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The Sensitivity of UK Marine Mammal Populations to Marine Renewables Developments

Authors: John Harwood and Stephanie King

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Lead Scientist: **John Harwood**

Project Manager: **Carol Sparling**

Scientific QA: **Nicola Quick**

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SUMMARY

This report should be read in conjunction with Harwood et al. (2014), which describes an interim protocol for implementing the Population Consequences of Disturbance approach for quantifying and assessing the effects of UK offshore renewable energy developments on marine mammal populations. Together, these reports should enable developers to be confident that the information on marine mammals that they provide in their ESs and HRAs is relevant and appropriate, and to help regulators and their advisors ensure that the assessments of the potential population consequences of these developments are consistent and comprehensive.

We review the potential threats to five marine mammal species (grey seals, harbour seals, minke whales, bottlenose dolphins and harbour porpoises) that may be associated with offshore renewable energy developments. We conclude that the most important risk is of distrubance from the noise associated with the construction of such developments, because of the large numbers of animals that may be affected and the fact that the resulting changes in behaviour may affect individual vital rates (the probabilities of survival and giving birth in a particular year for individual animals).

We develop a stochastic framework for modelling the population dynamics of these five species that can be used to assess the potential consequences of any changes in vital rates that may occur as a result of disturbance. We also suggest an appropriate set of demographic rates (the mean values of individual vital rates, averaged across all members of a population) for each of the different Management Units of these five species that have been identified by Anon. (2014).

We used this model framework to determine the sensitivity of the growth rate of these populations to changes in demographic rates that may occur as a result of disturbance. We confirmed the results of previous analyses that have suggested that the growth rate of marine mammal populations is most sensitive to changes in adult survival. However, when we reviewed the likely sensitivity of individual vital rates to the effects of disturbance, we concluded that the survival of dependent young (calves or pups) and fertility (the probability that an individual female will give birth in a particular year) were most likely to be affected for all species except harbour porpoise. Harbour porpoises carry much smaller energy reserves than the other four species and, as a result, adult and juvenile survival could also be affected by disturbance.

Finally, we consider how the approach described in this report and in Harwood et al. (2014) could be extended to other marine mammal species in UK waters. We conclude that the only species which could be assessed in the same way using currently available information is the killer whale.

1. Introduction

During 2013 a number of offshore renewable energy developments, which we define (following the terminology used within the UK Department of Energy and Climate Change) as wave and tidal stream devices, and offshore wind farms, entered the UK consenting process. This process includes the submission of environmental statements (ESs) and Habitat Regulations Appraisals (HRAs), and may require specific Environmental Impact Assessments (EIAs). Marine mammals are an important component of the marine fauna that is likely to be affected by these projects but, at present, there is no common framework that will allow developers, regulators and their scientific advisors to assess the potential impacts of these effects on marine mammal populations.

In 2005, the U.S. National Academy of Sciences convened a committee that examined how marine mammals respond to anthropogenic sound. The committee's report (National Research Council, 2005) provided a valuable conceptual framework, which it suggested could be used to structure future studies of the potential population-level effects of changes in behaviour of marine mammals. The committee named this model Population Consequences of Acoustic Disturbance (PCAD). In 2009, the U.S. Office of Naval Research (ONR) set up a working group to explore how the conceptual model developed by the NRC committee might be translated into a formal mathematical structure. A 2012 workshop co-funded by the Natural Environment Research Council's Marine Renewable Energy Knowledge Exchange programme, the Joint Nature Conservation Committee and the Countryside Council for Wales used the work of that group as a foundation for an interim framework that could be used to implement the PCAD approach for marine mammals in UK waters (Lusseau et al, 2012).

This research described in this report was also made possible by a grant from NERC's Marine Renewable Energy Knowledge Exchange programme. It represents one of the phases in the development of the PCAD approach, and should be read in conjunction with Harwood et al. (2014), which describes how an interim version of the PCAD approach developed at the workshop co-funded by NERC can be implemented. Together, these reports should enable developers to be confident that the information on marine mammals that they provide in their ESs and HRAs is relevant and appropriate, and to help regulators and their advisors ensure that

the assessments of the potential population consequences of these developments are consistent and comprehensive. This report's specific aims are:

- to review the potential risks to population units of grey seals, harbour seals, minke whales, bottlenose dolphins, and harbour porpoise from offshore renewables developments in UK waters, along the lines of the analyses carried out by Furness & Wade (2012), Desholm (2009) and Garthe & Hüppop (2004) for UK seabirds;
- to identify which individual vital rates (the probabilities of survival and giving birth in a given year for individual animals) are most likely to be affected by the construction and operation of marine renewables developments;
- to provide a generalised stochastic population model for the five priority species that can be used within the PCAD framework to investigate how changes in vital rates may affect the size and conservation status of the marine mammal populations within particular Management Units (MUs); and
- to describe how this approach can be extended to Risso's dolphins, white-beaked dolphins, common dolphins, white-sided dolphins and killer whales.

In the next section, we first provide a brief overview of the potential risks associated with offshore renewable energy developments for individual marine mammals. We then focus on one of the major risks: changes in behaviour as a consequence of acoustic disturbance that may affect individual vital rates (see Glossary). We then discuss how MUs for the different marine mammal species that occur in UK waters may be identified, and describe the PCAD framework, and its successor PCoD (Populations Consequences of Disturbance), in more detail. Next, we explain why the approach developed by Furness & Wade (2012), Desholm (2009) and Garthe & Hüppop (2004) for assessing the sensitivity of UK seabirds to the potential effects of offfshore renewable energy developments cannot readily be applied to marine mammals. We then develop a population modelling framework that can be applied to any marine mammal species in UK waters and use it to show how changes in population-level demographic rates (see Glossary for the distinction between demographic rates and vital rates) as a result of disturbance may affect the growth rates of marine mammal populations. Finally, we describe how that population modelling framework can be expanded to take account of environmental variation and uncertainty (see Glossary).

In sections 3-7, we review the available information on the likely response of each priority species to acoustic disturbance, and identify which individual vital rates are most likely to be affected by the construction and operation of offshore renewable energy developments for the five priority species. We also suggest appropriate values for the demographic rates (average survival and fecundity, age at independence, age at first reproduction) that can be used to model the dynamics of the populations within different MUs.

We consider how the PCoD approach could be extended to five other species: Risso's dolphins, white-beaked dolphins, common dolphins, white-sided dolphins and killer whales.

Finally, we provide a Glossary of the technical terms used in this report.

2. Basic Concepts

2.1 The potential impacts of offshore renewable energy developments on individual vital rates

The installation and operation of offshore renewable energy devices poses a number of threats to marine mammals that range from direct impacts on survival, as a result of collision or entanglement in wave and tidal energy devices, to apparently minor changes in behaviour in response to noise generated by their operation. As noted in section 1, we will not review these potential impacts here, because comprehensive reviews are available elsewhere. For example, ICES (2012) provides a detailed review of the effects of wave energy devices on marine mammals, with recommendations on research needs, monitoring and mitigation schemes. In this report, our concern is with the way such effects may alter vital rates, which we define as the probabilities that individual animals will survive, reproduce, breed and grow in a particular year (i.e. the components of individual fitness - see Glossary). We also describe how the uncertainties that are inevitably associated with estimates of effects on vital rates can be incorporated into predictions of future changes in population size.

Collision with, or entanglement in, wave and tidal energy devices may result in almost certain death. It is also possible, although unlikely given the source levels involved, that sound pressure waves associated with the installation of marine renewables devices may result in the death of animals that are very close to the device. The population consequences of these deaths can be modelled in the same way as deliberate killing or by-catch in fisheries. Again, there are well established ways of modelling the potential population consequences of this additional

mortality, such as the Potential Biological Removals (PBR) approach developed by Wade (1998) and modified by Boyd et al. (2010) for application to UK seal populations, and we will not discuss these further here. However, such interactions may also result in injuries that, although not immediately life-threatening, may affect an animal's vital rates.

A permanent shift in hearing threshold (PTS) at one or more frequencies as a result of exposure to underwater noise is an example of an injury that may affect an individual's vital rates. Southall et al. (2007) and Finneran & Jenkins (2012) provide guidance on the Sound Exposure Levels (SELs) or Sound Pressure Levels (SPLs) that may result in PTS for different marine mammal hearing groups. However, they recognise that these guidance levels are not based on direct observations but are extrapolations from experiments (usually only involving a small number of individuals from a handful of species) in which a temporary shift in hearing threshold (TTS) has been observed. Experiencing PTS may have a direct effect on the probability of survival of an individual if it reduces its ability to detect predators, and an indirect effect on survival and reproduction if it reduces the animal's ability to locate and capture prey. It may also have an indirect effect on the fertility (the probability of giving birth and raising an offspring successfully) because many marine mammals use sound to locate potential mates and to remain in contact with their dependent offspring.

The noise associated with the installation and operation of offshore renewable energy devices could also induce other physiological effects, such as TTS or changes in levels of stress-related hormones. For example, Rolland et al. (2012) reported a decline in levels of stress-related hormone metabolites in faeces collected from North Atlantic right whales in the Bay of Fundy, Canada, immediately after 11 September 2001, when average underwater noise levels decreased by 6dB in association with a dramatic reduction in boat traffic. This implies that high levels of underwater noise may result in elevated stress levels in some baleen whale species. High stress levels have been shown to decrease survival probabilities in humans (e.g. Russ et al., 2012). TTS may have short-term effects on the probability of survival and reproduction similar to those described above for PTS. However, the effects of TTS are, by definition, temporary and we therefore consider the onset of TTS as an extreme form of behavioural disturbance (see below and section 3.3).

Exposure to noise associated with the construction and operation of offshore renewable energy devices may also result in behavioural disturbance and acoustic masking (Richardson et al. 1995; Southall et al., 2007). Behavioural disturbance may have a direct effect on the survival

probability of an individual female's offspring if it results in her being separated from her calf. It may have an indirect effect on the probabilities of survival and reproduction if it results in a decrease in energy intake or an increase in energy expenditure. For example, observed declines in bottlenose dolphin populations subject to disturbance from tourist boats (see section 6.3) appear to be linked to a decrease in the amount of time spent resting by mothers with calves, because this is an activity with relative low energy demands (Lusseau, 2003; 2004). Behavioural disturbance may also result in animals being displaced temporarily from preferred, and potentially critical, habitats. This could have a direct effect on survival if individuals are displaced into an area where the risk of predation is elevated, and an indirect effect on survival and reproduction if they are displaced into an area where prey is less abundant or more difficult to capture, thus reducing their energy intake.

Many marine mammal species rely on specific sounds to locate their prey, avoid predators and communicate with other members of their social group. As a result, high levels of ambient noise could have direct and indirect effects on an individual's survival and reproduction because they may impair its ability to detect these sounds, a phenomenon known as masking. For example, Hatch et al. (2012) concluded that calling northern right whales in the Stellwagen Bank National Marine Sanctuary may have lost 62-67% of their calling space as a result of ambient noise and noise from ships. The effects of masking on vital rates are therefore similar to those of TTS, although they may affect many more individuals, because they are likely to extend over a larger area.

2.2 Behavioural Response to Acoustic Disturbance

The construction and operation of the offshore renewable energy developments that are planned for UK waters over the next decade will involve a number of different sound sources. The installation of wind or tidal turbines commonly involves pile driving, which will introduce high source-level pulsed underwater sounds into the environment. However, not all construction methods involve pile driving. Operational noise associated with these developments usually involves lower source levels than installation, but it is likely to result in a persistent rise in ambient noise levels in the development area.

The way in which individuals respond to these sounds is likely to be affected both by the context of the exposure and the individuals' experience, motivation and conditioning (Götz & Janik, 2010).

Although it is difficult to account for context, we can begin to quantify animal response to noise by utilising Southall et al.'s (2007) ordinal severity scale of behavioural response. This scale ranks behavioural responses according to their likely effect on vital rates.

Anon. (2010) advise that a disturbance effect under the European Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (usually referred to as the Habitats Directive) will occur if animals incur sustained or chronic disruption of behaviour that is likely to impair an individual's ability to survive, breed, reproduce, or raise young, or that is likely to result in that individual being displaced from an area for a longer period than would occur during normal behaviour. These behavioural disruptions coincide with behavioural responses with a score of at least 5 on Southall et al.'s (2007) severity scale. These include: extensive changes in speed, direction and/or dive profile; shifts in group distribution; the aggregation or separation of groups of animals; changes in vocal behaviour; active avoidance of the sound source; separation of females and dependent offspring; visible startle response; cessation of reproductive behaviour; and aggressive behaviour.

In sections 3-7 of this report, we provide a brief review of the current evidence of behavioural effects for each of the priority species, consider which vital rates may be most sensitive to the effects of disturbance, and provide some guidance on what demographic rates should be used to model the dynamics of the populations in the MUs recommended by Anon.(2014). Wherever possible, we report the received sound pressure levels (RLs), associated with behavioural responses and, wherever possible, provide a score on Southall et al.'s (2007) ordinal severity scale.

2.3 Sources of information on UK Management Units

Palsbøll et al. (2007) defined an MU as a "demographically independent population(s) whose population dynamics (e.g. population growth rate) depends largely on local birth and death rates rather than on immigration." They added " The identification of MUs is central to the short-term management and conservation of natural populations and is typically used to delineate entities for monitoring ... and regulating the effects of human activity upon the abundance of populations and species."

There have been a number of recent reviews of potential MUs for marine mammals in UK waters. ICES (2012) provided advice on MUs for "the most commonly encountered (cetacean)

species in the eastern North Atlantic", and Evans (2012) recommended a number of MUs for marine mammals in Welsh waters. ICES (2012) adopted Evans & Teilmann's (2009) definition of an MU as "a group of individuals for which there are different lines of complementary evidence suggesting reduced exchange (migration/dispersal) rates." This is rather broader than Palsbøll et al.'s (2007) definition. In addition, the Sea Mammal Research Unit has identified MUs for harbour seals and grey seals in Scottish waters in order to calculate the maximum number of seals that may be shot under licence (Boyd et al., 2010), and OSPAR has identified "population sub-units" for both seal species in the North Sea as part of their seal Ecological Quality Objectives (Heslenfeld & Enserink, 2008).

More recently, an Inter-Agency Marine Mammal Working Group of the Statutory Nature Conservation Bodies (SNCBs) has identified MUs (which the group defines as "the animals of a particular species in a geographical area to which management of human activities is applied") for a number of marine mammal species in UK waters (Anon., 2014).

In order to assess the potential effects of marine renewables developments on the marine mammal population within an individual MU, we need information on the size and history of that population. For grey and harbour seals, this information is provided by the annual reports of the Special Committee on Seals. For cetaceans the situation is more complicated. There have been three comprehensive surveys of cetaceans in the eastern North Atlantic using randomly placed line transect designs (SCANS, SCANS II and CODA). These have provided estimates of the density and abundance of the three cetacean species considered in this report for a number of survey blocks. The boundaries of these blocks were determined by practical rather than biological considerations and they do not necessarily correspond to the boundaries of proposed MUs. However, the Joint Nature Conservation Committee is leading the Joint Cetacean Protocol (JCP), a collaborative international project that will deliver information on the distribution, abundance and population trends of cetacean species in UK waters and the wider northeast Atlantic.

2.4 The PCAD and PCoD frameworks

As noted in section 1, a panel convened by the National Research Council of the United States National Academy of Sciences published a report on 'Marine Mammal Populations and Ocean Noise: Determining When Noise Causes Biologically Significant Effects' in 2005. The panel

developed what they referred to as a "conceptual model" that outlines the way marine mammals respond to anthropogenic sound, and how the population level consequences of these responses could be inferred on the basis of observed changes in behaviour. They called this model Population Consequences of Acoustic Disturbance (PCAD; Figure 2.1).

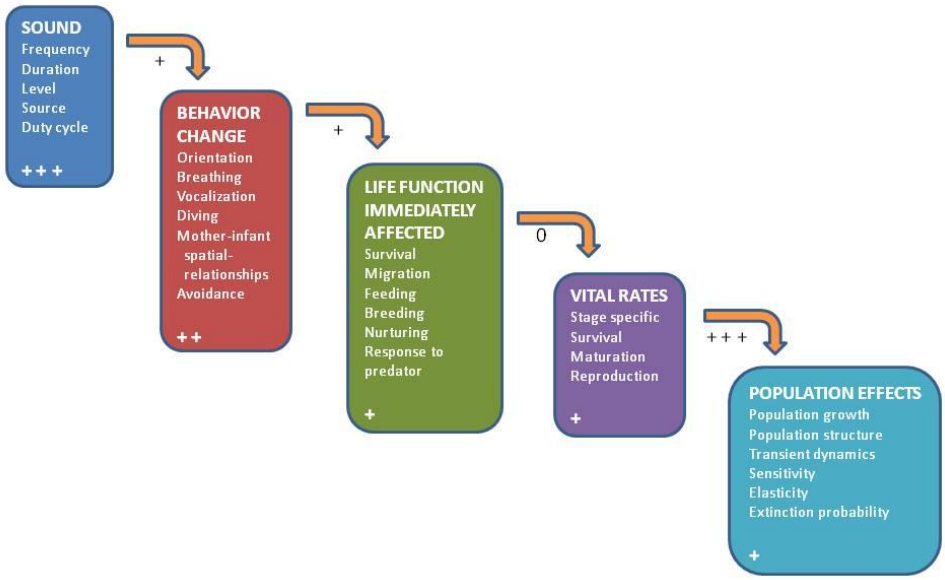

Figure 2.1 *The Population Consequences of Acoustic Disturbance (PCAD) framework developed by the National Research Council's panel on the biologically significant effects of noise. After Fig. 3.1 in National Research Council (2005). The number of + signs indicates the panel's evaluation of the level of scientific knowledge about the links between boxes,* 0 *indicates no knowledge.*

In 2009 the US Office of Naval Research set up a working group to transform this framework into a formal mathematical structure and to consider how that structure could be parameterised using data from a number of case studies. The ONR working group extended the PCAD framework to consider forms of disturbance other than noise, and to address the impact of disturbance on physiology as well as behaviour. The current version of that framework, which is based on case studies of elephant seals, coastal bottlenose dolphins, northern right whales and beaked whales, is now known as PCoD (Population Consequences of Disturbance). It is shown in Figure 2.2, and described in more detail in New et al. (2014).

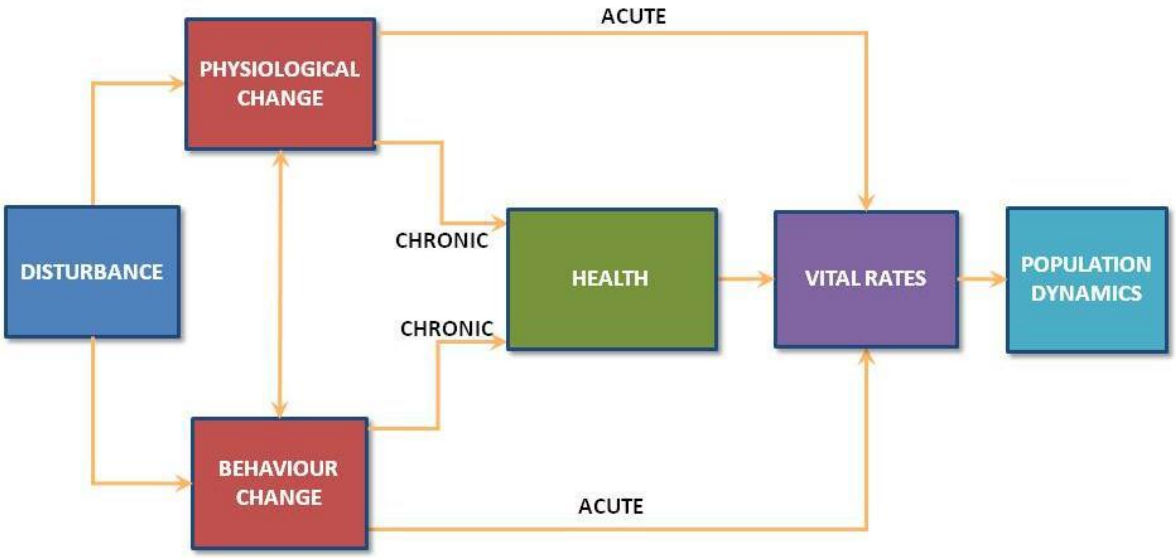

Figure 2.2 *The PCoD framework for modelling the population consequences of disturbance developed by the ONR working group on PCAD (modified from Fig.4 of New et al., 2014). The term "Health" is used to describe all aspects of the internal state of an individual that might affect its fitness. These could include, for example, the extent of its lipid reserves and its resistance to disease. "Vital rates" refers to all the components of individual fitness (probability of survival and producing offspring, growth rate, and offspring survival).*

The new framework shows how disturbance may impact both the behaviour and physiology of an individual, and how changes in these characteristics may affect that individual's vital rates either directly (an acute effect) or indirectly via its health (a chronic effect).

For example, exposure to high levels of sound may result in hearing damage (a physiological effect) as a result of PTS. This could have an acute effect on survival, because the affected individual might be less able to detect predators. It could also have a chronic effect on reproduction via the individual's health, because it might be less able to locate and capture prey. Similarly, behavioural changes in response to disturbance could have an acute effect on survival if they result in a calf being separated from its mother. They could have a chronic effect on reproduction, via body condition, if they result in the disturbed animal spending less time feeding or in activities that conserve energy, like resting.

As noted in section 2.1, one of the potential consequences of a behavioural response to disturbance is that animals may be displaced into areas where predation risk is high. There is considerable

evidence that the behaviour of marine mammals is shaped by the need to avoid predation. For example, bottlenose dolphins in Shark Bay, Australia avoid areas where there is a high risk of shark attack (Heithaus & Dill, 2002), and Alaskan harbour seals appear to avoid spending time in parts of the water column where they are likely to be vulnerable to attacks from sleeper sharks (Frid et al., 2007). As a result of these behaviours, bottlenose dolphins in New Zealand and Australia (see section 6.2) appear to be reluctant to vacate areas where disturbance is high, for neighbouring areas where there is a high risk of shark predation, even though this has a potentially negative effect on calf survival and inter-calf interval (Bejder et al., 2006). For these reasons, the ONR working group concluded that marine mammals are unlikely to be displaced into regions of high predation risk by disturbance, and that the main effects of disturbance on vital rates are likely to be through changes in individual health as a result of changes in behavioural time budgets.

New et al. (2014), and Schick et al. (2013) used case studies of elephant seals, and New et al. (2013) used a case study of bottlenose dolphins to show how changes in behaviour in response to disturbance could affect the energy reserves of adult females, and to estimate the implications of these changes for the probability of giving birth and offspring survival. The consequences of these changes for population dynamics could then be inferred from the number of animals that might be affected by disturbance and the size of the population of which they are a part. Nabe-Nielsen et al. (2014) used a similar approach to assess the potential impacts of wind farm operation on harbour porpoises in Inner Danish Waters.

Unfortunately, the kinds of information required to estimate the parameters of the full PCoD model used in these case studies are not available for the marine mammal populations considered in this report. We have therefore used a simplified version of the PCoD model, shown in Figure 2.3, which was developed at the workshop on 'Assessing the Risks to Marine Mammal Populations from Renewable Energy Devices' referred to earlier.

One way to obtain estimates of the parameters of the relationship between behavioural and physiological changes and individual vital rates illustrated in this model is to use an expert elicitation process (Runge et al., 2011; Martin et al., 2012) combined with the 4-step interval approach developed by Speirs-Bridge et al. (2010). Donovan et al. (in press) and Harwood et al. (2014) describe how this approach has been used to estimate the parameters of the PCoD model shown in Fig. 2.3 for five marine mammal species in UK waters.

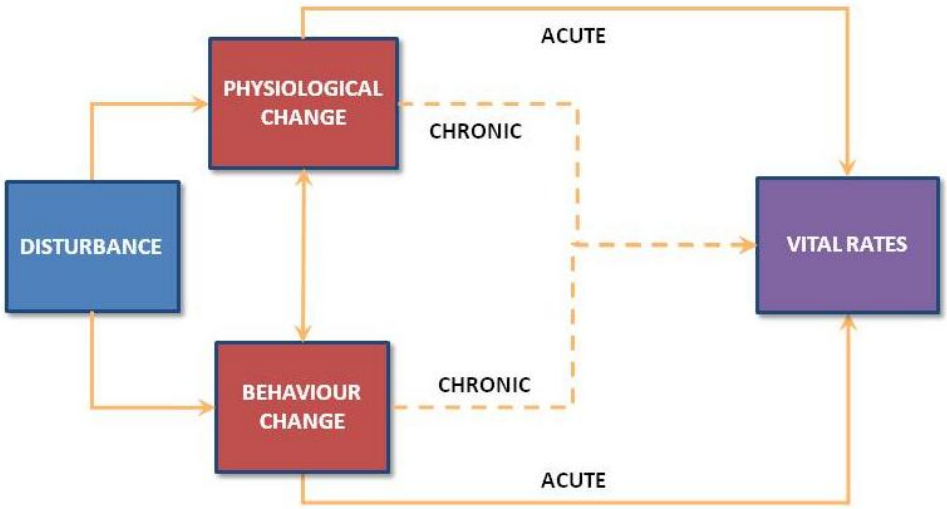

Figure 2.3 *A simplified version of the PCoD framework shown in Figure 2.2 that can be used as interim approach when empirical data on the effects of physiological and behavioural change in individual health is unavailable. The transfer functions that determine the chronic effects of physiological change and behavioural change on vital rates are represented with dotted lines to indicate that the form of these functions may be determined using the results of an expert elicitation process. The term "vital rates" refers to all the components of individual fitness (probabilities of survival and producing offspring, growth rate, and offspring survival).*

2.5 Sensitivity analyses for seabirds

There has also been considerable interest in the potential impact of marine wind turbines on seabird populations, and Garthe & Hüppop (2004) developed an index of the sensitivity of different seabird species to wind farms in the North Sea. This identified characteristics of their flight (manoeuvrability, altitude, percent time spent flying, nocturnal activity, sensitivity to disturbance) which were anticipated to make them more vulnerable to collision with turbine blades and scored these on a 5 point scale. They then scored the same species in terms of their habitat flexibility, adult survival rate and the conservation status of their North Sea populations. Desholm (2009) used elasticity analysis (see section 2.6) to estimate the sensitivity of populations of 38 North Sea migrant species to collisions with wind turbines. They concluded that the best indicators of each species' sensitivity were its relative abundance, and the elasticity of its population growth rate to changes in adult survival. Most recently, Furness &

Wade (2012) combined the approaches of Garthe & Hüppop (2004) and Desholm (2009) to rank the vulnerability of 40 Scottish seabird species to the effects of offshore wind turbines.

Collision with marine wind turbines is not an issue for marine mammals, but collision with, or entanglement in, wave and tidal devices is. Unfortunately, there is currently no empirical basis for assessing the vulnerability of individuals of different marine mammal species to collision or entanglement, although ICES (2012) provides some insights for wave devices, and SCOS (2012) for tidal devices. We were not, therefore, able to apply the sensitivity indices developed for seabirds to marine mammals. However, in subsequent sections we build on the demographic aspects of these indices to assess the sensitivity of marine mammal populations to the potential impacts of the construction and operation of marine renewables developments.

2.6 Sensitivity and elasticity analysis

Sensitivity and elasticity analyses examine the way in which the intrinsic growth rate of a population changes in response to variations in demographic rates (the average value of the individual vital rates within a population). These analyses are usually performed by constructing a projection model of the dynamics of the population of interest using a matrix formulation, and examining the response of the principal eigenvalue of this matrix (see, for example, Chapter 6 of Caswell (1989)), which provides an estimate of the population growth rate. Sensitivity analysis involves a simple comparison of the effect of the same small change in each demographic rate on the principal eigenvalue. In an elasticity analysis, the changes in the demographic rates and the corresponding changes in the principal eigenvalue are expressed as proportions.

We have developed such a matrix model for the five marine mammal species discussed in this report. The same model structure can be used for any of the other marine mammal species that occur in UK waters. The model divides each population into 9 age classes and a terminal stage class that includes all animals more than 9 years old. The classes are:

- pups or calves (depending on the species being modelled),
- one-year olds,
- two-year olds,
- etc. up to age 8,
- all animals aged 9 years and above, combined into a single stage class.

We amalgamated these classes into three age categories: pups/calves (animals that are dependent on their mothers for at least part of the year), juveniles (animals that are no longer dependent on their mothers but younger than the age at first breeding), and adults. We assumed that mean survival rates did not vary with age within these categories, and that the fecundity rate (the average probability that an individual which is alive at the start of the breeding season will give birth to a pup or calf) was the same for all individuals above the age at first breeding. Age at independence was defined as the year in which pups or calves become independent of their mothers. Age at first breeding was defined as the age at which an individual was likely to give birth to a pup or calf for the first time. These definitions impose an upper limit of 9 years on the age at first breeding.

This is a standard stage-classified matrix model (see Chapter 4 of Caswell, 1989) for a birth pulse population in which breeding is assumed to occur on the first day of each year. The elements of the age/stage vector for each year represent the number of female animals in each age/stage class at the end of the breeding season in that year, and all births are assumed to occur over a short time interval. The sum of the elements in the age/stage vector therefore represents the maximum size of the female segment population in that year. Maximum **total** population size is then estimated by dividing this sum by an estimate of the proportion of females in the population.

Equation 1 shows the matrix ($\boldsymbol{L}$) that would be constructed using these definitions for a marine mammal species in which the age at independence is 2 years, and the age at first breeding is 9 years. Sd is the survival rate for calves or pups, Sj is the survival rate for juveniles, Sa is the survival rate for adults, and F is the fecundity rate (the average probability that a mature female will give birth to an offspring at the start of the breeding season).

$$\boldsymbol{L} = \begin{bmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & Sa.F \\ Sd & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & Sd & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & Sj & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & Sj & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & Sj & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & Sj & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & Sj & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & Sj & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & Sj & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & Sj & Sa \end{bmatrix} \quad \text{Equation 1.}$$

The elasticity of the population growth rate to changes in individual demographic rates for long-lived species, such as marine mammals, with low fecundity rates is well known: changes in adult survival have the greatest effect, and changes in fecundity and pup/calf survival have the least effect. Young & Keith (2011) document a recent analysis for large cetaceans, and Fig 2.4 shows some typical results for harbour porpoise. However, it is rarely possible to estimate the change in a demographic rate that may occur as a result of an offshore renewable energy development. In practice, the Environmental Statement for a particular development usually provides an estimate of the number of animals that may experience injury and the number that may experience some form of disturbance as the result of one day of operation or construction work. Harwood et al. (2014) describe how these estimates can be used to predict the number of pups/calves, juveniles and adults that may die or fail to breed as the result of all operations within a year, subject to a number of strong assumptions. The effects of these predictions on demographic rates can then be calculated by dividing these numbers by the size of the relevant category of the population (pups/calves, juveniles, or adults).

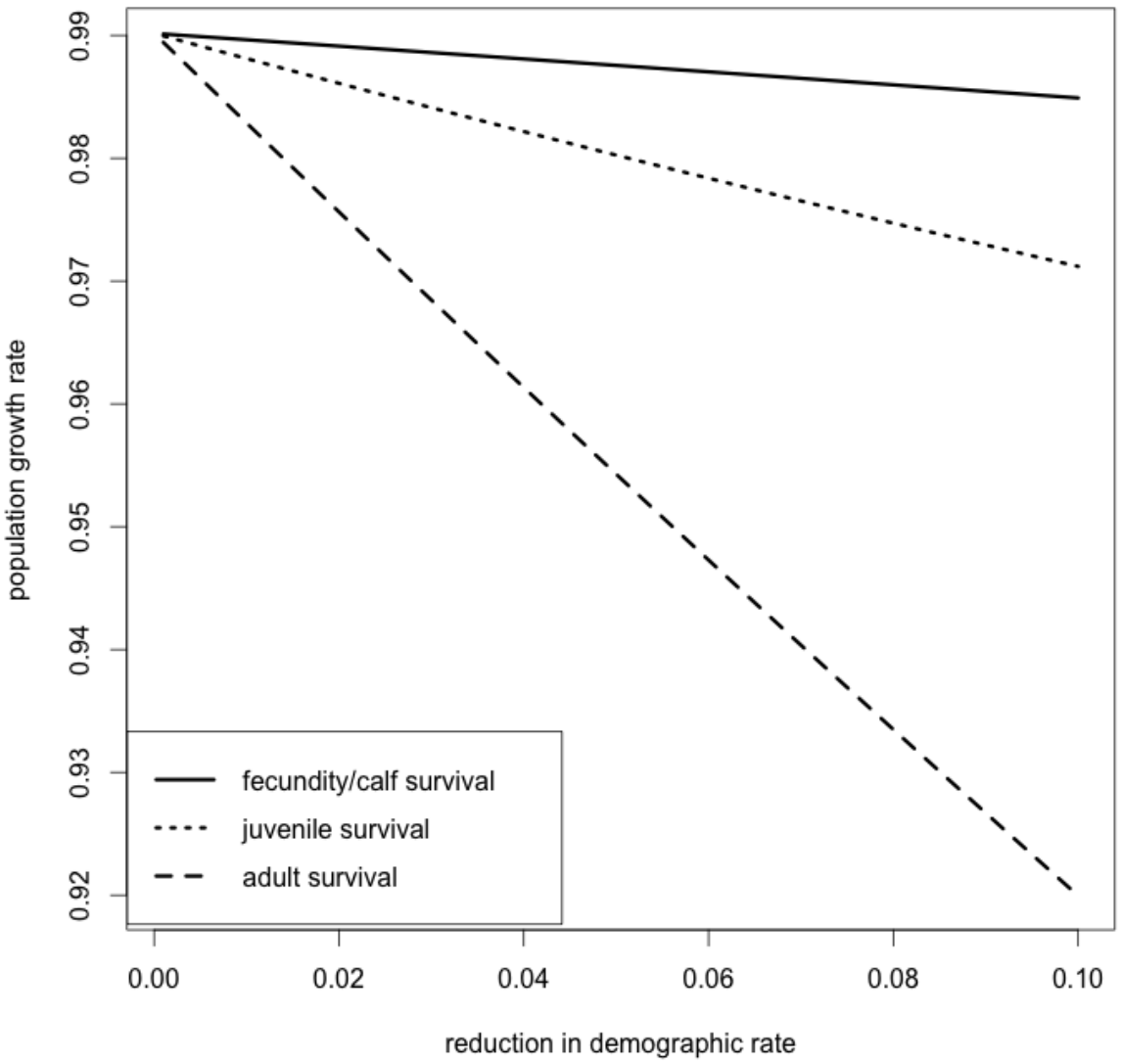

Figure 2.4 *Results of an elasticity analysis for a harbour porpoise population. The baseline population has a calf survival rate of 0.6, a juvenile survival rate of 0.85, an adult survival rate of 0.925, and a fecundity rate of 0.4. The age at independence is 1 year, and the age at first breeding is 5 years. This results in a population growth rate of 1.000.*

We therefore conducted an analysis of the sensitivity of the population growth rate of each of the five species to the number of pups/calves, juveniles or adults that might be predicted to die or fail to breed in a particular year as a result of the effects of an offshore renewable energy development. For convenience these numbers are expressed as a proportion of the size of the population in the MU that is likely to be affected. The values shown on the horizontal axis of

each of the following figures can therefore be converted to the actual number of animals predicted to be affected by multiplying them by the size of the relevant MU taken from Anon. (2014).

The results for each species are similar, so we have only illustrated the results for three different life history strategies that generate stable age structures (see section 5.4 of Caswell, 1989:) in which the proportion of pups or calves, juveniles and adults differ substantially. The three life history strategies are:

1. A harbour porpoise population with a low age at first breeding, a high adult survival and a low fecundity (Fig. 2.5). This population has a preponderance of adults.

2. A harbour porpoise population with a low age at first breeding, a relatively low adult survival and a high fecundity (Fig. 2.6). This population has approximately equal number of juveniles and adults.

3. A bottlenose dolphin population with a high age at first breeding, a high adult survival, and a low fecundity (Fig. 2.7). This population is mostly made up of adults, with only a small number of calves.

For all species and life-history strategies, an offshore renewable energy development that is predicted to result in the failure of a specified number of females to breed will have a smaller impact on population growth rate than one that results in the death of the same number of individuals; and a development that results predominantly in the death of adult animals will have a greater impact on population growth rate than one which affects any other age category, or one that affects all age categories equally.

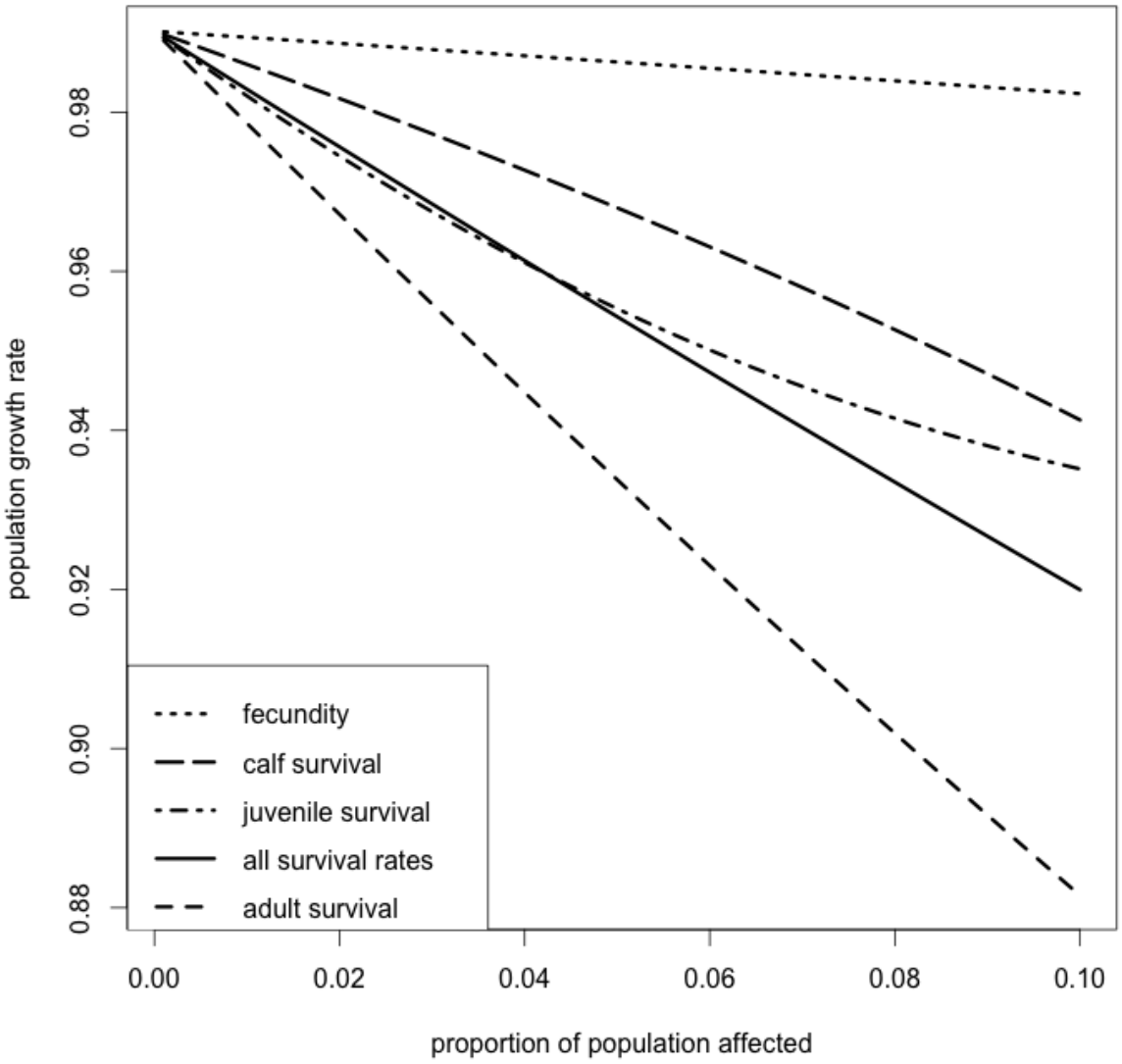

Figure 2.5 *The sensitivity of the growth rate of a harbour porpoise population to an offshore renewable energy development that is predicted to result in the death or failure to breed of a specified number of individuals. The baseline population has a calf survival rate of 0.6, a juvenile survival rate of 0.85, an adult survival rate of 0.925, and a fecundity rate of 0.4. The age at first breeding is 5 years. This results in a population growth rate of 1.000.*

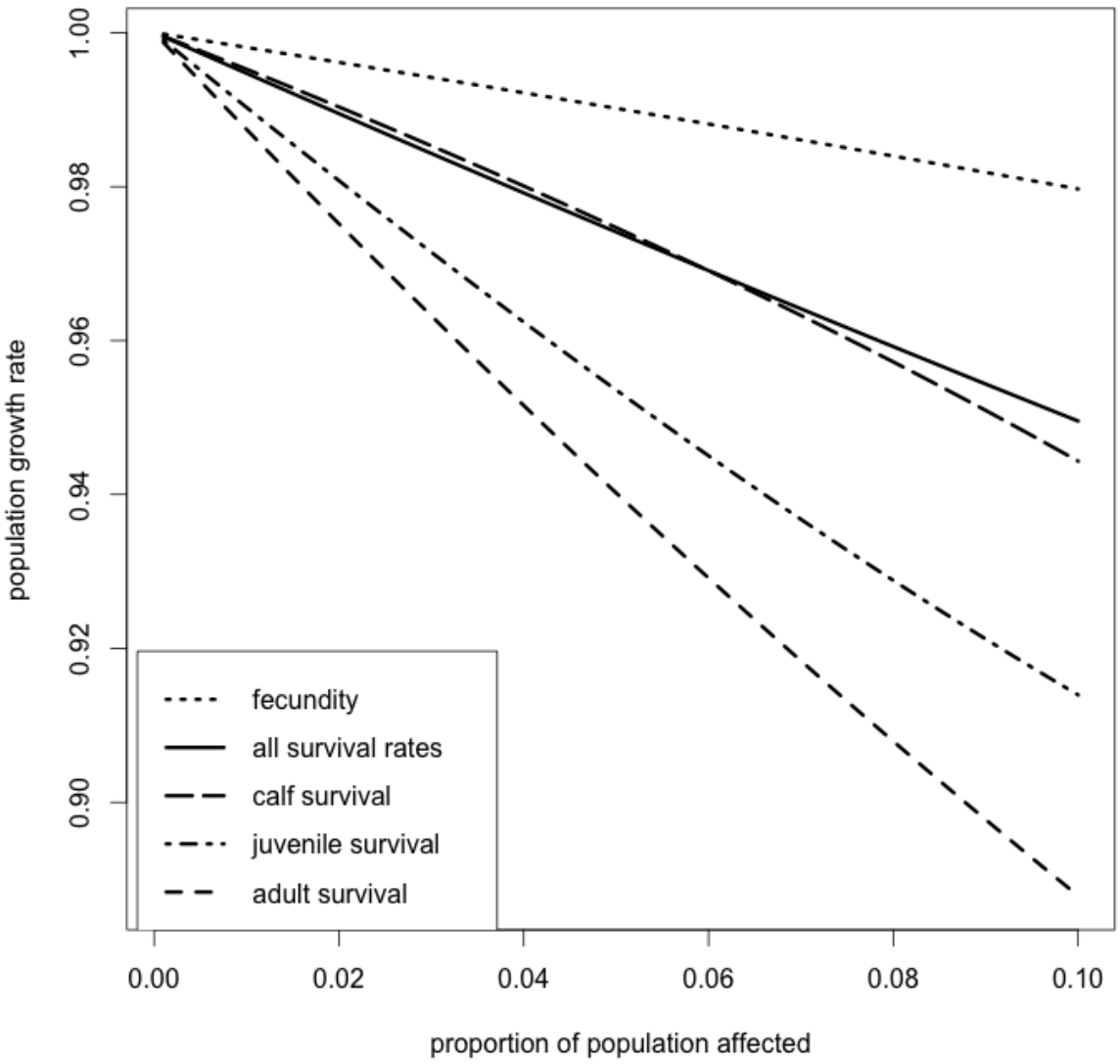

Figure 2.6 *The sensitivity of the growth rate of a harbour porpoise population to an offshore renewable energy development that is predicted to result in the death or failure to breed of a specified number of individuals. The baseline population has a calf survival rate of 0.6, a juvenile survival rate of 0.85, an adult survival rate of 0.85, and a fecundity rate of 0.96. The age at first breeding is 5 years. This results in a population growth rate of 1.000.*

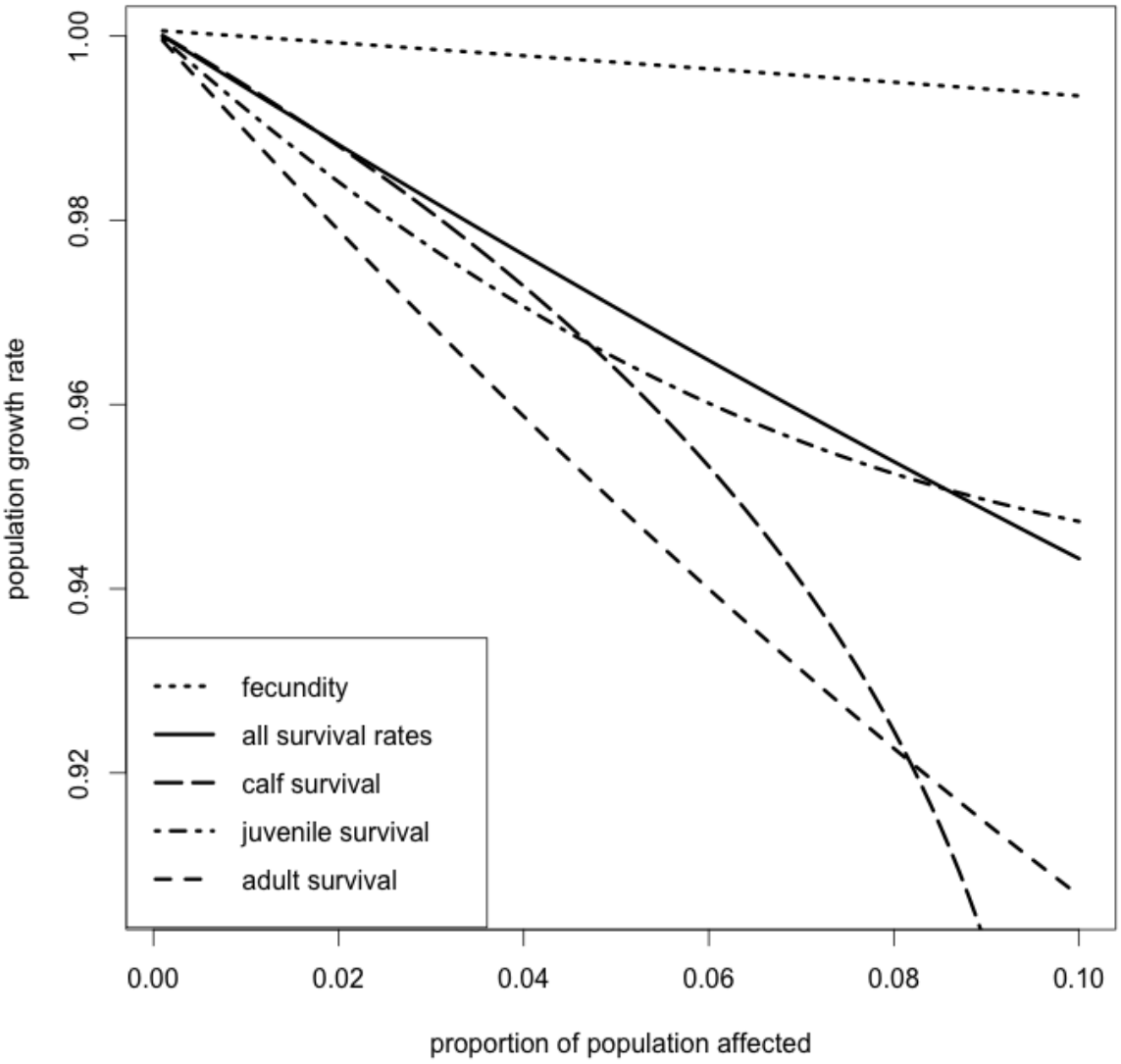

Figure 2.7 *The sensitivity of the growth rate of a bottlenose dolphin population to an offshore renewable energy development that is predicted to result in the death or failure to breed of a specified number of individuals. The baseline population has a calf survival rate of 0.8, a juvenile survival rate of 0.94, an adult survival rate of 0.94, and a fecundity rate of 0.25. The age at independence is 2 years and the age at first breeding is 9 years. This results in a population growth rate of 1.000.*

2.7 Stochastic population modeling, incorporating uncertainty, and evaluating Favourable Conservation Status

The sensitivity analysis described in section 2.6 is based on the following assumptions:

- demographic rates will remain constant over time,
- the number of animals affected by an offshore renewable energy development can be estimated without error,
- the size of the affected population is known exactly, and
- the effect of random variations in the numbers of animals dying and giving birth each year can be ignored.

In practice, none of these assumptions is likely to be entirely valid. A stochastic population dynamic modelling framework is therefore required to assess the effects of these sources of uncertainty, and we have adopted the modelling framework used for population viability analysis (PVA). Morris & Doak (2002) provide a detailed description of this approach. PVAs are generally carried out to estimate the probability that a population will become extinct, or fall below some critical population size, over a defined time horizon. However, the same general approach can be used to model any series of population events where small numbers and uncertainty are expected to play a large role.

The uncertainties that need to be addressed are:

1. Uncertainty about the size of the population in a particular MU;
2. Uncertainty about what proportion of that population will be vulnerable to the effects of a particular development;
3. Uncertainty in the predictions of the number of animals that will experience disturbance or injury as a result of one day of construction or operation;
4. Uncertainty about the effects of disturbance and injury on individual vital rates;
5. The effects of demographic stochasticity and environmental variation.

Items 1 and 3 are related, because calculations of the number of animals predicted to experience disturbance and injury depend, amongst other things, on the estimate of total population size that is used in the calculation. Predictions of the effects of disturbance and injury at the population level are ultimately affected by uncertainty about what proportion of the population is actually exposed to these threats on a particular day. Uncertainty about this proportion for the three cetacean priority species (harbour porpoise, bottlenose dolphin and minke whale) is being investigated as part of the continuing analysis of data collected under the Joint Cetacean Protocol. This analysis will not, however, capture uncertainty in the estimate

that could result from the use of different models for the propagation of the noise associated with construction or operation, or from the use of different ways of modelling the effects of differences in hearing sensitivity at different frequencies, such as M-weighting (Southall et al., 2007) or dB_{ht} (Nedwell et al., 2007).

Uncertainty in item 4 can be modelled by drawing at random from statistical distributions derived from the results of an expert elicitation process. For each iteration of the model, the software selects a set of parameter values at random from these distributions. This is equivalent to soliciting the opinions of one 'virtual' expert for each iteration. These values determine the number of days of disturbance required to have different levels of effect on vital rates, the effects of this disturbance on those vital rates, and the effects of injury on survival and fertility.

Year to year variations in environmental conditions are likely to affect the survival and fertility rates for all individuals in a population. As part of the expert elicitation process that was used to obtain estimates of the parameters of the transfer function relating disturbance to changes in vital rates, Harwood et al. (2014) also asked the same experts "by how much do you think survival or fertility is likely to vary from year to year for populations of this species in northern European waters in the absence of disturbance?" and invited them to choose one of six percentage values ranging from 0% to 50%. Because many survival and fecundity rates for marine mammals are close to 1.0 it is not possible for them to vary symmetrically around the mean from year to year. Harwood et al. (2014) therefore modelled environmental variation in each demographic rate using a Beta distribution, whose mean corresponded to the baseline value and whose variance was adjusted so that the lower 99% confidence limit corresponded to the mean percentage value chosen by the experts. They assumed that variation in demographic rates was uncorrelated, both among age/stage classes and among years. Table 2.1 summarises the values they derived.

Table 2. 1 *Values that can be used to describe environmental variation in demographic rates for five marine mammal species in UK waters, taken from Table A2.1 of Harwood et al. (2014). Each value represents the lower 99% confidence limit for the rate, expressed as a percentage of the mean. Harwood et al. (2014) did not ask experts for their opinion about the level of environmental variation in calf and juvenile survival rates for minke whales, because these age classes are rarely observed in UK waters. However, these values are required to simulate the full dynamics of the population, and they therefore suggest that the same values as those provided by the experts for bottlenose dolphins*

should be used for this species. Grey seal pups were treated as juveniles (i.e. independent of their mothers) because they do not usually enter the water frequently when they are pups (i.e. dependent on their mothers) and are therefore not likely to be exposed to the same level of disturbance associated with offshore renewable energy developments as adults and juveniles.

Species	Pup/calf survival	Juvenile survival	Adult fecundity
Harbour Seal	30%	30%	25%
Grey Seal	30%	30%	20%
Bottlenose dolphin	25%	20%	30%
Harbour Porpoise	25%	30%	25%
Minke Whale	(25%)	(20%)	20%

Demographic stochasticity is caused by the fact that, even if survival and fecundity rates are constant, the number of animals in a population that die and give birth will vary from year to year because of chance events. Demographic stochasticity has its greatest effect on the dynamics of relatively small populations. One consequence of demographic stochasticity is that two otherwise identical populations that experience exactly the same sequence of environmental conditions will follow slightly different trajectories over time. As a result, it is possible by chance for a population that experiences disturbance effects to increase, whereas an identical undisturbed population may decrease.

Under the European Community Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora, Member States are allowed to issue a licence to disturb protected species, provided this will not have a negative effect on their "favourable conservation status". Favourable conservation status is not particularly well defined, but EU (2006) suggests that a population which has experienced a decline in size of more than 1% per year would have an unfavourable conservation status. This suggests that predictions of the changes in population growth rate that are likely to occur as a result of disturbance would provide a useful metric for assessing the effects of this disturbance on the favourable conservation status of the species that are likely to be affected.

Harwood et al. (2014) explain how the stochastic modelling framework described above has been incorporated into a simulation package written in the R statistical computing environment (R Development Core Team 2010). In sections 3-7 of this report we provide suggested values for the demographic rates of the populations in each of the MUs identified by Anon. (2014) that can

be used with this package to provide advice on potential changes in favourable conservation status that might occur as a result of the effects of offshore renewable energy developments. We also review available information on the potential effects of disturbance on individual behaviour and vital rates for the five priority marine mammals. In Section 8 we consider how this approach might be extended to some other marine mammal species.

3. Grey seal

3.1 Demographic rates for UK Management Units

Although Anon. (2014) identified 11 MUs for grey seals in UK waters, telemetry studies (Russell et al. (2013) of UK grey seals have indicated that female seals may breed at colonies that are far removed from the MUs within which they are observed outside the breeding season. We therefore think that it is inappropriate to model the dynamics of each MU separately. We therefore suggest that a single population model is used for all MUs with demographic rates chosen so that the growth rate of the population is 1% per year, the overall growth rate of the British grey seal population in recent years (SCOS, 2012), p8). We based the demographic rates shown in Table 3.1 on the mean estimates in Thomas (2012) for colonies that are monitored annually, and then tuned them to achieve the desired population growth rate. We assumed that the pup survival was the main demographic rate that responsed to changes in population density, because Thomas (2012) found this to be the model that best fitted the survey data.

Table 3. 1 ***Suggested demographic rates for grey seal MUs in UK waters. age2 is the age at first breeding and age1 is the age at independence.***

Management Unit	Growth rate	age2	age1	Pup survival	Juvenile survival	Adult survival	Fecundity
All	1.01	5	1	0.235	0.94	0.94	0.84

3.2 Empirical evidence of potential effects of disturbance on behaviour and vital rates

The effects of exposure to non-pulsed sounds on pinnipeds in water are poorly understood. However, Götz & Janik (2010) performed sound playback experiments with grey seals in the wild and observed a moderate response (level 6: sustained avoidance behaviour) to Received Levels

(RLs) of 135-144 dB re 1µPa. The authors used three different sound stimuli in their experiments: acoustic deterrent device (ADD) sounds, psychophysical model (PPM) sounds based on a model of sensory unpleasantness in humans, and control sounds (white noise and sine-wave pure tones). Animals moved away from the sounds source for all three types of the sound, but the strongest avoidance was in response to the PPM sounds.

Anderwald et al. (2013) found that increased vessel traffic on the northwest of Ireland during the construction of an underwater gas pipeline had a negative effect on the presence of grey seals. They suggested that the animals were displaced because of the increase in low-frequency noise from the construction vessels, although no RLs were provided.

Captive grey seals showed strong avoidance responses to playbacks from 200 kHz and 375 kHz sonar systems response (Hastie et al., 2014). The strongest behavioural response was to the 200 kHz sonar system. Seals spent significantly more time hauled out, this is equivalent to a level 7. The maximum 1/3 octave band SPLs within the seals' hearing ranges was 165.7dB re 1 Pa at 1m (RMS) for the 200 kHz and and 160.3dB re 1 Pa at 1m (RMS) for the 375 kHz sonar systems. Mid-frequency sonar playbacks to the closely related hooded seals have also been shown to elicit active avoidance behaviour at RLs of 160-170dB (RMS) re 1µPa (Kvadsheim et al., 2010). The animals reduced the amount of time they spent diving and actively avoided the sound source (equivalent to level 6 response). However, they appeared to habituate to the sound in subsequent trials. There are no published data on the response of grey seals to pulsed sound.

There is no direct empirical evidence of the way in which disturbance may affect the vital rates of grey seals. However, grey seals are capital breeders (Jönsson, 1997; Stephens et al., 2009) that rely on energy stored in the form of blubber to sustain them through the breeding season. The presence of these energy stores should enable them to survive even extended periods of reduced feeding activity caused by disturbance. However, the resulting depletion of their energy store could result in a failure to breed (i.e. reduced fertility), or a reduction in the amount of energy transferred to their pups during lactation. The latter effect would result in pups being weaned at a smaller size than normal, and post-weaning survival of grey seal pups is known to be affected by their weight at weaning (Hall et al., 2002). In addition, there is some evidence that the fertility of individual females does vary from year to year (Pomeroy et al., 2010). These arguments suggest that fertility and pup survival are the two vital rates most likely to be affected by disturbance in grey seals.

4. Harbour Seal

4.1 Demographic rates for UK Management Units

In contrast to the grey seal MUs, there is good evidence that the MUs proposed by Anon. (2014) for harbour seals can be considered as discrete populations for modelling purposes. However, the only MU for which there is good information on demographic rates is the Moray Firth, and we have therefore had to base our suggested demographic rates for the other MUs on the available estimates for this MU. Cordes (2011) provided a fecundity estimate of 0.88 for this population, and an adult survival rate in the range 0.94 - 1.0. Thompson et al. (2013a) used a value of 0.97 for adult survival and a value of 0.7 for pup survival. Recent aerial surveys suggest that the earlier decline in this population "may have been halted" (SCOS, 2012), while modelling studies (Matthiopoulos et al., 2014) suggest that it is now increasing slowly. We therefore suggest using a growth rate of 1.0 for this population. The juvenile survival rate required to give this population growth rate with the values for pup survival, adult survival and fecundity used by Thompson et al. (2013a) is 0.46: substantially less than the survival rate for pups. This seems unlikely. We therefore suggest using an adult survival rate of 0.94 (which is still within the range of Cordes' (2011) estimates) and a pup survival rate of 0.6 for this MU. The juvenile survival rate required to give a population growth rate of 1.0 with these values is 0.61. We suggest that this combination of demographic rates should also be used for all of the MUs whose status is reported to be "unclear' in SCOS (2012).

We estimated the annual rate of decline for the Shetland and Hebrides MUs from the earliest and most recent survey counts in Table 3 of SCOS (2012), and we took the rate of decline for the East Coast of Scotland MU from Lonergan & Thompson (2012). The reasons for the decline in size of these MUs is still unclear. However, there has been a marked increase in the number of harbour seals found dead in UK waters with so-called corkscrew injuries (Bexton et al., 2012), suggesting there may have been a decrease in survival for adult and juvenile animals. In addition, Matthiopoulos et al. (2014) report that "breeding success" (which we refer to as fecundity) was "the most volatile demographic component of the (Moray Firth) population." We therefore considered two scenarios: that fecundity and pup survival had remained at their undisturbed levels in these populations and the decline was entirely the result of a decrease in juvenile and adult survival; that adult and juvenile survival had remained at their undisturbed

level and the decline was entirely the result of a decrease in pup survival and fecundity. We were able to replicate the observed decline in the Shetland MU using both scenarios, and these are shown in Table 4.1. However, we were unable to replicate the observed decline in the Orkney and East Coast Scotland MUs by reducing fecundity and pup survival alone. We have therefore only shown the results from the first scenario for these MUs.

The status of the North-east and South-east England MUs is unclear. Table 4 of SMRU (2012) described the "England" harbour seal population (which is predominantly composed of these two MUs) as "increasing", but subsequent text (p41) indicates that the count for the "English East coast population (Donna Nook to Scroby Sands) in 2011 was 8% lower than the 2010 count. However, this was 26% higher than the mean of counts between 2004 and 2008." The individual counts shown in the accompanying figure (Fig. 7, p42) are highly variable. Hopefully, more recent counts will have clarified this situation but, until these are published, we are unable to offer any suggestions for the best way to model the dynamics of the populations in these MUs.

Table 4. 1 *Suggested demographic rates for harbour seal MUs in Scottish and Northern Irish waters. age2 is the age at first breeding and age1 is the age at independence.*

Management Unit	Growth rate	age2	age1	Pup survival	Juvenile survival	Adult survival	Fecundity
Shetland modified using survival	0.945	4	1	0.6	0.58	0.89	0.88
Shetland modified using fecundity	0.945	4	1	0.16	0.61	0.94	0.24
Orkney & north coast modified using survival	0.915	4	1	0.6	0.56	0.86	0.88
Moray Firth	**1**	**4**	**1**	**0.6**	**0.61**	**0.94**	**0.88**
East Coast modified using survival	0.82	4	1	0.6	0.50	0.76	0.88
South-west Scotland	1	4	1	0.6	0.61	0.94	0.88
West Scotland	1	4	1	0.6	0.61	0.94	0.88
Western Isles	1	4	1	0.6	0.61	0.94	0.88
Northern Ireland	1	4	1	0.6	0.61	0.94	0.88

4.2 Empirical evidence of potential effects of disturbance on behaviour and vital rates

As for grey seals, our understanding of the effects of non-pulse and pulsed sound exposure on harbour seals in water is poor. However, it appears that harbour seals exhibit only mild avoidance (a level 6 response) at RLs of 90 - 130 dB re 1µPa for non-pulse sound (Jacobs & Terhune, 2002; Kastelein et al., 2006a; Southall et al., 2007), although captive animals (Kastelein

et al., 2006a) showed a much stronger avoidance response. However, the wild animals may have had a higher tolerance to sound exposure because of food motivation (Southall et al., 2007).

There is only limited empirical data on the response of other pinnipeds in the genus *Phoca* to pulsed sounds in water. The most extensive data come from studies of the response of ringed seals (reviewed in Southall et al., 2007) to airgun arrays at RLs of 160-200 dB re 1µPa. Most animals showed no response until levels were > 190 dB re 1µPa. These RLs appeared to elicit moderate responses (level 6: avoidance behaviour). However, the calculated SEL for these studies (175 dB re 1µPa^2-s) exceeded the estimated level for TTS in harbour seals. It appears that at least some pinnipeds may show little avoidance response unless the RL is high enough for mild TTS (Southall et al., 2007). This view is supported by the research of Hastie et al. (in press) who monitored the movements of 24 harbour seals using telemetry during the installation of an offshore wind farm. Some of these animals approached within 7km of piling operations and probably experienced RLs as high as 146.8 - 169.4dB re 1µPa.

Harbour seals are, like grey seals, capital breeders that can, in principle, use the reserves of energy in their blubber to survive extended periods of disturbance. In addition, adult females have more opportunity to feed during lactation than grey seals, and so they may be more able than grey to compensate for the effects of reduced energy reserves to ensure that their pups are weaned at a weight that does not adversely affect their survival. However, as noted in section 4.1, Matthiopoulos et al. (2014) found that the fecundity of harbour seals in the Moray Firth varied substantially from year to year, suggesting that fertility (the probability that an individual female will give birth in a particular year) is the individual vital rate that is likely to be most sensitive to the effects of disturbance.

5. Minke whale

5.1 Demographic rates for UK Management Units

There is very little empirical information on demographic rates for common minke whales in European waters. However, Hauksson et al. (2011) estimated a fecundity rate and age at first breeeding based on samples from the Icelandic whaling industry. Taylor et al. (2007) summarised information on life history parameters for most cetacean species collected worldwide. In Table 5.1, we suggest using their estimate of the adult survival rate for common minke whales, which is based on a longevity estimate of 51 years. They also provided an estimate of calf survival of 0.8 using a ratio of calf survival to adult survival calculated for southern right whales, humpback whales and bottlenose dolphins. If this value is used in population projections with a value of 0.88 (intermediate between the calf and adult survival rates) for juvenile survival, it implies a population growth rate of 1.049. However, survey results from the Northeast Atlantic (NAMMCO, 2011) indicate that this population has decreased in size since 2001. Until there is a reliable estimate of the growth rate for the European population, we suggest using values for calf and juvenile survival that result in a population growth rate of 1.0.

Table 5. 1 *Suggested demographic rates for minke whale MUs in UK waters. age2 is the age at first breeding and age1 is the age at independence.*

Management Unit	Growth rate	age2	age1	Calf survival	Juvenile survival	Adult survival	Fecundity
European waters	1	9	1	0.70	0.77	0.96	0.91

5.2 Empirical evidence of potential effects of disturbance on behaviour and vital rates

Empirical data on minke whale response to acoustic disturbance is sparse. There appear to be no studies of behavioural response to pulsed sounds but a few studies have observed behavioural responses to non-pulse noise. Palka and Hammond (2001) reported a level 3 response (minor changes in speed, direction and dive profile) of minke whales to vessel noise at RLs of 110-120 dB re 1µPa. The 3S2 (Sea Mammals, Sonar, Safety) project - an international project funded by a number of Norwegian, Netherlands, UK and US agencies and WWF - observed strong avoidance behaviour (level 6-7) by a minke whale to low frequency active

sonar playback (Tyack, 2012) as source levels were increased from 152 -214 dB re 1µPa. The animal stopped feeding, changed dive patterns and avoided the sound source when subjected to a RL of approximately 80-100 dB re 1µPa. The animal subsequently showed a further increase in swim speed and change in dive patterns when RLs increased to 140-150 dB re 1µPa.

Anderwald et al. (2013) found some of evidence of minke whale displacement as a result of increased vessel traffic during the construction of an underwater gas pipeline on the northwest coast of Ireland. Although no RLs were provided, minke whales are known to be sensitive to low frequency noise. Minke whales have also been shown to reduce dive times and increase erratic movement in the presence of whale-watching boats (Christiansen et al., 2013). These observations suggest that their feeding activities were disrupted, with implications for individual vital rates and reproductive success (Christiansen et al., 2013).

Common minke whales are capital breeders that rely on energy reserves accumulated during the summer months, when they are in UK waters, for maintenance and lactation during the breeding season. Behavioural disturbance in UK waters is therefore unlikely to affect survival, but could affect the rate at which energy reserves are accumulated, with subsequent consequences for individual fertility.

6. Bottlenose dolphin

6.1 Demographic rates for UK Management Units

Lusseau (2013) used the results of capture-recapture analysis of sightings of individually recognisable bottlenose dolphins on the East coast of Scotland to obtain demographic rates that could be used to model the population dynamics of the Coastal East Scotland MU (which has an estimated growth rate of 1.018 - Cheney et al., 2013), and the sub-population within the Moray Firth (which appears to have a growth rate closer to 1.0 - Cheney et al., 2012). We suggest that the latter values are used for the other MUs identified by Anon. (2014).

Table 6. 1 *Suggested demographic rates for bottlenose dolphin MUs in UK waters. age2 is the age at first breeding and age1 is the age at independence.*

Management Unit	Growth rate	age2	age1	Calf survival	Juvenile survival	Adult survival	Fecundity
All other MUs	1	9	2	0.8	0.94	0.94	0.25
Coastal East Scotland	1.018	9	2	0.9	0.947	0.947	0.3

6.2 Empirical evidence of potential effects of disturbance on behaviour and vital rates

A number of studies have investigated how dolphins respond to increased and sustained vessel presence. However, only a few studies have measured or predicted RLs (bottlenose dolphins, Buckstaff, 2004; Indo-Pacific bottlenose dolphins, Morisaka et al., 2005; white-sided dolphins, Palka and Hammond, 2001). In these studies the dolphin species exhibited minor to moderate changes in behaviour (levels 2-5) at RLs of 110-130 dB re 1µPa. These responses included changes in vocal behaviour (level 2, Buckstaff et al., 2004; level 5, Morisaka et al., 2005) and changes in orientation which was linked to simple avoidance behaviour (level 3, Palka and Hammond, 2001). The Indo-Pacific bottlenose dolphin exhibited a strong response to vessel noise with prolonged changes in vocal behaviour (a level 5 response).

Although these responses to vessel noise may appear minor in comparison to those shown by other species to pulsed noise with high source levels, their cumulative impacts may result in

significant long-term changes in behaviour. For example, even minor to moderate changes in group distribution or group size may have an impact on foraging behaviour and thus on fitness.

Other studies have assessed the effect of vessel presence on dolphin behaviour, although received noise levels were not quantified. Increased vessel traffic has been shown to disrupt resting behaviours and cause short-term displacement from preferred areas (Lusseau, 2003; 2004). Bottlenose dolphins off the coast of Zanzibar reduced the time they spend foraging, resting and socialising and increased the amount of time spent travelling during vessel presence (Christiansen et al.; 2010). These changes may have significant energetic implications. In addition, dolphin movements in Doubtful Sound, NZ, became increasingly erratic with high vessel presence (Lusseau, 2006), and these could ultimately have an impact on the stability of group composition. They may also impede a mother's ability to stay with her dependent offspring. The response of bottlenose dolphins to increased vessel presence in Shark Bay, Australia has led to a decline in local abundance (Bejder et al., 2006). Thus, short term behavioural responses to vessel noise may become biologically significant if animals are exposed for sustained periods of time (Bejder et al., 2006).

There have been a number of studies of the response of delphinid species to a selection of non-pulse sound sources other than vessel noise in both captive and wild settings. Captive bottlenose dolphins have been shown to refuse to initiate trained tasks (a level 6 response) when subjected to noise exposures of 170-180 re 1µPa (Nachtigall et al., 2003), and to actively avoid experimental situations (a level 8 response) at noise levels of 180-200 re 1µPa (Finneran & Schlundt, 2004). Houser et al. (2013) found that captive bottlenose dolphins rapidly habituated to repeated exposures to simulated mid-frequency sonars when the RL was < 160 dB. However, no habituation was observed when the RL exceeded 175 dB, and all dolphins refused to participate (a level 7 response) in trials when the RL reached 185 dB.

Bottelnose dolphins have also been observed to avoid the sound source, decrease their vocal activity and change their swim patterns (a level 7 response) in response to noise exposures of 140-160 dB re 1µPa (Niu et al., 2012). In the wild, animals have been observed to avoid acoustic pingers and change their vocal behaviour (a level 6 response) with a source level of 165 dB re 1µPa (Leeney et al., 2007).

It is important to note that the level 7 and 8 responses to non-pulse noise were shown by captive animals (Niu et al., 2012; Finneran and Schlundt, 2004), which may not be representative

of animals in the wild. For example, Risso's dolphins have been reported not to respond to non-pulse noise at RLs of 135 dB re 1µPa (Southall et al., 2010).

There is limited data on the behavioural response of bottlenose dolphins to pulsed noise. Finneran et al. (2000) observed moderate changes (level 4 and above) in the behaviour of captive bottlenose dolphins to recordings of explosions at RLs of 196 dB re 1µPa (peak-to-peak) and a captive beluga whale to recorded explosions at an RL of 220 dB re 1µPa (peak-to-peak). Substantial variations in behavioural responses between individuals were observed in these studies.

Other mid-frequency cetaceans have been reported to show moderate level responses (level 6; Miller et al., 2005) or no response (Madsen & Møhl, 2000; Madsen et al., 2002) at RLs of 120 – 180 dB re 1µPa.

As noted above, Bejder et al. (2006) documented a decline in the relative abundance of a sub-population of bottlenose dolphins in Shark Bay, Australia that experienced high levels of disturbance, and Lusseau & Bejder (2007) describe how the change in behaviour that was observed in response to this disturbance could lead to a reduction in reproductive output through decreased calf survival and increased inter-calf interval (i.e. individual fertility). It therefore seems likely that these are the two individual vital rates that are will be most sensitive to disturbance in UK bottlenose dolphin populations.

7. Harbour porpoise

7.1 Demographic rates for UK Management Units

As part of the development of a model of the potential effects of by-catch on the harbour porpoise population of the North Sea, Winship & Hammond (2008a) estimated survival (which they assumed to be constant across adults and juveniles), age at maturity and maximum birth rate that were compatible with data from by-caught animals and survey data. Moore & Read (2008) used a similar approach for harbour porpoises in the Northwest Atlantic, but they modelled age-specific survival. We used Winship & Hammond's value of 4 for the mean age at maturity to estimate age at first breeding, and their two values for survival (0.85 and 0.925) as alternatives for adult survival. However, we chose a lower value of 0.6 for calf survival, based on Moore & Read's analysis. We then tuned fecundity to achieve the population growth rate of 1.0 suggested by Fig. 7 of Winship & Hammond (2008a) using either of these values.

Table 7. 1 *Suggested demographic rates for harbour porpoise MUs in UK waters. age2 is the age at first breeding and age1 is the age at independence.*

Management Unit	Growth rate	age2	age1	Calf survival	Juvenile survival	Adult survival	Fecundity
North Sea, based on low adult survival	1	5	1	0.6	0.85	0.85	0.48
North Sea, based on high adult survival	1	5	1	0.6	0.85	0.925	0.24

7.2 Empirical evidence of potential effects of disturbance on behaviour and vital rates

There are limited data on the behavioural responses of harbour porpoise to vessel noise. Palka and Hammond (2001) showed that harbour porpoise avoid vessels at RLs of 110-120 dB re 1µPa by changing their swimming patterns (a level 3 response).

Other studies (e.g. Olesiuk et al., 2002) have looked at the response of captive and wild animals to Acoustic Deterrent Devices used to scare seals away from marine fish farms. In controlled settings, harbour porpoise exhibit strong avoidance behaviour, with changes in swimming and

breathing patterns (a level 6 response), to RL of 90-120 dB re 1µPa (Kastelein et al., 1997; 2000; 2001; 2005; 2006b). Similar level 6 responses have also been observed in response to RLs of 130-150 dB re 1µPa (Teilmann et al., 2006). A captive harbour porpoise also exhibited brief behavioural responses, such as changes in swim speed, to playbacks of helicopter dipping sonar at RLs of 124-140 dB (Kastelein et al. 2013). Wild animals have shown strong avoidance behaviour (a level 6 response) at RLs of 80-160 dB re 1µPa (Culik et al., 2001; Olesiuk et al.; 2002; Johnston, 2002).

Recent studies have also reported behavioural responses by harbour porpoises to pulsed sound (Kastelein et al., 2008; Tougaard et al., 2009; Brandt et al., 2011; Lucke et al., 2009). These studies observed level 6 and above responses at RLs of 100-120 dB re 1µPa and 175-196 dB re 1µPa (peak-to-peak). In particular, harbour porpoise appear to be sensitive to noise associated with pile driving (Dähne et al., 2013; Teilmann & Carstensen, 2012). Teilmann & Carstensen (2012) found that some harbour porpoises continued to avoid areas where pulsed sounds associated with piling had exceeded these levels for at least 10 years. Thompson et al. (2013b) found that harbour porpoise stopped vocalising when exposed to RLs of 165-172 dB (peak-to-peak) from seismic survey arrays in the Moray Firth. However, animal detections returned to pre-disturbance levels a few hours after the sound exposure.

Although some studies have found substantial among-individual variation in the response of harbour porpoises to noise exposure, many harbour porpoises do appear to respond to noise exposure at much lower RLs than other marine mammal species (Southall et al., 2007).

The harbour porpoise is one of the smallest cetaceans and it is therefore unable to store large reserves of potential energy in its blubber. As a result, harbour porpoises, unlike the four other species considered here, are probably unable to survive extended periods of behavioural disruption, if these affect their feeding ability. Indeed, Koopman (1994, cited in Koopman, 1998) suggested that “the potential energy stored in the blubber layer would be sufficient to sustain a starving porpoise for only a few days”. This suggests that the survival of all age classes may be sensitive to the effects of disturbance.

8. Extending the approach to other marine mammal species

Any attempt to extend the approach outlined in the previous sections for assessing the potential effects of disturbance to populations of other marine mammal species faces a number of challenges. First, there is limited information on the response of most species to anthropogenic noise. The exceptions are the killer whale and Risso's dolphin. There is considerable information on the behavioural response of killer whales to vessel noise (Lusseau et al., 2009) and the potential consequences of this for energy acquisition and expenditure (Williams et al., 2006). In addition, Miller et al. (2014) accumulated sufficient data from controlled exposures of killer whales to a simulated navy sonar (a non-pulse sound) that they were able to fit a dose-response relationship. Similar studies of the response of Risso's dolphins fitted with telemetry devices to the same sound source (Southall et al., 2010) should provide sufficient information by the end of 2014 to fit a dose-response for this species.

The second challenge is to identify which vital rates are most sensitive to the effects of disturbance. Again, the best evidence is for killer whales, where data from the Pacific coast of Canada suggests that reproductive performance (Ward et al., 2009) and calf survival (Ford et al., 2010) in this species are linked to prey availability. These results imply that these are the individual vital rates that are most likely to change as a result of disturbance.

The final challenge is to develop realistic stochastic models of the population dynamics of these species. Kuningas et al. (2013) have recently published estimates of survival and reproductive rates for killer whales in the Northeast Atlantic, and Murphy et al. (2009) provided estimates of some reproductive parameters for common dolphins in the same region. In addition, Winship & Hammond (2008b) developed a model of the population dynamics of common dolphins in the Northeast Atlantic in order to assess the impacts of bycatch, which they fitted to survey data and life history data from stranded and bycaught females. Unfortunately, similar information is not available for the other marine mammal species (white-beaked dolphin, white-sided dolphin, Risso's dolphin) reviewed by Anon. (2014), and it is unlikely to become available in the near future.

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Glossary

Acute effect	The indirect effect of a change in behaviour or physiology on vital rates (q.v.) via individual health (q.v.)
Body condition	A measure of an individual's energy stores. In marine mammals, usually blubber thickness or total body lipid. One component of health (q.v.)
Chronic effect	The direct effect of a change in behaviour or physiology on vital rates
Demographic rates	The average survival and fecundity rates, and ages at independence and first breeding experienced by all members of a population in a particular year
Demographic stochasticity	Variation among individuals in their realised vital rates (q.v.) as a result of random processes
Environmental variation	Variation in demographic rates (q.v.) among years as a result of changes in environmental conditions
Expert elicitation	A formal technique for combining the opinions of many experts. Used in situations where there is a relative lack of data but an urgent need for conservation decisions
Fecundity	The average of individual fertility rates for all members of a population
Fertility	The probability that an individual adult female will give birth to a viable offspring in any particular year
Fitness	A relative term reflecting the potential contribution of the genotype of an individual to future generations. The fittest individuals leave the greatest number of descendants relative to the number of descendants left by other individuals in the population
Health	All internal factors that may affect individual fitness (q.v.) and homeostasis, such as body condition (q.v.), and nutritional, metabolic, and immunological status
Management Unit (MU)	The animals of a particular species in a geographical area to which management of human activities is also applied (Anon. 2014)
Population size	The number of animals of a species estimated to occur in a particular Management Unit (q.v.) as defined by the IAMMWG (Anon. 2014)
Uncertainty	Incomplete information about a particular subject. In this report, we are only concerned with those components of uncertainty that can be quantified
Vital Rates	The probability that an individual will survive from one year to the next, the probability that an individual adult female will give birth in one year

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