the absence of 1), spatially explicit informaton on density of each species at the project site, in conjunction with information on water column use, can act as a proxy.
3) Turnover/residency of individuals at the site;
4) *Avoidance/evasion or attraction rates (and how they may vary with the number of devices);*
5) *The consequences of collisions for individuals (ie the proportion that result in mortality or significant effects on the survival and fecundity of individuals);*
6) *The size of the relevant population management unit for each species and an understanding of the level of acceptable mortality.*

The latter three of these are in italics to highlight that it would be impossible for pre-application site specific survey data to provide information on these. Information to inform points 1-3, however, could be gathered using site specific survey. Information on point 4 could be collected at a site, but only after operation commences. Moreover, validating assumptions of collision risks requires the ability to accurately detect that collisions are occurring, which is not straightforward. It is important to note that most assessments to date have considered single demonstrator devices or small arrays, and there is uncertainty as to how collision risk scales with the number of devices at a site. It is unlikely to be a simple linear increase due to repeated responses to individual devices and learning by animals encountering multiple devices. The scale of avoidance may change with larger arrays (animals avoiding the entire array, reducing the probability of encounter of additional devices).

[1] This assumption may be overly precautionary but until empirical data exists to show otherwise, this assumption will be made. The Sea Mammal Research Unit are currently researching this issue and a preliminary study suggests that only a proportion of strikes are likely to result in mortality. It will, however, likely be difficult to determine the probability of an injury leading to subsequent reductions in survival and fecundity.

Evasion behaviour may also alter risk; the potential for avoidance of one device to take an animal on a path where encounter of additional devices may be more likely. However, it is unlikely that the basic site specific information requirements listed as 1-3 above will differ as projects scale up.

It is important to note that estimates of collision risk are extremely sensitive to assumptions made about the avoidance rates of animals. We might expect this to vary between different species, individuals and device types as well as with site-specific factors affecting an animal's ability to detect and respond to turbines, such as turbidity, flow speeds, noise levels (ambient and device-generated) and ambient light levels. Current uncertainty around this may be a significant consideration in the decisions surrounding survey requirements to address collision risk. It is important to note that in the absence of empirical information on avoidance and evasion, a confident quantitative prediction of collision risk will not be possible, regardless of how precise or robust site specific density estimates are.

4.3 Specific impacts: Disturbance

There is the potential for animals to become physically displaced away from a location as a result of a wave or tidal stream development or disturbed and therefore prevented from carrying out important life functions (e.g. breeding or feeding). Displacement can be considered one potential consequence of disturbance but it is important to note that animals can be disturbed with resulting consequences for survival and fecundity without being displaced. Displacement is often predicted as a result of acoustic disturbance during construction and maintenance/operation of devices. But disturbance (including displacement) could be a result of a response to the general physical presence of devices (during operational phases) and/or vessels and activity (during construction and/or maintenance). Any assessment of this impact needs to take into account the potential scale and therefore magnitude of the disturbance (over how large an area might this occur and how many animals may be affected). The potential consequences of the disturbance for individual animals, and consequently for the population, need to be considered. Consequences of disturbance could include: 1) displacement from important habitat, 2) disturbance at a breeding site leading to reduced breeding success, 3) disruption of social interaction, including mother-calf/pup relationships, 4) displacement resulting in a 'barrier effect' across an important 'corridor'. Although a barrier effect could be considered a consequence of displacement away from an area previously used for transit, and is commonly assessed as a separate impact, and is so here.

All of these effects may be considered as distinct impacts and often different data sets will be required if they are to be understood adequately. However, understanding the functional use of an area (why marine mammals are present in an area and what they are using it for) is fundamental to understanding which of these impacts might occur and what the potential consequences might be.

Assessment of these types of impact is often difficult given the uncertainty associated with predicting individual responses and difficulties of linking these from individual level (survival and reproduction) to population level consequences (growth or decline of a population). Assessments are often qualitative and based on an understanding of the relative importance of the site for the marine mammal species found there. However, in order to provide quantitative assessments and predict how many animals may be affected in each case and the consequences of any effect, an understanding of *how many* and *in what way* animals are using the area will be required to robustly model the consequences of the disturbance/displacement for individuals and subsequently for populations. Given the requirement to place impacts in the context of the effects on populations there is similarly the requirement for information on the size and status of the relevant marine mammal management units (IAMMWG, 2015).

It is clear that there are a number of key information requirements to allow robust quantitative predictions of the impact of disturbance and habitat displacement:

1) Site specific density for each species at a scale appropriate to the predicted impact footprint (this may be much larger that the licenced project area);
2) Turnover of individuals at the site (which defines the size of the vulnerable population);
3) The behaviour of animals at the site – what the site is used for (and therefore what the potential consequences may be for displaced or disturbed individuals);
4) *The availability of alternative habitat to meet the needs previously served by the project site and the cost of switching. This could also include impacts (such as competition) on animals within the areas into which the original animals are displaced;*
5) *Link functions between levels of disturbance and important vital rate parameters for individuals (e.g. the survival and fecundity of individuals);*
6) *The size of the relevant population management unit for each species and an understanding of the level of acceptable additional mortality (including cumulative impacts);*
7) *The conservation status of the population / management unit and other potential impacts/threats.*

Similar to the previous section, the latter four are in italics to highlight that it would be impossible for site specific survey data to provide information on these. Information to inform points 1-3, however, could be gathered using site specific survey or telemetry (depending on species).

4.4 Specific impacts: Indirect impacts mediated through changes in prey availability

The construction and operation of wave and tidal stream energy developments might cause changes to marine mammal prey availability. This can be as a result of a permanent loss of prey habitat due to the project footprint on the seabed, direct mortality of prey species, changes to hydrodynamic features resulting in prey being displaced elsewhere (or attracted in), avoidance of the development by prey, or increases in prey availability as a result of aggregation around structures. This latter effect could lead to both positive (increased foraging opportunites) and negative (increased collision risk if the prey aggregation is attractive) impacts on marine mammals. In order to assess the potential for these impacts there is a requirement for information on firstly whether or not the site represents an important foraging resource for marine mammals and secondly the potential for these direct impacts on prey. As with the impact pat hways detailed previously, there is a need to quantify the magnitude of any potential impact and to understand the potential consequences for individual marine mammals and populations. Because to some extent the consequences to individuals (reduced foraging opportunities) are similar to the impact of direct displacement from a foraging area, the information requirements are similar, although the requirement to understand whether or not marine mammals are exploiting prey at the site is most important. If animals instead use the site e.g. as a transit route or resting area then the impact of displacement from this area is much less likely to be significant. Information needs include the following: The behaviour of marine mammals at the site – specifically whether animals are foraging there. If they are, then there are a number of additional requirements:

1) What prey species they are feeding on, in what proportions; The predicted impacts on prey species of the development;
2) Site specific density estimates for each marine mammal species at a scale appropriate to the predicted impact footprint;
3) The degree of residency/turnover of individual marine mammals at the site (defines the size of the population potentially at risk);
4) *The availability of alternative foraging habitat to meet the needs previously served by the project site and the cost of switching. This could also include impacts (such as competition) on animals within the areas into which the original animals are displaced;*
5) *Link functions between levels of disturbance and important vital rate parameters for individuals (e.g. the survival and fecundity of individuals);*
6) *The size of the relevant population management unit for each species and an understanding of the level of acceptable mortality (including cumulative impacts);*
7) *The conservation status of the population / management unit and other potential impacts/threats.*

The latter four points are in italics to highlight that it would be impossible for site specific survey data to provide information on these. Information to inform points 1-3, however, could be gathered using site specific survey or telemetry (depending on species).

4.5 Specific impacts: Entanglement with mooring lines

The survey considerations for an assessment of the risk of entanglement impacts are similar to those for collision risk, in that an assessment of the risk of this impact occurring requires information on the likely rate of encounter between marine mammals and the device which presents the risk. For entanglement with mooring lines, a prediction of the rate at which marine mammals are likely to come into contact with the hazard will be important. However, pre-application survey can inform very little about the likelihood of entanglement happening, other than perhaps highlighting whether species which may be particularly vulnerable to entanglement are present (e.g. larger baleen whales). The properties of the mooring lines and how detectable they may be are more likely to influence risk; current recommendations are that assessments should be largely qualitative and based on an assessment of the risk posed by the mooring design alongside an understanding of the biological use of an area in terms of species present, densities, distribution and behaviour (Benjamins et al., 2014). It is unlikely that entanglement risk alone will drive pre-application monitoring requirements, so information on species present etc. will likely be gathered anyway for other impact pathways (either by survey or from existing literature), although information on the physical characteristics of moorings will need to be provided to the regulator independently.

Table 1. Information requirements for specific impacts which may result from wave and tidal stream projects

	Density	Horizontal Distribution	Vertical Distribution	Behaviour – what are they doing at the site in terms of movement and activity	Turnover/residency	Population size	Connectivity to designated sites
Collision	Yes	Yes -but only at device sites	Yes-but only at device sites	Yes – movement patterns and swim speed etc. will affect collision risk	Yes	Yes	Yes
Disturbance	Yes	Yes	Dive behaviour may be useful for assessing foraging	Yes	Yes	Yes	Yes
Indirect impacts – prey species	Yes	May indicate foraging	Dive behaviour may be useful for assessing foraging	Yes – restricted to whether they are foraging or not	Yes	Yes	Yes

4.6 Other issues influencing data requirements

4.6.1 Existing data – what is sufficient?

It may be possible to satisfy the information requirements of a robust EIA and HRA using existing information for sites and species where the suite of information requirements is well understood. The use of existing data may be seen as an adequate approach where an initial risk assessment (carried out at the scoping stage of projects) has concluded that there is a low risk of impacts leading to significant population level effects. However, the assessment of the usefulness and appropriateness of existing data can be extremely difficult and highly subjective. It is impossible to be prescriptive about how much information is sufficient; how recent is recent enough, how close to the project site is close enough and how precise is precise enough. There may be circumstances in which values from the literature for a given species from other sites may be sufficiently informative (e.g. incorporating the depth distributions of seals tagged elsewhere into collision risk modelling for a given site), the degree to which this is appropriate may depend on a number of factors such as the consistency across available datasets and the similarity of sites. The suitability of existing data will need to be assessed on a site and impact specific basis and will involve a degree of expert judgement and subjectivity. This is incorporated into the decision making process outlined in the next section.

A particular consideration in relation to tidal-stream energy sites is that “tidal rapids” are relatively small subsets of the distribution of most marine mammal populations and they are unusual habitats which are physically and biologically unlike the wider habitat of the species in many respects. In general they have received little research attention or survey effort, not least because they are challenging areas in which to operate. Thus, extrapolation from other better studied, non-tidal areas is sometimes difficult to justify.

Where the risk of an impact occurring is uncertain but there are firm grounds for being confident that they can be mitigated, the option of deploying with mitigation might be considered. An adaptive management approach, such as the approach taken at SeaGen in Strangford Lough (Savidge et al., 2014), could be developed where mitigation is applied and subsequently relaxed (or strengthened, depending on the findings) as operational data are collected and our understanding about actual impacts increases. In theory, and depending on project scale, an alternative approach could also be taken where the first step is to deploy and monitor and the need for subsequent mitigation being assessed after a short period of monitoring. This latter approach is only likely to be permitted where it has been demonstrated that a short period of operation will not result in any significant impacts and may require a staged approach to scaling up to larger projects (e.g. the approach taken at MeyGen where an initial stage of up to six turbines has been consented with the requirement to collect environmental data to inform the consenting for subsequent phases of up to 269 turbines). Both of these options hold a significant degree of risk for the developer in that by committing to adaptive management and unknown future mitigation, there is a degree of uncertainty about long term monitoring and mitigation costs that are unlikely to be attractive to investors. There is also the risk that should impacts be identified during the early monitoring phase, the ultimate mitigation is decommissioning and removal of the project. Therefore, there is obvious benefit to the developer in collecting sufficient information prior to consent to understand the future options for adaptive management and mitigation. There is also some risk to the regulator (or their advisors) in that the impacts may be larger than predicted or that mitigation may prove not to be effective in practice. Although in theory these risks are shared with developers and could lead to significant reputational damage and ultimately it would be the regulator(s) that would need to answer to the European Commission should a significant impact occur on any European protected site or species.

4.6.2 New data – how can we decide whether proposed survey methodologies are likely to be fit for purpose?

Given the information requirements for different impact pathways explored in the previous sections, there is a need for an appraisal of whether proposed survey methodologies are fit for purpose. It is crucial that developers and regulators avoid becoming 'locked into' a monitoring strategy that is not appropriate in terms of the amount and quality of data it delivers. For example, to determine whether a line transect survey campaign is likely to deliver relevant density data, which would provide a more robust prediction of impact than relying on 'worst' case estimates from the existing literature, it is worth considering beforehand what might constitute 'acceptable' levels of data quality, and how much survey effort might be needed to achieve such levels.

An important initial step to achieving robust density estimates should be to determine the desired coefficient of variation (CV) or the degree of variability in relation to the estimated densities of marine mammals. The CV is defined as the ratio of the standard deviation σ to the mean μ and is an indicator of precision; where higher CV values imply greater uncertainty (i.e. more scatter around the mean). If CV values are high, the uncertainty surrounding the mean density estimates may become sufficiently large as to make the estimates essentially useless for management.

For example, a density estimate of 1 animal/km^2 that has a CV of 10% would result in a 95% confidence interval around the mean of [0.8, 1.2]. For a CV of 50%, the 95% confidence interval around that same mean density estimate of 1 animal/km^2 would be [0, 2]. In the latter case, this implies that the true mean density would fall between 0 and 2 animals/km^2 for 95% of the time – in other words, anywhere between no animals at all, and double the actual density. At such levels of uncertainty, the density estimate loses its utility in informing decision-making.

It is therefore useful to consider setting acceptable levels of uncertainty beforehand. For example, it may be considered desirable to undertake a survey of sufficient effort such that the estimated density of animals/marine mammals would have a CV of 10%, 20% or 30%. In the case of line transect surveys, guidance has been developed to determine how much survey effort would be required to achieve a density estimate with the appropriate CV (Buckland et al., 2001). This would often require undertaking an initial pilot survey to obtain information on encounter rates from which to calculate a preliminary CV if survey effort or extent is prescribed by cost or logistical constraints. Low detection rates during the pilot survey may result in an unacceptably large CV. In such a case, a decision needs to be taken on whether the survey campaign is worth undertaking at all, or how much merit such data will have for informing relative abundance, or whether more effort is required and is possible, before more resources are committed. This decision should be made by the developer together with the regulator.

In some cases, a 'reverse engineering' approach may be an appropriate way to determine whether pre-application survey is likely to increase the ability to provide a confident prediction of impact. This involves first setting a threshold level of 'acceptable' impact, and then working backwards to calculate the site density that would be required to achieve above this threshold – an assessment could then be made of existing estimates of density and the requirement for further surveys.

This approach relies on two things. 1) a definition of what is an unacceptable level of impact and 2) a method to predict the population consequences of individual impacts (see Box 2).

Box 2: Setting thresholds and assessing population level impact

A variety of methods can be used to define acceptable levels of impact – in Scotland the Potential Biological Removal (PBR) approach (Wade, 1998) has been used to put the predicted mortality to seals from renewable energy projects into a population context, a practice that follows on from the Scottish Government using the PBR approach to issue licences to shoot seals to protect fisheries and aquaculture[1]. In Wales, a collision threshold approach, based on the calculation of PBR for various marine mammal populations, was the basis for the consent of Tidal Energy Limited's DeltaStream Project (CCW, 2009). The PBR approach has an implicit assumption that the target population size is equal or greater to the Optimum Sustainable Population (OSP), which is the number of animals in a population that will result in maximum productivity bearing in mind the carrying capacity of a population. For marine mammals this is thought to be between 50 and 58% of carrying capacity (Taylor and DeMaster, 1993). It is important to note that using PBR a population could still undergo a sizeable decline and still remain at or above the OSP. In addition, under a PBR approach, successively smaller 'takes' are 'available' for additional developments.

The Interim Population Consequences of Disturbance (PCoD) framework (Harwood et al., 2014; King et al., 2015) which provides a stochastic population modelling approach, was originally developed to assess impacts associated with exposure to underwater noise as a consequence of offshore wind farm construction but can be adapted to help assess the population consequences of mortality resulting from collision with tidal devices. Stochastic population modelling can be useful to explore the population level consequences of a given level of impact but it does not incorporate any assumptions of what is considered a desirable population outcome and there would still be a requirement for a definition of acceptable thresholds.

Alternative approaches to assessing the population consequences of individual level impacts include other take based methods such as the IWC catch limit algorithm or alternative stochastic population models similar to the Population Viability Analyses (PVA) used to predict the population consequences of avian collisions with wind turbines. All of these approaches are relatively straightforward when considering the consequences of a predicted number of collisions (although still depend on agreement of what is an acceptable population consequence) but where sub-lethal effects are concerned (e.g. disturbance), it becomes more complicated. There are many uncertainties surrounding the consequences of disturbance on individual survival and fecundity, therefore it is more difficult to translate individual impacts to population level consequences.

5 Determining survey requirements: assessment of the degree of the potential risk of impacts

5.1 Process of assessment

Figure 1 presents a schematic diagram of the process typically followed for determining pre-application survey requirements for a wave and tidal stream project. This representation of the process is currently not defined or documented formally in any guidance – here we attempt to provide a structured framework for the decisions at each stage in this process.

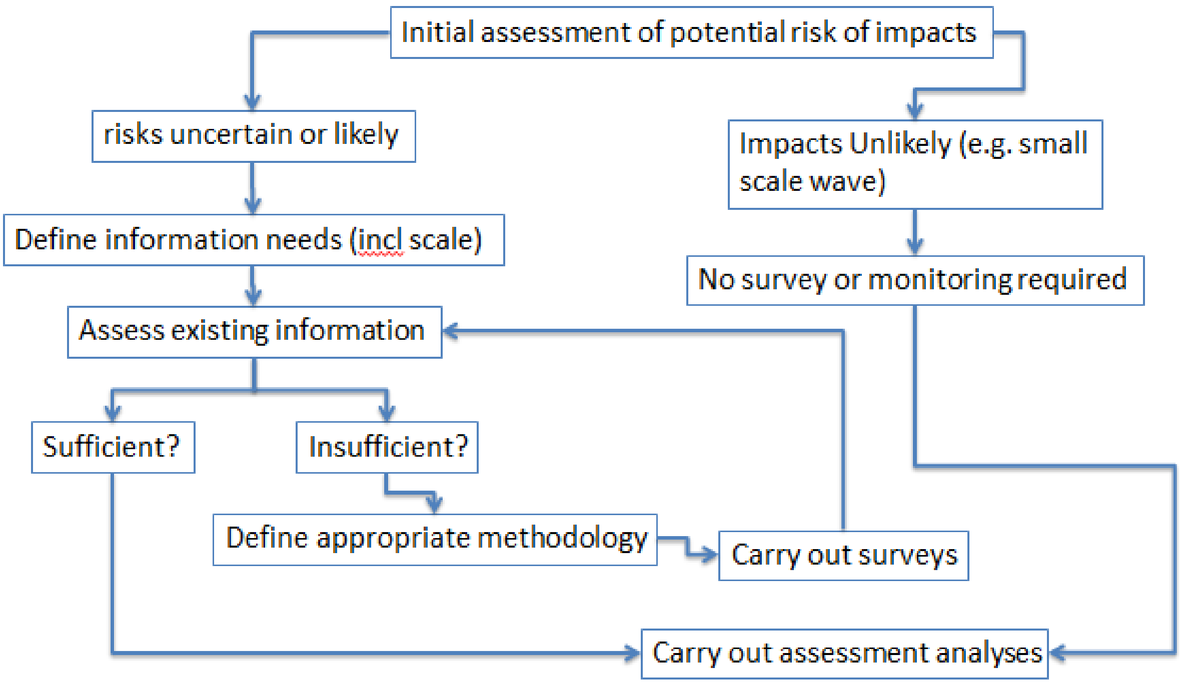

Figure 1. Schematic representation of the process followed to assess the need for pre-application surveys during consultation.

5.2 Risk Assessment Framework to guide information requirements

It should be possible to carry out an initial assessment, at the scoping stage of projects, of the degree of risk posed by a particular project to inform the need for gathering of site specific data. This approach is intended to be similar in principle to Marine Scotland's Draft Survey, Deploy and Monitor policy[2] but is developed specifically for a more detailed consideration of marine mammals

[2] http://www.gov.scot/Topics/marine/Licensing/marine/Applications/SDM

and extends to provide guidance on the type of survey that may be useful to inform the EIA and HRA in each case. This assessment should take account of a variety of features, e.g.:

- The type of device and its physical and mechanical features and the resulting likely impact pathways – for example a horizontal axis- rotor tidal stream turbine with large, fast moving parts that could cause injury and mortality to a marine mammal poses a different level of risk of impact than a surface floating wave energy structure with no such apparent pathway for injury. This would involve a review of the evidence base, including research and knowledge of impacts for a given device or similar types of device. This assessment would need to be carried out for each specific impact pathway identified.
- Scale of project – a single demonstrator device is likely to pose a much smaller risk of impacts than a large array of devices;
- Site sensitivity – the extent to which the proposed project site is used by marine mammals, the importance of the site for those animals and the degree of legislative protection afforded those animals (either as EPS and/or as part of an SAC population) will affect the degree of risk of impacts.
- Duration of project - clearly the risk of significant impacts will be higher for projects of longer duration.

It will be important to recognise that these categories are not independent of each other, for example a small scale deployment of a technology with a low risk of impact in an SAC should not be judged as a higher risk purely because of its location in isolation from other factors. Similarly, the current status and condition of any relevant marine mammal population should be taken into account; a small project which might result in a small impact on a declining population would probably pose a more substantial problem than a larger project impacting on a very healthy and increasing population. Similarly it is difficult to assess the risk posed by a particular technology completely independently of consideration of the deployment site, particularly for the potential for disturbance and barrier effects. It is therefore expected that this process will be somewhat iterative and will involve discussion between the developer, the regulator and statutory advisors. However defining some guiding principles for the process and decisions within it, at least provides a transparent and structured framework around which to base these discussions.

It is anticipated that this process would be followed by developers at the scoping stage of projects and as part of any 'evidence plan' development. It is expected that it will allow developers to take responsibility for decisions on pre-application data gathering, to fully understand the rationale behind any proposed data collection and understand the costs and benefits of any survey work. This will allow developers to understand the risks of not collecting sufficient information to inform an adequate EIA and HRA and the subsequent restrictions which might result, in the form of mitigation measures and other licence conditions.

5.3 Matrix based Risk Assessment

This section sets out a proposed framework for making decisions about the information requirements, (and therefore survey needs) to inform EIA and HRA processes for wave and tidal stream projects based on the features introduced in the previous section. We recognise that it is challenging to establish a process for determining a level of risk of impacts which adequately combines all these features, but consider it important that such a process is as transparent as possible.

There is also a need to avoid being too prescriptive in defining this process, as the level of understanding of impacts is likely to develop with time. Any framework for assessing riskiness needs to be flexible to adapt to new information – for example, the ability to change the assignment of a particular technology type to a lower risk category as data become available. Such classifications may be necessarily subjective but at least under the proposed approach, they can be determined as part of a transparent framework and there would be a requirement for detailed justification to back up any low risk classification.

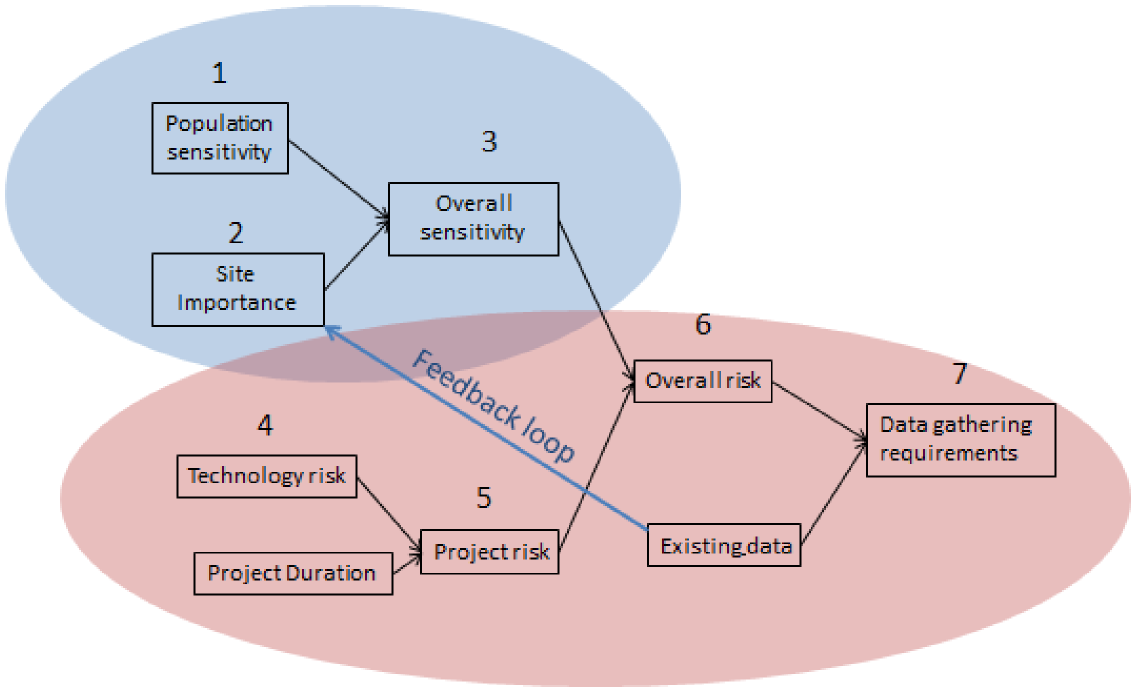

Figure 2. Schematic diagram of staged matrix approach to determining pre-consent data gathering requirements. The numbers (1-7) refer to the stages which are detailed further in subsequent tables. The red shaded area represents the technology/duration assessment parts of the process which need to be carried out separately for each identified impact pathway, the blue shaded area show the receptor/location assessment parts which need to be carried out separately for each species/population potentially impacted.

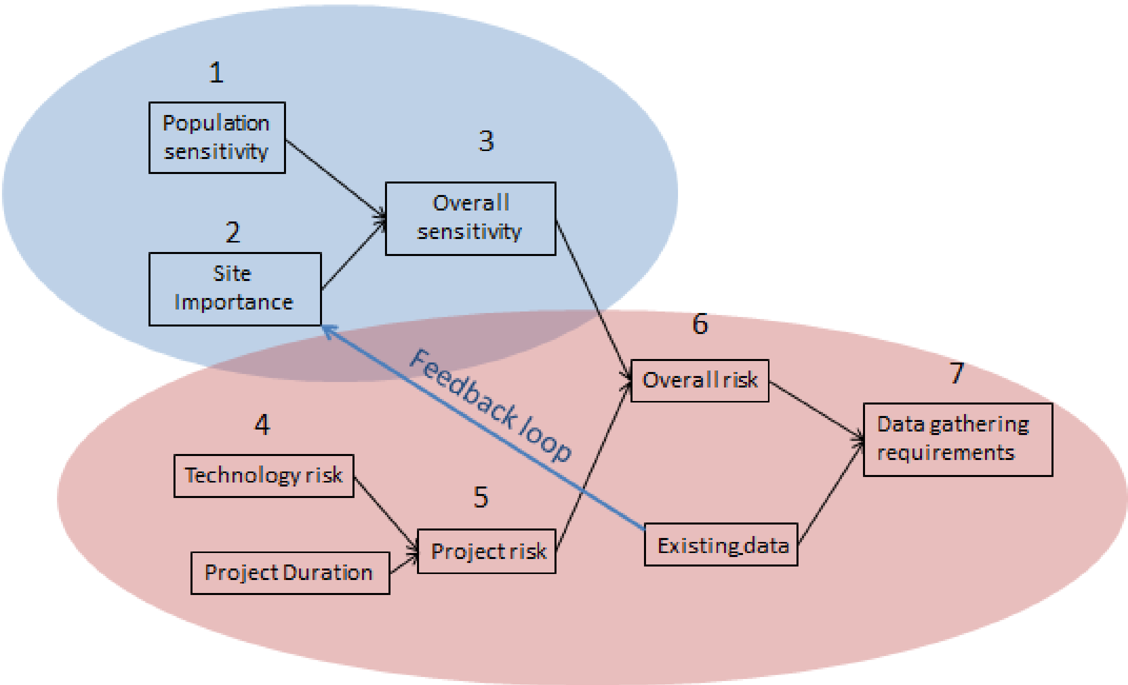

Figure 2 presents a summary of a staged matrix-based approach to combining assessments of project risk of impacts and mammal sensitivity to those impacts for determining subsequent survey requirements. The individual stages of this approach are detailed further in Tables 2-7. We propose that a classification of overall 'risk' (stage 6 in Figure 2) is assigned to a proposed project. This assessment of risk is based on the combined outcomes of two separate strands of risk assessment, one (blue-shaded) determining the overall sensitivity of the potential receptors (key marine mammal species) through a consideration of the sensitivity of the population(s) in question and the importance of the proposed location for those species, and the other (shown in red) determining the classification of risk posed by the project, taking into account the scale of the project, the technology type, and the duration of the project. Then the resulting classification of overall risk is used to define the data gathering (survey) requirements for a project. Whilst the receptor/location assessment is likely to be the same regardless of the impact pathway concerned, the technology/duration assessment will differ depending on impact pathway. Stages 5-7 will therefore need to be developed for each impact pathway identified as potentially significant at the scoping stage, as the data requirements may differ considerably between different impacts (e.g. disturbance vs collision).

Stage 1: SENSITIVITY OF MARINE MAMMAL POPULATIONS.

Each management unit (MU) for marine mammals in Welsh waters (as defined by the UK Inter-Agency Marine Mammal Working Group, IAMMWG, (2015)) has been pre-assigned a level of sensitivity of either Low, Medium or High based on an appraisal across a number of features. The main factors that contribute to the sensitivity of a marine mammal population to impacts are the current population size and distribution, current and recent trends in demographic parameters (fecundity, juvenile and adult survival) and life history variables such as age at maturity and longevity. The ability to adapt to change and the degree of existing threats are also important.

Table 2 provides the predetermined sensitivity for each MU. Note that for all species, there is only a single MU of relevance to Wales, hence presented as a species in the table. Although other marine mammal species are present in Welsh waters and may be species of concern for some developments, it is expected that that for most developments, the primary species of concern will be harbour porpoise, grey seals and/or bottlenose dolphins. This is because these are the most abundant species in Wales and are the only marine mammal Annex II species present in Wales. Other species are unlikely to drive survey requirements in the absence of these three species, however there may be some exceptions, such as Risso's dolphins around Bardsey Island (Llŷn Peninsula) and common dolphins in the southwest of Wales (e.g. Outer Bristol Channel and Pembrokeshire).

Table 2. Sensitivity classification of Welsh marine mammal populations

Species	Sensitivity	Rationale
Grey seal	**Low**	Moderately large population Favourable condition (increasing population) Moderately fast maturing species Moderately long lived Wide ranging species
Bottlenose dolphin	**High**	Small population Favourable condition (stable population) Moderately slow maturing Moderately long lived Not a highly mobile population
Harbour porpoise	**Low**	Large population Favourable condition (unknown whether stable or increasing) Moderately fast maturing species Moderately long lived Wide ranging species
Common dolphin	**High in some areas**[3]	Moderately large population Favourable condition (stable population) Moderately slow maturing species Moderately long lived Wide ranging species

Stage 2: IMPORTANCE OF THE PROJECT LOCATION FOR MARINE MAMMALS

This stage assigns a classification of Low, Medium or High to the project location based on its ‘importance’ to marine mammals. Importance is assessed on the basis of a number of features:

- The degree of connectivity between the proposed project location and sites designated under national and international nature conservation legislation, in particular Special Areas of Conservation (SACs) where marine mammal species are named as a qualifying feature and have corresponding conservation objectives.
- Prior knowledge on the density, abundance, distribution and persistent presence of European Protected Species at a project location.

[3] Common dolphin are only likely to be a key consideration in the outer Bristol channel due to high reported densities there (Baines and Evans, 2012).

- Evidence for 'functional importance' e.g. previous studies suggesting that the project location provides a particular function for a marine mammal species such as a known pupping or calving site, or a foraging 'hotspot'.

We have not defined prescriptive levels of connectivity or distance from protected sites as thresholds would be somewhat arbitrary, a degree of expert judgement will be needed to decide upon classifications depending on the biology of the particular species and available evidence such as from photo-identification studies. Similarly, judgements about the importance of a location for a particular species will depend on the biology of the species and the nature of the habitat concerned. This stage of the risk assessment is likely to require iterative discussion between the developer, regulators and advisors.

It is important to highlight that there is a clear dependency between this assessment and the availability of existing data– i.e. there needs to be some previous understanding of the importance of an area or connectivity with protected sites, based on existing data, to justify a lower sensitivity classification. Where there is insufficient information at the first stage to determine sensitivity then a precautionary approach would probably assign a classification of 'high', unless justification can be provided for a lower classification. A degree of pragmatism will be applied to this judgement. For example, if a project were proposed in a highly populated or frequented coastal area, even in the absence of site specific marine mammal survey data, it might be reasonable to suggest that if it were an area of high and persistent presence of marine mammals, this would likely be common knowledge amongst coastal and marine users. The absence of site specific survey data would not in this case lead to an automatic assumption that it could be an important marine mammal area in the absence of empirical data. Conversely if an area were highly remote with little previous knowledge from marine and coastal users, it would be reasonable to default to a precautionary set of assumptions and require data collection if there was a risk of significant impacts from the technology or project perspective.

There is therefore a need for a feedback loop in the process that would allow an initial period of data collection to refine the assessment of the need for data collection to continue. This reassessment could take place after pre-defined periods of time e.g. after 6 months or after 1 year, or once certain agreed data have been collected. This could result in an initial uncertainty based high risk classification to be reduced on the basis of 6 months or a year of data collection. Of course this initial data collection may not always reduce the classification from a higher to a lower grade as the data collected may reveal that the precaution was justified.

Table 3. Stage 2: Classification of the importance of the location for marine mammals

Importance of location	Description
Low	Distant from SAC, evidence of no or limited connectivity, transient EPS may be present occasionally. No evidence (despite previous survey effort or high human use of area) of the site having any particular importance for marine mammals.
Medium	Some evidence of connectivity with SAC, persistent use by low-medium densities of EPS. Animals may be using the area to feed or breed but it is not their primary foraging or breeding area.
High	Inside an SAC or high levels of connectivity with an SAC population, known area of persistent high EPS density. Evidence that there is a high degree of feeding or breeding at the location.

Stage 3: OVERALL SENSITIVITY

This stage is a matrix which combines the classifications from stage 1 and 2 to provide an overall sensitivity encompassing marine mammal population sensitivity and the importance of the location.

Table 4. Overall Sensitivity based on the outputs from stages 1 and 2.

		Sensitivity of population (output from stage 1)		
		Low	Medium	High
Importance of location (output from stage 2)	Low	Low	Low	Medium
	Medium	Low	Medium	High
	High	Medium	High	High

Stage 4: TECHNOLOGY RISK ASSESSMENT

This stage of the risk assessment process requires an assessment of the likelihood that the device(s) associated with a proposed project will result in impacts on marine mammals. The key impact pathways considered here are;

- Collision with devices or moving components
- Disturbance and habitat displacement
- Barrier to movements

Other impact pathways such as indirect effects on prey species, or toxic contamination have not been included here. Whilst they may be a consideration within the wider environmental assessment process, they are unlikely to drive survey requirements for marine energy projects in isolation from the key impact pathways listed above.

Until there is a better empirical understanding of the factors influencing whether wave and tidal stream devices are likely to result in collision, disturbance / habitat displacement or barrier effects on marine mammals, this stage of the risk assessment will need to be undertaken through iterative discussion between the developer, regulators and advisors. For each impact pathway a series of key principles are proposed, to guide discussions and seek to ensure that the rationale underlying the allocation of Low, Medium or High risk is clear and consistent, whilst remaining flexible. These guiding principles are provided below. Hypothetical examples of projects which might be considered to be Low, Medium and High risk are provided to illustrate how the guiding principles might be applied.

Collision with devices or moving components

Key guiding principles for assessing collision risk;

- Predominantly likely to be an issue for tidal energy devices, but might also be an issue for wave energy devices with highly mobile components.
- Likely to be an issue for devices which have exposed rotor blades.
- Likely to be more of an issue for devices with ducted rotor blades due to the limited opportunity for animals to evade contact with rotor blades.
- Likely to be an issue for highly mobile devices (e.g. kites) or devices with highly mobile components.
- Unlikely to be an issue for devices without exposed rotor blades or other exposed mobile components.
- Unlikely to be an issue for devices where moving parts are completely enclosed and protected.
- Risk of collision is likely to increase with blade tip speed or speed of moving parts.
- Risk of collision is likely to increase with the number of devices; although the exact nature of this relationship is not known (i.e. it is not necessarily linear).

A number of other factors and device and array characteristics may affect collision risk, including the size of devices or moving components in relation to body size of marine mammal receptors or the layout of an array for multiple devices. The position in the water column will also affect risk depending on the depth distribution of animals – for example, telemetry data from 20 tagged grey seal pups from Anglesey, Bardsey Island, Ramsey Island and on the Anglesey Skerries found that grey seals spent the majority of their time either at the surface or at the bottom of a dive with little time spent in the mid water depths (Thompson, 2012). This would mean that a device situated at the surface or on the seabed could pose a greater risk to this species than one in mid-water. Whereas by contrast most studies on coastal bottlenose dolphins have reported little time spent in waters deeper than 10m (Hastie et al., 2006 and Cockeran & Martin, 2004) meaning that a bottom mounted device in depths >10m could pose less of a risk to this species.

At the time of writing, understanding about the likely influence of these and other factors is very limited and certainly not sufficient to support any general assumptions or principles. However, the guiding principles listed above do not preclude the consideration of factors not listed, which might reduce collision risk. Where a technology developer considers that a particular feature or characteristic of their device reduces collision risk for marine mammals, supporting evidence should be provided. This might include empirical data collected from deployments elsewhere, or outputs from modelling exercises to predict the influence of device features or characteristics. Whilst this type of information might usually be expected to be provided within an Environmental Statement for a project, there are clearly benefits to the developer in providing this evidence at an earlier stage in the pre-application process to allow the initial risk assessment for their proposal to be refined.

As a result of this uncertainty and lack of empirical data, we have not included explicit thresholds for array size – this will allow a degree of flexibility and for each project to be assessed in the context of the existing information relating to impacts at any given time.

Table 5. Hypothetical project examples to illustrate the application of guiding principles for assigning Low, Medium and High collision risk.

Collision risk	Project description
Low	• Arrays of devices with completely enclosed blades and no exposed moving parts. • A small to medium array of devices with extremely slow-moving exposed moving parts.
Medium	• Single devices or small arrays of tidal devices with exposed blades (including ducted turbines), or devices with other fast moving parts.
High	• A medium or large array of tidal devices with exposed blades (including ducted turbines), or devices with other fast moving parts.

NB: It should be noted that the risk of long-term effects on mammal populations increases with project duration. This relationship is addressed in Stage 5 of the assessment so is not included here.

Disturbance and habitat displacement

Key guiding principles for assessing the risk of disturbance and habitat displacement impacts;

- Unlikely to be an issue for single devices or very small arrays, unless in particularly sensitive locations (see Box 3 below).
- Likely to be an issue for 'noisy' devices (i.e. with gearboxes or other noisy components).
- Risk is likely to increase with the number of devices or project footprint (i.e. total area of array), although the exact nature of this relationship is not known.

- The risk of an impact during construction and decommissioning may be different than during operation

Box 3: Relationship between risk of disturbance, habitat displacement and barrier impacts and the location and functional importance of project area.

It is important to note that the consequences of disturbance or habitat displacement are likely to be heavily influenced by the location and functional importance of the proposed project area for marine mammals and the availability of equivalent or alternative habitat. For example the following factors are likely to increase risk;

- Projects located in areas known to be functionally important for marine mammals, such as for breeding or foraging activity.
- Projects in confined straits or channels potentially important for transit.
- Arrays of devices occupying an entire area for which there is no equivalent habitat available or if switching to the equivalent habitat would incur cost.

Whilst these factors are also incorporated into Stage 2 of the overall risk assessment process, they will be a consideration when assessing the level of risk for this impact pathway.

Table 6. Hypothetical project examples to illustrate the application of guiding principles for assigning risk of disturbance and habitat displacement impacts as Low, Medium and High

Disturbance / habitat displacement risk	Project description
Low	• Single devices and small arrays, with low noise construction methods and low operational noise in open sea locations, including areas of functional importance. • Medium arrays of devices located in areas where existing data demonstrates no functional importance.
Medium	• Single devices or small arrays partially occupying (physically and/or acoustically) an area of apparent functional importance, with limited equivalent habitat available. • Medium or large arrays partially occupying (physically and/or acoustically) an area of apparent functional importance, but where equivalent habitat is available with low apparent switching cost.
High	• Medium or large arrays of devices occupying (physically and/or acoustically) an area of apparent functional importance or where functional importance is unknown, but plausible and where switching cost is likely to be high or limited equivalent habitat available.

NB: It should be noted that the risk of impacts for all types of devices and impact pathways increases with project duration. This relationship is addressed in Stage 5 of the assessment so is not included explicitly here.

Barrier to movements

Key guiding principles for barrier effects;

- Unlikely to be an issue in open sea areas unless arrays are very large.
- May be an issue for even single devices or small arrays in restricted areas (e.g. narrows, channels or straits) - data from SeaGen in Northern Ireland suggests single device did not create barrier but it is difficult to generalise to other species and sites .
- More likely to be an issue for large arrays or for medium arrays in confined narrows or important transit areas for marine mammals.
- Risk of impacts increases with increasing size of array and increasing levels of noise emissions.

Table 7. Hypothetical project examples to illustrate the application of guiding principles for assigning the risks of a barrier effect as Low, Medium and High.

Barrier risk	Project description
Low	• Single devices or small arrays in open sea locations.
Medium	• A single device or small array partially occupying (physically and/or acoustically) a strait or narrows. • An array of noisy or large devices located in an open sea area partially occupying (physically and/or acoustically) an area of apparent or plausible importance for transit.
High	• An array of devices fully occupying (physically and/or acoustically) a strait or narrows. • An array of devices fully occupying (physically and/or acoustically) an area of apparent importance for transit.

NB: It should be noted that the risk of impacts for all types of devices and impact pathways increases with project duration. This relationship is addressed in Stage 5 of the assessment so is not included here.

Stage 5: PROJECT RISK ASSESSMENT

Table 8. Project risk, combining the technology risk (from Stage 4) with project duration. This assessment is carried out separately for each impact pathway

		Technology Risk (defined at stage 4 above for each impact pathway)		
		Low	Medium	High
Duration of project	Low (1-3 years)	Low	Low	Medium
	Medium (3-10 years)	Low	Medium	High
	High (>10 years)	Medium	High	High

Stage 6: OVERALL RISK ASSESSMENT

Table 9. Overall risk, combining the project risk (from Stage 5) with sensitivity (stage 3). This assessment is carried out separately for each impact pathway

		Project Risk (defined at stage 5)		
		Low	Medium	High
Overall sensitivity (defined at stage 3)	Low	Low	Low	Medium
	Medium	Low	Medium	High
	High	Medium	High	High

Stage 7: DETERMINING INFORMATION REQUIREMENTS

For each impact pathway we have developed a decision tree process to determine the likely information requirements, based on the assessment of overall risk as low medium or high.

Collision risk (Figure 3)

A classification of low risk after following the process outlined in the previous section would result in little requirement for additional survey, it is likely that a qualitative impact assessment detailing the justification for low risk would be sufficient. If the project was assigned either a medium or high risk overall the developer may have the option of mitigating the risk to acceptable levels. If the developer was prepared to commit to shut down device(s) on the detection of approaching marine mammals should an EIA and HRA be unable to demonstrate that there won't be a significant impact based on existing data, any collision risk would be reduced to insignificant and therefore consent could be granted conditional on this mitigation, without the requirement for pre-application surveys, leaving the developer the option of investing more in refining collision risk estimates post-consent. If the developer was unable to make this commitment at this stage then there would be a requirement for a quantitative collision risk assessment. The information required for this is set out in in detail in section 4.2 but primarily relates to information needed to predict the likely encounter rate between marine mammals and the devices, so an indication of site specific density or transit rate is required along with an understanding of depth distribution of animals in relation to the position of the device in the water column. Information on how these vary over time (over the tidal cycle, day/night or seasonally) will be useful to refine estimates of risk but in the absence of this information, the upper confidence limits of averages will generally be used in any predictive models. In the absence of sufficient existing information, some degree of survey will likely be required (although see considerations below and in section 4.6.2 regarding precision and uncertainty).

The question of how much data are sufficient to inform a risk assessment is a difficult one and is best answered on a project and site specific basis. However the spatial and temporal resolution of previous surveys, the overall area of coverage, and the methods used to collect and analyse the data and how long ago the data were collected will be material considerations. General rules of thumb can be applied (e.g. data should be from within the last five years, though there may be exceptions to this).

The precision of the estimates that are likely to result from possible surveys are also a consideration – see the discussion in 4.6.2. If it is unlikely that surveys will provide density estimates with the required precision to provide a sufficiently precise estimate of collision rates (e.g. because the site is very small) then it may be that an approach based on existing data will be appropriate. In this case, a 'reverse modelling' approach is possible, whereby one could assess the population level consequences of range of collision rates – this could be combined with the collision probability calculated using the component parts of existing models to allow a 'threshold' encounter rate to be set which could be viewed in context of the existing information for a site – i.e. if a worst case assessment of collision based on a worst case interpretation of existing data (e.g. upper confidence limit of an appropriate regional density estimate) would not have a significant effect on a given population unit then it is unlikely that further pre-consent survey would be required.

Although not explicitly linked in Figure 3. It is possible that a deploy, mitigate and monitor approach will still require a collision risk estimate to determine the power and duration of the monitoring required, although this will depend on the confidence in the mitigation approach and the nature of the monitoring employed.

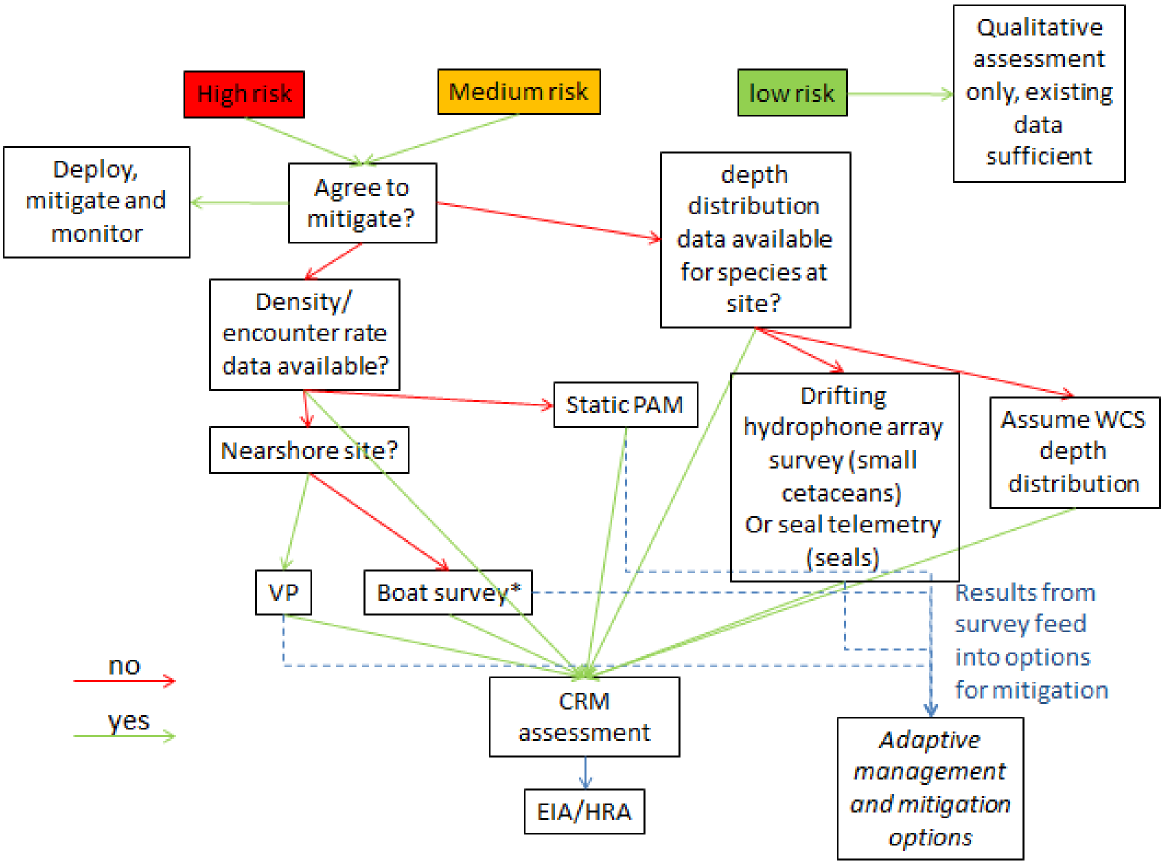

Figure 3. Stage 7: Decision tree to determine the approach to pre-consent data gathering for collision risk depending on the outcome of the risk assessment carried out in stages 1-6. *for boat surveys consideration should be given to the scale of the site – see text for details. The blue dashed lines indicate where information gathered from surveys will be useful to define adaptive management and mitigation options.

Displacement/Disturbance (Figure 4)

As with collision risk above, a classification of low risk would result in little need for additional survey, it is likely that a qualitative impact assessment detailing the justification for low risk would be sufficient. If the project was assigned either a medium or high risk overall, similar to collision risk, the developer would have the option to agree to mitigate the risk to acceptable levels. However, for a risk of disturbance or displacement, mitigation options are generally limited other than through changes to project design. It is therefore likely that in this case there would be a requirement for a quantitative assessment of disturbance/displacement as part of the EIA and HRA. The information required for this is set out in detail in section 4.3 but primarily relates to information needed to predict the magnitude of any displacement or disturbance, i.e. how many animals are likely to be affected, when and for how long. This will require information on the abundance and distribution of marine mammals across the project location and associated impact footprint. The consequences of any displacement or disruption of normal activities at the project location (and associated project footprint) will also need to be addressed; therefore an understanding of behaviour and functional use of the project location is also required. In the absence of site specific data to inform these assessments, a worst case scenario interpretation will be adopted – i.e. that the site is important for a range of life functions and that displacement of individuals will have a significant effect on their ability to breed and survive.

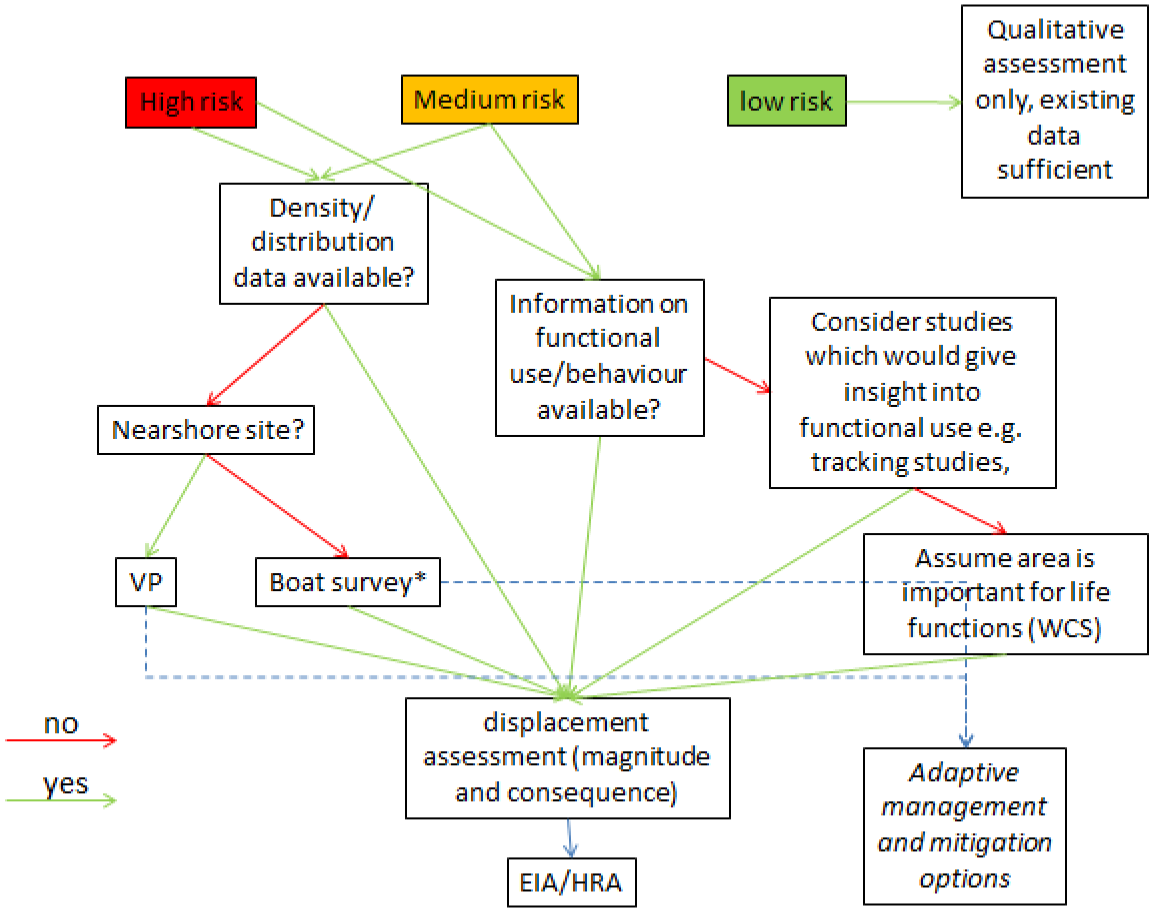

Figure 4. Stage 7: Decision tree to determine approach to pre-consent data gathering for displacement/disturbance depending on the outcome of the risk assessment carried out in stages 1-6. *for boat surveys consideration should be given to the scale of the site – see text for details. The blue dashed lines indicate where information gathered from surveys will be useful to define adaptive management and mitigation options.

5.3.2 *Worked Examples*

1) **Small scale tidal demonstrator project (up to 3 devices for 5-10 years) with some previous surveys in the area, tens of kilometres away from a number of SACs and pSACs designated for marine mammal features.**

a. **Collision risk**

Harbour porpoise	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	**High** based on relatively recent existing data suggesting high density (line transect boat surveys and static PAM deployments), close proximity to pSAC for harbour porpoise.
Stage 3: Overall sensitivity	Low x High **= Medium**
Stage 4: Technology Risk	Device risk high, scale small = **Medium**
Stage 5: Project Risk	Tech risk medium, duration medium **= Medium**
Stage 6: Overall Risk	Medium x Medium = **Medium**
Stage 7: information requirements	Recent density information available combined with medium risk = Consideration of site specific survey required to characterise worst case encounter rates for use in CRM, otherwise absolute WCS applied based on existing data (upper confidence limits). Collation of data from other studies to inform depth distribution otherwise WCS assumed in CRM. Developer may want to collect additional data to refine assessment below a WCS and/or understand options for adaptive management – i.e. it may be important to understand how density varies with the tidal cycle and seasonally.
Bottlenose dolphins	
Stage 1: Population/species sensitivity	**High**
Stage 2: Location sensitivity	Photo ID data suggests connectivity between project location and Cardigan Bay SAC, encounter rate relatively low based on large scale regional data but no systematic local survey = **Medium**
Stage 3: Overall sensitivity	High x Medium **= High**
Stage 4: Technology Risk	Device risk high, scale small = **Medium**
Stage 5: Project Risk	Duration high, tech risk medium **= Medium**
Stage 6: Overall Risk	High x Medium = **High**
Stage 7: information requirements	Little local information on encounter rates but recent regional average density information available

	combined with high risk = Consideration of site specific survey (static acoustic with ability to determine dolphin species) to characterise worst case encounter rates for use in CRM, otherwise regional WCS applied but there will be a degree of uncertainty as to the degree that regional estimates are appropriate at a local scale. Collation of data from other studies to inform depth distribution otherwise WCS assumed in CRM. Developer may want to collect additional data to refine assessment below a WCS and/or understand options for adaptive management. Because of the high level of sensitivity and HRA considerations, the onus is on the developer here to provide data to rule out the likelihood of a significant effect. The loss of a single individual of a population this size could be deleterious – in this situation agreement to mitigate (detect and deter, or detect and shut down) in the presence of bottlenose dolphins might be a good strategy.
Grey seals	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	Small degree of connectivity with SACs quantified by photo id and telemetry studies, existing survey data suggests low encounter rate but no systematic survey = **Medium**
Stage 3: Overall sensitivity	Low x Medium **= Low**
Stage 4: Technology Risk	Device risk high, scale small = **Medium**
Stage 5: Project Risk	Duration high, tech risk medium **= Medium**
Stage 6: Overall Risk	Low x Medium = **Low**
Stage 7: information requirements	No site specific data gathering required for grey seal assessment alone. Qualitative CR assessment required.

b. Displacement/disturbance

Harbour porpoise	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	**High** based on existing data suggesting high density (line transect boat surveys and static PAM deployments), proximity to pSAC.
Stage 3: Overall sensitivity	Low x High **= Medium**
Stage 4: Technology Risk	Device risk low, scale small = **Low**

Stage 5: Project Risk	Tech risk low, duration medium **= Low**
Stage 6: Overall Risk	Medium x Low = **Low**
Stage 7: information requirements	Low risk means that existing data will be sufficient to inform an assessment of the magnitude and significance of any displacement. Little data available on functional use of the site but scale and duration of project and technology risk low therefore little requirement for additional survey.
Bottlenose dolphins	
Stage 1: Population/species sensitivity	**High**
Stage 2: Location sensitivity	Photo ID data suggests connectivity with Cardigan Bay SAC, encounter rate relatively low based on large scale regional data but no systematic local survey = **Medium**
Stage 3: Overall sensitivity	High x Medium **= High**
Stage 4: Technology Risk	Device risk low, scale small = **Low**
Stage 5: Project Risk	Tech risk low, duration medium **= Low**
Stage 6: Overall Risk	High x Low = **Medium**
Stage 7: information requirements	Little local information on encounter rates but recent regional average density information available combined with medium risk = Consideration of survey to enable site specific quantitative assessment of impact, otherwise existing data used to inform WCS. Density/abundance of primary importance but developer should consider survey to inform likely functional use otherwise WCS assumed in assessment (site important for foraging/breeding/transit and that displaced animals will not feed or breed)
Grey seals	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	Small degree of connectivity to SACs quantified by photo id and telemetry studies, existing survey data suggests low encounter rate but no systematic survey = **Medium**
Stage 3: Overall sensitivity	Low x Medium **= Low**
Stage 4: Technology Risk	Device risk low, scale small = **Low**
Stage 5: Project Risk	Tech risk low, duration medium **= Low**
Stage 6: Overall Risk	Low x Low = **Low**

Stage 7: information requirements	No site specific data gathering required for grey seal assessment alone. EIA focus on demonstration of low project risk using regional or national datasets and justification for low technology risk.

2) Small scale (10MW) demonstrator wave project offshore, 5-10 years (not in SAC or pSAC)

a. Displacement/disturbance

Harbour porpoise	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	No local survey data but regional scale density estimates suggests medium density. Low proximity to pSAC = **Medium**
Stage 3: Overall sensitivity	Low x Medium **= Low**
Stage 4: Technology Risk	Device risk low, scale small = **Low**
Stage 5: Project Risk	Tech risk low, duration medium **= Medium**
Stage 6: Overall Risk	Low x Medium = **Low**
Stage 7: information requirements	Recent regional average density information available combined with low risk = little requirement for additional survey = existing data used to inform WCS.
Bottlenose dolphins	
Stage 1: Population/species sensitivity	**High**
Stage 2: Location sensitivity	limited connectivity with SAC, encounter rate relatively low based on large scale regional data but no systematic local survey = **Low**
Stage 3: Overall sensitivity	High x Low **= Medium**
Stage 4: Technology Risk	Device risk low, scale small = **Low**
Stage 5: Project Risk	Tech risk low, duration medium **= Low**
Stage 6: Overall Risk	Medium x Low = **Low**
Stage 7: information requirements	Little local information on encounter rates but recent regional average information available combined with low risk **=** little requirement for additional survey = existing data used to inform WCS.
Grey seals	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	low connectivity with SACs quantified by telemetry studies and photo ID, existing survey data suggests low encounter rate but no systematic survey = **Low**

Stage 3: Overall sensitivity	Low x Low = **Low**
Stage 4: Technology Risk	Device risk low, scale small = **Low**
Stage 5: Project Risk	Tech risk low, duration medium **= Low**
Stage 6: Overall Risk	Low x Low = **Low**
Stage 7: information requirements	No site specific data gathering required for grey seal assessment alone. Collation of data from other studies to inform impact assessment.

3) **Large scale (100MW) demonstrator wave project offshore, project duration >10 years (not in SAC or pSAC)**

b. **Displacement/disturbance**

Harbour porpoise	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	No local survey data but regional scale density estimates suggests medium density. Low proximity to pSAC = **Medium**
Stage 3: Overall sensitivity	Low x Medium **= Low**
Stage 4: Technology Risk	Device risk low, scale large= **Medium**
Stage 5: Project Risk	Tech risk med, duration high **= High**
Stage 6: Overall Risk	Low x High= **Medium**
Stage 7: information requirements	Recent regional average density information available combined with Medium risk **=** Consideration of survey to enable site specific quantitative assessment of impact, otherwise existing data used to inform WCS. Density/abundance of primary importance but developer should consider survey to inform likely functional use otherwise WCS assumed in assessment (site important for foraging/breeding/transit and that displaced animals will not feed or breed)**.**
Bottlenose dolphins	
Stage 1: Population/species sensitivity	**High**
Stage 2: Location sensitivity	limited connectivity with SAC, encounter rate relatively low based on large scale regional data but no systematic local survey = **Low**
Stage 3: Overall sensitivity	High x Low **= Medium**
Stage 4: Technology Risk	Device risk low, scale large= **Medium**
Stage 5: Project Risk	Tech risk med, duration high **= High**

Stage 6: Overall Risk	Medium x High= **High**
Stage 7: information requirements	Little local information on encounter rates but recent regional average density information available combined with high risk = Consideration of survey to enable site specific quantitative assessment of impact, otherwise existing data used to inform WCS. Density/abundance of primary importance but developer should consider survey to inform likely functional use otherwise WCS assumed in assessment (site important for foraging/breeding/transit and that displaced animals will not feed or breed)
Grey seals	
Stage 1: Population/species sensitivity	**Low**
Stage 2: Location sensitivity	low connectivity to SACs quantified by telemetry studies, existing survey data suggests low encounter rate but no systematic survey = **Low**
Stage 3: Overall sensitivity	Low x Low **= Low**
Stage 4: Technology Risk	Device risk low, scale large= **Medium**
Stage 5: Project Risk	Tech risk med, duration high **= High**
Stage 6: Overall Risk	Low x High= **Medium**
Stage 7: information requirements	Little local information on encounter rates but recent regional average density information available combined with medium risk = Consideration of survey to enable site specific quantitative assessment of impact, otherwise existing data used to inform WCS. Density/abundance of primary importance but developer should consider survey to inform likely functional use otherwise WCS assumed in assessment (site important for foraging/breeding/transit and that displaced animals will not feed or breed)

6 Survey methodologies

This section details the range of survey methodologies likely to provide the information required to understand the main impact pathways as detailed in Section 4. Table 10 provides an appraisal of each methodology against each information requirement. This appraisal is presented separately for seals and small cetaceans. The subsequent text provides a summary of how available survey methodologies can be applied to understand these impact pathways. Each methodology is then described in more detail in Appendix One.

Table 10. Appraisal of various survey techniques against the possible information requirements defined in Section 0.

SEALS								
Survey technique	**Local density**	**Horizontal Relative Distribution**	**Vertical Distribution**	**Variation over tidal cycle**	**Behaviour** (what are they doing at the site in terms of movement and activity)	**Turnover/residency**	**Population size**	**Connectivity to designated sites**
Line transect boat surveys	Yes but limited by spatial and temporal scale of effort	Yes	No	Yes – with intensive effort	Limited	No	Not at scale of development site	No
Telemetry	No	Yes	Yes	Yes	Yes	Yes	No	Yes
Shore based VP surveys	Yes (for limited inshore waters)	Yes (for limited inshore waters)	No	Yes (for limited inshore waters)	A limited amount depending on VP location	No	No	No
Haul out counts	Yes but not at sea, unless coupled with telemetry data	No	No	Yes – but not at sea	No	No	Not at scale of development site	No
Photo ID	No	No	No	No	No	Yes	Yes	Yes

SMALL CETACEANS								
Survey technique	**Local density**	**Horizontal Relative Distribution**	**Vertical Distribution**	**Variation over tidal cycle**	**Behaviour**	**Turnover/ residency**	**Population size**	**Connectivity to designated sites**
Line transect boat survey	Yes	Yes	Requires Development	Yes	Limited	No	Not at scale of development site	No
Static PAM	Possibly with Additional Work	Yes	If appropriate Arrays developed	Yes	Limited	No	Not at scale of development site	No
Drifting hydrophone array	Possibly but biased survey coverage a major issue	Possibly but biased survey coverage a major issue	Yes	Possibly but biased survey coverage a major issue	Yes	No	No	No
Drifting PAM detectors	Possibly in conjunction with arrays but biased survey coverage a major issue	Possibly but biased survey coverage a major issue		Possibly but biased survey coverage a major issue	Yes			
Shore based VP surveys	Yes (for limited inshore waters)	Yes (for limited inshore waters)	No	Yes (for limited inshore waters)	Limited	No	No	No
PhotoID	No	No	No	No	No	Yes for marked species such as BND	Yes for marked species such as BND	Yes for marked species

6.1 Collision Risk

The objective of collecting site specific survey data to inform collision risk is twofold; to quantify potential collision risk and to allow an understanding of the options for adaptive management.

Specific information requirements:

- Information on animal flux through the swept area (and how this varies across the tidal cycle)

As discussed in Section 4, this is not often collected and collision risk models often use an estimate of average density at the site and information on dive behaviour as a proxy. Animal flux could be measured using individual tagging studies in conjunction with information on the number of animals using the site as has been done for a small number of sites (e.g. Thompson et al., 2015). Where this is not possible surveys can be carried out which attempt to provide estimates of density. In order to estimate density and, in particular, how it varies across the diurnal cycle, the tidal cycle and throughout the year, surveys need to be carried out regularly and frequently with sufficient effort. Shore based visual surveys can be carried out for relatively low cost if the site is within reasonable distance from a suitable vantage point. However if the site is more than 1-2 km offshore then shore based surveys will not be possible. Line transect boat based surveys can be carried out but it will be costly to achieve enough survey effort to adequately characterise variation across the tidal cycle and over the day/night cycle. Monthly surveys can probably provide a reasonable measure of seasonal variation but it can be very challenging to achieve the necessary levels of effort during the winter months when weather conditions are poor and days are short. Precision of estimates are important to consider – see earlier discussion on the CV of density estimates (Section 4.6.2) and therefore the ability of any survey to provide a more precise estimate than that which might be assumed given a worst case interpretation of existing data should be considered. This may be particularly pertinent to small sites where even intensive effort would provide small sample sizes.
Static passive acoustics could provide information on the relative rate of encounter of different cetacean species at an extremely high temporal resolution. Although methods to determine absolute density from these data are not yet sufficiently developed, they should provide good information on the temporal patterns of occurrence.

- Vertical distribution of animals

Information on the distribution of animals in the water column and on their dive behaviour is important to allow quantitative collision risk estimates but these data have rarely, if ever, been collected by developers. This information is generally factored into predictions from existing data on the species concerned, often from data collected in non-tidal habitats. Data are available for a few tidal sites from telemetry studies or from surveys using drifting hydrophone arrays. Telemetry studies of seals have shown that the depth distributions of grey and harbour seals are fairly consistent, and in most areas around the UK, seals forage benthically. Therefore most of their time is spent either at the surface or at the bottom with very little time spent at intermediate depths (e.g. Thompson et al., 2012; Thompson, 2013,). Equivalent data are scarcer for cetaceans, although data derived from drifting hydrophone arrays in tidal areas are available for a few sites; Gordon et al.,

(2011) off the coast of Anglesey, SMRU surveys in the Great Race, Corryvreckan and Sound of Islay (unpublished but summarised in Hall et al., (2014)). The development of a drifting array buoy as a result of a NERC Knowledge Exchange grant may allow developers to collect these data at tidal energy sites in the future.

- Turnover of individuals

Information on the degree of residency will alter how outputs from collision models are interpreted. Information on turnover cannot come from traditional boat or shore based surveys and could only come from individual based studies using techniques such as photo-ID or telemetry.

- Connectivity with protected sites

This information is important for HRA considerations in cases where the project location may be at some distance from a protected site but may still be used by animals from that site. There are a number of techniques that can provide relevant information. For seals, telemetry is an obvious methodology and there is a large database of track information from tagged seals at a number of SACs around the UK held by the Sea Mammal Research Unit. Not all SACs are covered in this database however. For cetaceans, tagging is not carried out in the UK; therefore photo-id of marked individuals is an effective technique for examining movements of bottlenose dolphins (e.g. Cheney et al., 2014, Veneruso and Evans, 2012). Photo id can also be used for seals (SCOS, 2015). Porpoise are difficult to photograph and lack individual distinguishing marks and therefore this technique has not been used successfully with this species.

6.2 Disturbance

The objective of any site specific survey to inform an assessment of the probability, magnitude and significance of disturbance is also twofold: to quantify how many animals might be affected and to allow an understanding of the consequences and significance of this level of disturbance at individual and population level. Consideration should also be given to a future requirement to monitor the extent of any predicted disturbance post-consent.

Specific information requirements:

- Information on animal abundance and distribution across the project site and potential impact footprint.

To predict how many animals may be affected by displacement or disturbance it is necessary to estimate local abundance or density and how it varies over the project site (and associated impact footprint) and throughout the year and with tidal state. Shore based visual surveys can provide this information and be carried out for relatively low cost if the site is within reasonable distance from a suitable vantage point. However, if the site is more than 1-2 km offshore then shore based surveys will not be feasible. Boat based surveys are possible but the size of the site (and associated impact footprint) may limit the amount of effort and there limit degree of precision that can be achieved for any density estimate and may limit the ability to detect any spatial patterns in distribution. Precision of resulting estimates are important to consider – see earlier discussion on the CV of density estimates (Section 4.6.2) and therefore the ability of any survey to provide a more precise estimate than that which might be assumed given a worst case interpretation of existing data. This may be particularly pertinent to small sites where even intensive effort would provide small sample sizes.

Static passive acoustic monitoring could provide information on the relative rate of encounter of different cetacean species, providing very good temporal coverage with a high probability of being able to detect temporal patterns. Current methods do not provide absolute density. An array of devices should be able to provide information on the spatial variation in a density estimate, although recent work has shown that spatial variation in porpoise encounters can be very high even within a very small area.

- Behaviour of animals /functional importance of a site

By behaviour we mean what animals are physically doing at a site, including any information that allows an assessment of what the site is used for. This could be whether animals are feeding, information on movement patterns, whether and how animals are interacting (including mother-calf/pup interactions). This information is important for informing an assessment of the potential consequences of any displacement or disturbance. For example, habitat displacement will have very different individual and population level consequences if a project location is an important foraging or breeding ground compared to if the area is a transit route. The amount of behavioural data that can be gathered during traditional surveys designed to provide data on density or distribution is limited to observations during sightings such as whether individuals are travelling in a directed manner or are associated with a calf. Feeding is often observed at the surface but a lack of such observations cannot be taken to infer that an area is not important for feeding. Behaviour can be inferred from tagging data (e.g. Russell et al., 2015). A small number of sites may have data available from telemetry studies or from surveys using drifting hydrophone arrays to inform how the site is being used and infer behaviour. As discussed above the development of a drifting array buoy as a result of a NERC Knowledge Exchange grant may allow developers to collect these data at tidal energy sites in the future. Acoustic data can also provide information on the behaviour of cetaceans e.g. the presence of echolocation buzzes used when acquiring prey provides an indication that animals are foraging (e.g. DeRuiter et al., 2009)

- Connectivity with protected sites

This information is important for HRA considerations when the project location may be at some distance from a protected site but may still be used by animals from that site. Relevant techniques are likely to be telemetry and photo ID. Indirect impacts

To understand the potential magnitude and significance of indirect impacts, through effects on prey, the main consideration is whether the project location and associated impact footprint is an important foraging area for marine mammals. This requires the same suite of techniques as described above for disturbance. An understanding of the species they are feeding on, the likely effects of the project on these species, and the numbers of marine mammals using the project location are also important. It is unlikely that site specific survey will provide data on diet composition and this information is likely to be inferred from a combination of published literature on diet preference/composition and an understanding of the prey species composition at the site. As above, connectivity with protected sites is also a requirement to determine the effects on populations associated with these sites. It is unlikely that indirect impacts alone will drive pre-consent monitoring requirements, so the main information requirements for indirect impacts will likely be gathered anyway for other impact pathways (either by survey or from existing literature).

7 Conclusions and Recommendations

This report provides a coherent and transparent pathway for a decision making process for determining pre-consent data gathering requirements for wave and tidal stream projects in Wales. It also links survey recommendations to the impact pathways of specific concern for wave and tidal projects (collision, displacement/disturbance) and provides information on the appropriate methodologies which are capable of answering specific impact related questions, rather than adopting a one-size fits all approach to pre-consent site characterisation survey.

The main recommendations from this work are as follows:

- A risk-based assessment should be made at the pre-application stage of wave and tidal stream energy projects to determine the information requirements for a project. This should be based on existing information on the project location and the sensitivity of the marine mammal populations likely to be found there, in combination with what is known about the risks of impacts occurring as a result of the project in terms of the technology proposed and the scale and duration of the project.
- This process will highlight which impact pathways might result in potentially significant effects and which should be the focus and the drive for any additional data gathering and pre-installation survey. The information in this report is not intended to be completely prescriptive, but is intended to guide the developer, the regulator and statutory advisors through the process of identifying the most important information requirements to inform the EIA and HRA and how best to go about gathering such information.
- This will ensure that any surveys that are carried out will be focused on specific routes of impact and will have a higher probability of successfully informing and improving the impact assessment process.
- A careful assessment can then be made (using the information presented in Section 6 and Appendix 1) of the likelihood of site specific site based investigation providing this information, along with an appraisal of the existing information. A specific and appropriate information gathering process can then be designed and implemented.
- This process will ensure that developers take responsibility for making informed decisions about survey and data needs for their own assessments. This will require a careful cost-benefit analysis, including consideration of the risks involved in being unable to provide a robust impact assessment against the costs of data collection. This will also require an appraisal of the potential operational restrictions resulting from poor or inadequate baseline data.
- At present it is impossible to determine explicit guidance on parts of the staged matrix based assessment of project risk. For example explicit definition of what constitutes a small, medium and large array of devices is difficult without empirical information to support such

distinctions. We recommend that the assessment process is regularly reviewed and updated as information such as that linking project scale to risk of impacts develops.

- Thresholds of acceptable impact at the population level should be defined – this is important for providing a benchmark for the predictions made during project impact assessments.

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9 Appendix 1- detailed accounts of each survey methodology

9.1 Visual (& passive acoustic) vessel-based line transect survey

Vessel based survey is the best established and perhaps most immediately obvious approach for surveying marine mammals. Traditionally animals have been detected visually by teams of observers and increasingly acoustic detections are also being made using towed hydrophone arrays. Often boat based surveys are most productive when both visual and acoustic data are collected at the same time. Methods for conducting vessel based surveys are well developed with standardised methodologies. Most boat based surveys use a line transect approach and this is facilitated by powerful, easy-to-use software for designing line transect surveys and analysing survey data, in particular the DISTANCE program (Thomas et al., 2010; Thomas, 1999).

During line transect surveys the survey vessel transits along a predetermined trackline while observers search the waters ahead and to the side of the survey vessel. The line transect methodology and DISTANCE analysis approach is based on the fact that the ability to detect animals will decrease with distance. Thus, if the actual distribution of animals is random with respect to the trackline and the real density of animals at different ranges from the track is equal, then the number of detections will actually fall with distance from the trackline, reflecting an animal's decreasing probability of detection as range increases. During surveys ranges and bearings to animals are determined (either by eye, using range measuring devices such as reticule binoculars, or, ideally measured accurately with video/photogrammetric techniques e.g. (Leaper and Gordon, 2001). Ranges from the trackline can be calculated from these data by trigonometry and functions fitted to the resulting histograms of detections with range to determine the "sighting function". From this the "effective strip width" can be calculated. The effective strip width is the range at which the number of sightings at greater ranges is equal to the number of "missed sightings" at closer ranges. Thus the number of sightings made at all ranges during the survey would be the same as the number of sightings made within the effective strip width if the sighting probability was uniform over the strip width and equal the sighting probability on the trackline, ie at a range from the trackline of 0m. This procedure compensates for the effect of range on sighting probability. However, we can't assume that even on the trackline all animals will be seen. It may be that animals dive and are underwater while they are within visual range or that the observers simply fail to see them. This proportion of animals on the trackline which are not observed is referred to as g(0) and it is necessary to estimate g(0) to be able to calculate an absolute abundance estimate from the sighting data. Because a variety of poorly known factors are likely to affect this probability, including some that may be specific to particular vessels, observation teams or locations, surveys should attempt to directly measure g(0) during the particular surveys that they will be applied to. The normal way of doing this is to use two independent teams of observers. The probability of detection can then be determined by comparing, on a case by case basis, the proportion of duplicate detections, which is the proportion of sighting occasions when both observers detect the same animal.

While this methodology is well established and accepted the practical difficulties of collecting these data in the field should not be underestimated. Very rarely have surveys carried out by developers at marine renewable energy developments generated absolute density estimates. All marine mammals are difficult to spot at sea and the relatively small and shy species common in Welsh tidal sites (seals and porpoises) are particularly challenging visual targets, especially in high energy marine environments such as tidal and wave energy project locations. Searching for animals is mentally tiring and to operate effectively and consistently observers will need to be rested frequently. Implementing the dual observer method to measure g(0) requires larger teams and the provision of two independent platforms to ensure that observers are isolated from each other (i.e. are unaware of detections made by the other team). These considerations lead to requirements for larger, more expensive vessels and for large field teams.

Leaper and Gordon, (2012) describe a method for calculating g(0) for sighting of seals made during surveys at offshore windfarm sites using information on the dive behaviour of seals derived from telemetry records collected from seals in the same area. If this approach was to be applied at tidal sites it would be important to use telemetry data from such sites as animal's diving behaviour in these unusual habitats likely differs from that in other parts of the animals range.

Line transect methods have typically been used in fairly large bodies of water which aren't subject to strong tidal currents. Wave sites will generally have these characteristics and don't pose any particular issues for the application of standard line transect methods, except of course that these sites are, almost by definition, usually fairly exposed bodies of water.

Tidal sites have a number of characteristics that can make them challenging for boat based surveys. Another issue that particularly applies to wave and tidal sites is that they are relatively small when compared to other sites of industrial interest (Round Three offshore wind, dredging sites etc.). Amalgamations of multiple years of monthly surveys accordingly often generate too few animal sightings, even for common species, to lead to sufficient robust and accurate estimates of abundance or density.

Another consideration stems from their strong and variable tidal currents. Typical survey vessel speeds are in the order of 8 knots. Tidal currents at these sites may often reach 6 knots or more. This makes it essential to consider the frame of reference for the survey and how tidal current can affect survey effort. Is the survey aiming to measure animal densities in the water body, in which case the frame of reference moves with the current, or should the frame of reference be the seabed? If we were attempting to determine the density of jelly fish in water mass then it would be clear that the frame of reference would be the sea and the moving water body. Vessel speed through the water and tracklines which maintained constant headings would be appropriate. If the survey was of barnacles fixed to the sea bed then the frame of reference would be fixed in relation to the land and the vessel should cover fixed track lines at a constant speed over the bottom - even though the speed through the water and the vessel's heading would vary as the current changes.

Which of these two scenarios best applies to marine mammals in tidal areas is not obvious. These animals are clearly not fixed to the bottom; however they certainly do not drift passively with the current. Indeed, often they do seem to maintain stationary against tidal streams. Further, if the surveys are intended to inform collision risk assessments then density is not strictly the appropriate metric for some collision risk models. Rather we would like to know the encounter rate between animals and fixed turbine installations and a sea bed based frame of reference should be more appropriate for this. These considerations have generally been overlooked in surveys of development sites in strong tidal areas but they warrant more detailed attention.

Gordon et al., (2011) did give some consideration to these issues in designing a series of intensive boat based surveys in Welsh Tidal rapids habitats carried out in the summer of 2009. They also had to take into account logistical considerations and how practical it would be for a boat and helmsman to achieve a desired survey in a strong and constantly varying tidal stream. They considered that a bottom based frame of reference was likely to be most appropriate (for the reasons outlined above), thus the vessel was required to follow pre-determined tracklines over the bottom. They also decided that the vessel should maintain a constant speed through the water. There were two reasons for this. The first was that it would be difficult for a boat to continuously adjust its water speed to maintain a constant velocity over the bottom. The second was that changing boat speed, though adjusting engine revs and/or propeller pitch, would alter background noise levels and this would affect detection rates on the passive acoustic system that the vessel was also using and would also probably influence the extent of movements in response to the vessel by the subject animals.Finally, through simulation and geometry they designed survey tracks in relation to current direction which would minimise the effects of current speed on speed over the ground. This could be approximated by always surveying down current and by choosing an angle relative to current direction for the survey tracks for which bottom speed was the same at slack water and during the strongest currents.

In many instances it has been shown that animals may make use of different parts of a tidal rapids habitat at different states of the tide (e.g. Gordon et al., 2011; Pierpoint, 2008). In these areas, densities can vary by orders of magnitude between sites only hundreds of meters apart and can vary to a similar extent between stages of the tide only a few hours apart. It is clear therefore that to understand how animals use a tidal site, surveys need to be repeated many times so that all areas are covered equally at all stages of the tide. Further, if the surveys are intended to inform assessments of collision risk then it is the periods when the tide is running most strongly, the turbines turning most quickly and the risks from collisions are highest from which the survey data are most relevant (Indeed it might be argued that, from the perspective of collision risk the distribution of animals when current is too low to drive the turbines are of no interest). Gordon et al., 2011 staggered survey start times through the tidal cycle to achieve a near even spatial and temporal coverage at intensively surveyed sites off Anglesey. Although this generated an intensive period of effort and provided coverage across the tidal cycle, it was done over a short space of time (~1 month) and therefore couldn't characterise seasonal variation.

Tidal rapids pose several additional problems. The highly energetic environment raises a number of safety concerns for both survey vessels and personnel. This may lead to the use of larger vessels than might be required to carry out a similar survey in non-tidal waters and greater restrictions on the weather conditions considered workable.

Strong tides result in a range of features at the surface (such as standing waves) that can affect the ability to sight marine mammals. Generally too, higher sea states develop for a given wind speed than in non-tidal waters. Ultimately this means lower sighting efficiency and a smaller proportion of survey time when survey conditions allow useful data to be collected.

Tidal sites can often be rather small and often close to shore. This means that ‘edge effects’ at the periphery of a survey block will be a greater concern. For example, when a vessel turns at the edge of a survey block and begins to survey, the first section of trackline (out to the extent of visual detection) will receive less sighting effort because observers will not be able to monitor it for as long as other sections of the same length within the core area. In open water surveys the solution is to discard effort and sightings in these “buffer” areas. This of course “wastes” effort and this “wastage” is proportionally higher when survey blocks are smaller. The issue is more problematic if, as may be the case in tidal areas, the survey block extends close to land. In this case it will not be possible to have a discard buffer and methodological solutions to the uneven coverage in these areas will need to be developed and implemented.

9.2 Boat Based Acoustic Survey

Vessel based acoustic surveys are carried out by towing an array of hydrophones behind the survey vessel and monitoring for acoustic signals using either the human ear or computers, detection algorithms and software. The extent to which Passive Acoustic Monitoring (PAM) techniques can be used to survey marine mammals largely depends on the acoustic behaviour of the species concerned: the rate and predictability with which they vocalise and the range at which these vocalisations can be heard given the received noise levels (which will include noise from the survey vessel, ambient background noise and flow noise around the towed hydrophone). Towed surveys have not proven useful with UK seals. However, dolphins and harbour porpoises produce vocalisations that can be readily detected on towed hydrophone arrays. The methodology has been most fully developed for the harbour porpoise. There is every reason to believe that it could be easily adapted for use with dolphins but it has not yet been routinely used for these species in the UK.

Porpoise vocalisations consist of very high frequency clicks produced in a narrow frequency band centred at around 130 kHz. Detecting such high frequency clicks is possible via the capture and analysis of sound at an appropriate sampling rate (~500 kHz) using relatively affordable hardware and computers. The systems used routinely for survey and mitigation in the UK nowadays are based on the hardware developed for the SCANS II survey in 2005 (Hammond et al., 2013). Data are analysed using modules in PAMGUARD a freely available open source software suite (Gillespie et al., 2008).

Typical porpoise surveys have used simple stereo towed hydrophone arrays consisting of two elements separated by around 30cm and mounted within a long streamlined housing. This is towed behind the vessel on between 100 and 200m of strengthened cable. The long tow cable removes the hydrophones from vessel noise and the worst of wake turbulence and also allows the array to sink lower in the water. Typical tow depths are ~7m. Signals from the hydrophone are brought to an equipment station on the vessel where signals are filtered and amplified before being digitised using a high speed digital acquisition card and analysed and recorded by PAMGUARD software. Typically during surveys raw acoustic data are recorded directly to hard drives to provide a complete record and PAMGUARD also runs a click detector, classifier and time bearing display to provide real time indication that the system is working correctly and there are no problems with electrical or environmental noise. Operating in this mode an acoustic survey system requires little attention and can run unattended with occasional checks from trained PAM operators (employing MMOs also trained as PAM operators would provide the most cost effective way of carrying out surveys – PAM could be checked during line changes or observer watch rotations.)

The difference in the time of arrival of clicks at the two hydrophones can be analysed to provide a bearing to vocalising animals and analysis of changes in these bearings with time (target motion analysis) provides a value for the range of the target animal from the trackline. In this way the information required to calculate a detection function for distance based line transect methods is provided. Although there are limitations of this technique in that the range provided is not necessarily a horizontal range (ie the animal can be under the boat) and therefore the distances provided are not the same as distance from the trackline – this can introduce error in estimates. New modules to streamline target motion analysis with such data were incorporated into PAMGUARD in 2012 with funding from the renewables industry.

There are several advantages in using PAM which justifies its incorporation into towed hydrophones surveys at renewables sites. Acoustic detection rate, unlike visual detection probability, is little affected by sea state meaning that effective surveys can continue in rougher sea conditions - typically up to force 5. Acoustic detection is not affected by fog and poor visibility and can therefore continue at night. It provides data on distributions during the hours of darkness. It is possible that knowing the spatial distribution of animals at night when turbines will be difficult to detect visually might be even more important than knowledge of daytime distributions, and we have no reason to be confident that the two distributions will be the same. PAM can be highly automated and requires little in terms of additional field personnel. This saves costs and in addition, because detections are largely made by software, for porpoises at least, human variability is removed from the process and data should be more consistent. A complete record of the acoustic data can be collected as sound files on hard drives. This allows detection and analysis to be carried out in more controlled conditions by a single analyst ashore and for “difficult” data to be revisited if necessary. It also means that entire datasets can be reanalysed if detection and classification methods improve. Often PAM data can be collected more effectively from smaller, quieter and less expensive survey platforms. However survey effort is often driven by visual survey requirements first meaning many unsuitable (i.e. noisy) vessels can be used for PAM surveys.

There are also some disadvantages of PAM. As we have already noted, PAM does not provide useful data for all species. Additional equipment will be required, although in practice the required towed hydrophone systems have been straightforward to fit to a range of vessels and the cost of rental or purchase are modest compared to vessel charter and personnel costs. When towed hydrophone surveys are conducted in non-tidal waters it is reasonable to assume that the hydrophone follows the track of the vessel. PAMGUARD uses a threading model and knowledge of the length of the tow cable to estimate the hydrophone orientation at any time and the real world bearings to animals is calculated on this basis. This assumption is harder to justify in turbulent tidal areas where the orientation of both the survey vessel and the hydrophone at any time will be affected independently by tidal features such as rips, eddies and whirlpools. However, solutions to this issue are currently under development.

Passive acoustic detections are always affected by background noise. However, underwater noise can be measured from the acoustic record and its effect on detection probability is well understood theoretically and can also be explored experimentally and through simulation. This consideration does however put an emphasis on choosing quiet vessels as survey platforms. As animals are more likely to respond at greater ranges to noisy than quiet boats vessel noise should be a consideration for choosing platforms for visual survey as well. That said, maintaining consistent vessel noise by using the same vessel and a constant water speed is a higher priority for acoustic surveys. During their towed hydrophone surveys in Welsh waters Gordon et al., (2011) found that there were some discrete patches where background noise levels were extremely high. They hypothesised that these were areas where sediment was moving in tidal currents. Sections of trackline with very high noise levels accounted for a very small percentage of the total survey effort and were simply removed from the dataset.

While there could be occasions when surveys might be conducted using towed hydrophones without any visual effort (if porpoises were the only species of interest for example, or surveys were carried out at night) the normal situation would always be to collect visual and acoustic data concurrently. In these cases visual survey teams and acoustic detection systems can be considered as two independent observation platforms. Duplicate detections can be compared between the two platforms allowing g(0) to be calculated for both visual and acoustic detections. Gordon et al., (2011) trialled this approach with data collected at tidal sites in Wales. Leaper and Gordon, (2012) have explored the application of this approach more formally for visual and acoustic datasets collected during long term surveys at large offshore wind farm sites in England and Scotland and estimated g(0) and as a result, estimated absolute abundance estimates, although more work is required to improve the matching of visual and acoustic detections and to correct range estimates to be equivalent to distance from trackline. Although g(0) can be obtained using two independent visual detection teams the additional personnel that this requires and the size of vessel required to provide two truly independent platforms makes this a difficult and costly undertaking.

Vessel based surveys are valuable because they can provide a known and unbiased coverage of an area of interest and data collected on them can be analysed to provide absolute density estimates. The combined visual and acoustic vessel based surveys conducted at Welsh tidal sites by Gordon et al., (2011) provided the first comparable density estimates of which we are aware. They revealed that densities and detection rates for porpoises at these sites were higher than recorded from most other UK sites. Visual observers can collect sightings data for all marine mammal species (including seals) as well as seabirds. The same vessels can also tow hydrophones which can collect useful data on odontocetes. Acoustic methods often complement visual techniques compensating for some of their weaknesses. Tidal rapid habitats present some new challenges for boat surveys. Most of these can be overcome with appropriate planning and methodologies but they certainly make surveys in tidal areas more difficult and costly. Survey vessels and teams of observers are always expensive so the total amount of time that can be spent in any particular area in the course of a boat based survey is often extremely limited. For this reason they are less well suited to revealing temporal changes, including tidal diurnal and seasonal cycles and changes in local densities before and during man-made developments or activities.

9.3 Aerial Line Transect Survey

Aerial line transect surveys, both conventional surveys using human observers (e.g. during SCANS II surveys) and digital surveys using photographic and video equipment to record data (e.g. EMU, 2011), are generally employed for surveying large areas of sea, as large distances can be covered in short periods of time. Therefore at the current scale of wave and tidal development, they are not particularly cost effective or widely used.

Digital methods, because of their increased flight height are becoming preferred for surveys of offshore wind farms, where turbine height may pose safety concerns for conventional survey. However this is less likely to be a consideration for surveys of wave and tidal sites. Other advantages of digital methods are the reduction in observer bias, whereby sightings may be missed by visual observers, and the creation of a permanent, auditable record of the survey. Increased flight height is also considered an advantage for bird survey where the possible disturbing effect of low flying aircraft is a concern, although this is less of a concern for marine mammals. Avoidance of the bias associated with potential attraction or disturbance from boat surveys is another advantage of both types of aerial survey. Furthermore because digital methods result in the recording of sightings across the whole width of the surveyed area, analysis to account for reduced detectability with distance is not required.

There are disadvantages associated with digital aerial methods; for marine mammals the ability to detect animals below the surface in good sighting conditions could lead to an overestimate compared to traditional methods where only surfacing animals are counted. Methods for dealing with ‘availability’ bias i.e. to convert sightings at the surface to total abundance by accounting for animals submerged are currently in development for digital aerial surveys.

Aerial survey methods may best be employed for the future strategic survey of large areas of sea to provided baseline understanding of marine mammal abundance and distribution on a regional scale (e.g. Marine Scotland HiDef surveys: http://www.gov.scot/Resource/0045/00458980.pdf) and may become more appropriate for site characterisation of areas earmarked for large, commercial scale arrays. However, the resulting data will not provide information on the fine temporal or spatial scale needed to inform individual project environmental assessment.

9.4 Static PAM

Bottom mounted static acoustic recording or detection devices have the potential to collect continuous long term datasets at specific locations over extended periods. The species most likely to be detected at Welsh tidal sites (porpoises and dolphins) produce sounds at ultrasonic frequencies and this poses storage problems if raw sound recordings are to be collected (as sampling has to be at very high frequencies). However, the click vocalisations of these species are easy to detect and are distinctive (especially in the case of porpoises) and it has been possible to make autonomous long term detection devices. The most widely used of these has been the POD, designed and produced by Nick Tregenza of Chelonia Ltd. PODs are modestly sized, robust, self-contained units that can collect data continuously for several months at a time.

They have been widely used to monitor for the presence of porpoises and dolphins in a range of different habitats. It is no exaggeration to say that this has been a revolutionary technology in terms of porpoise monitoring. Because static devices collect such a large amount of data in a single location they provide powerful datasets for revealing temporal trends including diurnal, tidal and seasonal patterns (Gordon et al., 2011; Todd et al., 2009), and changes in detection rates in response to anthropogenic disturbance such as that from pile driving (Tougaard et al., 2009) and acoustic deterrent devices (Northridge et al., 2010; Carlstrom et al., 2009). What has proven much more difficult however has been to obtain absolute density data from these devices, they provide no information on range so additional data are required to determine detection functions. Attempts have been made to calibrate the effective detection range of pods for example by comparing pod detections with data on porpoise locations from another independent source such as shore based tracking observations (Kyhn et al., 2012) or locations derived from floating arrays as has been trialled in the SAMBAH surveys for porpoise in the Baltic (Len Thomas and Jamie Macaulay, University of St Andrews Pers Comm.). However, these have been very involved undertakings that produce calibration information that is site specific. They would be particularly difficult to replicate at tidal sites.

Using static devices at wave sites should pose no particular difficulties, other than those listed above. However, there are a number of challenges when attempting static passive acoustic monitoring in tidal currents. Several of these relate to moorings. Obviously, a more substantial mooring is required to prevent the equipment being swept away by the current. Static monitoring devices are usually configured to float up from the bottom, either through their own buoyancy or assisted by additional floatation. As the current increases however any floating device will tend to be “swept down” by the current, changing its orientation and depth, potentially flattening it on to the bottom. These factors are likely to affect detection range and could lead to equipment damage. Any buoys reaching the surface, recovery buoys for example, will be influenced in a similar way. In tidal waters such buoys may only break the surface for a short period around slack tide. Wood et al., (2014) carried out a two year porpoise monitoring project at a strong tidal site in Canada using CPODs mounted in a buoy specially designed to carry packages in strong currents (Open Sea Instruments Sub Buoy). An acoustic release was used for recovery eliminating the need for any surface buoys. However there were a number of issues with this deployment. Given their size, the Sub Buoys create quite a bit of drag such that at high tidal flow, they were pushed down and angled away from the tidal stream. Because of this, the CPOD was no longer facing straight up, which may have affected detection rates. In addition, at high current velocities, the unit seemed to oscillate and move back and forth quite a bit. This may have led to some CPOD failures.

Moorings in tidal currents can collect debris, kelp is a particular problem late in the year when large holdfasts break loose. Creel fishermen setting pots in strong tidal areas consider it important to raise and clean their equipment regularly and this may mean that long term deployments will require regular visits to prevent biofouling reducing the effectiveness of monitoring. While there may seem to be obvious advantages in deploying static devices for extended periods to reduce the cost of servicing trips two factors argue against this in tidal sites. One is the likelihood that deployments will collect debris and be subject to high levels of wear. The other is that data are only recovered when a device is retrieved. If a device is lost the data are lost with it. The severe physical conditions at tidal sites makes equipment loss more likely, so more frequent retrieval and download of data may actually be more cost effective.

Another suite of issues relate to noise. Background noise is relatively high and spatially variable within tidal sites (Carter et al., 2008) and can become louder as the tidal current increases (Gordon et al., 2011). An additional source of noise for moored devices will be flow noise caused by water flowing around the device itself. This can be reduced by placing hydrophones in streamlined housings, as is done with towed arrays, unfortunately existing devices, such as the POD, are rather poorly streamlined, though it should be reasonably straightforward for future designs of detectors or recorders to use streamlined hydrophones, indeed SMRU have been carrying out trials of these with encouraging results (Jamie Macaulay, pers comm). Any noise that overlaps with signals in time and frequency will reduce the ability of either the human ear or a detection algorithm to make detections and will thus reduce effective range. It is essential therefore to record noise in an appropriate manner so that its effects can be accounted for. When using CPODs it is important to manually set the click buffer to maximum in areas where high background noise is expected. Not doing so will result in loss of monitoring time once buffers fill up quickly with background noise (Booth, in press).

Some locations in tidal rapids can be so noisy when the tide is running that any acoustic detection would be very difficult. Gordon et al. (2011) were able to locate discrete very high noise locations during towed hydrophone surveys at Welsh tidal sites and clearly, some sort of pre-survey of this type to identify noisy locations would be prudent before choosing specific deployment sites.

Static devices only provide detection data for the areas surrounding them within their effective range. The acoustic range for porpoises being picked up by pods is in the order of two hundred meters or so. Clearly, an excessively large number of devices would be required to completely cover even a small study site. Thus, what researchers must aim to do is provide sufficient devices to provide a representative sample. How many devices are required to achieve an adequate coverage has never been formally calculated in any survey of which we are aware, but it’s clear that the greater the spatial variability within a survey site the greater the number of sampling locations and devices required. Thus, if the true density was completely uniform a single device would provide effective coverage. Although spatial distributions of small cetacean within tidal sites have only been explored at a very few sites the indications from these studies is that spatial distribution is highly variable. In some cases for example relatively small areas with particular tidal features such as jets and over-falls seem to be highly favoured (Wood et al., 2014; Wilson et al., 2013; Gordon et al., 2011; Pierpoint, 2008). The implications of this are that if static devices were to be used to collect data on spatial distributions in tidal areas a very large number of them would be required.

In spite of these problems several projects have successfully deployed static acoustic monitoring devices in strong tidal currents. Wood et al., (2014) were able to achieve good temporal coverage at seven sites over a two year monitoring period. They did find that their ability to monitor was severely compromised at some sites due to very high background noise levels but were able to show clear seasonal and diurnal trends in acoustic detections. Another project that is particularly relevant in the context of this report is that of Gordon et al., (2011). In part because the work was carried out at Welsh tidal sites, but also because one of the study areas was also intensively covered by towed hydrophone surveys at the same time as the static deployments allowing a comparison of data provided by towed and static methods. In this project six TPODS were deployed (an earlier version of the CPOD) at two tidal sites: the Skerries for an average of three weeks and at Ramsey Sound for approximately six weeks. One device at the Skerries failed to collect any data while two devices at Ramsay sound were lost. Very clear tidal and diurnal cycles were evident in these data, especially the longer datasets from Ramsay sound. Some sites only a kilometre or so apart showed entirely opposite tidal cycles in habitat use. The PODs deployed in the Skerries were in a study area which was covered intensively (several times a day on average) by towed hydrophone surveys. Thus it received a level of survey effort orders of magnitude higher than that normally expended during tidal site characterisation surveys. In spite of this, the towed data did not show temporal patterns as clearly as did the static data dataset.

Ongoing and Possible Future Developments.

It's very likely that the capabilities of passive acoustic monitoring system will continue to improve. Storage capacity will increase, electronic packages will get smaller, more reliable and cheaper and with very little effort streamlined devices suitable for deployment in tidal current could be developed. These features should serve to make it easier to collect improved versions of the presence absence data of the type collected with static devices so far. Another area in which work is ongoing (at SMRU and possibly elsewhere as well) is in development of small bottom mounted arrays. By analysing data from these with the software already developed for drifting arrays (see section 9.2) it will be possible to calculate bearings to sound sources. Further, crossing bearings from two or more such bottom arrays will provide locations of vocalising animals. This capability should lead to a qualitative step change in the type of data that can be collected using bottom mounted devices. For example, several of these arrays mounted at a candidate site for a turbine could provide site specific information on animal flux in different parts of the water column, exactly matching the data required for collision risk modelling. The ability to determine ranges to detections from such arrays would also open up new possibilities for determining densities from such data. Systems like this certainly require additional development but all of the necessary components and the supporting software seem to be in place.

9.5 Drifting PAM detectors

Drifting passive acoustic detectors (often described as "drifters") consist of a self-contained acoustic detection unit (C-POD or other) attached to a drifting drogue and surface float (Wilson et al., 2013).

The drifters are deployed upstream of a tidal area and recovered for redeployment once the current has carried them past the site of interest. Drifters can be deployed from small boats, and multiple drifters can be deployed at the same time to increase spatial coverage. Each drifter contains a positioning unit (either a self-contained GPS or more sophisticated satellite-transmission capability, allowing near-real-time tracking), which enables any detections of vocalising marine mammals (such as harbour porpoise) to be mapped. This approach occupies an intermediate position between static moored detectors (good temporal, poor spatial resolution) and vessel- or plane-based surveys (good spatial, poor temporal resolution). It is also likely to be considerably less expensive than the latter.

Repeated redeployments of drifters across the tidal cycle will result in a distribution map of porpoise echolocation activity, associated with particular flow speeds and/or tidal phases. Because trajectories and speeds of drifters are essentially uncontrolled, survey metrics are likely to be based on time spent within cells of a spatial grid rather than linear travelling distance. As a result, these data are not directly compatible with distance sampling methodologies in terms of forming the basis for an absolute density estimate. However, this approach does allow for high-resolution mapping of relative densities of vocalising porpoises and other marine mammals in response to tidal currents. Flow speeds (obtained as a by-product of drifter positioning data) can also be used to optimise hydrographic flow models. Environmental covariates collected concurrent or in parallel with drifter deployment (flow speed, background noise, tidal phase, bathymetry, distance from shore etc.) can also be used in environmental models to assess their relative significance in (indirectly) driving marine mammal presence and can also identify sites suitable for position static acoustic recorders (Wilson et al., 2013).

9.6 Drifting vertical hydrophone arrays

As noted earlier, to be able to estimate collision risk for a marine mammal it is necessary to know their underwater movements and dive behaviour. It should be appreciated that tidal rapid habitats are a small and very unusual subset of the range of any of the species found there. Thus, there is no basis for assuming that underwater behaviour recorded in other parts of the animal's range will be representative of how they will behave in tidal rapid sites. For some species, such as seals, it is possible to collect information on underwater behaviour using telemetry. This isn't a very promising approach for small cetaceans however as it is extremely difficult to attach tags to these animals. For porpoise for example, most telemetry has been carried out in areas (such as the Baltic and the Bay of Fundy) where animals regularly become entrapped in fixed fishing traps and tags can be attached when porpoises are caught and released from these traps. These tagged animals have rarely, if ever, ventured into tidal rapid sites. A better approach then is to develop a system which can be taken to the particular sites of interest and used to measure underwater behaviour of the animals found there. Dispersed hydrophone arrays with four or more elements are capable of localising animals from time of arrival differences of their vocalisations between each hydrophone in the array and series of these locations could be assembled into tracks providing information on underwater moments and dive behaviour. As part of their trials of methods to collect data relevant for management from marine mammals at Welsh tidal sites Gordon et al., (2011) constructed a simple 4 element vertical array to test the feasibility of this general approach in tidal rapids. This array was deployed from a drifting vessel and heavily weighted so that it maintained a near vertical orientation. Initial results provided data on dive depths and ranges, and were very encouraging (Gordon et al., 2011). More sophisticated methods for analysis were developed by (Macaulay, 2010). The methodology has been further developed over recent years with support from the Scottish Government. The most recent boat based system utilises a 10 element, vertically oriented, three dimensional array. It is able to track the movements of vocalising animals in 3D and has been used to collect new underwater moments and dive data needed for collisions risk models from several different tidal rapid sites in Scottish waters. The software to track porpoises from these data has all been incorporated into PAMGUARD ensuring that it is freely available for others to use and/or develop further.

This system is currently capable of providing the underwater behavioural data required for collision risk modelling (Macaulay et al., 2014; Macaulay et al., 2013). However, two considerations may limit its widespread use. The first is that the current system is quite complicated and requires a technically competent team to set it up and collect data. The second is that a reasonably substantial vessel is needed (most data has been collected from a 20m motor yacht), and this inflates field costs. Through its Knowledge Exchange scheme the Natural Environmental Research Council are funding work to develop a compact, autonomous buoy-based system that could be deployed from a modest inflatable vessel and record data from an eight element-vertically oriented array for several hours at a time. This development is ongoing but initial trials have been successful and provided data to yield 3D underwater tracks from porpoises in tidal rapids. The final output from the project will be a report detailing how to make the system from readily available standard components and a suite of open source easy to use software within PAMGUARD to analyse the data it provides.

Odontocetes emit clicks for echolocation and in many cases particular patterns of clicks (feeding buzzes) are emitted when animals are foraging and closing with prey. Feeding buzzes are thus reasonably good indicator of feeding attempts and by analysing when and where they occur in space and in the water column we can begin to get some insights into how animals forage within the water column and over surveyed areas.

One or more autonomous buoys could be used to collect acoustic data on the distribution of porpoises in tidal areas in the same way as the simple floating detectors described in section 9.5. Because the multichannel buoy provides range and depth data it will be possible to calculate a detection function for localised tracks so that properly quantitative Distance based methods could be applied to obtain density estimates. One could also envisage that such buoys be used in conjunction with simpler and cheaper single channel recorders (such as the Sound Trap or the SM3M) or detectors to provide range data to calibrate these simpler units too. In addition of course such buoys will provide behavioural data. However, the fundamental problems with the use of drifting devices in tidal currents remain: they provide unplanned, uneven and potentially highly biased coverage of the area and they maximise the survey frame of reference problem described in Section 9.1 in that they provide no trackline effort through the water and a rather peculiar coverage with respect to the bottom.

At the moment, the key information that floating arrays are able to provide is quantification of underwater movements and dive behaviour which are particularly important for informing collision risk. They appear to be the only practical method for providing this important information for small cetaceans.

9.7 Land based visual survey

Surveying marine mammals from shore has been used extensively as a means of studying marine mammals in coastal areas (Acevedo 1991; Würsig, Cipriano & Würsig 1991; Best, Sekiguchi & Findlay 1995; Goodson & Sturtivant 1996; Harzen 1998; DeNardo et al., 2001; Mendes et al., 2002; Hastie et al., 2004) but has rarely provided density estimates. Depending on the methods and equipment used, land based observations can be relatively low-cost and are usually entirely non-invasive; further, they have been successfully used to study the majority of species likely to be encountered in Welsh waters, including bottlenose dolphins (Hastie et al., 2003b; Hastie et al., 2004; Bailey & Thompson 2006), harbour porpoises (Koschinski et al., 2003), minke whales (Johnston et al., 2005), and seals (Koschinski et al., 2003; Zamon, 2001).

Land based observations provide information on the spatial and temporal distribution of animals within a defined coastal area of interest. This has been used to provide marine mammal data at a series of levels including occurrence (e.g. Hastie et al., 2003b), relative abundance (e.g. Gailey et al., 2007), and, in a limited number of studies, absolute abundance (e.g. George et al., 2004; Hammond, 1984). Further, given the relatively low cost of land based observers, it is possible to collect relatively large amounts of data that make it ideal for analysing short term temporal patterns in these metrics (e.g. tidal, seasonal, or time of day). With location information for animals within the area of interest, it is also possible to collect information on the spatial distribution of animals (e.g. Hastie et al., 2003b), their movements (Bailey and Thompson, 2006), surface behaviour (Hastie et al., 2004), and habitat use (e.g. Hastie et al., 2003b).

Data are usually collected by individuals or small teams of observers located on an elevated site overlooking an area of interest. The area can be sampled by visually searching for marine mammals at the water surface and noting the time, location, and behaviour of each animal that is sighted. This is often carried out as 'snapshots' at set temporal intervals; in reality these are visual scans of the area which can vary in length depending on the size of the area. Auxiliary data such as the weather conditions, observer ID, and sea state can also be collected to inform any analyses.

To examine the distribution or movements of marine mammals within an area of interest, a method of data collection that produces accurate spatial information is required. A number of studies have made use of land based surveying instruments such as theodolites to position surfacing marine mammals at sea (Würsig & Würsig, 1979; Jefferson, 1987; Kruse, 1991; Best, Sekiguchi & Findlay, 1995; Harzen, 1998). These can provide accurate locations but it can be time-consuming to take readings thus limiting the data to positions of small groups or sub-groups rather than individuals. Real time GIS tracking is possible using theodolites connected to a PC running appropriate software, although it is getting more difficult to use theodolites with such systems as the manufactures are making it harder for third party software to communicate with their devices. More recently, photographic or video techniques have been adapted to accurately locate animals from shore.

These are based on techniques that use information on the height of the observation station and reference spatial locations (e.g. the horizon or a far shore) within the frame to determine the geographic coordinates of surfacing marine mammals (DeNardo et al., 2001; Gordon, 2001; Hastie et al., 2003a). This has a further advantage in that it provides a permanent record of the behaviour of the animals for further analysis. Software has been developed specially for calculating positions of sightings (e.g. PAMGuard video range[4] module and VADAR[5])

Some behavioural data can be collected using land based observations; for most species, this is limited to recording when the animal is at the water surface and this may provide some information of habitat use for some species (Hastie et al., 2004), although for other species which surface only briefly this may not provide much insight. Movement information can often collected using the spatial techniques (e.g. theodolites or video ranging techniques) described above to collect consecutive locations at the surface. Together, such behavioural data can provide useful information when looking to interpret the functional use of the area.

There are a number of limitations associated with surveying marine mammals from land. Depending on the species of interest, the effective search radius can vary from 1-2 km for small species (e.g. harbour porpoise in Koschinski et al., 2003) to ~10km for large cetaceans with conspicuous blows e.g. humpback whales in Noad et al., (2008). This approach is also generally dependent on the presence of a suitable elevated observation site; generally, the higher the elevation the further the distance that can be searched. In reality, the probability of observing an animal decreases as an unknown function with range from the observation site; this will also be affected by environmental conditions such as precipitation, fog or sea surface conditions (e.g. Sea state) or sun glare. This potentially has significant consequences when studying the distribution of animals; if the effects of detection distance are not considered, it is likely that the importance of areas closer to the observer may be over-represented (and vice versa). Therefore, to effectively estimate the density or distribution of animals in an area from a land-based location, it is necessary to model the distribution of sighting probabilities as well as the distribution of sightings (Hammond, 1984). For example, Arranz et al., (2014) used information on sighting distance and sighting angle to estimate both the detection function and the probability density function (pdf) of animal distribution. Further, Hammond (1984) combined sighting data from shore with concurrent aerial survey data to establish the sighting probability with range from shore. More recently, spatially explicit capture-recapture (SECR) methods (Borchers et al., 2014) in combination with double observer approaches are being applied to land based observation data and are likely to provide an effective means of estimating absolute abundance and density of marine mammals from shore.

The challenges involved in the visual detection of some cetacean species make it difficult to obtain information about their distribution and habitat preferences using traditional sampling methods. Although high powered optics can be used, species identification can be challenging for some

[4] http://www.pamguard.org/11_PluginModules.html#visual_methods

[5] http://cyclops-tracker.com/about-cameras.html

species (e.g. some delphinids and grey and harbour seals). Further, estimating the number of individuals (particularly for schooling species) can be extremely challenging from shore, and any information is limited to the water surface (i.e. no information on the depth distribution of animals).

9.8 Telemetry

Assessing the potential effects of offshore developments requires information on the distribution of animals and an understanding of how they use the water column. A basic requirement for developing such an understanding is an accurate description of the movement patterns and behaviour of animals. Approximate density information can be obtained from visual sighting surveys or observation programmes, but these are restricted to observations at the surface, in line of sight and in good weather during daylight hours. Telemetry devices offer high resolution data on haul out activity, at-sea locations and dive behaviour of seals that is otherwise impractical or impossible to attain. A range of telemetry systems have been widely used to study movements, dive behaviour, swimming activity and haul out patterns in both grey and harbour seal. The spatial and temporal resolution of different systems mean that none of them will provide a complete description of the full range of relevant behaviours. Here we describe the main telemetry systems that have been used on seals in UK waters. To date there has been little work on telemetry studies of cetaceans in UK waters.

9.8.1 VHF radio

Historically, VHF radio tags have been used to study at-sea behaviour (Thompson et al., 1989). Such transmitters usually supply only presence information, although coded data radio transmission systems have been developed recently (described under UHF/GPS below). Data collection relies on line of sight between the transmitter and the receiver so is limited in range. However, if the aim is simply to monitor haul out use at specific sites, VHF telemetry may be sufficient.

9.8.2 Argos satellite tags

Data logging transmitters relaying information via the Argos satellite system (Argos 2008) have been widely deployed to track seals (McConnell et al. 1999), large whales (Mate, Mesecar & Lagerquist 2007), and small odontocetes (Sveegaard et al. 2011). The standard commercially available transmitters incorporating a range of sensors including pressure/depth, swim speed and more recently oceanographic quality conductivity and temperature sensors have been used to study the movements, dive behaviour, haul out patterns and local oceanographic environment of free ranging seals. The Argos System provides location estimates based on Doppler shift in signals received by low earth orbiting satellites, as such it has the advantage of providing global coverage. However, the combination of restricted transmission rates and the intermittent surfacing behaviour of marine mammals means that the location data are sparse (perhaps one to six locations per day) and of low precision (with location errors of several km in a high proportion of location fixes (Vincent et al., 2002). Location accuracy can be improved by inclusion of FastLoc GPS sensors. Even with GPS location data the low transmission/reception rates mean that fine scale movement patterns will not be reliably detectable in the track data. They are therefore suitable for general movement and distribution studies but not useful for investigating fine scale movements around marine renewable devices.

9.8.3 GPS/GSM tags

The Global Positioning System (GPS) provides highly accurate location fixes and its use in terrestrial animal studies is common (e.g. Tomkiewicz et al., 2010). However, standard GPS receivers require a significant signal reception period to collect sufficient information to provide a location fix. The short, irregular surfacing intervals combined with problems of interruption due to antenna immersion mean that standard GPS receivers will not work on marine mammals. The recent development of Fastloc GPS sensors that obtain snapshots (< 0.2 s) of GPS satellite transmission when the animal surfaces has effectively overcome this problem. The snapshot data is processed and condensed into a short message containing the pseudo-range data which are time-stamped and stored for subsequent transmission by an appropriate radio transmission or satellite relay system. The pseudo-range data are then post-processed to provide a series of accurate GPS fixes. Fastloc data can be relayed within Argos uplinks (transmissions that are successfully received and relayed by the satellite segment), but this imposes severe restrictions on the amount of GPS fixes that can be relayed. Lonergan et al., (2009) emphasised that accurate track recreation depends not just on fix precision, but also on the number of fixes per day.

Animal-borne GSM mobile (cell) phone devices have recently been developed to allow transmission of large quantities of archived GPS pseudo range data. Since 2004 GPS/GSM phone tags (developed by Sea Mammal Research Unit (SMRU)) have been deployed on seals. GPS/GSM tags collect and store Fastloc GPS and behavioural data (usually depth data) until the seal swims within suitable GSM network coverage. The stored data are sent ashore to a commercial mobile phone. This allows high data rates to be achieved – and at low energy and financial cost. In their usual configuration the tags store data for up to two days before attempting to relay them ashore due to the energy overhead associated with establishing each GPRS session. However data latency could be potentially reduced. SMRU recently deployed prototype GPS/GSM tags transmitting a GPS fix at the start of each surfacing. The received data are archived but can be accessed within 1 hour of transmission.

GPS/GSM tags also record and relay detailed depth profiles within each dive. However these profiles are time based. An attempt to geo-reference them relies upon a linear interpolation between GPS fixes at the start and end of each dive. This introduces uncertainty into the track and thus to the locations at which dive depths occurred. Since grey and harbour seals have dive durations in the order of 3 – 5 minutes, this uncertainty may extend to some hundreds of meters.

9.8.4 Passive acoustic monitoring (PAM) of acoustic transmitters

Passive acoustic monitoring systems have been developed for 3D localisation of actively vocalising marine mammals such as harbour porpoises. Grey and harbour seals do not regularly and predictably vocalise underwater, but they could also be locally tracked with a passive acoustic array if fitted with acoustic pingers (McConnell et al., 2014).

Acoustic pingers are routinely used to track fish (Cooke et al. 2011) and coded data transmitting pingers have been used to track both grey and harbour seals (Thompson & Fedak 1993, Bjorge et al 1995) using steerable directional hydrophones mounted on small vessels. The transmitters have been used to send dive depth, swim speed, stomach temperature and heart rate information, coded in interpulse intervals (Thompson & Fedak 1993, Bjorge et al 1995).

Fine scale 3D movements of seals have been previously studied using coded acoustic pingers and fixed 3D arrays. For example movements and habitat use by Weddell seals diving under ice were studied using a system developed by VEMCO Ltd in the 1990s (Hindell et al. 2002). A number of companies offer bespoke solutions with the potential to track suitably tagged seals. For example, Wright et al. (2007) monitored the locations of tagged harbour seals in an estuary using a fixed array of 15 acoustic receivers.

3D arrays deployed to track small cetaceans can be used to track tagged seals. The tag signals can be optimised for localisation and will have a full 3D transmission pattern and should therefore be easier to localise than small cetaceans with more variable calls and directional transmission patterns.

9.8.5 GPS/UHF Real time tracking

Fastloc GPS data can be relayed by a variety of radio transmission systems. A purpose built UHF data transmission system that combined the capacity to provide near real-time at sea positioning of animals with data storage and periodic transmission to archival base stations on shore, was recently developed by Pathtrack ltd and deployed on harbour seals by SMRU. Animal-borne tags captured GPS data which were processed by the tag using the Fastloc algorithm. UHF telemetry (in the 869.4-869.65MHz frequency band) was then used to broadcast these Fastloc data at the first opportunity when animals were at sea. These data were also stored in the tags so that they could be downloaded by UHF to fixed base stations once animals had hauled out ashore and within range of a station for a pre-determined period. UHF stations were small stand-alone, solar powered devices.

When an animal surfaced at sea the tag took a “snapshot” of GPS data and then immediately broadcast the previously collected GPS information from memory using UHF. This broadcast information, which usually related to the location for the previous surfacing, was thus available to be received in real-time on the tracking vessel. Here, the Fastloc algorithm provide an accurate GPS locations that was presented on a Google Earth screen together with the vessel’s current position.

Tests of the system in good weather conditions suggested that with the direction finding aerials mounted at ~6m, signals could be reliably decoded at ranges of up to 16 km.

The combination of two way communications between the tags and the base stations and multiple methods for retrieving archived data from base stations and tags resulted in a system that was flexible and adaptable and provided accurate, high resolution seal tracks in near real time. Although the real time nature is not necessarily a benefit for pre-installation surveys, the high resolution location data, potentially on every surfacing, is very useful for understanding fine scale usage of tidal energy sites (SMRU, unpublished data).

9.9 Haul out counts

Accurate information on seal populations is required at a range of spatial and temporal scales:

1. At a regional scale:
   - to allow developers to characterise the sensitivity of a location
   - allow an assessment of the importance of the location for the particular species/populations
   - to meet the regulatory requirements of assessing impacts on populations.
2. At fine spatial and temporal scales to provide informaton on local density for each species at the sites of turbines or in the vicinity of construction activities to estimate the population at risk.

Monitoring seal numbers and/or behaviour at haul out sites is a relatively straightforward task. Various methods have been used depending on the spatial scale and temporal resolution at which data are required. It is however often not clear what spatial-temporal extent and resolution is required in order to identify the potential impacts of a development. Predicting the level of impact effects will likely involve an analysis of the connectivity between haul out sites in an area, the short term movement rates between inter-connected sites and the responses of animals to individual and repeated disturbance events.

On a larger spatial scale it is important to determine which haul out sites are important for breeding and moulting but also to appreciate and account for the fact that haul out site usage is a dynamic process. Local population increases or declines can change the way in which haul out sites are used and long-term survey data will be needed to assess the importance of or assign causation to changes in the numbers of seals at particular sites.

A range of monitoring techniques may be employed to provide information at the range of scales required.

9.9.1 Regional/Wide area population estimates:

Grey seal pup production.

Grey seals are widespread around Wales and breed at several sites, mainly in the south-west but also along the west coast and north coasts. Grey seals breed at specific sites where white coated pups are born and remain ashore, other than during short swimming excursion, for several weeks. This enables us to count the pups, which allows the productivity of the population to be assessed annually and, given information on fecundity and survival rates, also allows estimation of the size of the total population.

Aerial surveys to count the number of grey seal pups produced at all major colonies have been carried out in Scotland and eastern England since 1967 (Vaughan, 1971). The techniques involved in conducting aerial surveys to monitor seal populations in the UK are well established (SCOS, 2014). A standardised methodology is used to obtain annual pup production estimates for all main breeding sites. The current method uses vertical aerial photography with a high resolution digital camera on a vibration-damped, motion compensating cradle mounted in a light fixed wing aircraft. Grey seal aerial surveys are carried out during the breeding season when pups are born on land and are easily detectable using aerial photography. This allows for a series of counts of the numbers of pups on the colony at weekly intervals throughout the breeding season. Count data are then used in a model that incorporates time related functions of birth, stage transition and leaving probabilities to estimate the total number of pups born at each colony. These data are then summed to produce regional estimates of overall pup production in any given year. The annual surveys provide data on the status of established breeding colonies and also allow monitoring of potential breeding sites to detect new colonies.

These aerial survey techniques were developed for monitoring pup production on the open beaches and low lying islands that represent to large majority of grey seal breeding sites in the UK. However, in Wales a significant proportion of the grey seal population breed on beaches in caves or at the base of large, often overhanging cliffs where aerial surveys are ineffective. To date, all grey seal pup surveys in Wales have been conducted by land based observers or from small boats or kayaks to allow access to cave beaches (Westcott & Stringell 2003). Regular annual count data are available for open air sites such as the breeding beaches on Ramsey and Bardsey Islands, but the expense, logistical and safety considerations of small boat work in caves mean that only sporadic counts are available for cave sites.

Standard aerial survey methods could be applied to the major colonies in Pembrokeshire and at Bardsey and the Skerries, but in practice it would be difficult and expensive to improve on the data available from the current ad-hoc monitoring programmes in Wales. The most cost effective use of resources would probably be to provide additional support or guarantee continuing support for the current monitoring efforts with the aim of improving coverage and/or frequency of surveys at identified priority sites.

9.9.2 Non breeding counts

Aerial surveys

Outside the breeding season grey seals are widely distributed around the coast, using a large number of haul out sites ranging from tidal sand banks to remote offshore tidal rocks. Site use may be sporadic, varying with season, tidal phase, tidal cycle and weather. Such variations are effectively unpredictable given our current state of knowledge so that monitoring haul out sites will require frequent, wide-area coverage to provide sufficient information to interpret geographical and or temporal changes in numbers.

The vertical aerial photography used for pup surveys is ineffective for such wide area searching surveys. The widely used alternative is a combination of visual observation and oblique aerial photography from either a light fixed-wing aircraft or from a helicopter. Using this technique both harbour and grey seal haul outs are photographed using a digital SLR camera with an image stabilised lens. The images can then be used to count animals and identify seals to species and/or classify group composition.

The method is relatively simple and easy to use for seals on well-defined sites and especially on sand banks (SCOS, 2013). However, seals hauled out on rocky shores are surprisingly difficult to detect by eye and there is an increased risk of missing animals during surveys of rock shore habitats typical of large parts of the grey seal range in Wales.

In Scotland and on the English east coast non breeding surveys are carried out for both grey and harbour seals during the summer when they are widely dispersed. Oblique photography and visual surveys from fixed wing aircraft are mainly restricted to estuarine haul out sites (SCOS, 2013). Stretches of coastline on which it is difficult to detect seals are surveyed by helicopter (operating at an altitude of 150-250 m and at a distance of 300-500 m offshore) using a thermal imaging camera (Barr and Stroud IR18, thermal accuracy 0.1oC) with a dual telescope (x2.5 and x9 magnification). The thermal imager is mounted on a pan-and-tilt-head and operated out of the helicopter window, with an effective range greater than 3 km. Both the thermal image and a 'real' image (from a digital video camcorder) are displayed continuously on a monitor within the helicopter to allow real time assessment of coverage and recorded to allow accurate post survey counts of seals. In addition, high resolution digital photographs are taken of most groups of seals to confirm species identity and counts.

This technique enables rapid, thorough and synoptic surveying of complex coastlines and is used routinely around the north and west coasts of the British Isles (Cronin et al., 2007; SCOS, 2014). The main drawback is cost. Helicopters are expensive relative to fixed wing aircraft and the thermal imagery system is more expensive than standard digital photography.

Boat-based surveys

Boat based surveys are most useful when the sites to be surveyed are confined to a relatively small area and are difficult to get to by land or where no vantage point is offered.

For small areas, surveys by boat can be relatively cheap compared with aerial surveys and may allow frequent or regular repeat counts of an area to be made throughout the year. This can provide estimates of the short term variability in numbers of animals using an area and may allow seasonal changes in haul out numbers to be assessed.

The main operational drawback of boat based surveys is that they are slow and do not produce synoptic counts of large groups of haul out sites or extensive sections of coast. Boat surveys are reliant on good weather conditions, require trained crew and may be logistically difficult with restricted effective ranges. They may also pose a disturbance risk and care should be taken to minimise disturbance when approaching a haul out site or breeding colony.

Land based observations

Land based monitoring of seals outside the breeding season is unlikely to provide accurate/useful synoptic information over a large area. Within even a relatively small section of coast there are likely to be several haul out sites, some of which will be difficult or impossible to observe. However, monitoring specific, easily observable sites can provide useful information on local numbers of seals and how site use varies over time.

Ground counts

Ground counts of seals are carried out regularly at local sites throughout the UK but the methodology employed can differ markedly between sites. Where vantage points are available that allow counts to be made of a representative proportion of a haul out site or breeding colony from distance, relatively little disturbance is caused. Ground count surveys are relatively inexpensive and do not require specialist staff or equipment.

As with boat based surveys, approaching the haul out site or breeding colony pose a risk of disturbance and this should be taken into account when designing survey programmes.

9.9.3 Local area/population monitoring.

With the exception of grey seal pup production surveys, all methods described above for regional population monitoring are potentially applicable to local/small scale population monitoring. However in many cases the costs involved in aerial survey or even boat surveys may outweigh the advantages of synoptic rapid coverage.

Low cost methods such as regular ground based surveys from vantage points may provide sufficiently high resolution data from a sufficiently large proportion of sites within a small defined area. Such surveys can be supplemented by or replaced by continuous monitoring methods.

Remote camera systems

Time-lapse photography or continuous video recording can be used to monitor the numbers of seals using specific sites. Both types of system have been used to remotely monitor pinniped species such as Steller sea lions (*Eumetopias jubatus*) (Kulinchenko et al., 2004; Maniscalco et al., 2006), northern elephant seals (*Mirounga angustirostris*) (California State Parks, 2009), grey seals (Strathspey Surveys, 2006; NOAA, 2011) and harbour seals (Hoover-Miller et al., 2004; Andersen et al., 2012). Typically for pinniped studies, these camera systems are used to count animals and to collect behavioural data (Thompson & Harwood, 1990; Maniscalco et al., 2007) and have recently been used to monitor seals at marine developments (Brasseur et al., 2009; Edren et al., 2010). In addition, if images are of sufficient quality they may provide opportunities to identify individual seals from their pelage patterns (Hiby, 2012).

Remote video cameras are currently used to monitor breeding grey seals on the Isle of May SAC (Strathspey Surveys, 2006) and live video footage is transmitted to the Scottish Seabird Centre 17km away. Stills taken from this video footage have been used to match images of individual grey seals using photo-ID software developed at SMRU (Hiby, 2012).

This technology is applicable to remote sites using wind or solar power. Image data can be transmitted via microwave signal to a receiver several kilometres away. Such systems using both video and time-lapse photography have been used extensively to assess cave use and estimate local populations of endangered Mediterranean monk seal (Hiby & Jeffery, 1987b; Dendrinos et al., 2007).

Remote cameras are limited in that they only cover relatively small areas of shoreline. Depending on the topography of the site to be surveyed and the purpose of the survey, more than one camera may be required. However, once set up they allow continuous coverage of the site using a survey technique that causes no disturbance.

Systems range in complexity from simple time lapse digital still cameras in weather-proof housings through to complex microwave-transmitted, video systems. Cost will vary accordingly. Where the survey effort at a site is intended to provide a long-term monitoring capacity the initial cost of setting up a remote system may be less than the protracted cost of repeat surveys using alternative methods.

Photo identification

Photo-identification is used to individually identify animals where these have unique, long-lasting, naturally occurring patterns. Natural markings data can be used to estimate survival rates, site connectivity and patterns of large scale movement and under certain circumstances, abundance.

Photo-ID is a research tool that has been widely applied to free ranging cetaceans since the 1970s, and free-ranging seals since the 1990s. To be effective it requires that:

- Animals are approachable from boats or come sufficiently close to shore or other vantage points for high quality pictures to be taken
- Animals have sufficient unique markings to be reliably distinguished from one another
- Markings are long-lasting enough to be retained across the sampling period
- The community/population of animals using the study site is manageably small (preferably tens or hundreds of individuals rather than many thousands) especially in relation to the point 5)
- Animals are likely to be re-identified on repeat surveys

While widely applied, the advent of digital photography has massively increased the effectiveness of the technique and also reduced the cost. It can be applied at a variety of levels from anecdotal observations (e.g. “Muddy” the bottlenose dolphin was seen 3 years in a row) all the way through to precise estimates of abundance with confidence intervals, dispersion rates and so on. As with other analytical techniques relying on field data, there are a number of common pitfalls and biases that practitioners should be aware of.

In relation to the general requirements for initial assessment (Section 4.1) the technique is excellent at confirming the presence of the species as reported i.e. it generates pictures of animals that can easily be verified to the species level. So long as the animals have the five characteristics outlined above, repeat surveys (spaced through seasons) can also provide opportunities to investigate levels of residence and estimate the numbers of animals using a site, and investigate connectivity between sites, which is particularly useful in a HRA context. As a method photo-ID from boats can be conducted on special surveys or tacked onto line-transect or other at-sea work so long as the vessel and skills of the crew are suitable. Similarly it can be performed from fixed vantage points depending on range (shore or at-sea structures), or at seal haul out sites, either as a sole activity or opportunistically.

In itself, behaviour is not usually a by-product of photo-ID but estimates of group size, the presence of calves and social associations can result from the data gathered. In terms of connectivity, individuals can be assessed as to whether they link to SACs. Of course this requires that comparable photo-ID is also carried out in the conservation area(s) and the data are actually available for comparison.

For more specific assessments of risk such as collision with tidal stream turbines (Section 4.2), photo-id has more limited application. Of most use will be the opportunity to determine the population size and flux of animals likely to be at risk and therefore inform encounter rate / collision risk modelling. For most modelling attempts to date, animal densities or passage rates have been the most useful metrics and are not typically outputs of photo-ID. However total population size is possible to estimate and can be particularly valuable when considering whether estimated take rates can be deemed acceptable.

At a finer scale and post-deployment, photo-ID can be used to see if animals of a known population occur near or approach a turbine or array and potentially determine actual passage rates of individuals. Of course, this would require careful consideration of the ethics of potentially disturbing animals when they are in the development site. However for sites where there are suitable non-boat vantage points, photo-ID could be feasible. Such investigations could also provide opportunities to determine the fate of individuals once they've had close-approaches whether repeatedly returning, disappearing or conceivably stranding with traumatic injuries. Despite being highly informative, such investigations are inevitably going to be anecdotal simply because of the number of hours required for observations against likely interaction rates. Also such opportunities will be site and population specific and consequently relatively rarely available tools.

As with device interactions, assessment of disturbance and habitat displacement (Section 4.3) would only be a secondary use of photo-ID. It is only particularly valuable if remote vantage points are available as it would be difficult to distinguish the impact of the presence of a boat taking pictures from any animal turbine interactions. However, if remote vantage points are available then it may be possible to determine whether individuals avoid or remain in a site after encountering a device compared with pre-construction equivalents.

While photo-ID is widespread, it is also labour intensive, potentially intrusive and restricted to daylight hours. There are potentially other individual specific monitoring methods. Traces of DNA left in the environment are currently rather too nascent for real application but the use of individual-specific communication calls (e.g. dolphin signature whistles) could have application. Fixed acoustic monitoring equipment capable of capturing waveforms could be deployed to obtain similar information to conventional photo-ID but from fixed locations day and night. At this time the method is theoretical and has the significant caveat of requiring that individuals call using their individual identifiers when they are in the area of concern. The rate at which animals call using their individual identifiers at wave and tidal sites is currently unknown but (like the opportunistic photo-ID) the use of individual calls method may be useful if acoustic data are already being collected.

Given the requirements for photo-id listed above, it is clear that the technique is more suitable for some species than others. Animals with uniform or poorly defined pelage markings are difficult to distinguish. Grey seals, and adult females in particular, bear striking black white and grey asymmetric patterns that are individually identifiable and have been photo-identified across the UK, the Baltic and, more recently, mainland Europe. Hiby et al., (1996) found that more than 90% of adult female grey seals had pelage patterns which allowed individual identification.

It should be noted that determining sex of grey seals from photos alone is not easy. Sub-adult males often have pelage patterns similar to adult females and until they are mature lack the pronounced ‘Roman nose’ and rugose neck and shoulders typical of breeding males. Unless they are photographed on breeding colonies with a nursing pup, or with a clear view of the underside, adult females and sub adult males will be difficult to tell apart. From a recapture point of view, sub adult males are much less likely to be recaptured (annually) as most adult breeding males’ pelage tends towards uniformly dark. Thus the effect they would have in a photo-id database would be to inflate the number of single encounter animals.

Given that there are enough well marked animals to allow photo-id as a method, the photos themselves become important in that they must allow the distinguishing features of each animal to be seen. In practice this means photos have to have a minimum amount of information in them. This usually translates into some estimate of “image quality”, but this summarises different factors – image size and resolution, lighting, angle to camera, portion of body in view, state of pelage. Photographers must be able to take images of animals which show the parts of the animal most useful for identification. Hiby et al., 2012 describe using head, neck, flank and chest pattern extracts and the likely errors that can be expected in making matches through extracted patterns from photos of adult female grey seals.

Bottlenose dolphins are often easily identified by the natural markings on their dorsal fin. These marks include nicks, rake marks, skin lesions and depigmented areas. These marks vary in their longevity and therefore this affects the duration over which individuals can be followed.

Having established that well-marked animals represent a large part of the population, it is necessary to establish a sampling regime appropriate to the nature of the question being asked, taking into account the availability of animals, geographic localities, dispersion and site use within the area of interest.

Photographic equipment

Camera equipment has to capture an image that, for example, allows the head and neck of a seal, (or the dorsal fin of a dolphin) to take up at least one quarter of the frame, depending on the resolution of the camera so that the granularity of the image has to be finer than the scale of the patterns being discriminated. The photographer has to be close enough to the animal to obtain the shot. Often this means having a telephoto lens of 300mm or considerable zoom capabilities. Good quality photographs are possible from a fixed position using a digiscope adaptor for SLR cameras on a spotting scope on a tripod. With such a system, useable photos can be obtained at ranges of around 100m. Beyond this, astronomical telescopes can be used, but even with these, the maximum ranges in good light are around 150-200m.

Taking photos

In all cases, it is imperative to record the associated details of sampling occasion, and each picture taken, to allow best use of the data. Recording date, time, approximate number and age/sex category of animals present, frame numbers, side of animal in frame, and crucially whether the lefts and rights of a particular animal have been obtained, is extremely important. Mistakenly linking left and right sides of an animal causes significant problems later.

Alternative Imaging platforms

Camera traps deployed at haul outs or remotely operated fixed cameras can provide images of seals if they haul out close to the cameras. Passive systems which record automatically either at fixed intervals or by motion-activation are limited by field of view and extraneous triggering. They may also attract unwanted attention from members of the public and incur data protection issues if people are recorded. Live feed systems are expensive to install, maintain and service. They also require line of sight transmission capability.

Unmanned aerial systems (UAS) can obtain images in areas that are otherwise inaccessible. To allow images for photo-id, multirotor UAS platforms with independently controlled cameras are required. They must be quiet enough not to disturb seals, flown within CAA guidelines and therefore operated within 500m of where the seals are. At present, flight times are unlikely to be longer than 20 minutes. They fly best in wind conditions up to a (steady) 18 knots. In many coastal locations where winds are more likely to be gusty and variable, this can be challenging.

Sampling regime

The frequency and spatial range of sampling is set by the questions being addressed. Analysis can be simplified by having a sampling design that produces the highest capture probability practical with resources available. Existing knowledge of grey seal behaviour allows some insight into guiding this. For example, the majority of adults using breeding colonies return there to breed during their reproductive lives, although some individuals may occasionally use “adjacent” colonies (Pomeroy et al., 1994, 2001; Twiss et al.,1994). Out of the breeding season, animals photographed on haul out sites in summer on the UK’s east coast tended to be found at the same location, irrespective of the intervening time interval (Hiby et al., unpubl). Telemetry tracks of grey seals tagged on the east coast suggest that they are central place foragers and that typical foraging trips last for 3 days, followed by 1-2 days ashore (McConnell et al., 1999; Russell et al., 2015).

If multiple locations are sampled (with the proviso that they are surveyed at approximately the same time in relation to the movement capabilities of the animals) information on movements of animals can be obtained.

Determining annual survival rates require several years’ resighting data on groups of animals. Most analyses use a variant of the Cormack-Jolly-Seber method to obtain time-dependent survival estimates. If surveys can be carried out multiple times within years, use of a robust design can give extra information on abundance and an indication of temporary migration.

Abundance estimations, often based on the Petersen estimator, use the dilution of captured and marked animals in a subsequent capture event to estimate population size but are sensitive to violations of model assumptions. These include assumptions that marked and unmarked animals mix together perfectly, that a representative sample of animals is obtained, marking does not affect future recapture and that all animals have an equal chance of recapture at each sampling event. These assumptions are rarely satisfied, with capture hetereogeneity a common problem, although these can be accounted for in models available in programs such as MARK. The population under consideration can be regarded as open or closed, with respect to birth, death, immigration and emigration; closed population models may be used if small time intervals are considered, but in practice open population models are more appropriate where animal movements may be uncertain. Stevick et al., (2003b) use a series of two-sample estimates to account for capture heterogeneity. More recent Bayesian approaches to analysis of resighting data use the same basic models but implement them such that competing explanatory models can be assessed for example using Akaike Information Criterion (AIC) (Smout et al., 2011, 2012).

Error rates:

With small groups of animals (typically no more than around 100) it may be possible to match images by eye. However, in doing so, a filtering process is often involved to use only images above a threshold quality. Rejection of images which are actually matches causes a false rejection rate error - Hiby et al.(2013) found that even poor quality images may provide useful information. False positive matching may occur where different individuals are recorded as the same one. This can be minimized by setting a high threshold for making matches. Unless the whole process is automated, consistency is difficult. The ExtractCompare programme allows semi-automated comparison of pattern extracts with the final match decision resting with the human operator. Even with the assistance of semi-automated methods, the false rejection rate (failure to make matches) for images may seldom be better than 0.14 (Hiby et al., 2013).

Cost of processing image data:

Many photo-id studies underestimate the time and effort required to process the data collected. Extracting patterns for analysis can be time-consuming - a skilled operator may process 80 images per day.

Photo-ID can be either a manual task (where the researcher matches photographs of individuals by eye) or can be computer-aided. Manual matching is viable for a relatively small catalogue of pictures (hundreds). However, computer-aided matching software allows for much larger catalogues (hundreds of thousands) which can be organised into databases. They are also more scientifically robust, with the software allowing efficient comparisons and consistently finding matches that would be much more difficult to find by eye.

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