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ASSESSMENT METHODOLOGY FOR IMPULSE NOISE

A case study on three species in the North Sea

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SUMMARY

This report describes the outcome of a project commissioned by Rijkswaterstaat aimed to develop a policy supporting advice on how to assess the impact of impulsive noise on marine life in the North sea. The project builds upon previous international work in this field. Underwater noise is a transnational problem which can only be handled through international co-operation. Alignment of methods and discussions on the merits and drawbacks of various options in the assessment framework are therefore crucial. This assessment will provide input to the international discussion on how Good Environmental Status (GES) for impulsive noise can be assessed and will be used for the reporting obligations to OSPAR and for the Marine Strategy Framework Directive (MSFD) in the North Sea.

The report was written by experts from Rijkswaterstaat, TNO, WMR and Arcadis. It describes the implementation of a method proposed by OSPAR (Merchant et al. 2018a) to assess the impact of impulsive noise on marine life. In the study a stepwise assessment framework is applied and evaluated through case studies on impulsive noise assessment.

This project showed that the framework is useful in determining effects of impulse noise on indicator species. The stepwise assessment provides a clear and structured approach to assessing impulsive noise and consists of 10 steps. For each step the procedures were followed as closely as possible, and limitations were identified. The major steps are the identification of indicator species for the assessment, the choice or calculation of effect distances for those species and the interpretation of the final outcome (risk curves and risk indicators).

Despite its usefulness, our study showed that there are a number of important shortcomings of the method that require further discussion and research. Firstly, the selection criteria proved difficult to apply to select species. Based on literature on existing criteria for selecting indicator species, a set of selection criteria were chosen to enable the selection of indicator species to be used in this study. It proved difficult to fully apply all the criteria in the final species selection. In practise there is a lack of knowledge and data for many marine species found in the North Sea. Therefore, species were chosen based on the best data availability and suitability. The process and final selection of the indicator species was the result of discussions among experts and has been well documented in this report. This resulted in the selection of the harbour porpoise, the harbour seal and Atlantic cod as indicator species. Due to a general lack of data for invertebrates only species of higher trophic levels could be chosen for this study. This means that the indicator species choice is highly biased towards species at a higher trophic level.

For all three species data on distribution and habitat were collected and gathered into distribution maps. These were made for the determined assessment area, an extension of the Dutch North Sea, and time frame, 2015 – 2017 as at the start of the project for these years information was available in the impulse noise register. Based on data from the impulse noise register pressure maps were made that were combined to form exposure maps. These could not be created for harbour seals as data proved unsuitable.

An important input needed to create these pressure and exposure maps, is the distance from the source at which effects on the indicator species can be shown (the effect distance). Even for the relatively well-studied species considered here, documented effect distances can be used. However our study showed that effect distances are often not available or validated for the environment in which they are applied.

In order to determine effect distances sound propagation models can be used. However these models require dose-response curves and our study has shown that for most species these curves are lacking. In recent years most knowledge on impulse noise has been acquired through projects for offshore wind energy with pile driving. However sound propagation from seismic surveying requires further validation. It is plausible that the same framework can also be applied to seismic surveying, but different components need validation, like propagation modelling, dose-effect curves, duration of disturbance and its effect on animal fitness.

With the data from the exposure maps, risk curves and risk indicators were created. The use of risk curves and risk indicators to evaluate the Good Environmental Status seems to be useful to combine both the spatial and temporal aspects of the exposure to underwater noise as required by the Marine Strategy Framework Directive. However more experience is needed in the use of these curves and more examples

need to be generated to understand their meaning. These examples should cover more species and more variation in noise exposure.

The MSFD requires that the status of the ecosystem is assessed, and not the exposure of individual animals. The requirement can be translated into the effect on the population of the indicator species or on the suitability of the habitat for the species under the exposure to noise. Translating the exposure curves for harbour porpoise and cod could not be done within the current project, but this could be possible in the future.

To translate the effect of underwater noise from individual animals to population consequences various models are available. The two most popular approaches that have been developed are the iPCoD model and the DEPONS model. There is currently no clear preference for either model. To include the changes in animal distribution due to the exposure to impulsive noise now also Individual Based Models (IBM) are under development for other species.

Currently, lack of information on both species distribution, abundance, habitat use and their behavioural response to impulsive noise is one of the major problems identified in the impulse noise assessment. Dedicated research targeting identified data gaps should be carried out. Beside a general lack of information, there is a specific lack of spatial information and available GIS data. Therefore, internationally available data should be collated and analysed together in a joint effort.

1 INTRODUCTION

1.1 Underwater noise

Sound is a common phenomenon in the underwater marine environment and is vitally important for many marine animals. For many marine animals, the auditory senses are very important for survival, which is especially true in areas of their habitat where visibility is very low due to high turbidity (for instance in the Wadden sea) or due to the absence of light (for instance in the open ocean).Many marine animals use sound to navigate, find food, communicate with potential partners, perceive danger and as a warning against various threats. For example, many marine mammals use echolocation to detect their prey, certain fish species produce a variety of sounds with their swim bladders to communicate, and some invertebrates can make loud sounds to deter potential enemies and communicate with each other.

Underwater sound is much less attenuated than airborne sound therefore marine sound carries over larger distances than sound in air. For example, in the deep ocean, the low frequency sound of certain whale species may carry over thousands of kilometres and in a shallow sea, like the North Sea, sound can easily travel tens of kilometres. This makes sound a very effective communication channel and a source of information about the environment, location, and behaviour of marine animals.

Underwater sound is a transnational phenomenon since sound sources, sound propagation and the affected animals cross marine boundaries of the national exclusive economic zones (EEZ). The many human activities that contribute to sound in the North Sea are also crossborder activities. Some of the sources are localized (pile driving and explosions), but other sources are mobile (shipping and seismic surveys). Even sound from localized sources can cross borders and become a multinational disturbance.

Many of the terms related to underwater sound are defined by ISO 18405:2017. Unless indicated otherwise OSPAR adopts ISO 18405:2017 as a starting point. Furthermore, definitions are compatible with those adopted by EU Member States in connection with MSFD (TG Noise; Dekeling et al., 2014; update 2020 in prep.).

In this report the term “noise” is used when discussing sound that has the potential to cause negative impacts on marine life. The more neutral term “sound” is used to refer to the acoustic energy radiated from a vibrating object, with no particular reference for its function or potential effect. “Sounds” include both meaningful signals and “noise” which may have either no particular impact or may have a range of adverse effects (*From: “Towards thresholds for underwater noise”, TG Noise in preparation, 2020).*

Underwater noise can generally be divided into two categories: impulsive and continuous. Impulsive noise has a short duration and is caused by activities like pile driving, seismic surveying, explosions and sonar activity. Continuous noise has a longer (continuous) duration and is caused amongst others by shipping and operational wind farms. This project only considers impulsive noise.

1.2 Background and activities

In 2008 the European Commission approved the Marine Strategy Framework Directive (MSFD: 2008/56/EC), requiring all EU Member States, to reach or maintain a Good Environmental Status (GES) by 2020.

“Marine strategies shall apply an ecosystem-based approach to the management of human activities, ensuring that the collective pressure of such activities is kept within levels compatible with the achievement of good environmental status and that the capacity of marine ecosystems to respond to human-induced changes is not compromised, while enabling the sustainable use of marine goods and services by present and future generations.” Article 3, MSFD

This ecosystem approach means that the effects on the whole ecosystem should be assessed and not the effects on individuals. In practice for underwater noise the effects on the population of key species or on the habitat for those species will be assessed.

GES is described by eleven descriptors and all the Member States must set criteria and methodological standards for each descriptor in their marine waters. Member States are required to define the GES they use. In the framework directive underwater noise is marked as a pollutant. Descriptor 11 focuses on the energy in the marine environment, including underwater noise and describes two types of underwater sound, divided into two indicators:

- loud, low and mid frequency impulsive sounds (D11C1) and
- continuous low frequency sound (D11C2).

The EU/MSFD Technical Group on Underwater Noise (TG Noise) is commissioned to further develop the two indicators of underwater sound. Currently TG Noise is working on the development of a common methodology to assess potential effects of underwater noise, as a first step towards development of Threshold Values for D11C1 and D11C2.

Member States (MS) should take measures to ensure that GES is achieved and maintained, and this should be monitored as well. Policy makers need to manage the environmental risks related to underwater noise. In a 6-year cycle the marine environment should be assessed for all 11 descriptors. Descriptor 11 relates to energy, including underwater noise and is subdivided into two criteria, one for impulsive noise (D11C1) and one for continuous noise (D11C2).

The MSFD also requires that member states co-ordinate their activities on a regional level through regional sea conventions to reach GES. For the North Sea this is coordinated through the OSPAR Convention. In the last decade much work has been done on underwater noise. For this project the relevant activities were performed within the EU working group TG Noise (Technical Group), the OSPAR group ICG Noise (Intersessional Correspondence Group) and national work related to offshore wind energy.

TG Noise
TG Noise is a technical group under the Common Implementation Strategy of the MSFD to give an expert advice to the European Commission. An important report by TG Noise is the Monitoring Guidance (Dekeling et al., 2014) which was published in 2014. An updated version is expected in 2020. At this moment TG Noise is also preparing a publication on threshold values (TG Noise 2020). In Figure 1 the workflow for TG Noise to arrive at threshold values is depicted. At this moment TG Noise has arrived at step 4. This project is mainly related to the activities in box 4 of Figure 1.

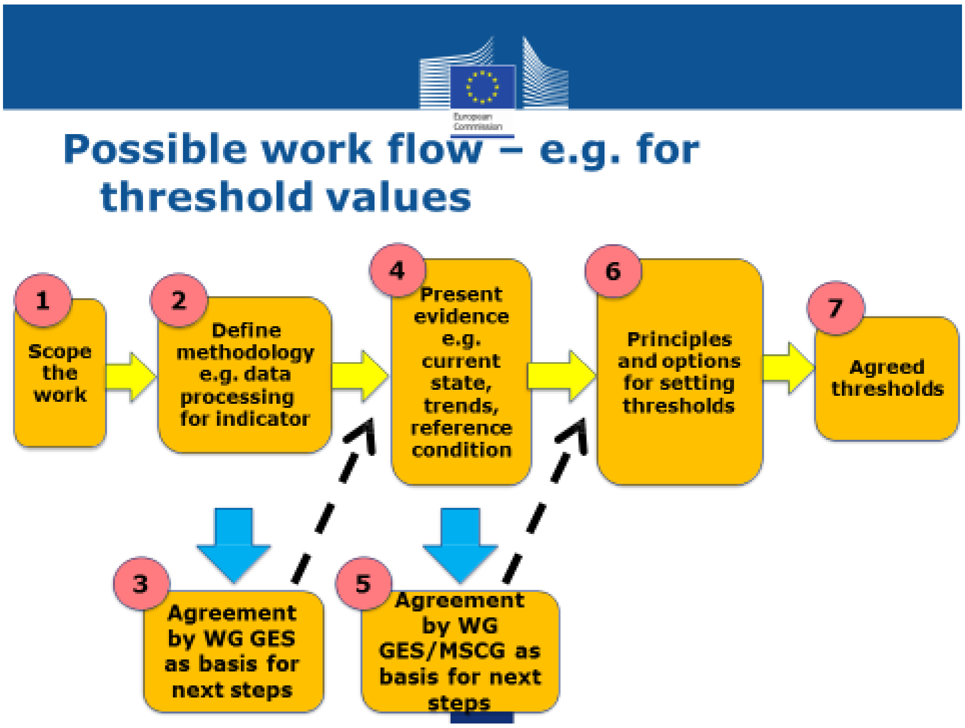

Figure 1: Workflow of TG Noise for its work on threshold values.

OSPAR ICG Noise
The OSPAR working group ICG Noise is looking at the following topics:

- Monitoring of underwater noise
- Developing an assessment framework, defining indicators and assessment of noise
- Inventory of mitigation measures

For the monitoring of impulsive noise a registry has been developed at ICES (International Counsel of Exploration of the Sea). The register contains all activities that produce impulsive noise. This registry plays an important role in this framework and will be analysed in more detail in chapter 3 of this report. At this moment the registry contains data from the years 2015 till 2018.

A pressure indicator has been developed by ICG Noise to assess the pressure by impulsive noise on the environment (described in CEMP Guidelines (EIHA 2017)). It is expected that an indicator for the 'risk of impact' for impulsive noise will be adopted by OSPAR in 2020. The stepwise assessment framework, that is described in this report, follows this proposal closely.

Assessment framework for Offshore wind energy (NL)
Rijkswaterstaat developed a framework to assess the cumulative impact of the piling activities for offshore wind farms (Heinis, et al., 2019). The framework also follows the stepwise approach proposed by OSPAR and uses the interim PCOD model, developed in 2013 by SMRU Marine and the University of Saint Andrews, to estimate population effects as an extra final step in the framework.

1.3 Purpose of the project

The proposed assessment framework by OSPAR has not been applied often, and only with a very limited dataset and applied for a limited number of species. Therefore, its practical use still has to be proven and experience has to be gained. The Netherlands wishes to get hands-on experience itself.

The purpose of this project is therefore to present policy supporting advice on how to assess impulse noise impact for the MSFD in the North Sea. In the study the stepwise assessment framework is applied and evaluated through case studies on impulsive noise assessment. The outcome of this project can be used to improve the proposal for a risk-of-impact indicator by ICG Noise.

The project intends to build on previous international work in this field and to gain experience in implementation of the assessment framework. Different elements of the framework will be evaluated and several case studies will be applied in order to assess:

- Whether the framework can be implemented practically.
- Where methodological errors or data gaps occur.
- Whether the approach matches scientific insight in the effects of impulsive noise on marine life.
- How the approach should be interpreted for the Dutch Continental Shelf (DCS).

Specific questions to be discussed in the project are:

- How to evaluate ecosystem impact from impulsive noise?
- Should this be done at:
  - Population level, i.e. is a significant part of the population affected, or can a significant change in population size is expected?
  - Habitat level, i.e. how much habitat loss is caused?
- How to select species to be considered within the assessment?
- How do models fit in the assessment framework to translate pressures to population impacts?
- What is needed by the end users, i.e. policy makers?
- What are remaining knowledge and information gaps?

Finally, the project will provide a discussion and suggestions for future assessment of impact of impulsive noise on the marine ecosystem.

With this project Rijkswaterstaat hopes to help the international discussion on how Good Environmental Status (GES) for impulsive noise can be assessed. Underwater noise is a transnational problem which can only be handled through international co-operation. Alignment of methods and discussions on the merits and drawbacks of various options in the assessment framework are crucial for progression.

After this introductory chapter in chapter 2 the stepwise approach to the assessment of impulsive noise will be described. In the chapters 3 to 12 the different steps of the scheme will be discussed and illustrated with an implementation of the assessment. Finally, in chapter 13 the results and conclusions will be summarized and in chapter 14: looking forward, the lessons learn for future impulse noise assessment and policy are presented.

2 METHOD

To assess the impact of impulse noise a stepwise approach is followed. This approach has been based on the assessment methodology developed by TG Noise and published by Nathan Merchant (Dekeling, et al , 2014; Merchant, et al., 2018).

The assessment steps are illustrated in Figure 2. Steps 1 to 9 were introduced in the previously named papers and closely follow the approach applied in the Dutch KEC (Heinis et al. 2019). The authors decided to add step 0: define stressor, as the process could apply to multiple stressors, and step 10: determine population effect, as a population effect is a concrete outcome policy makers can base regulation on. Some steps are taken in parallel, at the same stage of the process and can possibly be in a different order. These processes are shown aligned on the same line, e.g. 1,2,3.

The assessment starts by determining what exactly needs assessing: defining a stressor (0). In the next phase the impacted species (1), the assessment area (2) and the temporal resolution (3) are chosen. These steps occur simultaneously as for instance the area chosen has a certain biodiversity and therefore a range of species to choose from. In the next phase information is gathered on the distribution of the chosen species (4) and the impact from the stressor (5), e.g. noise occurrence. In step 6 these datasets are combined to form exposure and risk maps (6), from which exposure curves (7) and risk indicators (8) can be calculated. The error margin of the result is then determined in step 9, after which an estimate of the population impact can be done with a certain confidence (10).

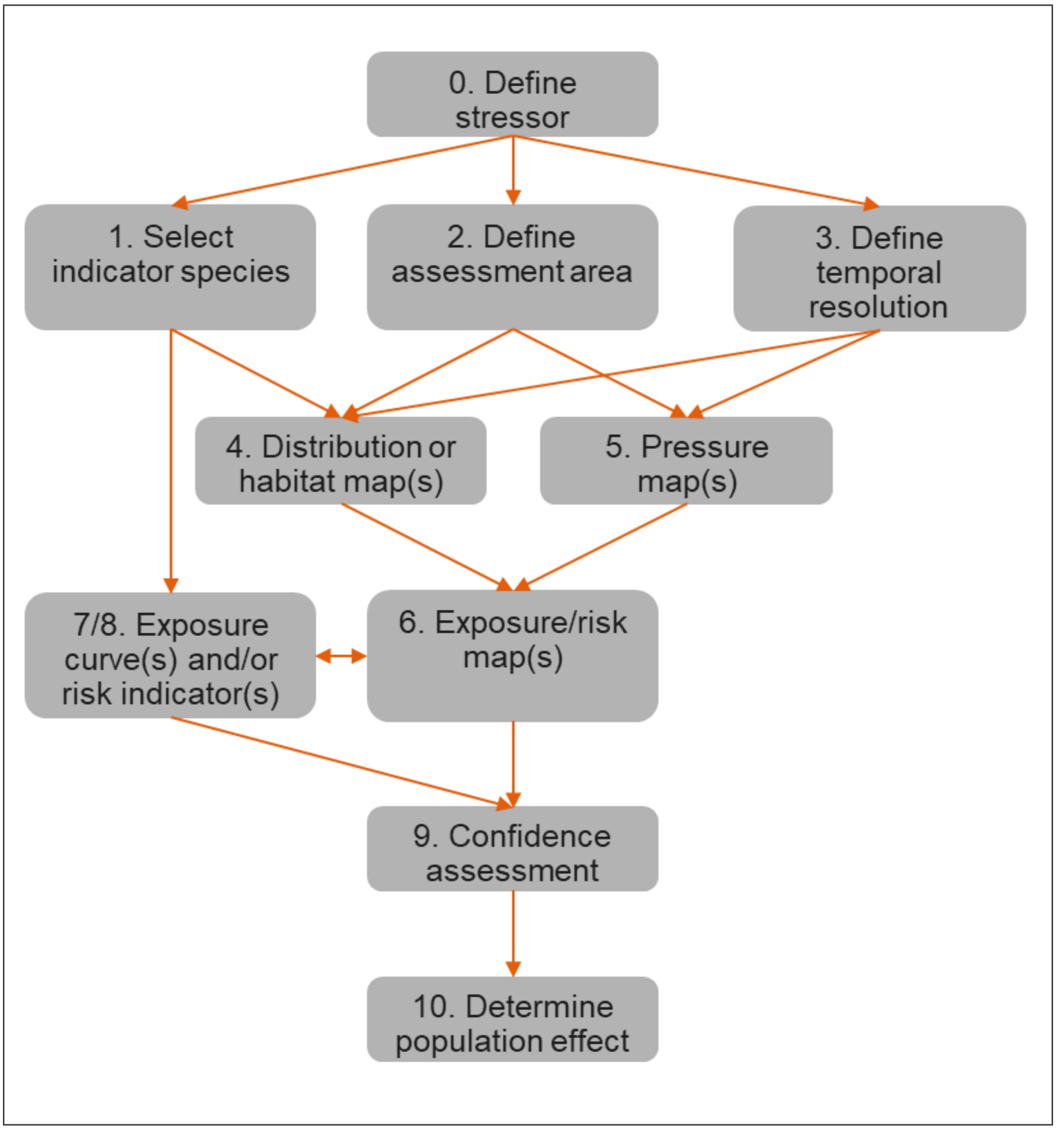

Figure 2: Assessment framework

Within this report the framework and the underlying methodology will be described in detail by applying the methodology to a case study on the impact of impulse noise in the Dutch North Sea. In the following chapters each step including inputs, choices, and outcomes, will be demonstrated.

3 STEP 0: DEFINE STRESSOR

In paragraph 1.1 the definition of impulse noise has been introduced. In this paragraph this stressor is explained in more detail.

Underwater sound consists of vibrations that propagate through the underwater environment. Due to its effective propagation in water, sound an important medium for perceiving the environment for most, if not all, forms of marine life. Sound pressure describes the difference between the sound wave pressure and the ambient pressure, and is perceived by marine mammals, and some fish species. Particle motion describes acceleration, displacement and velocity of particles (due to sound pressure), and is sensed by fish, and many other lower-order species, such as cephalopods, crustaceans, etc.

Impulsive sounds are of short duration and with a rapid onset. Typical anthropogenic sources producing loud impulse sounds are airguns, pile-driving, underwater explosives and sonar working at relevant frequencies. Although these are typically considered as being the dominant sources, other sources such acoustic deterrent devices (ADD), boomers, sparkers and scientific echo sounders are also considered (identified as 'generic impulsive sound sources').

To support the reporting and assessment of impact of impulse sound sources, the OSPAR Commission (which implements the 1992 OSPAR Convention for the Protection of the Marine Environment North-East Atlantic), has tasked the International Council for the Exploration of the Sea (ICES) to set up a register to collect impulsive noise producing activities by EU member states in a systematic manner. This Impulse Noise Register contains information about the temporal and spatial distribution of activities that produce loud underwater noise (Figure 3).

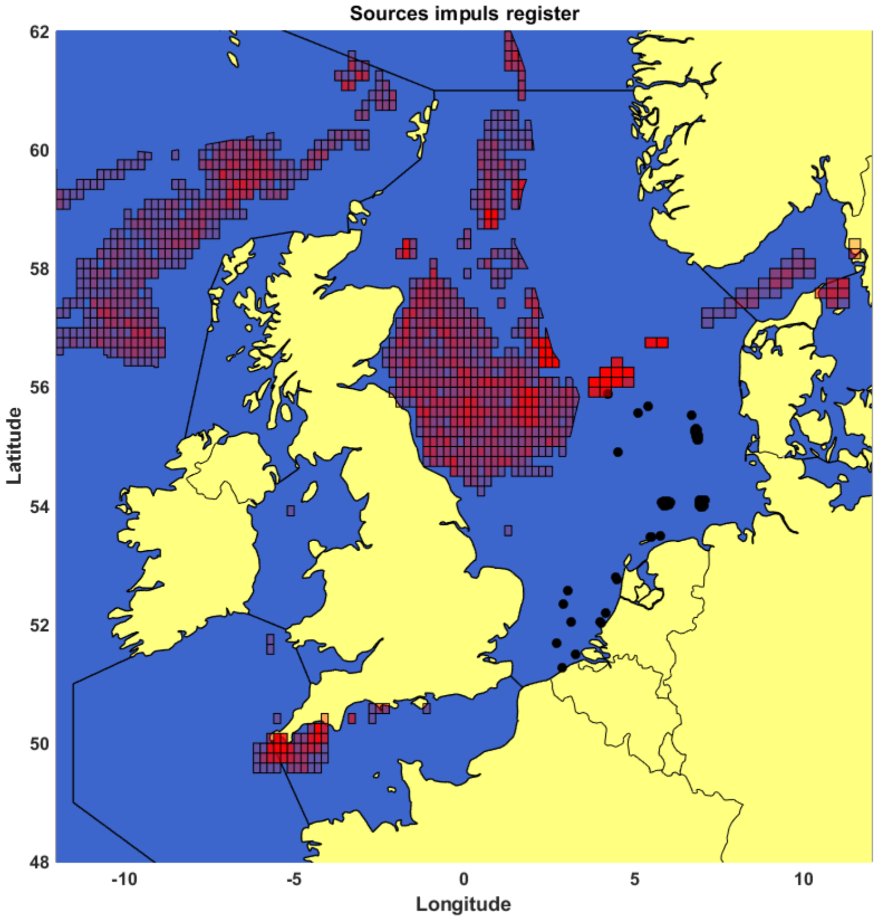

Figure 3: Distribution of impulsive sound sources in the North Sea contained in the impulsive noise register for 2015 (von Benda-Beckmann et al. 2017).

4 STEP 1: SELECT INDICATOR SPECIES

In the previous chapter step 0: define stressor was completed. The identified stressor is: impulsive noise. In this chapter step 1: Select indicator species and the process taken to select the species is described (Figure 4). This step was taken simultaneously with step 2 and 3 that will be discussed in the next chapters.

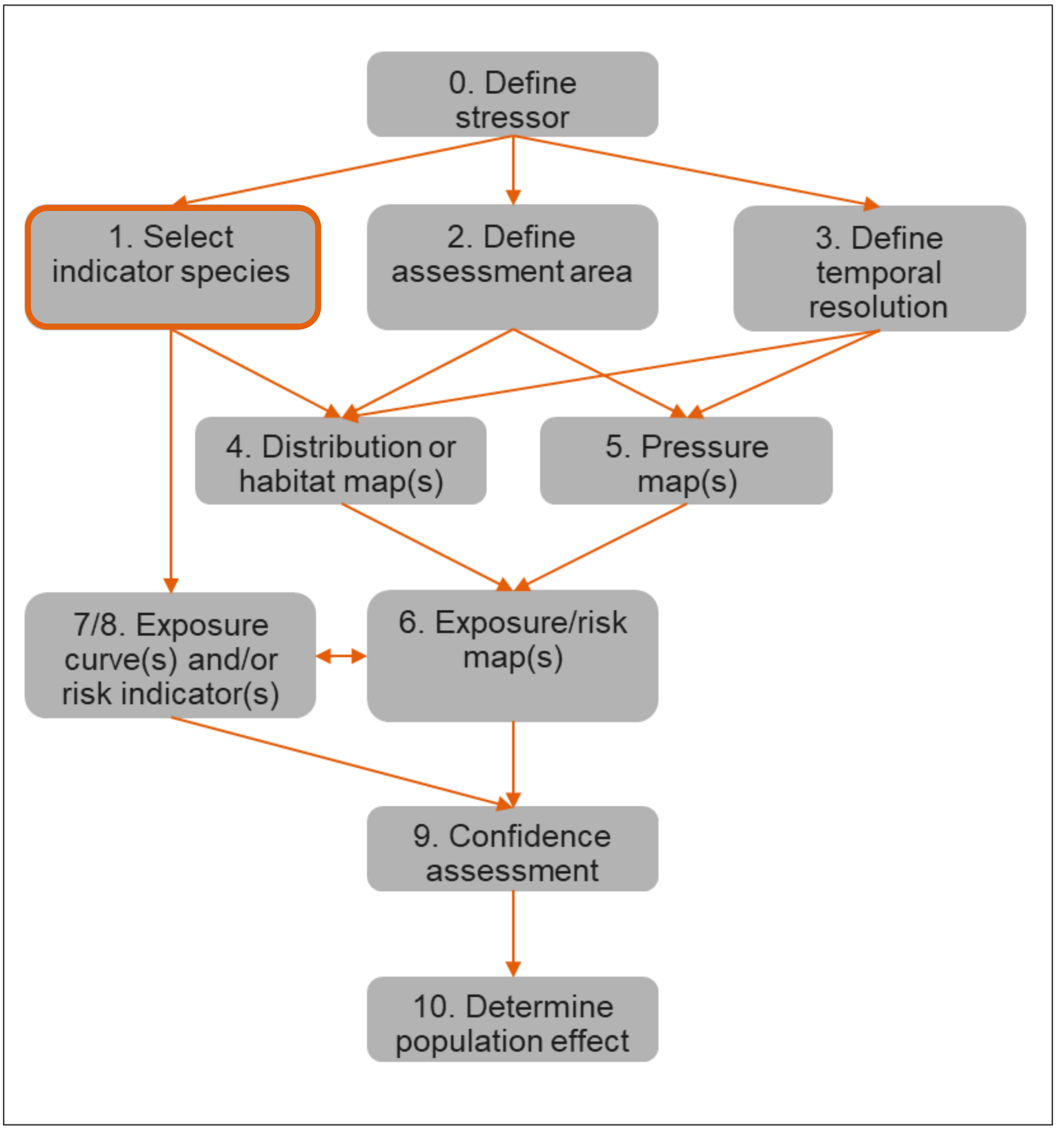

Figure 4: Stepwise approach, current step (highlighted orange).

Chapter 4 is divided into several paragraphs. Firstly (4.1) the definition of an indicator species is discussed. Than the potential impact of impulsive noise on animals is discussed (4.2). Then the four steps of species selection are described in paragraph 4.3 selection criteria. The steps are executed in the following paragraphs:

1. A description of the North Sea ecosystem (4.4)
2. Vulnerability of species to impulsive noise (4.5)
3. Data availability and suitability (4.6)
4. Selection of the indicator species (4.7)

4.1 What is an indicator species?

Indicator species (IS) are used to monitor changes in the environment and can be single species or groups of species (Siddig, et al., 2016). They are selected based on their sensitivity to a particular environmental attribute, such as in our case impulsive sound, and used to assess the effect the attribute has on the IS (Siddig et al. 2016). Ideally an indicator species is representative for a larger group of species or is an indication for the status of the marine environment.

In this chapter we provide the rationale for the selection of indicator species to quantify the effects of impulsive underwater noise on the marine ecosystem in the North Sea. The selection of potentially suitable candidates includes several considerations, such as the sensitivity and vulnerability of the species to impulsive sound, occurrence in the North Sea, management and conservation status and data availability.

4.2 Effects of impulsive noise

Richardson et al. (2013) define *noise* as sound that impairs reception of signals of interest or sound that affects the animal in such a way that normal behaviour is disrupted. *Effect* is defined as a (measurable) change in the species or taxa of interest. The term *impact* is used when effects cause significant risks or negative changes in populations or ecosystems.

As described in section 1.1 and chapter 3, different parameters characterize sound such as sound level and frequency content, and for impulsive sound this also includes pulse length and the number of pulses. In figure 5 a schematic representation is given of the different direct effects of impulsive underwater noise on marine organisms based on their distance to the sound source. However the model used in figure 5 does not consider that sound propagation at sea is three-dimensional and its interference, reflection and refraction patterns will lead to more complex sound fields, in particular when several sound sources are simultaneously active. The simplified assumption is that the effect of sound on animals lessens the further away from the sound source they are, due to geometrical spreading and attenuation. Depending on the distance and noise level sound is expected to have a different effect on animals described by different impact zones (Madsen, Johnson, et al., 2006; Richardson et al., 2013, Figure 5). If, how and when an animal reacts to sound differs not only between species but can also vary between individuals and their internal state (for example their reproductive status).

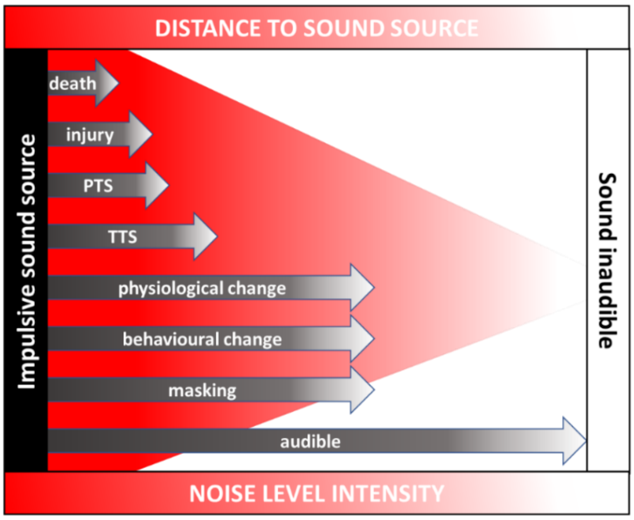

Figure 5: Schematic representation of the different direct effects of underwater noise on marine organisms in relation to distance to the impulsive sound source. PTS and TTS are defined as permanent and temporary threshold shifts in hearing respectively. Effects vary widely between different taxa and can overlap. Adapted from (Gomez et al. 2016a)

The first impact zone is the one closest to the sound source where animals can suffer physical injury which can be so strong that animals can be killed, either immediately or later. Depending on the distance and exposure time, animals can experience temporary threshold shifts (TTS) or permanent threshold shifts (PTS) of their hearing (see Figure 6 and Table 1).

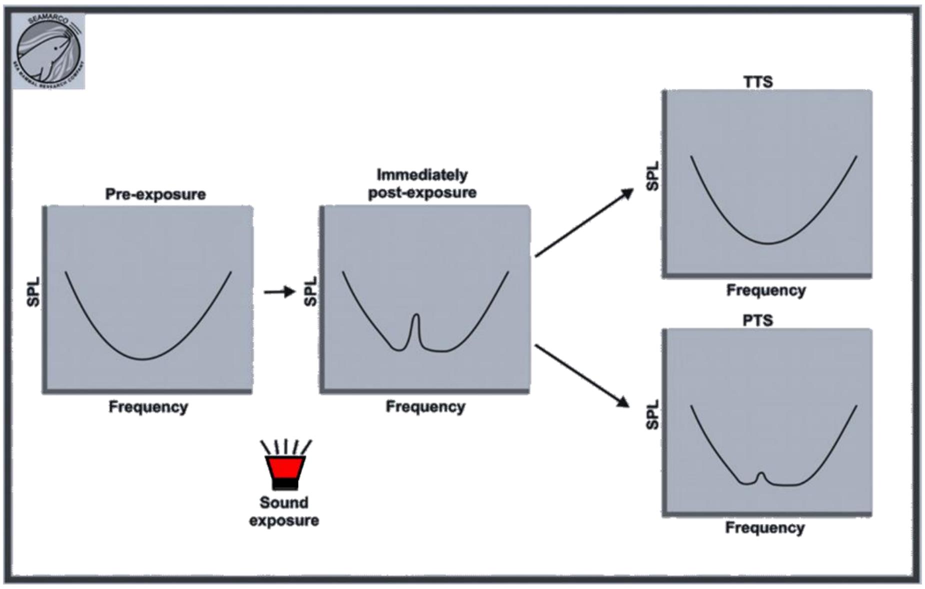

Figure 6: Schematic overview of Temporary Threshold Shift (TTS) and Permanent Threshold Shift (PTS). Exposure to sound results in a shift in the auditory threshold (in a specific frequency range). If this is temporary the hearing ability will return to the pre-exposure level. The sound pressure level (SPL) describes the auditory sensitivity for a specific frequency, higher SPL indicating lower sensitivity. Used with permission from Seamarco.

Further away from the sound source lies the zone in which animals can have behavioural or internal physiological reactions to the sound, such as an increase in stress hormones. These reactions are difficult to measure in the field. Therefore, it is challenging to determine the size of this zone. In the zone of 'masking' the sound overlaps with the acoustic signals animals are producing and receiving, which among other factors depends on the ability of the animal to adapt their hearing. The largest zone is the zone of 'audibility' which describes to what distance animals can discern the signal from the ambient noise.

It is important to point out that depending on the sound source, the sound characteristics, the length of exposure, the environmental conditions and the animal, impact zones will overlap and transitions between zones cannot be sharply defined.

The reaction of individual animals is often considered on the short-term even though long-term effects can be substantial. Permanent hearing loss of certain frequencies can reduce the ability to hunt for food, a long-term exposure to stress hormones can lead to a reduced immune response (Atkinson, et al., 2006) and impulsive sounds can potentially mask communication essential for successful reproduction (Slabbekoorn et al. 2010). Quantifying these cascade effects is very challenging.

Once enough individuals are affected by noise, impacts can be measured at a population level, for example because of reduced reproductive success or increased mortality. For most marine organisms measuring changes in population parameters is more challenging than measuring direct reactions to sound in an experimental or semi-experimental setting. The sea is a highly dynamic environment, adding to the complexity of finding out what factors contribute to observed population level changes. It is therefore often impossible to prove a direct causal relationship between a specific stressor such as impulsive sound and changes in population factors. As stated in the OSPAR assessment for impulsive sound "there is uncertainty as to whether and how these effects of sound on individuals are translated to the population or ecosystem scale" (OSPAR 2017).

Table 1: Overview of potential effects of impulsive sound on marine animals (adapted from Gordon et al. 2003, Popper et al. 2019)

Potential effect		Description
Physiological change	Death & Injury	Immediate death or injury to an animal can be caused for example through the rupture of body tissues. Later death due to an injury is also possible, e.g. through secondary infection. In cases of irreversible injury, it can lead to a decreased fitness of the animal and an increased chance of death.
	TTS & PTS	Permanent threshold shift (PTS) is the reduction in auditory sensitivity from which there is no recovery. Temporary threshold shift (TTS) is a reduction in auditory sensitivity with eventual recovery (see figure 5). TTS and PTS can be caused by metabolic exhaustion or mechanical damage to hair cells in the inner ear. Depending on the taxa, damage to the ear can be reversible or not. TTS and PTS can have secondary effects such as a reduced ability to navigate, communicate or forage. For some marine species permanent hearing loss is an impairment that will lead to death.
	Stress reactions	Other physiological changes include changes in hormones. Increase in stress hormones can lead to an integrated stress response and reduced overall fitness. This can for example include negative changes in the immune or reproductive system. In some species an increase in respiration rates is a proxy for an increase in stress (Barreto & Volpato 2011)
Masking / Perceptual changes		Auditory masking is the amount by which the hearing threshold for one sound is raised through the presence of another (Fries et al. 2007). For some marine species sound is biologically highly relevant, and masking of biologically significant noises can affect communication signals, echolocation, orientation and avoidance of predators or human threats such as shipping.
Behavioural changes		Behavioural changes describe a disruption of normal behaviour. This can include predator avoidance, for example some species will try to hide and “freeze” in reaction to a perceived threat. Others will show flight behaviour, such as moving away from the sound source and an increase in swimming speed. These reactions can interrupt behaviour such as feeding, reproduction or care for offspring. Reactions can differ between species and even individuals (female or male, juvenile or adult) and they are highly dependent on the internal state of the animal.
Indirect effects		Indirect effects have not received a lot of attention to date. Population impacts on a species will have consequences on the North Sea ecosystem. This is especially true for taxa that have key roles in the food webs, both as prey and predators.

4.3 Selection criteria

As a starting point, the indicators for selecting the potential candidate (indicator) species were based on the five criteria outlined by HELCOM (2017) to identify noise sensitive species:

1. **Hearing sensitivity.** For a species to be susceptible to impacts of noise outside of the immediate vicinity of the sound source it must be able to detect sound.

2. **Impact of noise**. A species might be able to detect and produce sound within a range of frequencies, but it may not be very sensitive to noise disturbance. In that case the impact on this animal is likely to be small. On the other hand, an animal may react to noise even if the frequency spectrum is outside the frequency of best hearing or sound production of the species. In this case it can have a substantial impact on this animal.

3. **Threat status.** Populations already threatened by impacts from other pressures, such as eutrophication or hazardous chemicals, may be more susceptible to detrimental effects from noise. Threat status includes information on the red list status of species.

4. **Commercial value.** Noise effects on species with high commercial value can potentially affect the economy of large scales industries such as the fishing industry or the smaller scale recreational industry relying on the presence of marine mammals.

5. **Data availability.** If little or no knowledge is available on either hearing sensitivity or noise impact, or if little or no data are available on spatial distribution, a species is not considered useful at this stage and thus not included. Data supplied at a later stage may warrant a species to be considered a priority species.

The criteria for HELCOM are aiming to find noise sensitive species, considering all types of sound (impulsive and continuous), and are specific to the species occurring in the Baltic Sea and the regional conservation and management frameworks.

These criteria are also reflected in the two criteria used for selecting indicator species by Merchant et al. 2018b: (1) acoustic sensitivity and (2) conservation, ecological, or economic importance. The authors decided that two criteria should be added to this, namely that indicator species may be considered: representative (or precautionary) examples of broader taxa. An overarching condition we applied was the ‘data availability and reliability’. Here we assessed if the scientific evidence was sufficient and of adequate quality to allow application of what is known to the North Sea scale

Using the before mentioned criteria as a guideline we chose the following selection criteria for the North Sea indicator species selection:

1. Vulnerability of species to impulsive sound. This includes:

Acoustic sensitivity: can the organism detect the impulsive sound?

As described in more depth further below, impulsive sound (or sound in general) can be perceived through different mechanisms. Application of this criterion means there is sufficient scientific evidence that the species can detect impulsive sound.

Vulnerability to sound: is the organism vulnerable to the sound?

Vulnerability to sound can be determined through several approaches. The aim is to obtain dose-response curves that allow to link specific sound exposure to an effect, e.g. changes in physiology or behaviour. For many taxa evidence of vulnerability is only limited to small data sets or individual animals. It is important to interpret these results with care, in particular when extrapolating these results to make assumptions about the vulnerability of a population or species in general..

2. Spatio-temporal distribution: does the organism occur in the North Sea with spatio-temporal overlap to impulsive noise?

The selected species in question should be distributed in the North Sea and data on density over time should be available. In addition, the species should occur in areas that are most likely to experience exposure of impulsive sound.

3. Conservation and/or management frameworks: this includes organisms that are listed as priority species for conservation as well as any that are subject to management due to exploitation.

For many taxa there are existing conservation and management frameworks. The aims of these are to reach conservation aims for species that might be at risk and to ensure sustainable harvesting of species that are used for human consumption. For these taxa determining the impact of additional human stressors is of a high priority.

4. Data availability and suitability for species. An overarching condition we applied was the data availability and reliability. Here we assessed if the scientific evidence was sufficient and of adequate quality to allow application of what is known to the North Sea scale.

Finally, the North Sea consists of a diversity of habitats with many different taxa. To ensure that we would not exclude any species or taxa, we first defined the most relevant habitats and their typical inhabitants and reviewed the available data regarding the above criteria.

Based on this we then picked three species that could be used as indicator species for the assessment following the process in figure 6.

Selection step 1: Describe the North Sea ecosystem

↓

Selection step 2: Assess the vulnerability of North Sea fauna to impulsive

↓

Selection step 3: Assess the data availability and suitability for North Sea

↓

Selection step 4: Select three indicator species based on the selection criteria

Figure 7: Steps in the selection process used for the selection of the three sample indicator species for the assessment

4.4 Species Selection Step 1. Description of the North Sea ecosystem

The North Sea was chosen as assessment area (see chapter 5). In Step 1 of the selection process, we provide a brief description of the most typical North Sea habitats and the key taxa that live within them (Figure 8 and Figure 9). This provides a basis for determining which species or groups of species would be best suitable as indicator species.

4.4.1 North Sea habitats

The North Sea was formed mainly by glacial activity during and after the last ice age. It is part of the European shelf, with a general depth of around 30m. Shallower sandbanks occur along the coasts, as well as offshore such as the Dogger Bank. Tidal movements and currents (e.g. Channel current from the South and Central North Sea current from the North) influence the waters, both at sea and along the coast. There are a few deeper trenches, such as the Norwegian Trench that reaches 700m (Paramor et al. 2009). We will consider the typical shelf habitats that occur up to a depth of 200m and not the deeper trenches. The following main habitat zones are found in the North sea (Figure 8):

1. Benthic zones (seafloor) consist of two parts for the North Sea. The ***intertidal zone,*** situated between the low and high tide lines, is not part of our assessment. The other part of the benthic zone is always submerged and called the ***sublittoral***. The substrate of the seafloor consists primarily of soft sediments with only small areas of gravel and pebbles (Paramor et al. 2009).

2. The **pelagic zone (water column)** on a shelf from 0 to 200m is also called the **neritic zone**, which is small in comparison to the oceanic zone which covers the deeper waters off the ocean shelves. Sunlight, in particular in the upper parts of the water column (< 20 m deep) in combination with sufficient nutrient input, allows phytoplankton to thrive (Joint & Pomroy 1993). Phytoplankton production is the main reason for the high productivity in the North Sea system.

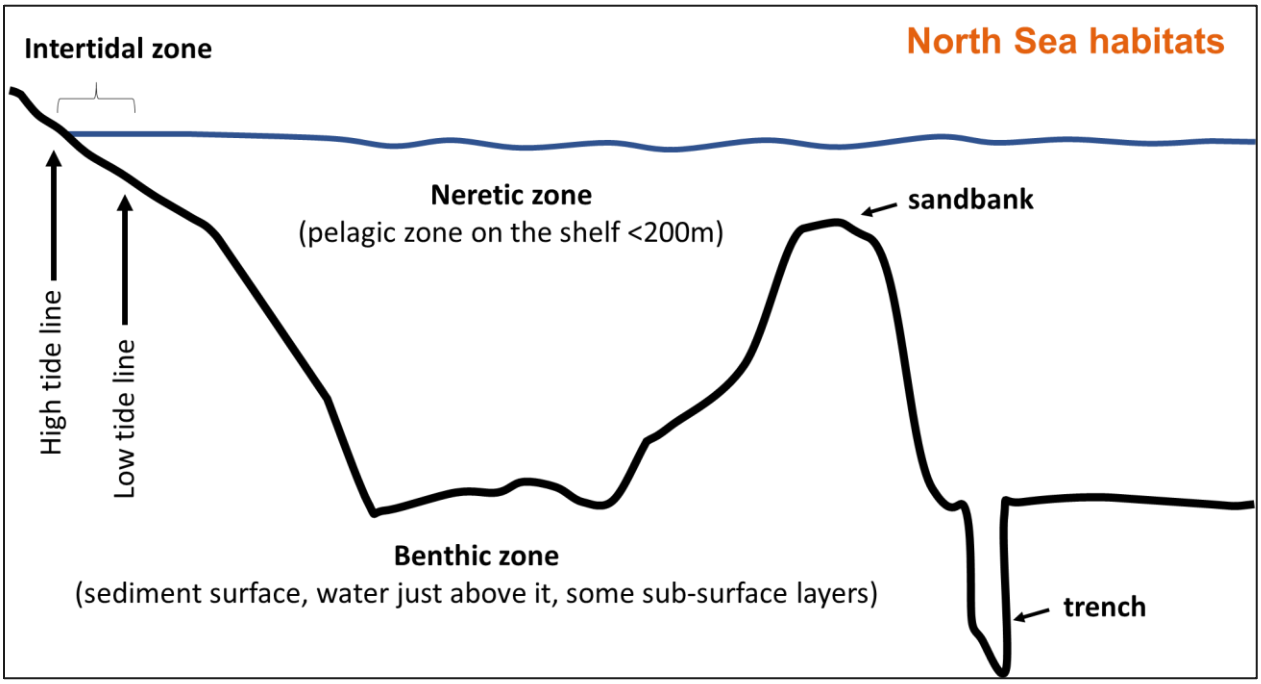

Figure 8: Schematic representation of the main types of North Sea habitats (figure: M. Scheidat).

4.4.2 North Sea fauna

The following faunal communities are found in the North Sea: benthos, zooplankton, nekton and seabirds. These groups are illustrated in Figure 9 and described in the paragraphs underneath.

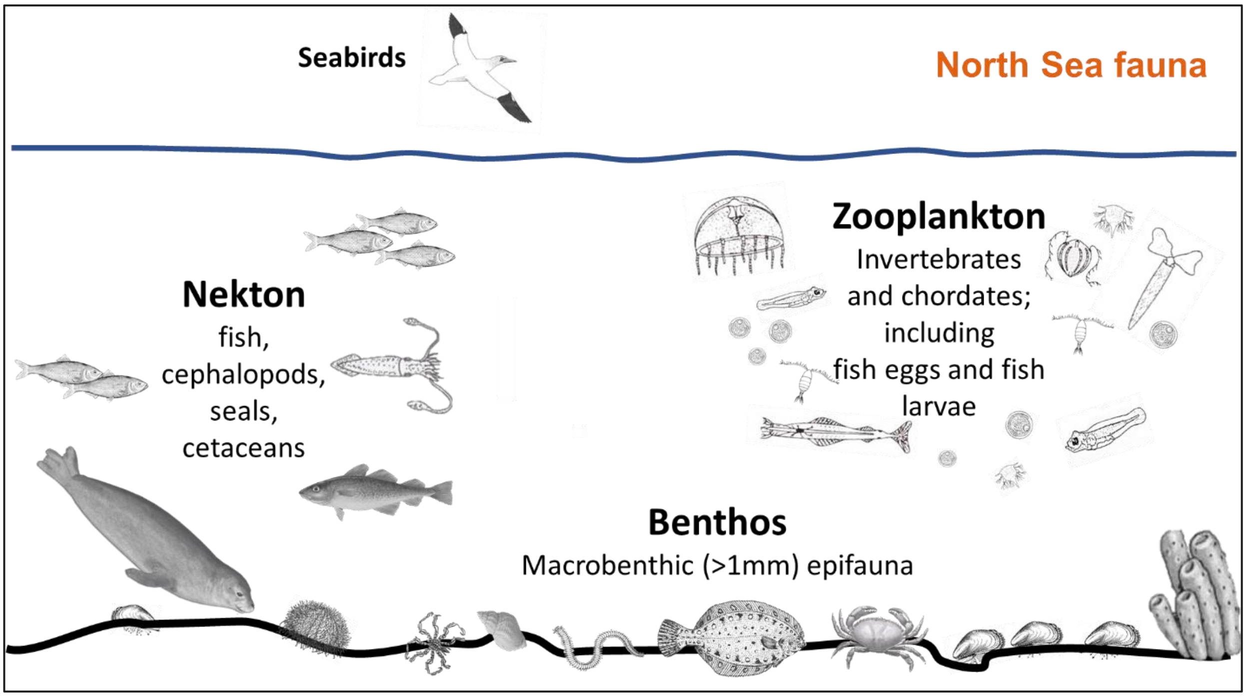

Figure 9: Overview of North Sea faunal communities included in the review on sensitivity to sound (figure: M. Scheidat)

4.4.2.1 Benthos

The organic matter produced in the shallow North Sea is deposited in considerable amounts at the seafloor, helping to sustain rich and diverse benthic communities: benthos (Heip et al., 1992). Benthos are organisms living in or on the sea bottom. Benthos consists of a wide diversity species and shows large-scale spatial patterns related to environmental variables (Heip & Craeymeersch 1995, Reiss et al. 2010). This makes them suitable for monitoring the effects of changes in the environment (Ramsay et al. 1998). Their abundance, biomass and species richness changes from the shallower inshore waters of the southern North Sea, to the offshore communities in the northern deeper waters of more than 50m in depth (Heip & Craeymeersch 1995).

Benthic fauna play an important role in cycling nutrients, decomposing detritus and serving as food for higher trophic levels. Some sessile species are important ecosystem engineers.

Benthic organisms are characterized by size as well as by where they occur. In his report we included those taxa considered macrobenthos, which is larger than 1mm. Most examples provided here are considered epibenthic organisms (occurring *on* the sediment), in contrast to infauna (living *within* the sediment). Macrobenthos have different strategies to obtain food. Some are deposit feeders, some suspension feeders and there are those that consume other fauna as predators. Some are sessile others have varying levels of mobility (Ramsay, et al., 1998).

The North Sea macrobenthos consists of a large number of different taxa, the most common being polychaetes, crustaceans, echinoderms (sea urchins and starfish) and molluscs (snails and bivalves) (Heip et al., 1992; Heip & Craeymeersch, 1995; Reiss et al., 2010).

4.4.2.2 Zooplankton

Marine zooplankton include a wide range of taxonomic groups of mostly small animals (invertebrates, chordates and fish larvae) that drift or weakly swim, primarily in the productive surface waters. It includes a wide range of species with varying size. Some species remain their entire life cycle in the planktonic state, others only occur in this state during their larval or juvenile stage. Zooplankton communities can be very complex and are unique for different habitats. As they are sensitive to changes in their environment they can be used to determine the health of an ecosystem.

For the North Sea the most common organisms in the zooplankton community are small crustaceans, such as copepods (*Calanus* spec.) that are 0.5 mm to 6 mm in size. These form the major food resource for many fish species, such as cod (*Gadus morhua*) and herring (*Clupea harengus*). Other members of this diverse community are “krill” (euphausiids), jelly fish (coelenterates), ctenophores, as well as larvacea, thaliacea (salps, doliolids) and colonial hydrozoa (siphonophores). It also includes larval stages of for example crustaceans (decapods), starfish and sea urchins (echinodermata) and fish (Boersma, et al., 2003).

Zooplankton has an essential role in the marine ecosystem. It forms the most important link between primary production by phytoplankton and higher trophic levels, both in pelagic and benthic habitats.

4.4.2.3 Nekton

Nekton comprises of those organisms that can actively swim and move independently of water currents. Nektonic species come in different sizes, the smallest include crustacean species, such as for example euphausiids (krill) which have been categorized as both macroplankton and micronekton, the largest include seals and whales.

While many micronektonic organisms conduct diel vertical migrations in the water column, moving up during the night and down during the day, their movement range is limited. More typical for this zone are larger nektonic species such as cephalopods (squid and octopus), bony fish, elasmobranches (sharks and rays) and marine mammals (Brodeur & Pakhomov 2019).

As diverse as the species occurring in this region are also their foraging strategies. For instance, herring use specialized gill rakers to sieve out large or smaller planktonic organisms. The minke whale is a baleen whale of the North Sea and filter-feeds on zooplankton and fish (Olsen & Holst 2001). Fast-swimming predators primarily feed on fish, with some (e.g. grey seals and killer whales) include marine mammals in their diet.

One of the characteristics of the larger nekton is that animals can adapt to changes in their environment. In case they encounter a shortage of prey or other adverse situations, they can move into other regions.

4.4.2.4 Seabirds

Seabirds are mainly dependent on the sea for their food and spend most of their life time outside of the breeding season at sea. The majority of the species feeds on prey in the neretic zone (nekton). At least 19 seabird species use the productive waters of the North Sea to forage for food for themselves and to provide their young. The most numerous in the North Sea are northern gannet (*Morus bassanus*), northern fulmar (*Fulmarus glacialis*), herring gull (*Larus argentatus*), lesser black-backed gull (*Larus fuscus*), black-legged kittiwake (*Rissa tridactyla*) and the common guillemot (*Uria aalge*).

In addition, there are bird species living either year-round or during the winter season in the North Sea coastal waters. Some of these use the coastal area for breeding (e.g. terns and cormorants), feeding and resting purposes. Others, like grebes, divers and ducks mainly winter in the coastal zone, where they feed on fish and benthic organisms, respectively.

4.5 Species Selection Step 2: Vulnerability to impulsive noise

To determine the vulnerability of marine animals to sound in general and specifically to impulsive sound it is important to first understand the different mechanisms of underwater hearing in an animal as these mechanisms determine how sound is perceived. Species differ in the type of sound they can process (sound wave and particle motion), their sensitivity to frequencies and their auditory thresholds. An audiogram shows the measurements of when the level a sound of a specific frequency is audible (Figure 11). Two methods are commonly used to obtain the needed information.

The first is the use of Auditory Evoked Potentials (AEP) or the Auditory Brainstem Response (ABR). An acoustic stimulus is given to an animal causing a response in the form of electrical pulses showing neural activity within the brain. The electrical pulses can be measured non-invasively by for example attaching electrodes to the animal (Nachtigall 2008). This method has been used extensively on marine vertebrates, in particular fish and marine mammals (e.g. Cook et al. 2006, Houser & Finneran 2006, Ladich & Fay 2013). A small number of studies have also applied this successfully to a few invertebrate species (Hu, et al., 2009; Lovell, et al., 2005; Mooney et al., 2010).

The second method is the psychophysical or behavioural approach. With this technique, the animal is trained to respond to an acoustic stimulus, for example by vocalizing, moving away from a listening station or touching a specific object (Kastelein et al., 2002; Kastelein et al., 2003; Kastelein et al., 2018; Nachtigall et al., 2005). If training is not possible a measurement of changes in heartrate or behaviour has been used (e.g. Offutt 1974, Mccormick & Popper 1984, Yan & Popper 1991). In direct comparison to AEP it is often considered to be more accurate (Houser & Finneran, 2006a; Schlundt et al., 2007; Szymanski et al., 1999). Its main disadvantage is that it can only be applied to animals that can be trained, thus limiting the number of individuals and the species that can be studied (Wolski et al., 2003).

The subsequent response of organisms to sound can manifest on a genetic and cellular level, as well as in physiological and behavioural responses. An important consideration are differences between individual animals, considering parameters such as age, sex or reproductive stage. For some taxa a large number of studies have been conducted providing a fairly good understanding of one or both of these two mechanisms (underwater hearing and the response to sound); for many others this is not the case (Williams et al. 2015).

The following section provides an overview on how invertebrates and vertebrates are able to perceive sound, their vulnerability to sound in general and to impulsive noise specifically. The review is not meant to be comprehensive but aims to showcase the type and quality of information available for the different taxa, in particular as relevant for the North Sea. For more in-depth information readers are referred to the extensive reviews available describing the effects of anthropogenic sounds on marine animals (Carroll et al., 2017; de Soto, 2016; Gomez et al., 2016b; Hawkins et al., 2014; Juanes et al., 2017; Popper & Hastings, 2009; Popper et al., 2003; Popper & Hawkins, 2019; Slabbekoorn et al., 2010; Weilgart, 2018; Weilgart, 2007).

4.5.1 Invertebrates

Invertebrates are generally considered less vulnerable to noise-related trauma than marine mammals and fishes, because they lack gas-filled spaces, such as for example swim bladders (Edmonds et al., 2016). Pressure waves generated by sound can cause rapid motion in any gas-filled structure and result in tissue damage (Hawkins & Popper, 2014). Invertebrates are considered to “only” be vulnerable to particle motion (see chapter 3), in some taxa causing damage to sensory hairs, antennae and statocysts, all structures that have a high biological significance (André et al. 2011, Mooney et al. 2012, Edmonds et al. 2016).

The number of studies that have investigated the effects of impulsive sound sources on invertebrates is limited (reviews available by (Carroll et al. 2017, Weilgart 2018). Information on sensitivity has been shown mostly for molluscs (cephalopods and bivalves) and crustaceans (decapods) (Carroll et al. 2017, Figure 10). **Error! Reference source not found.**Many of these studies are based on small sample sizes or limited to laboratory studies. Below we will highlight examples of bivalves, cephalopods and crustaceans and were applicable focus on North Sea species.

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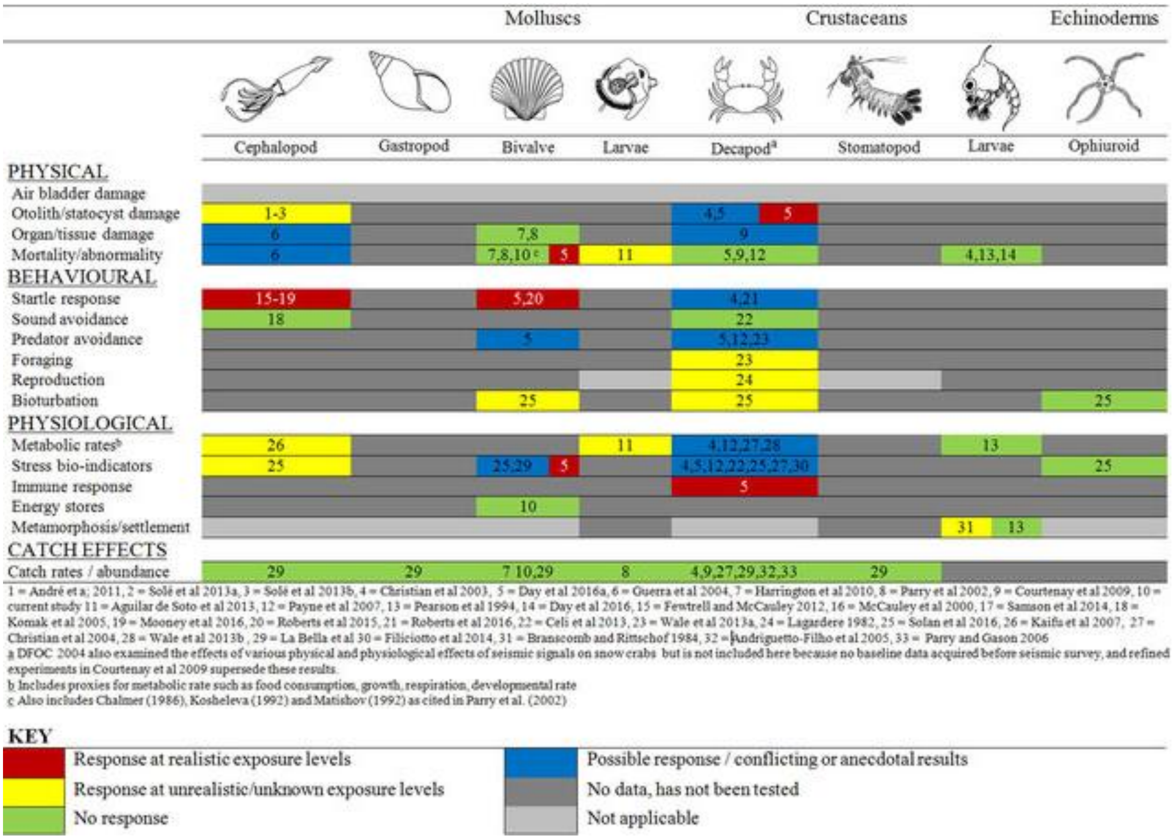

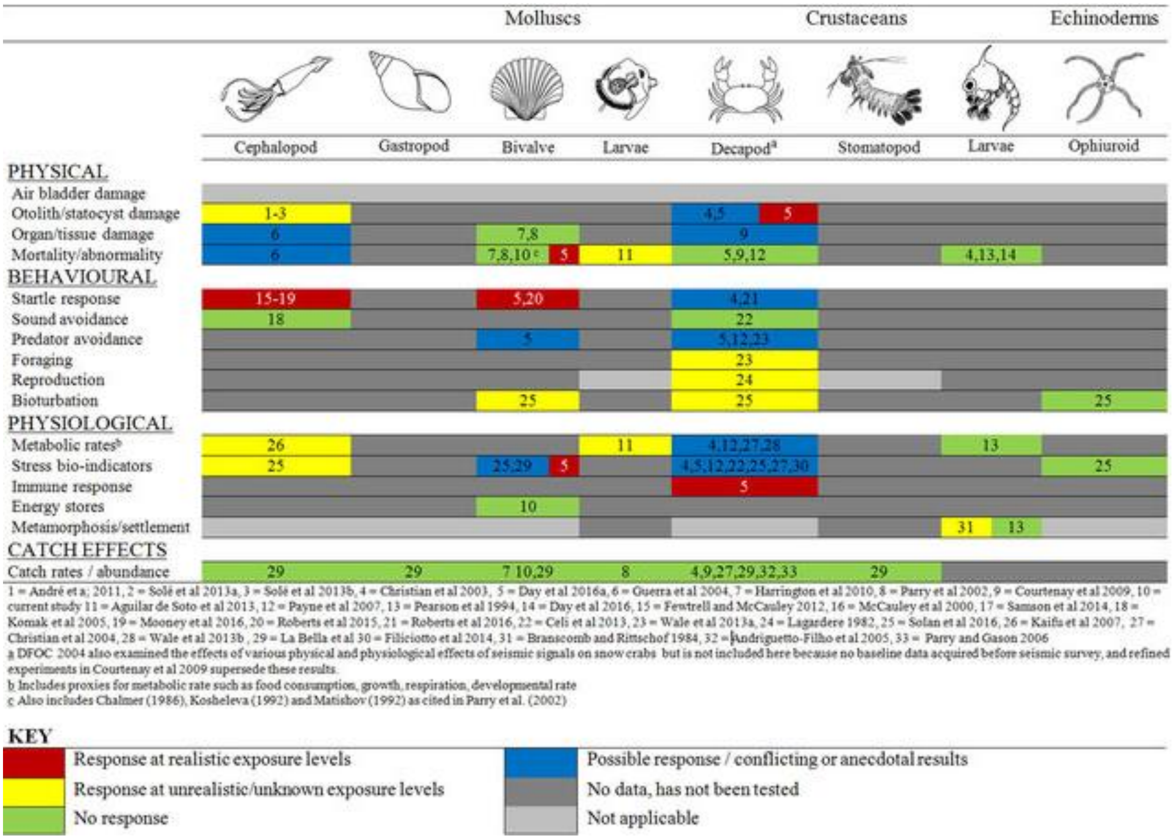

Figure 10: A summary of potential impacts of low-frequency sound on various responses of marine invertebrates. Impacts are classified according to the sound exposure treatments as realistic for seismic surveys or unknown/unrealistic. There are significant differences between seismic studies regarding sound exposure and the environment in which studies were conducted. From: Carroll et al. 2017

4.5.1.1 Zooplankton

Crustaceans and bivalves occur in zooplankton in adult (only crustaceans) as well as larval stages. Some adults will spend their entire lives in the water column, while others larval stages will leave this habitat at some point in their development. Larvae of the New Zealand scallop (*Pecten novaezelandiae*) showed effects to seismic pulses in the form of malformations and delayed developments (De Soto et al. 2013). McCauley et al. (2017) showed that experimental air gun signal exposure decreased zooplankton abundance when compared with controls, with a two- to threefold increase in dead adult and larval zooplankton. Impacts were observed out to the maximum 1.2 km range sampled. The authors hypothesized that the impulsive air gun signal created damage to the sensory hairs or tissue of the zooplankton leading to a loss or reduction in sensory capability that would reduce fitness and increase mortality due to predation. In contrast Parry et al. (2002) found no reduction of catch rate in plankton (including bivalve larvae) as a result of a seismic survey.

Other studies showed no impact of impulsive sounds on crustacean larvae with regard to survival, development and catch rate (Day et al., 2016; Pearson et al., 1994). Different types of sound have shown to impact the timing of settlement of larvae (Lillis, et al., 2013). Behavioural changes due to anthropogenic sound have also been documented (Branscomb & Rittschof 1984).

4.5.1.2 Mollusks

The blue mussel (*Mytulis edulis*) is an example of a filter-feeding sessile benthic bivalve occurring in the intertidal and subtidal waters of the North Sea (Asmus & Asmus 1993). Researchers have applied different methods to investigate behavioural (e.g. valve movement, algal clearance), physiological (oxygen consumption) and biochemical (structural DNA damage, oxidative stress) responses to sound (Wale, 2017; Wale et al., 2019).

Algal clearance rate is defined as the rate at which filter-feeders filter suspended particles from the water and it is considered a reliable indicator of feeding activity in mussels. Using a semi-open field experiment, (Spiga et al., 2016) found that mussels had significantly higher clearance rates during pile driving. The authors suggest that this could indicate a physiological change from a maintenance state to active metabolism due to noise stress. Investigating the effect of playback ship noise on the blue mussel (Wale et al. 2019) showed a different reaction with a reduced algal clearance rate. At the same time oxidative stress increased, as well as changes in DNA integrity. This can potentially reduce growth, reproduction and immune response (Wale et al. 2019).

Reactions of non-sessile bivalve species such as scallops (*Pecten* sp.) also included physiological and behavioural changes (Day et al., 2017).

The most highly developed mollusks are cephalopods, such as squid or octopi. Cephalopods can hear sound (Hu et al. 2009) and are sensitive to the particle motion component of it (McCauley & Fewtrell, 2008; Mooney et al., 2012; Solé et al., 2013). There is a number of studies demonstrating that cephalopod species such as *Sepia officinalis*, *Octopus vulgaris*, *Loligo vulgaris* and *Illex condietii*, as well as giant squid (Architeuthis) are susceptible to acoustic trauma (André et al., 2011; de Soto, 2016; Guerra et al., 2011; Solé et al., 2013).

4.5.1.3 Crustaceans

As for other invertebrates, limited data are available on the effect of impulsive noise on benthic crustaceans in general (Budelmann 1992, Weilgart 2018) and in particular for North Sea species. Edmonds et al. (2016) investigated the sensitivity of the edible crab (*Cancer pagurus*), the European lobster (*Homarus gammarus*) and Norway lobster (*Nephrops norvegicus*) to sound. The results indicated that physiological sensitivity could be found only for the particle motion effects of sound production. However, the authors conclude that due to the limited information available, no relationship between impulsive noise and effects on crustaceans can be found yet. The hermit crab (*Pagurus berhnardus*) did show a behavioural response to particle motion in controlled laboratory sound experiments (Roberts et al., 2016). A number of studies showed that *Crangon crangon* can react to noise with metabolic changes (Regnault & Lagardere 1983, Slabbekoorn et al. 2010).

Morris et al. (2018) investigated the impact of seismic survey activity on the catch rates of snow crab (*Chionoecetes opilio*). Their results did not show a negative impact on catch rates, but the authors suggest that effects might be too small to detect considering the high natural spatial and temporal variation in crab occurrence. Other studies were inconclusive in their results, possibly due to limitations in their study design (Boudreu et al., 2009; DFO Science, 2004). Morris et al. (2018) concluded that seismic activity did not negatively affect catch rates of snow crab in shorter term (i.e. within days) or longer time frames (weeks).

Additional studies showed that benthic crustaceans exposed to ship noise change their foraging and anti-predator behaviour, increase their oxygen consumption and affect their immune response (Celi et al., 2015; Wale et al., 2013b, 2013a).

4.5.2 Vertebrates

4.5.2.1 Fish

There are more than 33.000 species of fish, representing over half of all vertebrate species. In this section we will consider the largest and most diverse group: bony fish.

Fish eggs and larvae
Effects of impulsive sounds (by quantifying mortality) have been studied for larvae of different marine species. The studies showed different results, from no increased mortality to sublethal and lethal effects, including increased incidence of abnormalities and reduced growth (Bolle et al., 2012; Carroll et al., 2017; Day et al., 2016; De Soto et al., 2013; McCauley et al., 2017; Morgan et al., 1999; Nedelec et al., 2015).

Adult fish
Fish have developed a high diversity of morphological adaptations to perceive sound, which are not necessarily linked to specific taxonomic groups. The high interspecific variability in ear structure makes it especially challenging to classify their hearing capabilities, which led to their categorization as either being hearing generalists or hearing specialists (Popper et al., 2003; Popper & Hawkins, 2019).

Hearing generalists are typically fish species with no or a reduced swim bladder (such as flatfish) (Figure 11a). They tend to have a low auditory sensitivity detecting sound up to about 1kHz. This sensitivity is mainly related to the particle motion component of the sound field (Fay & Popper, 2012; Popper & Fay, 1993).

Hearing specialists are fish with fully functional swim bladders (Figure 11b), in some taxa connected to the inner ear by an extension of the swim bladder (bullae) (Figure 11c). They can also detect particle motion, but their hearing is more sensitive with generally lower hearing thresholds and a broader hearing range of sounds above 1.5kHz. They are also considered to be more susceptible than hearing generalists to injury due to impulsive sound (Fay & Popper, 2012; Popper & Hastings, 2009; Popper et al., 1993; Popper et al., 2019).

Species sensitive to sound pressure also respond to sounds over a wider frequency range than less sensitive species (*Figure 11*) but depending on their anatomy their sensitivity for various frequencies varies greatly between species.

Defining hearing ability in fish is complex due to the different adaptations for pressure and particle motion and how these systems respond to sound (Radford, et al., 2012; Wahlberg & Westerberg, 2005). In principle both components should be considered when quantifying fish sensitivity to sound, this is, however, extremely challenging. In addition, the high variability between species and individuals and the inherent difficulties in studying fish behaviour make it difficult to extrapolate results from laboratories to the field and from one species to another (Slabbekoorn 2016). For more detailed information on this subject we refer the reader to recent reviews of the current knowledge on the impact of sound on fish (Popper & Hawkins, 2019; Popper et al., 2019).

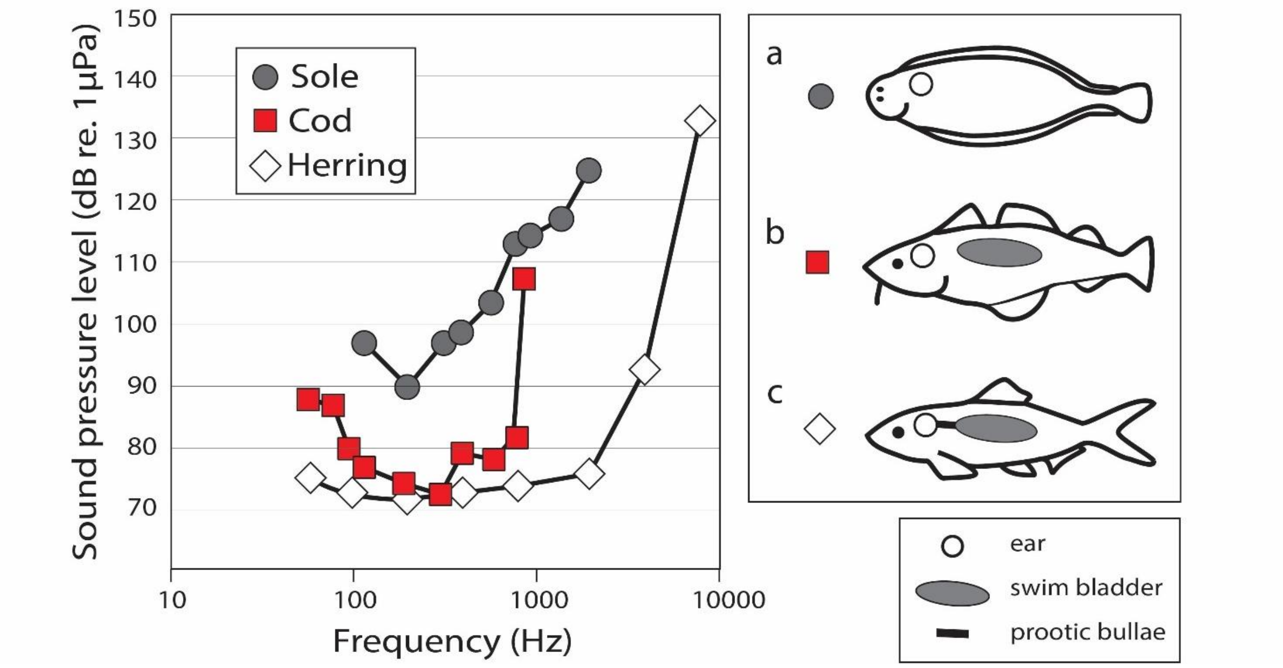

Figure 11: Overview hearing ability of sole, cod and herring and a schematic representation of the hearing systems. The sound pressure level (SPL) indicates the auditory sensitivity for a specific frequency, higher SPL indicating lower sensitivity. Sole has no swim bladder as an adult (a), cod has a swim bladder that is not connected to the ear (b) and herring has a connection between the swim bladder and the ear (c) (adapted from Dooling & Therrien 2012).

A number of studies have investigated how sound affects fish, both using laboratory and field studies. In the following section we showcase some examples of North Sea species for which data are available.

An example of a benthic fish species of the North Sea is the sole (*Solea solea*). Soles have no swim bladder in their adult stage. One study looked at behavioural responses of sole to playback of impulsive sound (piling) finding high individual variability in responses with a general increase in swimming speed after multiple sound exposures (Mueller-Blenkle et al. 2010). The authors point out that this increase in speed is not the type of behavioural response that is typically used by sole, as they are likely to hide in the sediment and stop moving when startled.

The European seabass (*Dicentrarchus labrax* L.) is a slow growing fish that undertakes seasonal migrations from inshore habitats to offshore spawning sites. In the North Sea it is targeted by commercial and recreational fisheries (De Pontual et al., 2019). The seabass has been subject of a number of studies on the impact of impulsive sound, primarily under captive or semi-captive conditions. These studies showed behavioural changes, such as increased swimming depth and changes in group cohesion, as well as physiological changes such as biochemical stress responses (Debusschere et al., 2016; Hubert et al., 2020; Kastelein et al., 2017; Neo et al., 2014; Santulli et al., 1999).

Like the seabass, the Atlantic cod has a fully functioning swim bladder. The cod produces different sounds, such as “grunts”, that play a role during mating (Rowe & Hutchings, 2008; Wilson et al., 2014). Several studies found a reduction in catch rates of cod due to seismic surveys (Engås et al., 1996; Løkkeberg et al., 2012b, 2012a). It was not investigated whether the decrease was caused by animals dying, changing depth or moving out of the area. An exposure to between 1 and 5 hours of pure tones (50 Hz to 400 Hz at 180 dB re 1 µPa) has led to damage to the sensory epithelium of cod (Enger 1981). In addition, cod showed behavioural responses, such as aggregation and changing swimming speed, to playback pile-driving noises (Mueller-Blenkle et al. 2010). There are also data indicating a stress response of cod leading to a disturbance of reproductive success (Sierra-Flores et al., 2015).

Clupeids such as the Atlantic herring (*Clupea harengus*) also have a swim bladder. However, a special feature is that they also have a prootic bulla (a gas-containing sphere) that physically links the swim bladder to the hearing system (Blaxter et al., 1981; figure 10c). Fish that have this connection are thought to have improved hearing, indicated by an increase of the frequency range and lowering of their hearing thresholds, which has been confirmed by audiograms showing a higher sensitivity to higher frequencies than cod and salmon (Popper et al., 2019; figure 10). Some studies, however, contradict the categorization of the Atlantic herring as a typical hearing specialist. In fact they seem to be less sensitive to sound than hearing generalists, but with a wider bandwidth than specialists (Mann et al., 2001). Studies on how herring responds to impulsive sound are limited. Caged herring have been exposed to low-frequency naval sonar (1.0 – 1.5 kHz) and no significant escape reactions were noted (Doksæter et al., 2012; Doksæter et al., 2012b). During spring spawning herring has been observed to dive to greater depths during the use of seismic sonar, and return to their original water depth when the sounds stopped (Slotte et al., 2004). An experiment was done to see if pingers reduce the catch rate of herring in gill nets (Culik et al., 2001). No change was found for high frequency sounds above 20kHz, but an increase in catch rate was found for 2.7 to 19 kHz for gill nets, indicating that these pingers attracted herring.

Sound is also an important aspect of herring communication, possibly to ensure group coherence during schooling (Wilson et al., 2004). As such they are likely also vulnerable to masking of their communication sounds.

4.5.2.2 Pinnipeds

As amphibious animals, pinnipeds have adapted to hearing both in air and under water. Their hearing sensitivity has been found to rival fully aquatic mammals such as cetaceans, as well as terrestrial carnivores (Reichmuth et al., 2013).

The two species occurring regularly in the North Sea are the grey seal (*Halichoerus grypus*) and the harbour seal (*Phoca vitulina*). Both belong to the group of ‘true seals’ or phocids, with one of their anatomical characteristics being the lack of outer ear structures. They use vocalisations primarily for communication, often in the context of territorial and reproductive behaviour (Schusterman & Van Parijs, 2005; Van Parijs et al., 2003).

The morphological adaptations in their ear structure allows them to expand their frequency range of hearing in water. Phocids not only have the broadest hearing range of all pinnipeds, but likely of most marine mammals (Kastelein et al., 2009; Reichmuth et al., 2013; Southall et al., 2000).

Both harbour and grey seals have shown reactions to impulsive sounds. Many studies investigated the effectiveness of so-called “seal scarers” or Acoustic Deterrent Devices (ADDs) used to prevent seals from approaching fishnets or fish farms or to deter them from activities that produce potentially harmful sound (Kastelein et al., 2017; Northridge et al., 2013). The effects of ADDs on deterring seals vary depending on the devices, the methods used and the sound types and levels produced (Mikkelsen et al., 2017), as well as the distance to haul-out sites (Harris, 2011; Janik & Götz, 2013) and potentially habituation (Mikkelsen et al. 2017).

At sea data from tagged harbour and grey seals have been used to investigate individual behavioural responses to piling noise. Harbour seals tagged in the Greater Wash wind farm area in the UK were found to stay away at distances from around 5 to 40 km when piling was conducted (Hastie et al. 2015a). In the same area Russell et al. (2016) showed that during piling seals were displaced up to 25 km, with a recovery time of two hours. Kirkwood et al. (2014) tracked 20 grey seals for 73 to 114 days during the construction of the windfarm Luchterduinen in the Dutch North Sea and 7 animals were in the area when piling occurred. Of 85 tracks that could be analysed, 64% indicated no change in movement direction, 22% showed potential aversive reactions and 6% showed a potential approach towards the piling site.

In addition there have been numerous studies of harbour seals in captivity demonstrating physiological and behavioural reactions to a range of different types of impulsive sound (e.g. Terhune 1988, Turnbull 1994, Kastelein et al. 2006, 2012).

4.5.2.3 Cetaceans

Cetaceans possess sensory abilities adapted to a life spent entirely in water. The ability to produce as well as perceive sound is necessary for all cetaceans to communicate, navigate, avoid predators and forage (Tyack 1998). Cetaceans are split up into baleen whales (mysticeti) and toothed whales (odontoceti). Most cetaceans occurring in the North Sea are toothed whales, such as harbour porpoise (*Phocoena phocoena*), white-beaked dolphin (*Lagenorhynchus albirostris*), Risso's dolphin (*Grampus griseus*) and bottlenose dolphin (*Tursiops truncatus*). Humpback whales (*Megaptera novaeangliae*) are visiting the North Sea at times, but the most common baleen whale is the common minke whale (*Balaenoptera acutorostrata*).

Baleen whales (mysticeti)
The hearing capability of baleen whales has not been measured yet, as behavioural audiograms cannot be obtained and determining the auditory brainstem response to date has been unsuccessful (Ridgway & Carder 2001). Current knowledge is obtained from anatomical studies, predictive models, vocalizations, taxonomy and behavioural responses to sound (e.g. Cranford & Krysl 2015, Southall et al. 2019b, Risch et al. 2019) indicates that baleen whales hearing ranges from 5–20 Hz to 20–30 kHz, with likely the best ability at detecting low frequencies below 1kHz (Weilgart 2007).

Most studies on the impact of impulsive sound on baleen whales have shown changes in their behaviour, in particular their communication and distribution (Castellote et al., 2012; Di Iorio & Clark, 2010; Dunlop et al., 2016, 2017; Kavanagh et al., 2019; Parks et al., 2010).

The minke whale is the most common baleen whale in the North Sea. It is most widespread in the summer season, when it's distribution stretches to the south of the Dogger bank (Hammond et al. 2002, 2013, De Boer 2010, Gilles et al. 2011, Geelhoed et al. 2014). A recent PAM study along the Scottish east coast showed May to November, with highest detection rates in June-July (Risch et al. 2019).

As with other baleen whales information on minke whale hearing capabilities is very limited and predictions of sensitivity to noise have been modelled or inferred from species-specific vocalizations, behavioural responses to sounds or anatomical measurements (Mellinger et al., 2000; Tubelli et al., 2012; Yamato, 2012). As direct measurements have not been possible, simulations have been conducted of the minke whale auditory pathway suggesting that the best frequency range between approximately 30 and 25,000 Hz (Tubelli et al. 2012).

Regarding impulsive sound effects, minke whale densities off Scotland have been observed to decrease during naval exercises, and mortalities of minke whales occurred during at least two mass stranding events that have been linked to military sonar exercises (Parsons 2017). Minke whales have been observed to respond to mid-frequency active sonar by reducing or ceasing calling and by exhibiting avoidance behaviours (e.g., movement away from the sound source) even at relatively low received sound levels (Kvadsheim et al., 2017; Sivle et al., 2015). Also, results of a behavioural response study showed minke whales strongly avoiding ADDs that were used as noise mitigation of construction activities (McGarry et al., 2017). Low frequency active sonar playback during the 3S2 project induced a strong avoidance behaviour by a minke whale, including a cessation of feeding behaviour, change of dive pattern, increase in swimming speed and an avoidance of the sound source (Sivle et al., 2015).

Toothed whales (odontoceti)
In contrast to baleen whales much more data are available on the hearing ability of toothed whales. This is due to the fact that a number of smaller toothed whale species have been held in captivity providing the opportunity to investigate their hearing capabilities, using both behavioural studies and measuring Auditory Evoked potentials. Most research has focused on two species, harbour porpoise and bottlenose dolphin, and even though sample size can be considered small, their audiograms are well documented (e.g. Kastelein et al. 2002a, Houser & Finneran 2006b, Popov et al. 2007, Houser et al. 2008, Finneran et al. 2008, Ruser et al. 2016). The vulnerability of toothed whales to impulsive sound is often derived through experiments in captivity. This allows the testing of TTSs as well as behavioural changes to noise.

Information on changes in behaviour and occurrence can also be obtained at sea. For example, by observation and monitoring during impulsive noise emissions, or by directly measuring behaviour and the sound that is received by the animal(s) in tagged individuals. Again, a large number of scientific studies document that toothed whales will negatively react to a range of impulsive sound sources; construction work at sea, such as pile driving during the building of offshore windfarms, the use of ADDs, detonation of ordnance at sea, naval exercises and seismic surveys (Finneran et al., 2000; Gordon et al., 2003; . Kastelein et al., 2012; Lucke et al., 2009; Miller et al., 2012; Pirotta et al., 2014; Von Benda-Beckmann et al., 2015).

Harbour porpoise is the most common toothed whale in the North Sea and has shown behavioural reactions to impulsive sound. The construction of wind farms have provided evidence of a reduction of porpoises in the area compared to the baseline, likely due to a change in distribution (e.g. Tougaard et al. 2003, 2009, Madsen et al. 2006b, Dähne et al. 2013). Porpoises have also shown to negatively react to the sounds of ADDs (Johnston, 2002; Mikkelsen et al., 2017; Olesiuk et al., 2002). Foraging rates have gone down in reaction to seismic surveys (Bergès et al., 2020; Pirotta et al., 2014).

An additional, indirect source of information is the investigation of stranded animals. Naval sonar has been linked to stranding events of a number of species, including beaked whales, long-finned pilot whales (*Globicephala melas*), dwarf and pygmy sperm whales (*Kogia breviceps* and *K. sima*) and melon-headed whales (*Peponocephala electra*) (Weilgart, 2017). Pathological investigations have shown acute and chronic tissue damage due to rapid decompression, indicating that animals might have suffered decompression sickness (Jepson et al. 2003).

4.5.2.4 Birds

The time spent underwater differs per feeding guild. Pursuit divers, which follow their prey underwater (e.g. greves, divers, auks) and benthos feeders (e.g ducks) spend more of their time underwater than species that make shallow plunges to catch their prey in the upper sea layer (e.g terns). The northern gannet has an intermediate feeding technique. This large seabird uses plunge dives to hunt for herring (*Clupea harengus*), mackerel (*Scomber scombrus*) and sandeels (*Ammodytes* spp.), reaching a depth of 30m. Gannets use underwater flight to follow and catch prey (Brierley & Fernandes, 2001; Garthe et al., 1999). At this point there is only very limited information available on a few individuals. The underwater hearing abilities and vulnerabilities of most marine birds is still unknown.

Bird sensitivity to noise in air, including impulsive sound, has been demonstrated for a broad spectrum of species under many different circumstances, with a focus on terrestrial species (e.g. Stone & Rubel 2000, Dooling & Popper 2016). However, studies on the effects of underwater sound on diving birds are limited to a few taxa. Penguins are extremely well adapted to life in the marine environment with their excellent ability to swim and dive. Several studies have shown the hearing capability of different penguin species and the impact of impulsive sound on their behaviour (Wever et al. 1969, Woehler 2002, Pichegru et al. 2017, Sørensen et al. 2020).

However, penguins don't occur in the North Sea. Data on the hearing ability of other diving birds, such as eider ducks, the common guillemot, the Atlantic puffin (*Fratercula arctica*), the great cormorant, the red-throated diver (*Gavia stellata*) and the northern gannet are available (Aran Mooney et al., 2019; Crowell et al., 2015; Hansen et al., 2017; Johansen et al., 2016). The great cormorant was found to have capabilities similar to marine mammals, with a hearing threshold in the frequency band of 1-4 kHz and the greatest hearing sensitivity at 2kHz with an underwater hearing threshold of 71 db re 1µPa rms and (Johansen et al. 2016, Hansen et al. 2017).

4.6 Species Selection step 3. Data availability and suitability

The review of North Sea habitats and their components gave a first overview of potential indicator species. Table 2 shows an overview of the main (Dutch) North Sea phyla and the data availability on their vulnerability to impulsive sound. It highlights the large data gap for invertebrates with limited data on specific species, such as the blue mussel or the edible crab, which is based on very few studies with at times contradictory results. Even though invertebrates form the basis of the North Sea food web the current lack of data does not allow us to choose one of their representatives for the assessment.

Table 2: Overview of data availability for the most common phyla (adapted from Bos et al. 2016) in the Dutch North Sea regarding vulnerability to sound.

(Sub) Phylum	common name(s) of selected taxa	Current data availability on vulnerability to (impulsive) sound
Porifera	Sponges	None
Coelenterates	Ctenophora & cnidaria; comb jellies and jelly fish	Limited (e.g. Solé et al. 2016)
Platyhelminthes	Flat worms	None
Entoprocta	Sessile filter-feeders	None
Bryozoa	Moss animals	None
Phoronida	Horseshoe worms	None
Nemertea	Ribbon worms	None
Annelida	Ringed worms	None
Mollusca	Mussels, snails, cephalopods (squid, octopus)	Limited
Cephalorhyncha	Cephalorhyncha	None
Nematoda	Nematodes	None
Arthropoda	crustaceans (krill, crabs), horseshoe crabs, spider crabs	Limited
Echinodermata	star fish, sea urchins, sea cucumber, sea lillies	Limited
Enteropneusta	Acorn worms	None
Tunicata	Sea squirts, salps	limited (e.g. McDonald et al. 2014)
Cephalochordata	Lancelet	None
Vertebrata	Fish (bony fish; sharks & rays), mammals (cetaceans, pinnipeds), reptiles (turtles), birds	Yes

Data availability on the vulnerability to (impulsive or other) sound is clearly biased towards vertebrates with most of the studies in this group focussed on bony fish and marine mammals.

For fish most research on hearing ability and effect of sound has been done in captivity, often with freshwater species. There are only four North Sea species for which a reasonable amount of information on their hearing, as well as how they react to sound is available: sole, Atlantic herring, European seabass, and the Atlantic cod. As described earlier, the data indicate that effect of sound waves is probably limited for sole due to the lack of a swim bladder in the adult stage. Clupeids in general and herring in particular have well developed hearing capabilities. However, research on if and to what degree impulsive sound affects them is limited and the results are inconclusive. The European seabass has been investigated primarily in semi-captive or captive conditions, while the Atlantic cod has been primarily investigated in the field. Both species show reaction to impulsive sounds, so from that aspect both fulfil the first two criteria (acoustic sensitivity and vulnerability to sound) to be a suitable indicator species.

Distribution data on sole, Atlantic cod and Atlantic herring is based on designated surveys covering the North Sea twice a year (IBTS – International Bottom Trawl Surveys) and in addition coordinated acoustic surveys (HERAS) are conducted for herring. ICES uses the results to provide fishery management advice for these commercially important species. All three species are widely distributed in the North Sea and meet the criterium of spatio-temporal distribution to be a suitable indicator species.

ICES provides advice on the management of seabass, even though it is not subject for a quota of total allowable catches (TAC). Unfortunately data quality is poor as it relies on landings data and the number of animals taken in recreational fishery and discarded is not well documented (ICES 2019). The species occurs in comparably low densities making it less suitable as an indicator species.

The other vertebrate group that has comparatively well studied representatives are the marine mammals. Two pinniped species occur in the North Sea regularly – the harbour seal and grey seal. Most studies on underwater hearing sensitivity and vulnerability to impulsive sound have been conducted on harbour seals in captivity, whilst grey seals are less well studied. Harbour seal fulfils the first two criteria (acoustic sensitivity and vulnerability to sound) to be a suitable indicator species.

Spatio-temporal distribution information can be obtained by using the tracks of tagged individual seals and applying spatial modelling. This has been done for both species, however, the analyses have currently not included all available data from the North Sea. Harbour seal distribution and habitat use has been modelled for Dutch waters and the decision was made to investigate how and if this could be used for the assessment. Grey seals are known to conduct foraging trips covering the offshore waters off the North Sea, whereas the harbour seal stays closer to the haul out sites. Thus, both species potentially fulfil the criterium on spatio-temporal distribution.

Harbour and grey seals are covered in several international conservation agreements. They are listed in both Annex II and V and in the European Habitats Council Directive (92/43/EEC). In the UK seals are protected under the Conservation of Seals Act 1970, The Marine (Scotland) Act 2010 and in the Wadden Sea area they are under the management of the Trilateral Wadden Sea agreement. The Marine (Scotland) Act 2010 introduced Seal Conservation Areas as a conservation measure for vulnerable local populations of harbour seals.

There are a number of cetacean species that occur regularly in the North Sea. For all but bottlenose dolphins and harbour porpoises information on the effect of impulsive sound is poor and is often inferred from what is known from other species. The acoustic sensitivity of bottlenose dolphins and harbour porpoises have been studied in captivity, whilst some studies on their reaction to impulsive sound have been conducted at sea. Both species therefore meet the first two criteria for a suitable indicator species.

Bottlenose dolphins have been studied very intensely in those areas where they occur coastally as resident populations (OSPAR 2017). Using photo-identification individuals can be tracked over a life time and the size and distribution of these populations can be assessed (Wilson et al., 2004). While this is in principle a great prerequisite for an indicator species, the very patchy distribution makes them only suitable for investigating local impacts. Sightings offshore are rare and data on possible offshore populations in the North Sea is poor. Harbour porpoises are abundant and widely distributed over the North Sea. Information on distribution and abundance is available due to dedicated survey efforts over the last decades. So, bottlenose dolphin does not meet the criterium on spatio-temporal distribution for a suitable IS, whereas harbour porpoise meets this criterium.

The harbour porpoise is a protected species under European legislation (i.e. the EU's Habitat Directive) in Annex II and Annex IV. The Netherlands has signed international (North Sea Conservation Plan ASCOBANS) and national agreements on the protection of the harbour porpoise (Dutch national porpoise conservation plan (Camphuysen & Siemensma 2011).

4.7 Species Selection step 4. Selection of three indicator species

A final overview of the criteria for the different species considered as an indicator species is shown in figure 3. All considered North Sea vertebrates are listed as priority species for conservation or are subject to management due to exploitation. Data availability and data suitability on acoustic sensitivity and vulnerability to sound limits the number of potential IS. Availability and suitability of data on the spatio-temporal distribution narrows this down to three indicator species for the assessment: **Atlantic cod, harbour seal and harbour porpoise** (Table 3).

Table 3: Overview of the data availability (DA) and suitability (DS) for the assessment for suitable indicator species/taxa (bony fish and marine mammals) for impulsive sound in the North Sea (NS). Selected IS are shown in bold and green. IBTS: International Bottom Trawl Survey, HERAS: coordinated acoustic surveys for herring. Green: data available and suitable; yellow: limited data available or limited suitability; red: no data available.

Species	DA/DS	Vulnerability	Spatio-temporal distribution	Conservation and/or management frameworks
Sea bass	DA	Laboratory and semi-captivity	no distribution maps from IBTS survey data	ICES advice
	DS	Behavioural changes, biochemical stress response	Low density in the NS	
Sole	DA	semi-captivity	IBTS survey data	ICES advice
	DS	inconclusive behavioural reaction	widely distributed in the southern and eastern NS	
Atlantic cod	DA	laboratory and field research	IBTS survey data	ICES advice
	DS	Behavioural changes; biochemical stress response	widely distributed in the NS	
Atlantic herring	DA	laboratory and field research	IBTS survey data and ICES – HERAS surveys	ICES advice
	DS	Behavioural changes	Widely distributed in the NS, linked to spawning grounds	
Grey seal	DA	field research	Partial distribution for NS based on tagging, seasonal data	National and international agreements
	DS	Changes in behaviour at sea	Widely distributed in the NS	
Harbour seal	DA	Laboratory and field research	Partial distribution for NS, seasonal data	National and international agreements
	DS	Behavioural changes; physiological changes	Distributed primarily in coastal areas	
Minke whale	DA	Anatomical inference; field research	Survey data, seasonal data	National and international agreements
	DS	Behavioural changes, increased mortality	low density; likely seasonal occurrence in offshore waters	
White-beaked dolphin	DA	No direct measurements	Survey data, seasonal data	National and international agreements
	DS	Unknown	low density in the North Sea	
Bottlenose dolphin	DA	Laboratory and field	Survey data and photo-identification, seasonal data	National and international agreements
	DS	Behavioural changes, physiological changes	low density in the North Sea; coastal small resident populations	
Harbour porpoise	DA	Laboratory and field	Survey data, seasonal data	National and international agreements
	DS	Behavioural changes, physiological changes	widely distributed in the North Sea	

5 STEP 2: DEFINE ASSESSMENT AREA

5.1 Assessment method

In the previous chapters step 0: define stressor and step 1: select indicator species were completed. The identified stressor is: impulsive noise. The selected indicator species are harbour porpoise, Atlantic cod (cod) and harbour seal. In this chapter the assessment area is defined (Figure 12). This step was taken simultaneously with step 1 and 3 that will be discussed in the next chapters.

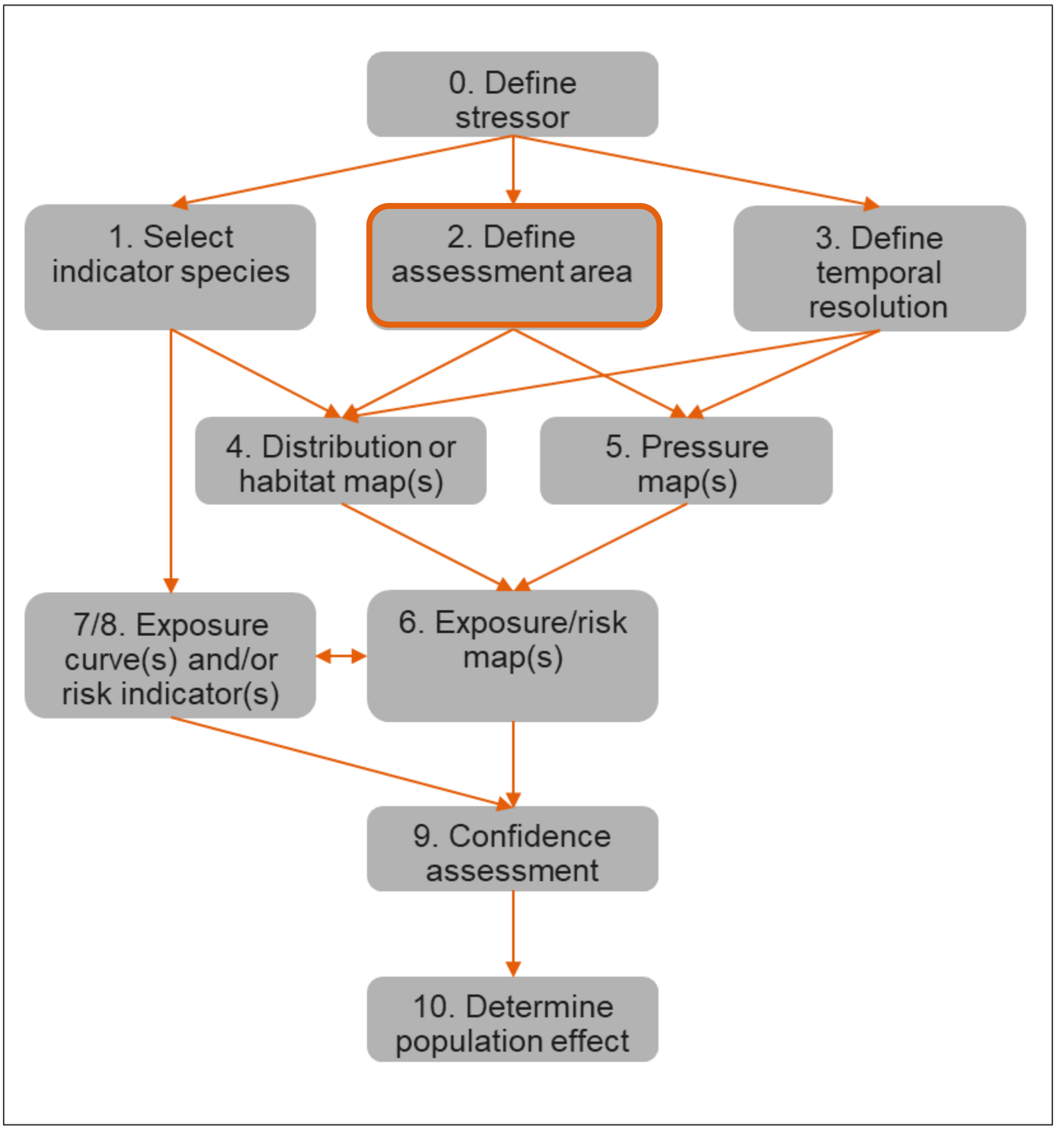

Figure 12: Stepwise approach, current step (highlighted orange).

5.2 Assessment area

The assessment area is primarily determined by the obligations to be fulfilled with the assessment. The assessment can be performed for the MSFD reporting on art. 8 (due in 2024) or as input to the OSPAR Quality Status Report (due in 2023). It has been agreed that the assessments made for the QSR2023 will be suited to be used for the MSFD assessment as well.

In these cases, the underwater noise (as the Netherlands are concerned) is usually assessed for OSPAR area II, the Greater North Sea. This area approximately covers the ICES areas Northern North Sea (IVa), Central North Sea (IVb), Southern North Sea (IVc), the Skagerrak and Kattegat (IIIa) and the Eastern Channel (VIId).

To determine the assessment area several other factors were taken into account:

- The report was commissioned by the Dutch government and the area therefore had to at least include the Dutch territorial waters.
- Geographically the area has to be limited to match noise distribution.
  - Norway (currently) does not contribute to the impulse noise register which means that the northern North Sea waters are missing data.
  - The Dutch waters are likely to be affected by or affect the surrounding Belgian, English, German and Danish waters.
- From an ecological perspective all systems are connected. But to find data and create maps a balance has to be found between:
  - Realistic ecological connectivity
  - Practical limitations to the assessment areas

All these factors combined, and particularly the lack of data around Norway, led to the following assessment area which includes the ICES area southern North Sea and most of the central North Sea:

Figure 13: The selected assessment area

6 STEP 3: DEFINE TEMPORAL RESOLUTION

6.1 Assessment method

In the previous chapters step 0: define stressor, step 1: select indicator species and step 2: define assessment area were completed. The identified stressor is: impulsive noise. The selected indicator species are harbour porpoise, cod and harbour seal. The assessment area is an extension of the Dutch North Sea. In this chapter step 3: Define temporal resolution (Figure 14) is described. This step was taken simultaneously with step 1 and 2 that were discussed in the previous chapters.

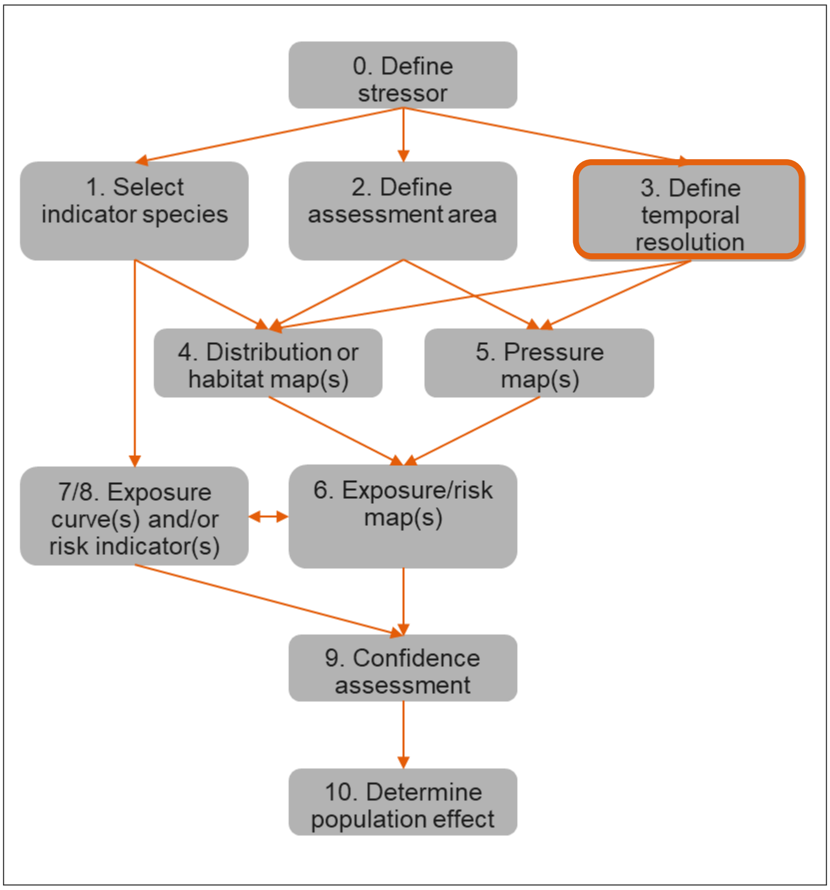

Figure 14: Stepwise approach, current step (highlighted orange).

6.2 Temporal resolution

The MSFD has a reporting period of 6 years. Not only the actual environmental status at a certain moment must be assessed, but also the trend over the reporting period (6 years).

To make a proper analysis, temporal variations in the input parameters must be accounted for as far as they are ecologically relevant.

- The distribution of species shows seasonal variations based on reproductive behaviour and food availability.
- For the activities that produce impulsive noise data are available with a resolution of one day for the years 2015-2017 (2018 data came available during this project and have not been used). These data show typically a large year to year variation.

Currently (at the start of the project in 2019) reports are being published for the period 2012 – 2017. In the impulse noise register data from 2015, 2016 and 2017 have been registered. Disturbance data are available per day. In this report all given examples are based on the sound distribution data from 2015. The temporal resolution might be further defined depending on the distribution data found in step 4, chapter 7.

7 STEP 4: DISTRIBUTION OR HABITAT MAPS

In the previous chapters step 0: define stressor, step 1: select indicator species, step 2: define assessment area and step 3: define temporal resolution were completed. The identified stressor is: impulsive noise. The selected indicator species are harbour porpoise, cod and harbour seal. The assessment area is an extension of the Dutch North Sea and the temporal resolution is defined as 2015 to 2017. In this chapter step 4: Distribution or habitat map(s) is described (Figure 15). This step was taken simultaneously with step 5 that will be discussed in the next chapter. Distribution and habitat is discussed per indicator species in the three paragraphs of this chapter.

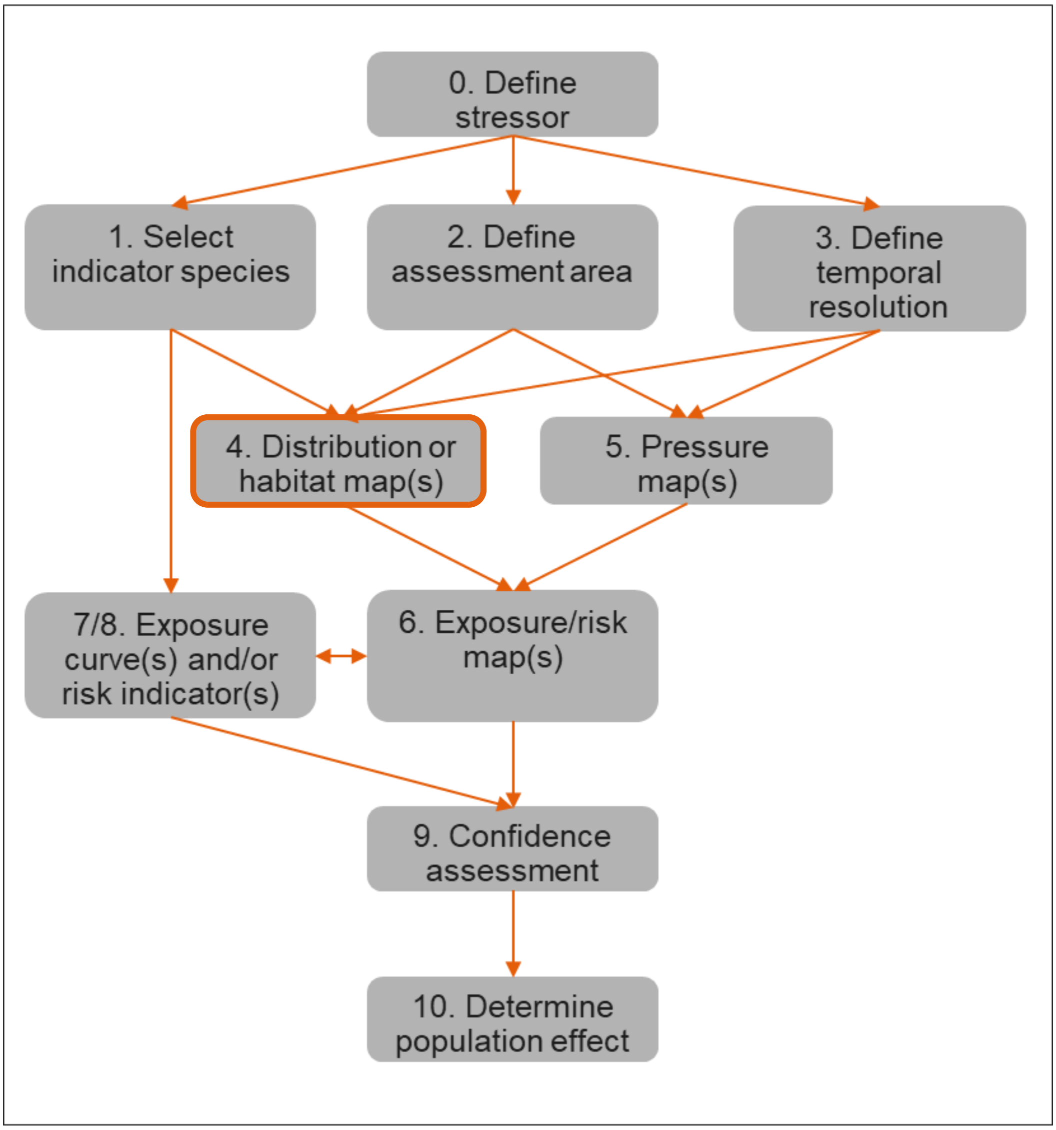

Figure 15: Stepwise approach, current step (highlighted orange).

7.1 Cod distribution

Data on the presence of cod in the North Sea is available from the international monitoring program called the North Sea International Bottom Trawl Survey (NS-IBTS). This monitoring program is a collaboration between eight countries (Denmark, France, Germany, Netherlands, Norway, UK Scotland, UK England and Sweden). Data from IBTS survey's (and comparable predecessors) has been collected since the first quarter of the 1960's. The data are collected by research vessels using a bottom trawl net. Methodology of the surveys is the same for each research vessel from the different countries. An in-depth explanation of the methodology can be found in the survey protocol (The International Bottom Trawl Survey Working Group 2015). The data on cod distribution are collected as number of individuals caught per unit effort (CPUE, Catch Per Unit Effort in n/hour). Data are available from the first and third quarter of each year and for 2015, 2016 and 2017. The data are collected in a raster with a resolution of 10.000 km^2.

As the data are gathered as a number per unit effort this makes it unsuitable for later exposure analysis. To make the data suitable for later analysis it is displayed as a percentage of the total population in the assessment area. Displaying the data like this requires to assumptions: it assumes an identical catch protocol and equal catch effort for every ICES quadrant over the relevant season. In addition, it is assumed that the amount of Cod present per quadrant is distributed evenly over the quadrant. Making these assumptions and displaying the data in this way makes it suitable for the exposure analysis and creation of exposure risk maps in chapter 9. Figure 16 and Figure 17 show cod distribution per ICES quadrant in 2015 for autumn and spring as a percentage of the total population in the study area.

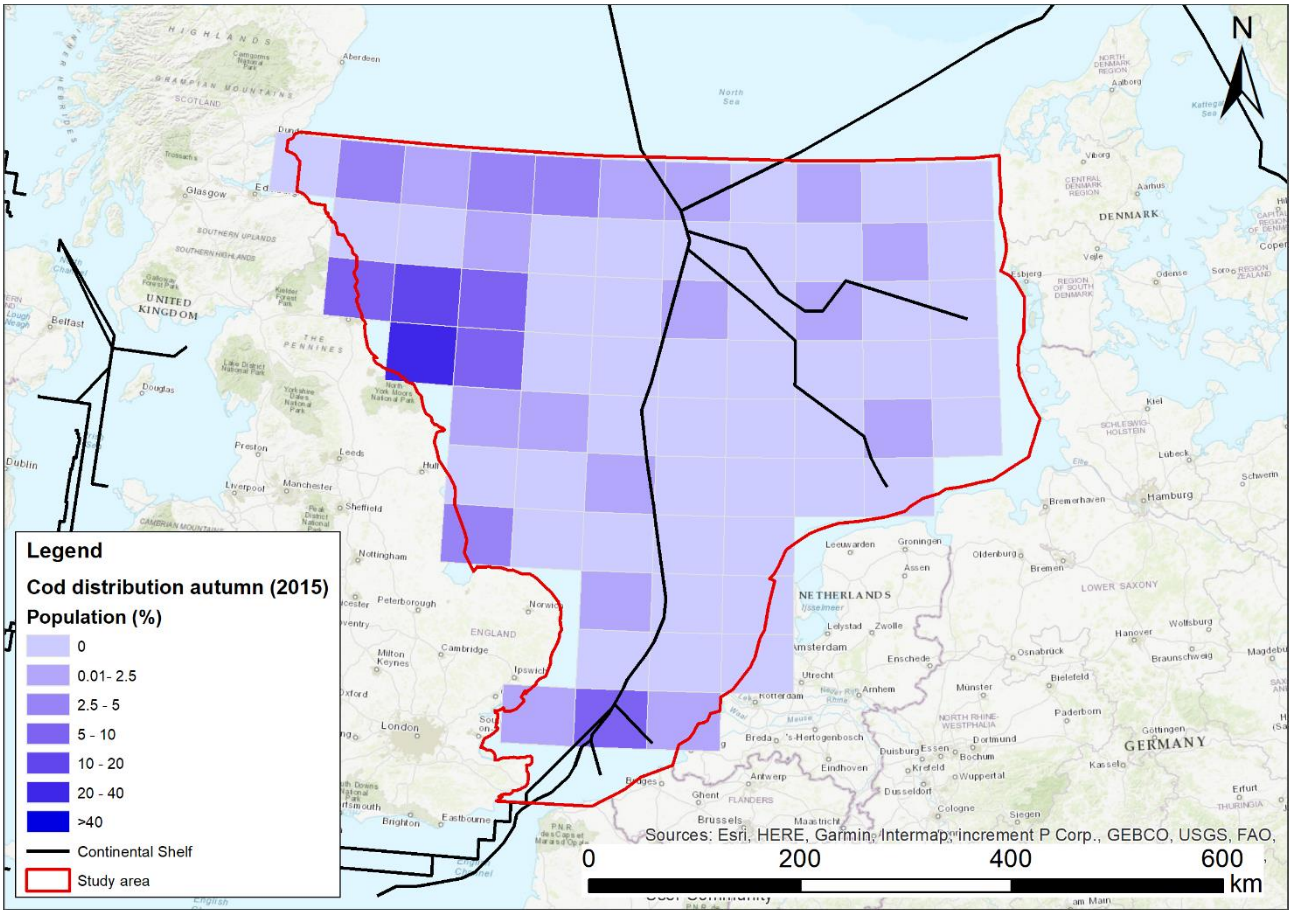

Figure 16: Modelled cod distribution autumn 2015. Grid cells display the percentage of the complete population within the assessment area. Data downloaded from: https://datras.ices.dk/Data_products/Download/Download_Data_public.aspx.

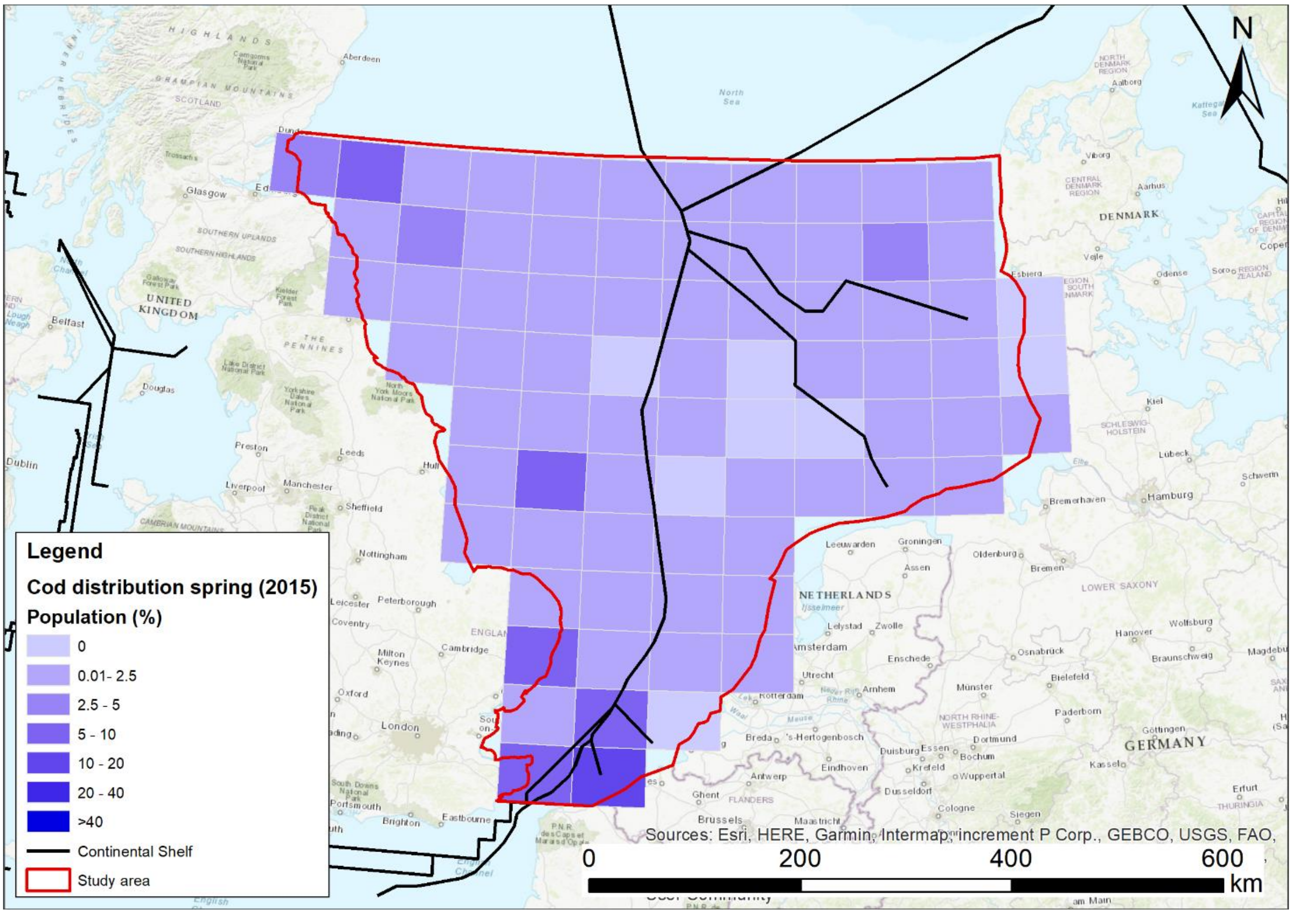

Figure 17: Modelled cod distribution spring 2015. Grid cells display the percentage of the complete population within the assessment area. Data downloaded from: https://datras.ices.dk/Data_products/Download/Download_Data_public.aspx.

7.2 Harbour porpoise distribution

Harbour porpoise distribution and abundance in the North Sea is monitored through dedicated aerial line-transect distance sampling surveys. These are conducted about every 10 years in coordinated summer surveys covering all of the North Sea and adjacent waters (Hammond et al., 2002, 2013; Ulrich et al., 2016). In addition parts of the North Sea are covered during national monitoring schemes, also covering additional seasons (Geelhoed et al., 2014; Gilles et al., 2011; Scheidat et al., 2012).

As aerial surveys are harder to conduct and therefore less frequent during winter, the data collection in winter is patchy. Distribution data of harbour porpoise during the winter are therefore considered unreliable. Therefore, data are available for spring, summer, and autumn but not for winter. Figure 18, Figure 19 and Figure 20 display the data in the calculated number of individuals per grid cell for spring, summer and autumn. The data input for these maps are the survey data from 2005 through 2013 (Gilles et al., 2016).

Based on additional survey data from 2014 to 2019 (Gilles et al., 2016, update in Gilles, Ramírez-Martínez et al. in prep) generated new model outputs for Harbour Porpoise distribution. As data were not complete/ reliable enough for both spring and autumn periods, new maps were only created for the summer period. This distribution map is shown in Figure 21. Figure 22 shows a side by side comparison of the summer distribution maps based on the 2005 – 2013 data and the 2014 – 2019 data. The data from 2014 – 2019 seem to show a shift in distribution slightly to the north west with a more prominent hot spot at the latitude of Newcastle.

The available data are used in chapter to perform the exposure analysis and create exposure maps for both the old (2005 – 2013) and new (2014 – 2019) data.

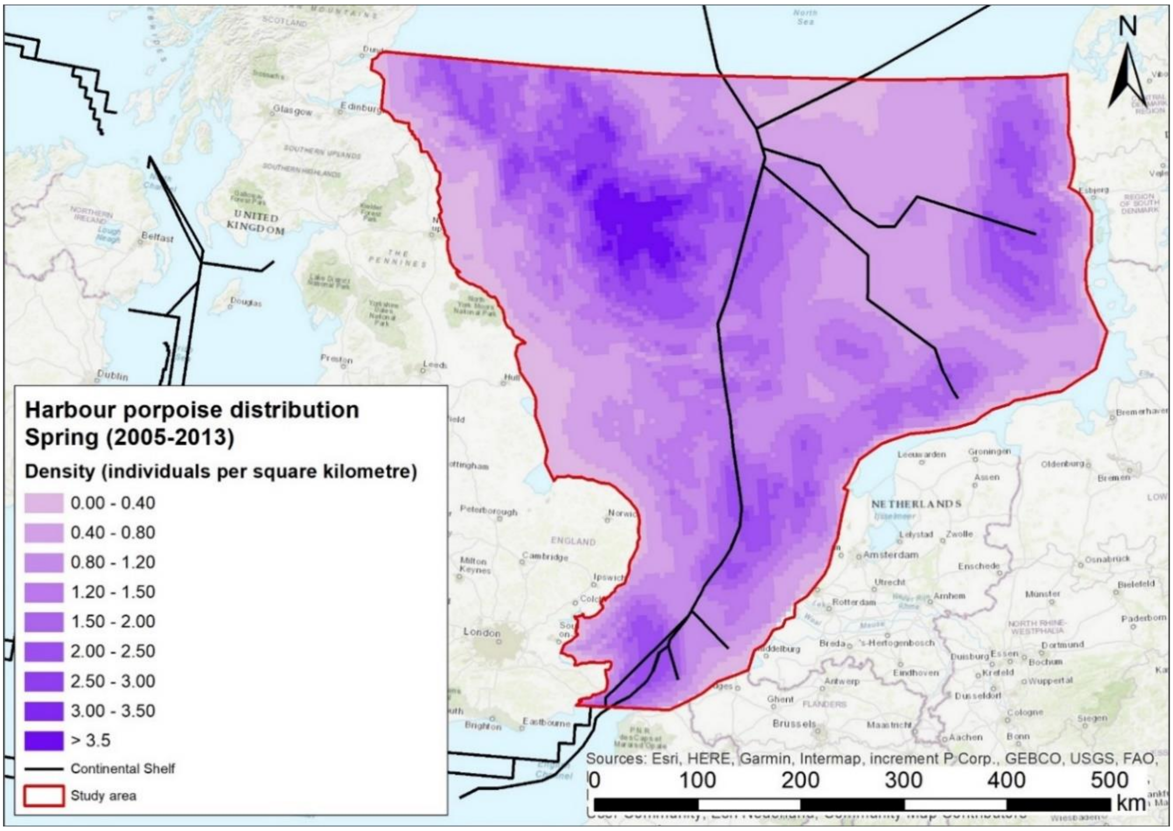

Figure 18: Modelled harbour porpoise distribution spring (2005 – 2013). Grid cells display the density of harbour porpoises in number of individuals per square kilometre. Data source (Gilles et al., 2016)

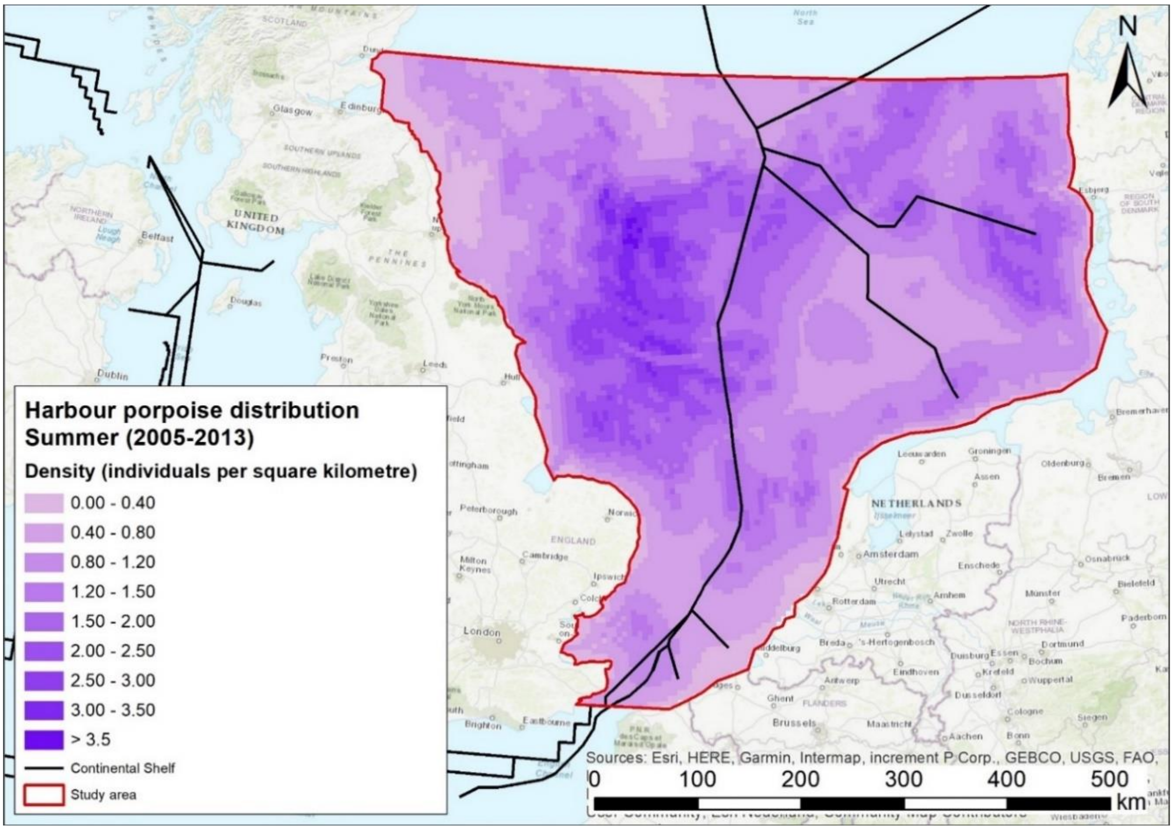

Figure 19:Modelled Harbour porpoise distribution summer (2005 – 2013). Grid cells display the density of harbour porpoises in number of individuals per square kilometre. Data source (Gilles et al., 2016)

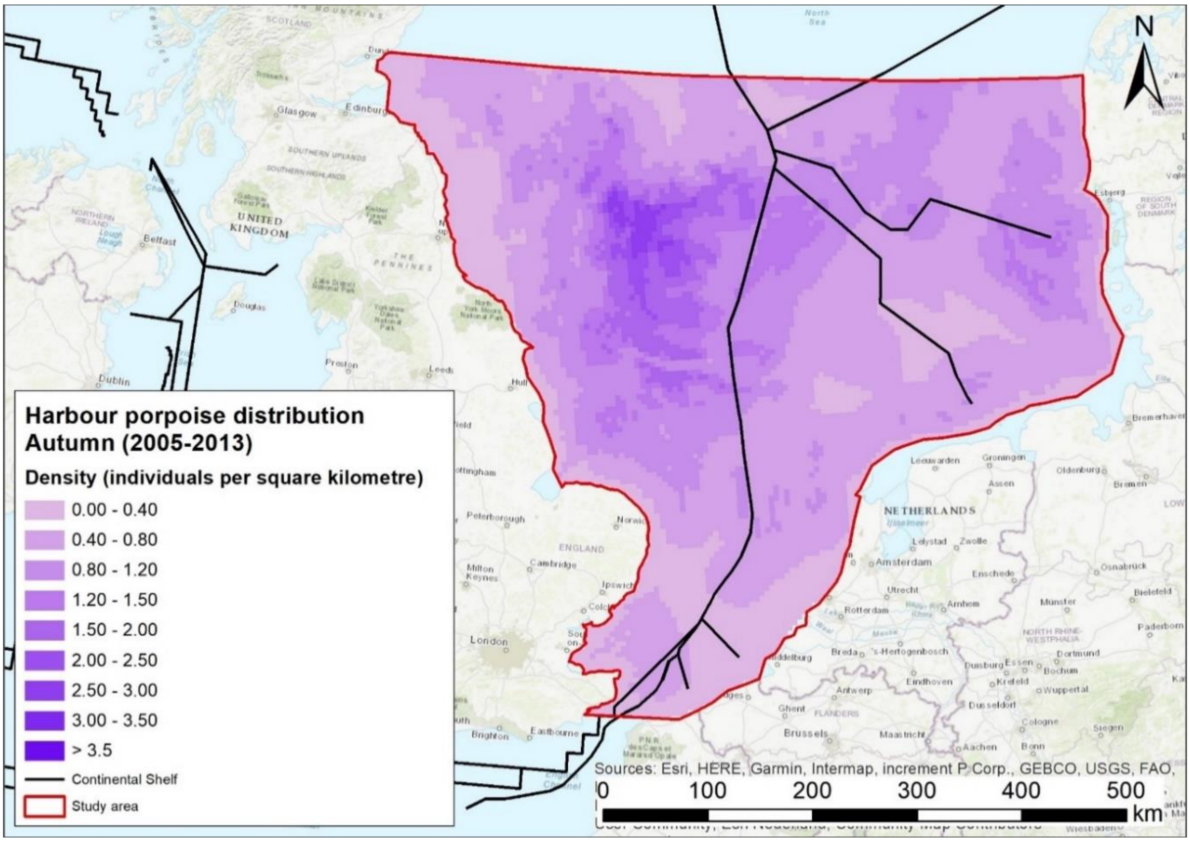

Figure 20: Modelled harbour porpoise distribution autumn (2005 – 2013). Grid cells display the density of harbour porpoises in number of individuals per square kilometre. Data source (Gilles et al., 2016)

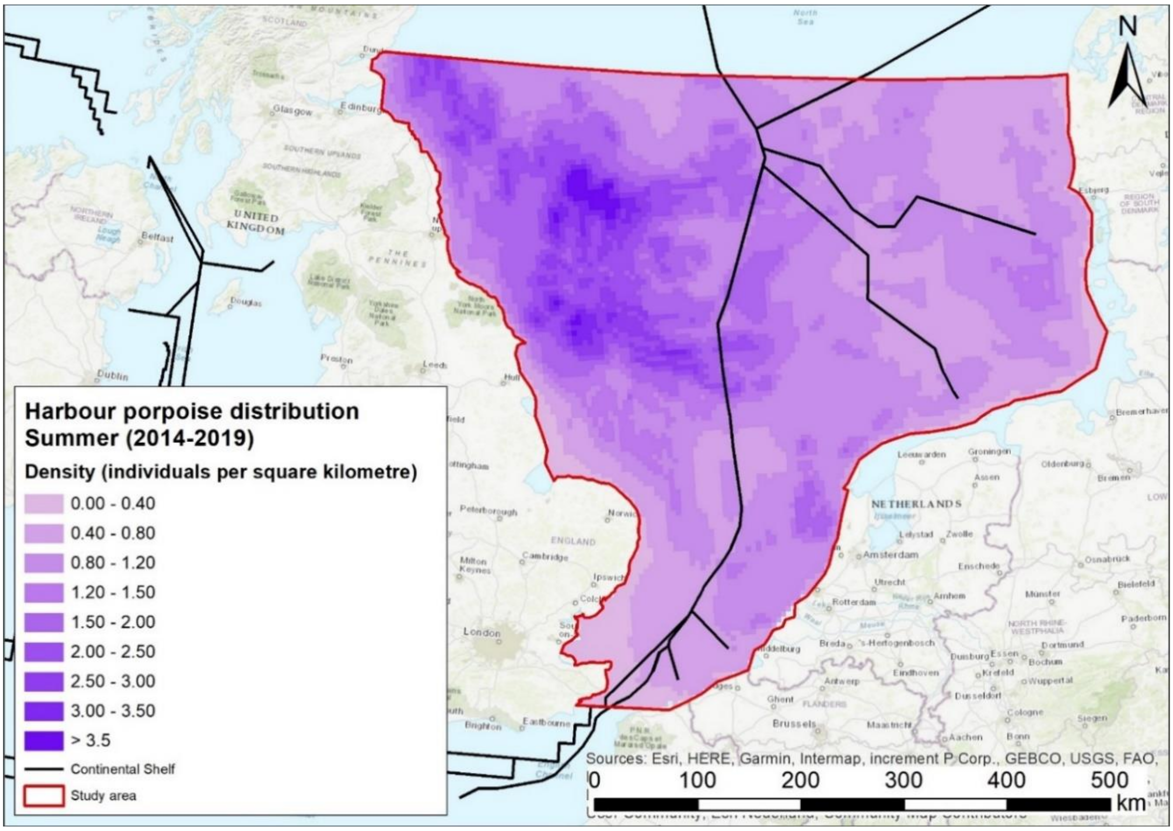

Figure 21: Modelled harbour porpoise distribution summer (2014 – 2019). Grid cells display the density of harbour porpoises in number of individuals per square kilometre. Data source; (Gilles et al., 2016, update in Gilles, Ramírez-Martínez et al. in prep)

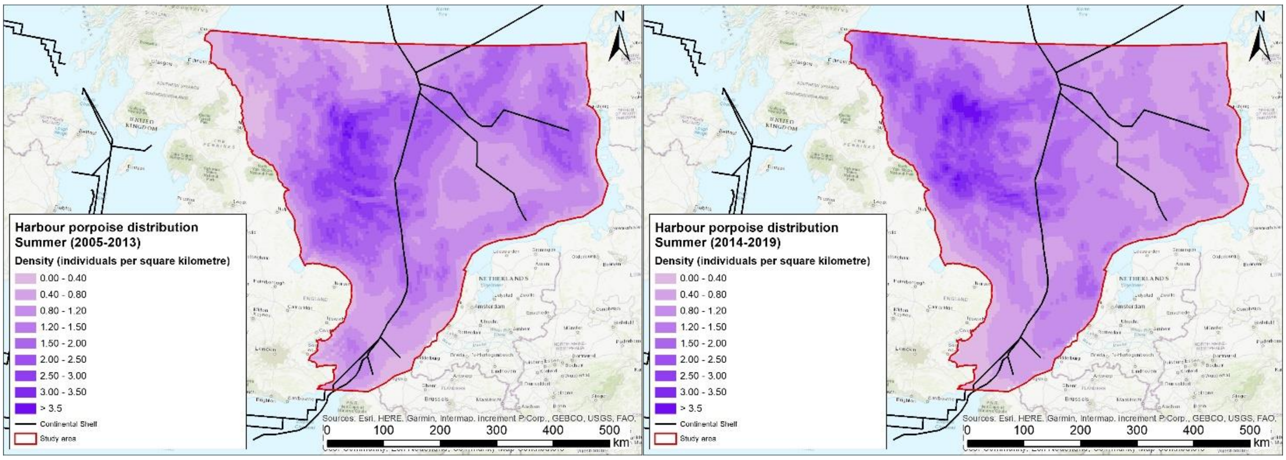

Figure 22: Side by side comparison of the updated distribution maps for harbour porpoise distribution in summer based on 2005 – 2013 survey data (left) and 2014 – 2019 survey data (right).

7.3 Seal distribution

Data on the distribution of harbour seals are available through GPS tracking of harbour seals from haulout sites located near the Dutch coastline. An in depth explanation of the followed methodology of these GPS tracking studies can be found in the corresponding scientific paper (Aarts et al. 2016c). Figure 23 shows the GPS tracking data that was collected between 2007 and 2015. These data were used by Aarts et al. (2016) to estimate seal distribution in relation to environmental factors. Data from aerial surveys of haul out sites (Reijnders et al., 2003) were used to add to these distribution model outputs. The model predictions on harbour seal distribution within the DCS for February are displayed in Figure 24. Predicted distribution in September is displayed in Figure 25.

Distribution data are only available up to 2015. After 2015 additional GPS data were collected that showed a more extensive use of the coastal zone that is not reflected in the distribution predictions. Furthermore, the distribution predictions are based on population estimated and locations of haul out sites at that time. Both population estimates and haul out sites have changed since then. As the predictions are static in time it's output can give a warped image of the intensity of use of a specific location as the location is visited very frequently over a time period. These factors introduce an uncertainty in the distribution predictions which makes the data to unreliable to use for accurate and present-day predictions of harbour seal distribution.

In addition to the above-mentioned uncertainties the available data on spatial distribution of Harbour seals are limited to the DCS. There is no uniform monitoring program that gives consistent and reliable data for the complete study area. This results in the fact that no data are available on the distribution of seals that, at this point in time, is complete enough to use as a basis of impulse noise exposure mapping. **Seals are therefore not incorporated in the exposure analysis and creation of exposure maps.**

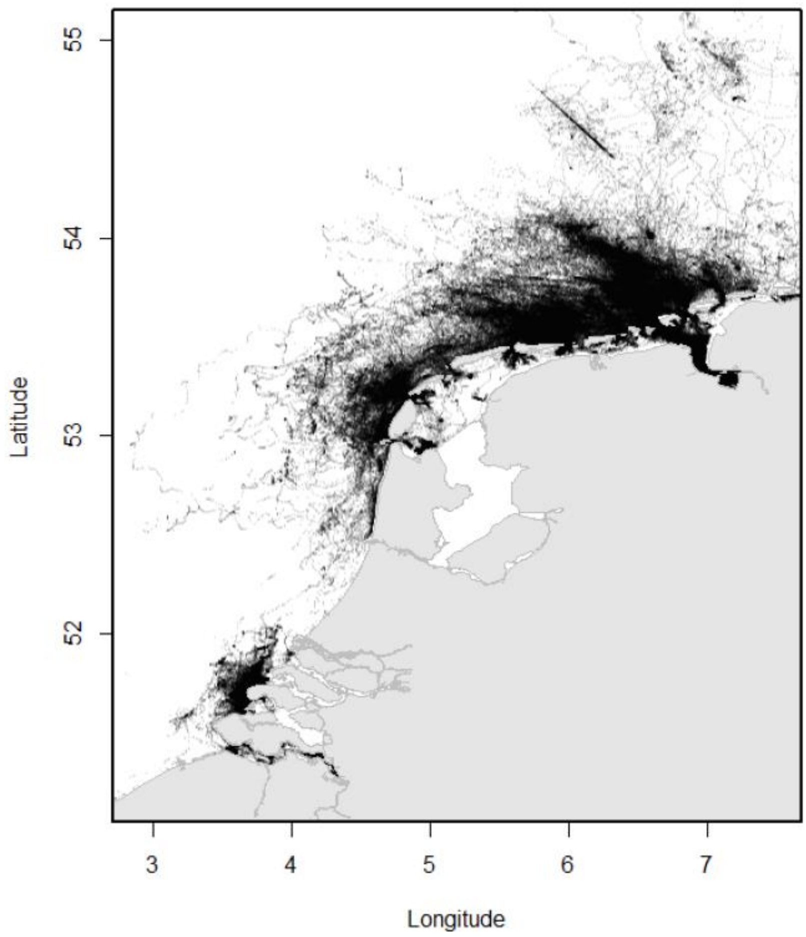

Figure 23: GPS tracking data from 219 harbour seals between 2007 and 2015. Source: (Aarts et al. 2016c)

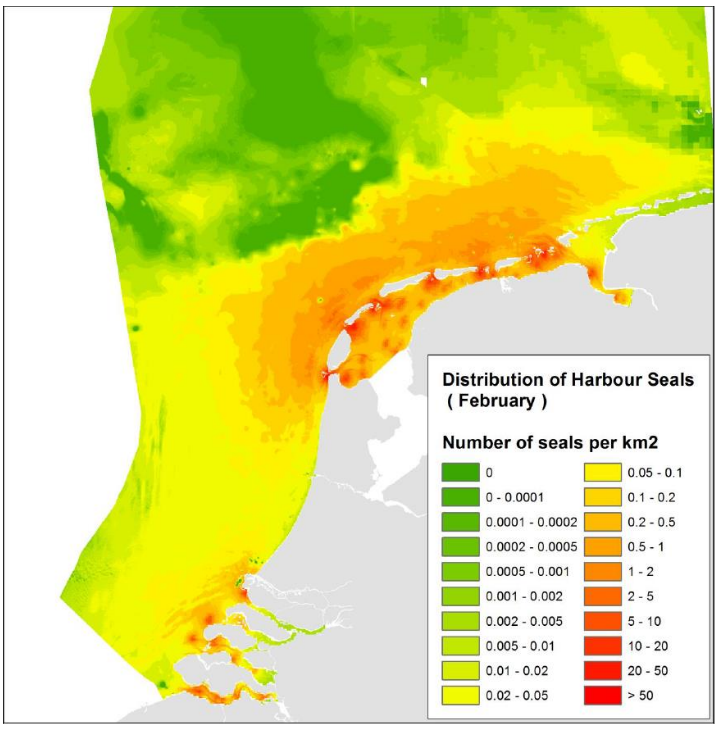

Figure 24: Predicted spatial distribution in February of harbour seals from Dutch haul-out sites. (Aarts et al. 2016c)

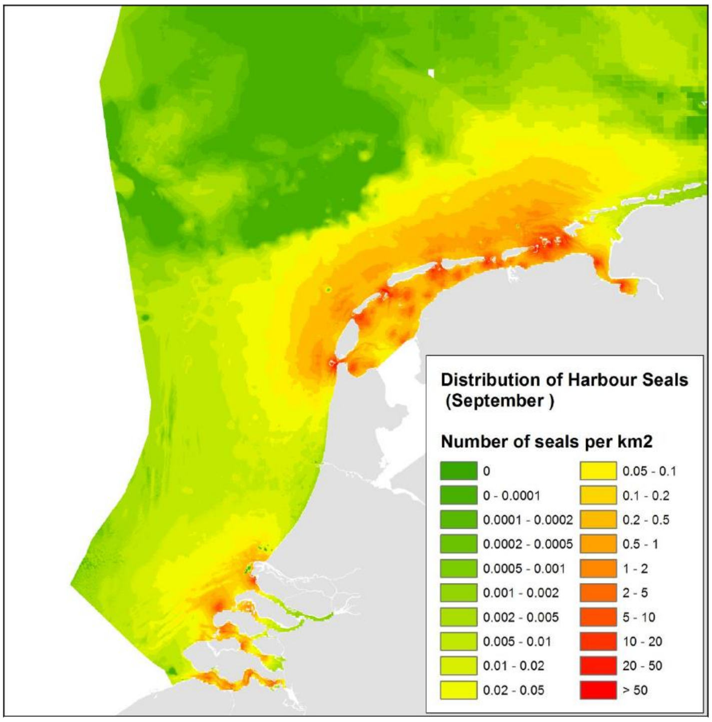

Figure 25: Predicted spatial distribution in September of harbour seals from Dutch haul-out sites. (Aarts et al. 2016c)

8 STEP 5: PRESSURE MAPS

In the previous chapters step 0: define stressor, step 1: select indicator species, step 2: define assessment area, step 3: define temporal resolution and step 4: Distribution or habitat map(s) were completed. The identified stressor is: impulsive noise. The selected indicator species are harbour porpoise, cod and harbour seal. The assessment area is an extension of the Dutch North Sea and the temporal resolution is defined as 2015 to 2017. Distribution maps were created for cod and harbour porpoise. Data for harbour seals are not suitable for further analysis. In this chapter step 5: pressure map(s) is described (Figure 26Figure 15). This step was taken simultaneously with step 4 that was discussed in the previous chapter, therefore harbour seals are still included in the chapter.

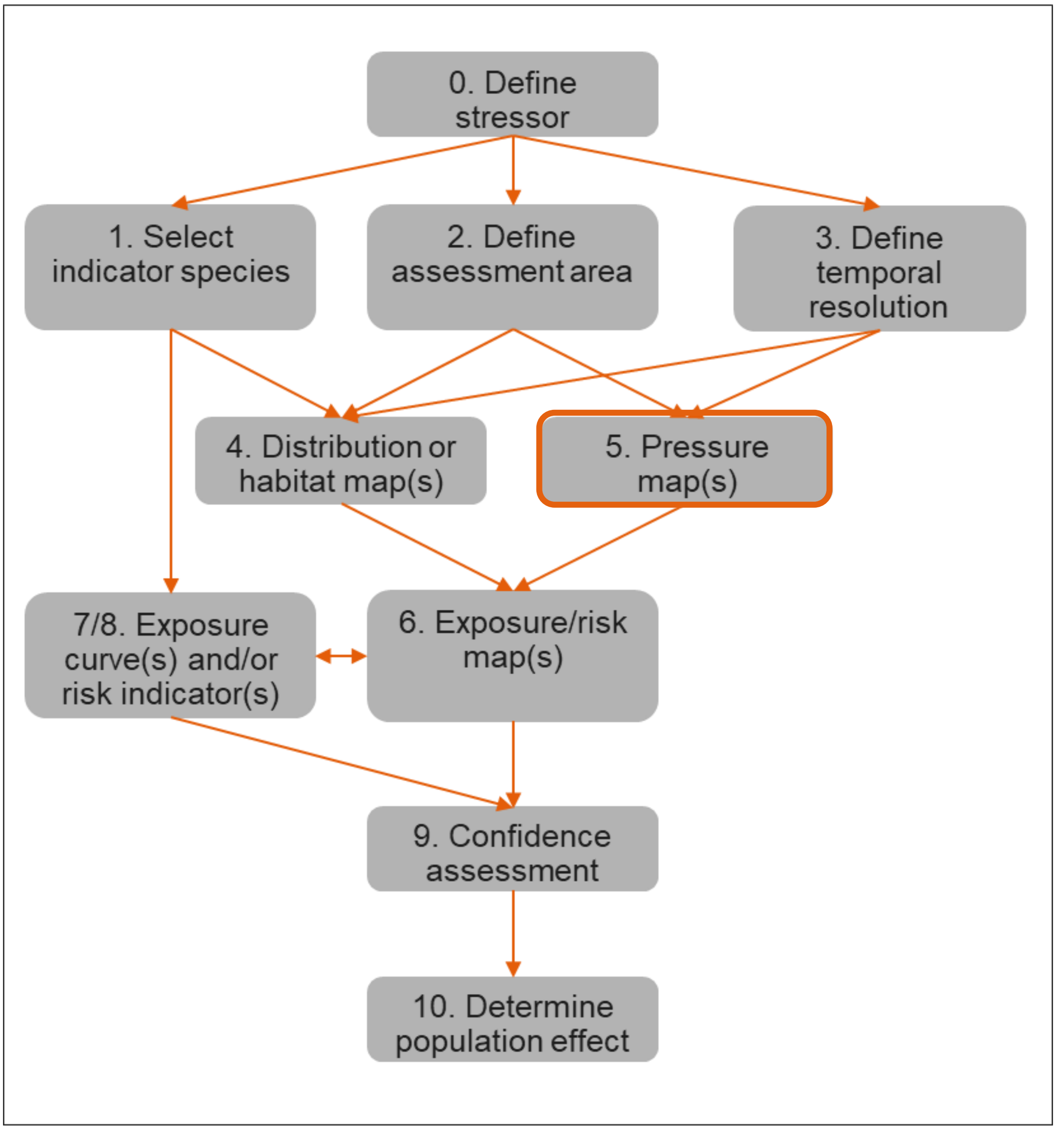

Figure 26: Stepwise approach, current step (highlighted orange).

To obtain pressure maps the relationship between sound and disturbance has to be identified. This topic is generally explored in paragraphs 8.1 and 8.2. In paragraph 8.3 dose-response values are then presented for each indicator species. In paragraph 8.4 pressure maps are presented for all noise sources in the impulse noise register (8.4.1) and then split by sounds source (8.4.3). In order to do this, propagation models were needed which are explained in paragraph 8.4.2.

8.1 Predicting disturbance in marine mammals and fish

Different approaches are currently being considered for assessing the impact of impulse noise using the impulsive noise register, either by considering a constant effective distance based on observed response distances (Merchant et al. 2018b), or by predicting effect distances for different sound sources and environments using sound propagation models (Von Benda-Beckmann et al., 2017).

A literature review indicates that information is available for only some sound categories in the impulse noise register, depending on the species (Table 8). For the three species, observed effect distances were categorized by source type and source category as defined in the impulse noise register. Quite a few combinations of source categories and source types lack information (Table 8). Most information is currently available for the harbour porpoise, less for seals and very little field data for cod. Ongoing field research on cod exposed to seismic sound and pile driving within the PCaD4Cod project is likely to result in empirical data on the short term.

Based on the porpoise observations, an increase in effect distance with increase in noise category can be seen for seismic surveys, as well as for pile driving with and without noise abatement mitigation, indicating that higher noise levels typically produce disturbance at larger distance. For the pile driving category, a wide range in reported effect distances exists. This variation can be due to environment (worse propagation in more shallow areas where small effect distances were reported) but could also reflect variability in animal responsiveness between construction sites and period.

Table 4: Summary of observed disturbance distances reported in the literature for different species (harbour porpoises, seals, and cod), broken down by source type (Sonar, airgun, etc.) and source category ("Very low" to "Very high") in the Impulse Noise Register.

Harbour porpoise	Sonar and acoustic deterrents	Airgun arrays	Pile driving	Explosion	Generic impulsive source
	Distance / km				
Very low		0.69 [9]			
Low		5 - 10 [8]	1.3 - 7 [5] 20 [3,4] 15 [2] 17 - 33 [1] 10 - 20 [6]		
(mitigated)			14 [2] 12 [10]		
Medium					
High		8 - 12 [7]			
Very high					

Seals	Sonar and acoustic deterrents	Airgun arrays	Pile driving	Explosion	Generic impulsive source
	Distance / km				
Very low	0.01 [11]				
Low			25 [12] 36 [13]		
(mitigated)					
Medium					
High					
Very high					

Cod	Sonar and acoustic deterrents	Airgun arrays	Pile driving	Explosion	Generic impulsive source
	Distance / km				
Very low					
Low					
(mitigated)					
Medium					
High		33 [14]			
Very high					

1 Brandt et al. (2018)

2 Carstensen et al. (2006)

3 Tougaard et al. (2009)

4 Dähne et al. (2013)

5 Graham et al. (2019)

6 Geelhoed et al. (2017)

7 Sarnocińska et al. (2020)

8 Thompson et al., (2013)

9 Van Beest et al. (2018)

10 Dähne et al. (2017)

11 Kvadsheim et al. (2012)

12 Russell et al. (2015)

13 Aarts et al. (2018)

14 Engås et al. (1996)

8.2 Dose-response relationship – predicting behavioural response using acoustic metrics

8.2.1 Predicting disturbance in marine mammals and fish

The approach in von Benda-Beckmann et al. (2017) relies on the assumption that the acoustic metrics used here can be used to predict the behavioural response distance (e.g. avoidance, cessation of feeding) of marine species. Therefore, the metric can be used to interpolate between conditions where effect distances were observed to different sources and environments. Purely acoustic characteristics of the sound may not be the sole predictor of behavioural responses in marine mammals. Other contextual factors (e.g. distance to source, individual variability, prey availability, prior experience to the sound), can drive the responsiveness of animals to the sound (Dunlop et al., 2018; Ellison et al., 2012; Gomez et al., 2016b; Harris et al., 2015; Hawkins et al., 2020; Miller et al., 2012; Slabbekoorn et al., 2010; Southall et al., 2007; Tougaard et al., 2015). Ideally, response distances should be based on observed responses to the source considered for wild animals in the region and time period of interest, as well as considering a potential effect of context (Gomez et al. 2016a, Merchant et al. 2018b). Given the lack of consistent observations for a wide range of source properties, contexts and environments, extrapolation from observed responses to situations for which no observations are available is not credible without a science-based approach to bridge the gaps in empirical data. The effective distance approach in which behavioural disturbance distances are determined purely on the few observed effect distances (Table 4) does not provide a framework for interpolating between exposure conditions.

It is proposed to use the SEL or SPL (sound exposure level or sound pressure level, depending on the sound source type) as a means of extrapolation between source categories and environments. For marine mammals, these quantities are systematically reported. The effect particle motion has on fish is poorly understood (Hawkins et al., 2020), and challenging to model accurately. We acknowledge that particle motion should be used as a predictor for disturbance for fish and other species such as invertebrates, but for this report, we will also rely on single-shot sound exposure level (SELss) as a predictor for disturbance for these taxa.

Various controlled exposure experiments in marine mammals have shown that there is an increase in responsiveness with increasing sound exposure both for wild animals (Harris et al., 2015; Miller et al., 2012; Moretti et al., 2014; Southall et al., 2019) as well as captive animals (Houser et al. 2013a and b). From observations it is clear that effect of activities that differed strongly in output levels lead to significantly smaller or larger effect distances (e.g. Dähne et al., 2013; Thompson et al., 2013; Sarnocińska et al., 2020; Wensveen et al., 2019). Even though it is argued that captive studies may not be representative due to positive reinforcement by feeding (Gomez et al., 2016b; Tougaard et al., 2015), there is nevertheless consistency between the levels at which captive porpoises respond to sound compared to animals in the wild (e.g. Wensveen, 2016). Captive studies can therefore serve as interim predictions of effect distances when field observations are lacking.

There exists significant variability within populations and between species which can be captured using dose-response relationships (e.g. Southall et al. 2007, Harris et al. 2015, Russell et al. 2016, Wensveen 2016, Graham et al. 2019). The main challenge is to derive generic dose-response relationships for species (or species groups) and source types that have not been studied in detail, or for which no field observations are available (see Table 7). Gomez et al. (2016) carried out a meta-analysis of studies on behavioural responses of marine mammals to different sound sources and found no clear evidence for the influence of acoustic metrics (SPL) on the severity of the response. In their study, species were grouped by hearing sensitivity (LF cetaceans, MF cetaceans; Southall et al., (2007). Dose-response relationships were present for individual species, but were masked by this coarse grouping as species with very different sensitivities to sound were considered as a single group (e.g. killer whales and pilot whales, or humpback vs minke whales (Miller et al., 2012; Sivle et al., 2012)). Responsiveness of species within the same hearing group (Southall et al., 2007) may vary strongly with these categories as they can also be determined by species specific anti-predator response templates (e.g. Visser et al. 2016)). Nevertheless, Gomez et al. (2016) makes a valid case that it is challenging given the current state of knowledge to derive science-based generic dose-response relationships that are valid for a wide group of species.

The role of source distance (in addition to level of the acoustic exposure) in driving the responsiveness is the subject of ongoing debate. Some studies have suggested that animals tend not to respond to distant sound sources at similar received levels on a limited sample (e.g. Deruiter et al., 2013), or lack data for exposure level (Falcone et al. 2017), Only a few studies have been carried out that systematically investigate the role of distance on the responsiveness of animals to sound exposure. In some cases studies report an effect of distance on the dose-response relationship on distances of a few km (blue whales exposed to sonar, (Southall et al., 2019); humpback whales exposed to airgun, (Dunlop et al. 2018)). Both the Dunlop et al. (2018) as well as the Southall et al. (2019) study did find a dose-response relationship at every distance, but that the probability of response (at equal SELss or SELcum) decreased with increasing distance. A different study found no difference in response thresholds up to the maximum distance test of 28 km (bottlenose whales exposed to sonar; Wensveen et al. 2019). Further studies are required to assess the relative importance of distance and received level.

8.2.2 Dose response versus single disturbance threshold

Generally a single threshold does not represent the variability in responsiveness reflected in observed dose-response relationships, which tend to express a probabilistic increase in response as a function of sound dose (Tyack & Thomas, 2019). Dose-response relationships can readily be included in the proposed framework. Consider any grid cell, for a given day di, a source sj active on that day has a probability to disturb this grid cell that is determined by the acoustic dose generated in that grid cell by that source. The species-specific dose-response relationship relating dose to probability of disturbance, $P_{\mathrm{dist}}(d_i, s_j) = P_{\mathrm{dist}}(\mathrm{SEL}_{ss})$. The combined probability that a grid cell is disturbed on that day is then given by (assuming independence of disturbance probabilities due to multiple exposures on a single day):

$$P_{\mathrm{dist}}(d_i) = 1 - \prod_{j}^{n} (1 - P_{\mathrm{dist}}(d_i, s_j))$$

The average number of days of disturbance in any cell can be estimated from the sum of the combined probabilities over all days:

$$< N_{\mathrm{dist}} >= \sum P_{\mathrm{dist}}(d_i)$$

The uncertainties in probability of disturbance can in principle also be obtained from Monte-Carlo simulation, drawing multiple realisations of the disturbance for each during the assessment period (e.g. a full year, or season).

In principle, single effective response threshold can also be derived from dose-response by weighting the affected area suggested by probability of response to occur (Tyack & Thomas, 2019). The choice of effective threshold depends on the sound propagation and dose-response adopted (Tyack & Thomas, 2019).

When considering typical conditions for pile driving it is noted that for unmitigated piling noise, effective thresholds are close to the levels corresponding to 50% probability of response (Figure 28). For seismic sound propagation, it depends strongly on the fall-off of the airgun sound at larger distances (> 10 km). The accuracy of acoustic models for predicting sound produced by distant airgun array is still lacking (Prior et al., 2020).

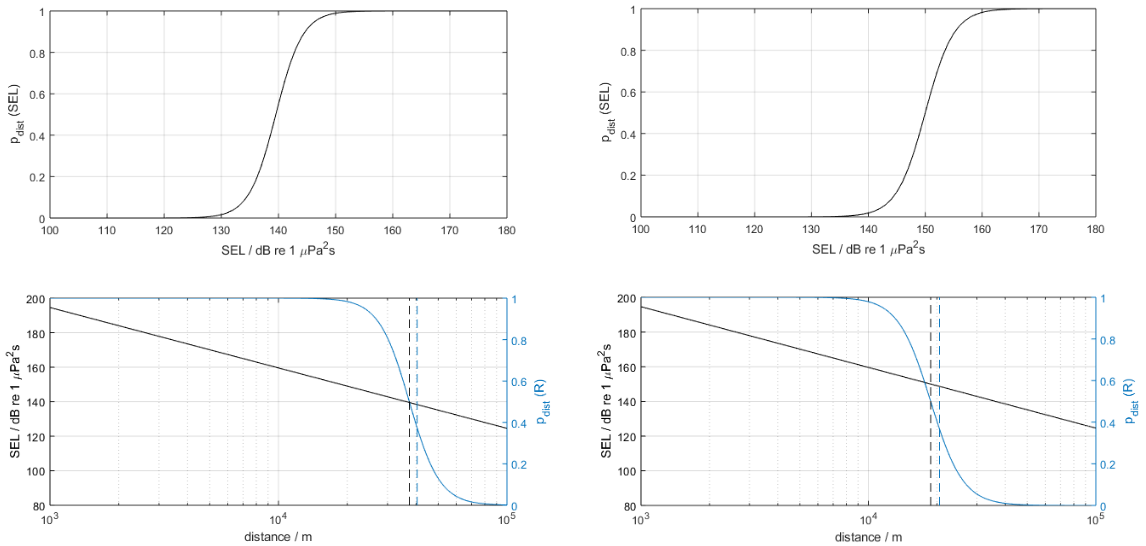

Figure 27: Effect of dose-response relationship (top) on predicted effect distance (bottom), in comparison to assuming using a single response threshold. For typical fall-off of SELss (black solid line) with distance for pile driving, the effective disturbance distance was found to be only slightly larger than the 50% disturbance distance. A dose-response relationship with the 50% probability of response at SELss = 140 dB re 1 μPa²s (adopted for harbour porpoises, KEC 2019) would predict effect distances of 40 km (left), whereas a dose-response with an increasing response for SELss > 140 dB re 1 μPa²s would predict effect distances up to 20 km (right).

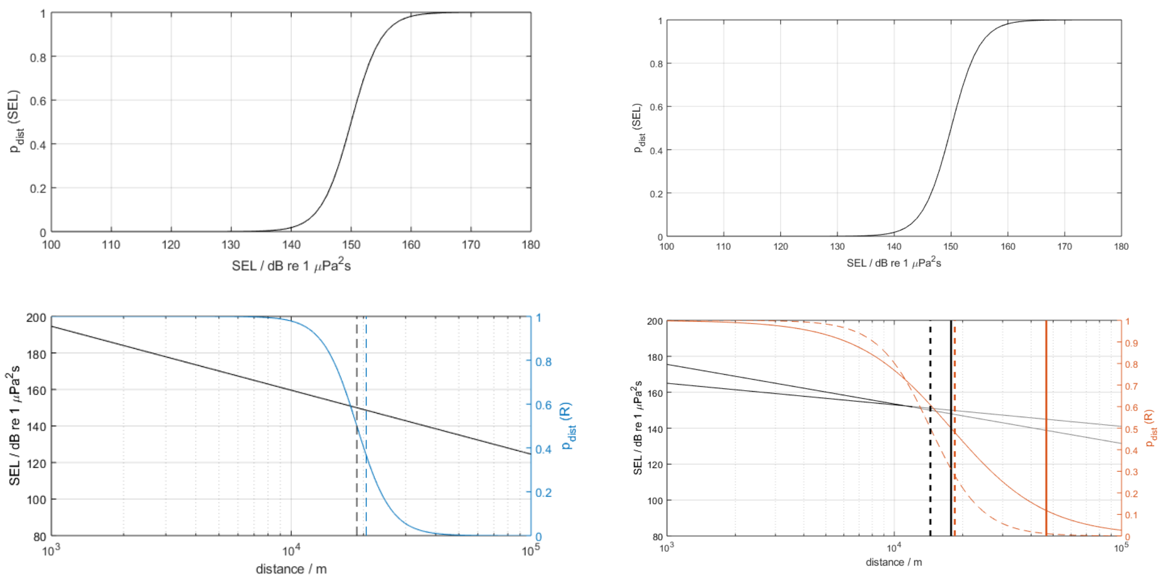

Figure 28: P Effect of dose-response relationship (top) on predicted effect distance (bottom), in comparison to assuming using a single response threshold. For typical fall-off of SELss with distance for pile driving (black solid line; left) and seismic surveys (black solid line; right), the effective disturbance distance was found to be only slightly larger than the 50% disturbance distance for pile driving noise, but could potentially be larger for gun sounds, depending on how the sound propagates to larger distances. The accuracy of acoustic models for predicting sound produced by distant airgun array is still lacking (Prior et al. 2020).

8.3 Selection of dose-response relationships for indicator species

In the process for deriving dose-response relationships, or identifying data that could be used for establishing dose-response relationships, the following points were considered:

- Observed behavioural responses that are used to motivate the dose-response relationship should constitute a significant response (e.g. habitat avoidance, cessation of feeding, disruption of social behaviour in fish schools), with the potential for affecting individual's fitness and ultimately affect the population (Dekeling et al., 2014; Dekeling et al., 2013; Heinis et al., 2019; National Research Council, 2005; Slabbekoorn et al., 2019; Southall et al., 2007).
- Aim for consistency with current Dutch thresholds used for assessing impulse sounds (Heinis et al. 2019)
- Where possible quantitative, if not possible, an alternative is to carry out a sensitivity study to indicate what thresholds for given species distribution lead to significant effect, or at least the amount of overlap this leads with species in space and time.

As previously discussed, the three indicator species considered here as examples likely have different sensitivities to impulse noise. The noise level necessary to trigger a response may be different as well as the duration of the response to the sound. Because of this the species-specific disturbance thresholds and effect durations are discussed for Harbour Porpoise, Cod and Harbour- and Grey Seals in the following paragraphs.

8.3.1 Cod

8.3.1.1 Hearing ability

Sound plays an important role in their reproduction and they produce short "grunts" during mating at a frequency of between 45 to 90 Hz (e.g. Finstad & Nordeide (2004)). Cod can also produce a range of other sounds, such as clicks occurring at a peak frequency of 6 kHz (Vester et al., 2004). Sound pressure thresholds for Atlantic cod were obtained using cardiac conditioning in which the change of heart rate in combination of a mild electric shock is used to assess if a pure tone is heard. The results showed that Atlantic cod are sensitive to sound from 30 to 470 Hz with the greatest sensitivity from 60 to 130 Hz (Chapman & Hawkins, 1973; Hawkins & Picciulin, 2019; Offutt, 1974).

8.3.1.2 Vulnerability to impulsive sound

A review of studies of cod behaviour in relation to underwater sound shows that data on cod responses to sound are scarce. In Thomsen et al. (2012, (Mueller-Blenkle et al. 2010)) captive cod responded (freeze response, no statistical change in swim speed measured) to piling playbacks of peak sound pressure level $L_{p,zp}$ = 140 dB - 161 dB re 1 µPa (note, the SEL for broadband impulse sounds is typically ~20 dB lower). However, this is considered to not count as a significant response. Sierra-Flores et al. (2015) found reduced reproductive success in cod when they were exposed to underwater noise with tonal low frequency (100 – 1000 Hz) sounds of SPL between 104 and 110 dB re 1 µPa. In the Nordkappbanken airgun array experiment, trawl catches of Atlantic cod (Gadus morhua) were reduced by 70% within the exposure area (5.5 km × 18.5 km). No noise measurements were available and estimates in Handegard et al. (2013) are deemed unreliable due to the lack of model validation and differences between the airgun surveys that were not accounted for, so it cannot be reliably determined what levels of the sound caused the reduction in catch rates. Startle study (Kastelein et al., 2008) suggests that seabass and mackerel more responsive than cod. Captive seabass piling playback (change in cohesion, startle responses, signs of habituation) (SEL = 131 dB – 147 dB re 1 µPa2s). Mackerel responded to pile driving playback (change in depth of fish school without scattering) (SEL = 142 dB re 1 µPa2s). Reduced catch-rates in relation to airgun arrays were observed for redfish, halibut, haddock occurred for measured levels between SEL = 120 and 170 dB re 1 µPa2s. Because of the poor high frequency hearing sensitivity of cod (see Section 4.3), it is considered very unlikely that sonar exposures with frequencies 1-10 kHz can affect cod behaviour. No information is available on the effect of underwater explosions on cod. Recent reviews by (Andersson et al., 2016; Hawkins et al., 2014; Popper & Hawkins, 2019) also concluded that too limited data were available to support disturbance thresholds. For now we therefore considered a range of possible thresholds, to help identify under what sensitivity significant effects on the population is expected to occur. Three thresholds were selected over a wide range of SEL (130, 150, and 170 dB re 1 µPa2s) which given what is known about fish sensitivity to broadband sound is assumed to cover the most sensitive to very insensitive assumptions. It is anticipated that more quantitative dose-response relationships for cod will become available through the PCaD4Cod project (Slabbekoorn et al. 2019) in the foreseeable future.

8.3.2 Harbour Porpoise

8.3.2.1 Hearing ability

The harbour porpoise is a very high frequency specialist, with the best hearing from 16 kHz to140 kHz (Kastelein et al., 2010; Southall et al., 2019). The maximum sensitivity occurs between 100 kHz and 140 kHz, corresponding to the peak frequency of echolocation pulses produced by harbour porpoises (120-130 kHz). Hearing sensitivity falls at a rate of about 10 dB per octave below 16 kHz and falls off sharply above 140 kHz.

8.3.2.2 Vulnerability to impulsive sound

The vulnerability of harbour porpoises to impulsive sound has been studied in a number of different studies both in captivity and in the field. Studies into the construction of wind farms have provided evidence of a reduction of porpoises in the area compared to the baseline (Brandt et al., 2018; Henriksen et al., 2003; Madsen, et al., 2006; Tougaard et al., 2003; Tougaard et al.,, 2005). Porpoises have also shown to negatively react to the sounds of ADDs (Olesiuk et al. 2002, Johnston 2002, Mikkelsen et al. 2017). Foraging rates have been shown to change in reaction to seismic surveys (Pirotta et al. 2014) and during wind farm construction (Bergès et al. 2020). In captivity, impact of sound on porpoises has been investigated in several studies (e.g. Lucke et al. 2009, Kastelein et al. 2013b c, 2014).

Studies investigating the sensitivity of the harbour porpoise hearing system to noise exposure, have shown that these animals have lower TTS onset thresholds than other marine mammal species (Lucke et al. 2009; Kastelein et al. 2014; Southall et al. 2019). Harbour porpoises are more sensitive to high frequency sound exposures, and less sensitive to sound at lower frequencies (Kastelein et al. 2013b c; Kastelein et al 2014; Kastelein et el. 2017; Southall et al., 2019) where they also have poor hearing (Kastelein et al., 2010).

The effects on harbour porpoise are also examined and discussed in “Kader Ecologie en Cumulatie” (Framework Ecology and Cumulation). This framework was established in 2015 (Heinis & de Jong 2015) and discusses the cumulative effects of the construction of offshore windfarms on Harbour Porpoise. In 2018 updates of this framework were published increasing the accuracy of the assumptions (Heinis et al., 2019). A threshold of 140 dB re 1 μPa^{2}s was adopted within KEC (2015, 2018) which was interpreted as the worst-case assumption for a disturbance threshold of avoidance/ disturbance, as Brandt et al. (2018) found the decline in Harbour porpoise densities to decrease for sound exposure levels of more than 143 dB re 1 μPa2s, with a complete decline at levels around 160 dB re 1 μPa2s. The threshold of 140 dB is also in line with the proposed thresholds by the TG noise (Dekeling et al., 2014).

Recent observations of response distances during the Gemini wind farm construction indicate that response distances were smaller (< 25 km; Geelhoed et al. (2018)) than predicted using the of 140 dB re 1 μPa2s threshold, which suggests a disturbance distance of up to 70 km (De Jong et al., 2019). This may be a consequence of the lower responsiveness of the animals, or because unweighted SEL is not a good predictor for disturbance in harbour porpoises. Incorporating a dose-response relationship with onset of responses at 140 dB re 1 μPa2s and high probability of response at 160 dB re 1 μPa2s as suggested by the Brandt et al. (2018) study, the figures shown in paragraph 0 provide disturbance distances that are more in agreement with the observed distance. The role of frequency weighting is also currently discussed (De Jong & Von Benda-Beckmann, 2018; Kastelein et al., 2019; Tougaard & Dähne, 2017; Tougaard et al., 2015; Von Benda-Beckmann et al., 2015), and requires further verification to assess whether frequency weighted SEL provides a better prediction of observed effect distances than other metrics. Dose responses data becoming available using field data can be derived porpoises (e.g. Graham et al. 2019, Whyte et al. 2020) and can in principle also be derived; (Brandt et al., 2018; De Jong et al., 2019; Geelhoed et al., 2018; Sarnocińska et al., 2020). The development of dose-response relationships for broadband impulse sounds is currently foreseen using measurements from the Borssele wind farm construction site.

For sonar exposures, no field data are available, only for captive harbour porpoises exposed to 1-20 kHz sonar signals (Kastelein, et al., 2018, 2014, 2015). The disturbance criterion of SPL > 130 dB re 1 μPa as proposed by the TG Noise, (Dekeling et al., 2014, 2013) appears to be slightly lower than levels for which captive porpoises show avoidance behaviour to 1-10 kHz sonar sound, but higher than levels at which captive porpoises show an increase in respiration rate (Kastelein et al., 2018, 2019, 2014, 2015).

No information is available on harbour porpoise behavioural responses to underwater explosions (Von Benda-Beckmann et al., 2015). Following the TG Noise advice, the potential effect of underwater explosions on behaviour is currently estimated based on levels that cause TTS onset (Dekeling et al., 2014).

8.3.3 Seals

8.3.3.1 Hearing ability

Harbour seal sound production has been well studied under water. Male harbour seals use low frequency (250 Hz to 1.4 kHz) “rumbles”, in particular during the mating season (Bjørgesæter et al., 2004; Van Parijs et al., 2000, 2003). Information on the hearing capability of harbour seals is available from a number of studies providing behavioural audiograms as well as auditory brainstem response techniques (Kastelein et al., 2013; 2009; 2009b; Terhune, 1989). Harbour seal critical ratios have been investigated for frequencies between 100 Hz and 2500 Hz. The results show ratios increasing with frequency with values between 13 dB at 200 Hz and 17 dB at 2.5 kHz. This suggests that harbour seals are good at detecting low frequency signals in noise, but are not specially adapted to specific frequencies (Southall et al., 2000).Harbour seal sound production has been well studied under water. Male harbour seals use low frequency (250 Hz to 1.4 kHz) “rumbles”, in particular during mating season (Andersson et al., 2015; Van Parijs et al., 2000). Information on the hearing capability of harbour seals is available from a number of studies providing behavioural audiograms as well as auditory brainstem response techniques (Kastelein et al., 2009; Ronald A. Kastelein, 2009a, 2009b; Reichmuth et al., 2013; Terhune, 1989). Harbour seal critical ratios have been investigated for frequencies between 100 Hz and 2500 Hz. The results show ratios increasing with frequency with values between 13 dB at 200 Hz and 17 dB at 2.5 kHz. This suggests that harbour seals are good at detecting low frequency signals in noise but are not specially adapted to specific frequencies (Southall et al., 2000).

8.3.3.2 Vulnerability to impulsive sound

Initial response threshold for seals to broadband impulse sounds discussed in KEC (2014), were based on studies on captive harbour seals in relation to pile driving noise playbacks (Kastelein et al., 2013). In recent years, several field studies of seal responses to pile driving noise have provided data that can be used to determine dose response for harbour seals (Russell et al., 2016; Whyte et al., 2020) and grey seal studies (Aarts et al. 2018).

Russell et al., (2016) measured a temporary displacement of harbour seals up tot distances of 25 km. Model prediction indicated that harbour seals were displaced at SELs of between 142 and 151 dB re 1 µPa2s. The unweighted SELss threshold (142 dB re 1 µPa2s) from the pile driving playback sound study with captive seals (Kastelein et al, 2011) seems to be consistent with the threshold (142-151 dB re 1 µPa2s) from the field observations of Russell et al., (2016). Similarly, Whyte et al., (2020) report disturbance distances up to 25 km, at an average unweighted SELss of 145 dB re 1 µPa2s .

Harbour seals tagged in the Greater Wash wind farm area in the UK were found to stay at distances from around 5 to 40 km when piling was conducted (Hastie et al. 2015b). Hastie et al., (2016) monitored the movements of 24 harbour seals using telemetry during the installation of an offshore wind farm. Some of these animals approached within 7 km of piling operations and the maximum predicted peak-to-peak sounds pressure levels were as high as 146.8 - 169.4dB re 1µPa.

At sea data from tagged seals have been used to investigate individual behavioural responses to piling noise. The studies vary in their results, ranging from no measurable response as well as significant changes in behaviour at least up to 36 km from the pile driving site (e.g. Edrén et al. 2004, Brasseur et al. 2010, Kirkwood et al. 2015, Aarts et al. 2018).

Aarts et al., (2018) noted changes in diving behaviour by grey seal at up to roughly 40 km to the Gemini and Luchterduinen offshore wind park construction site. Since grey seals are predominantly benthic feeders, cessation of bottom dives is associated with cessation of feeding (Aarts et al. 2018). Based on model predictions, the Aarts et al. (2018) study indicates that responses occurred on average at SELss of 133 dB re 1 µPa2s (1 m above the bottom, unweighted, assuming no wind-effect). For SELss exceeding ~137 dB re 1 µPa2s, the majority of exposures (10 out of 18) showed a significant behavioural response in one of the dive or movement variables. Comparing these numbers suggests a difference in responsiveness between grey seals and harbour seals. However, modelled data in Aarts et al. (2018) study were based on an earlier version of Aquarius, which has been shown to significantly underpredict the levels at tens of kilometres compared to the Gemini dataset (de Jong et al. 2018). A rough extrapolating the U8 data from Gemini, a higher value of SELss ~ 145 dB re 1 µPa2s is expected at 30 km (distance at which 95% likely change in distance to pile was observed).

Overall, the recent field data indicate that disturbance responses start to occur on average at levels exceeding roughly SELss > 142 dB re 1 µPa2s, with increasing probability of disturbed behaviour (avoidance or cessation diving behaviour) with higher SELss, likely a results of individual and contextual differences within the population. The effect of frequency weighting (De Jong & Von Benda-Beckmann, 2018; Tougaard et al., 2015; Götz & Janik, 2011) on these results cannot de directly assessed, as information spectral content of the exposure is not readily available.

Limited data are available on seal responses to sonar, and responses to sonar sound have only been studied for animals in captivity. A study on captive hooded seals showed avoidance responses to sonar sounds within a frequency range of 1-10 kHz revealed that seals respond to sonar at peak sound pressure levels $L_{p,\,pk}$ = 160-170 dB re 1 µPa (Kvadsheim et al., 2012; the SPL would be approximately 3 dB lower, i.e. 157 – 167 dB re 1 µPa). Kastelein et al., 2017 harbour seal response to seal scarer operating at frequencies between 1-20 kHz. Swimming with the head out of the water or complete haul out during the exposure occurred when the SPL reached 160 dB re 1 µPa, however the number of jumps already increased when sound was presented at received SPLs of 139 to 148 dB re 1 µPa. Gordon et al., (2019) conducted a series of controlled-exposure experiments (CEEs) by investigating how ADD signals impacted the tracks of harbour seals. The results showed that ranges of <1 km (predicted SPL of 134.6 dB re 1 µPa) elicited a change in behaviour, such as moving away from the sound source, with a maximum response range of 3123 m (predicted SPL of 111 dB re 1 µPa). The responses however were not always resulting in substantial movements away from the source.

Information on seal behavioural response to underwater explosions is not available. Note that due to the frequent clearance of UXOs on the North Sea, it is possible that opportunistic tagged seal data may provide information on the potential responses of seals to explosions. Similar to harbour porpoises, currently TTS onset thresholds (Southall et al., 2019) could be used to evaluate behavioural responses to explosions.

8.4 Maps based on the impulse noise register

8.4.1 All registered impulse noise

The impulse noise register provides information on the source type, source category, and location. As described in Von Benda-Beckmann et al., (2017) several assumptions need to be made in order to connect these to acoustic models. These assumptions deal with the lack of spatial information (for airgun arrays, generic impulsive sound sources, and sonar activities), as well as well as the choice of a representative source description based on the rough source categories in the register. The examples provided in this report are based on the simplified modelling in Von Benda-Beckmann et al. (2017). More realistic source and propagation model that can be used to predict more accurately the sound levels from sources in the Impulse Noise Register are briefly summarized below.

Based on the disturbance thresholds mentioned in paragraph 8.3.1 for Harbour porpoise, the disturbance days were calculated. Figure 29, Figure 30 and Figure 31 show the disturbance days using the 140dB disturbance threshold in the North Sea for spring, summer and autumn respectively. As mentioned in paragraph 8.3.1 three different disturbance thresholds were used for Cod. Based on these three values the disturbance days were calculated for cod for two seasons in which monitoring data were available. The disturbance days for Cod incorporating the different thresholds are displayed from Figure 32 to Figure 37.

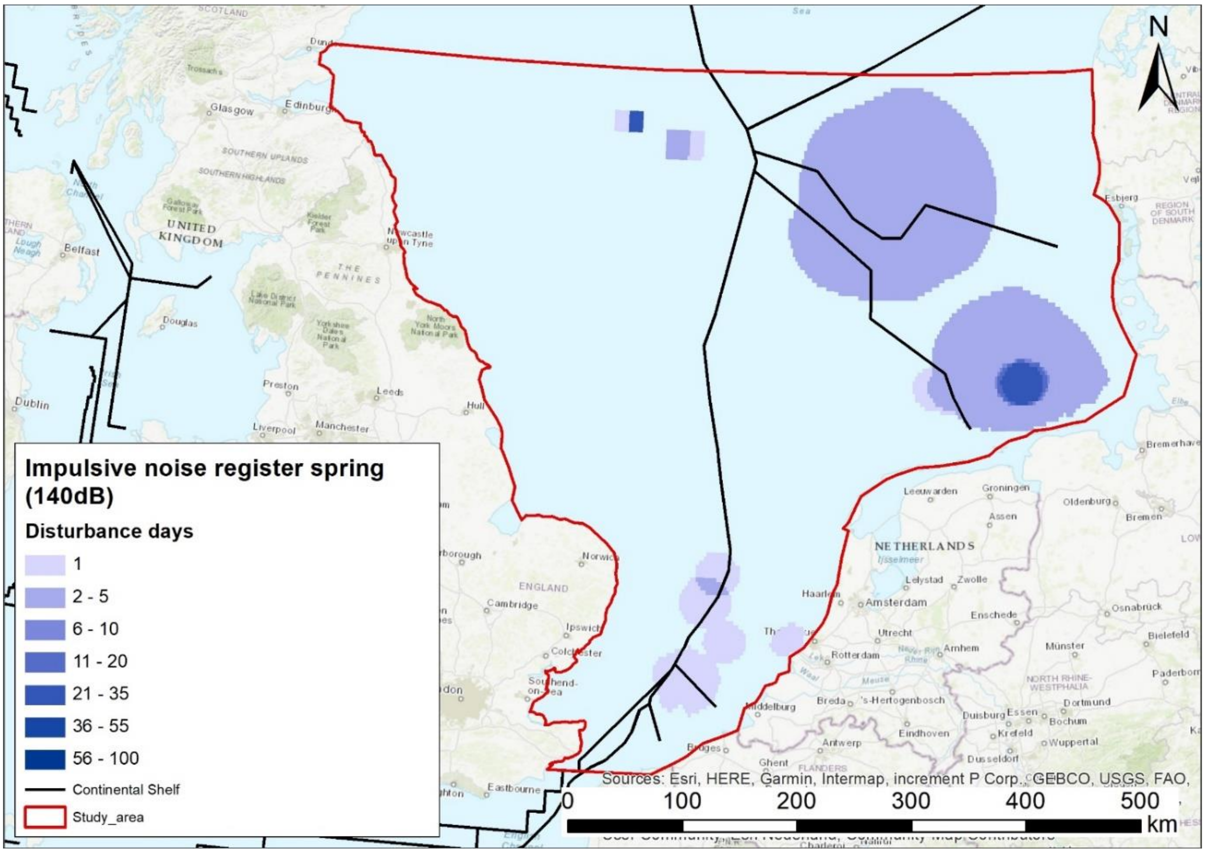

Figure 29: Disturbance days within the study area as a result of the total sum of noise sources for spring based on the 140 dB disturbance threshold used for Harbour porpoises. Data shown are data from the Impulsive Noise Register.

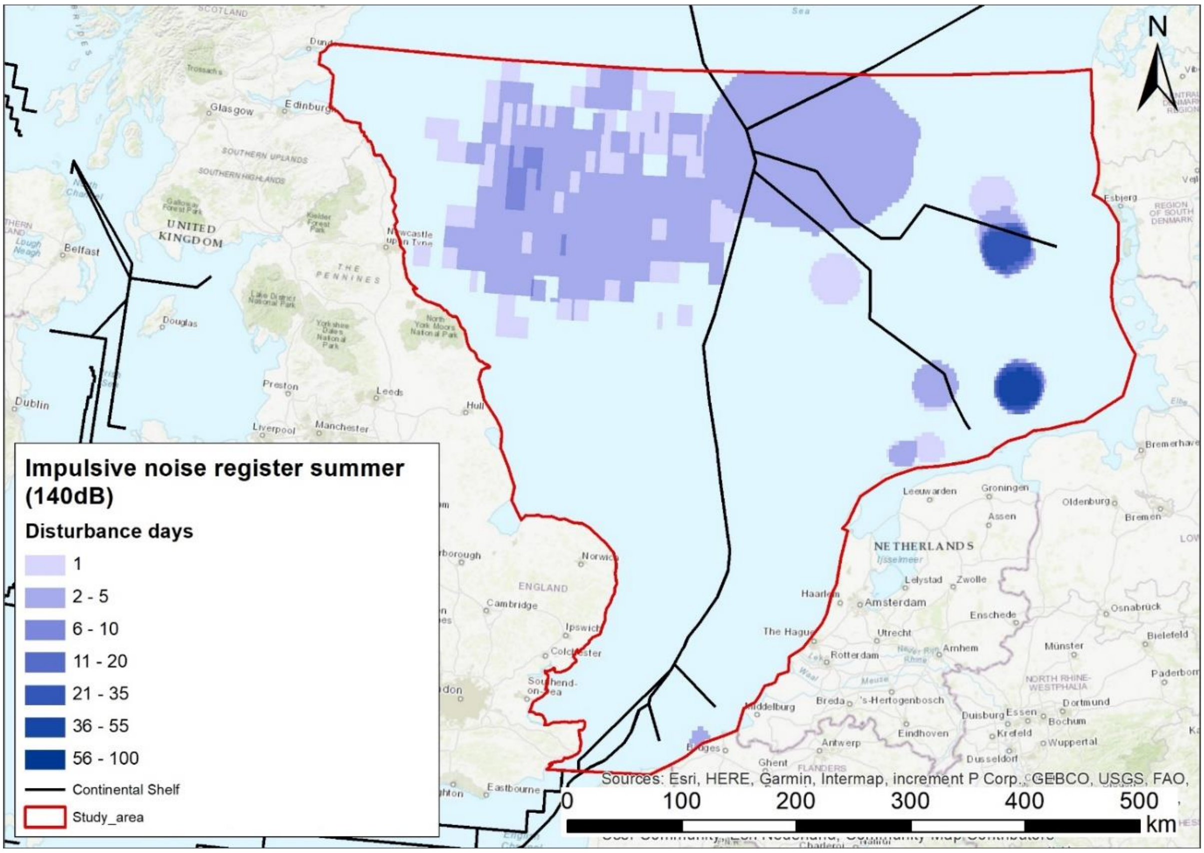

Figure 30: Disturbance days within the study area as a result of the total sum of noise sources for summer based on the 140 dB disturbance threshold used for Harbour porpoises. Data shown are data from the Impulsive Noise Register.

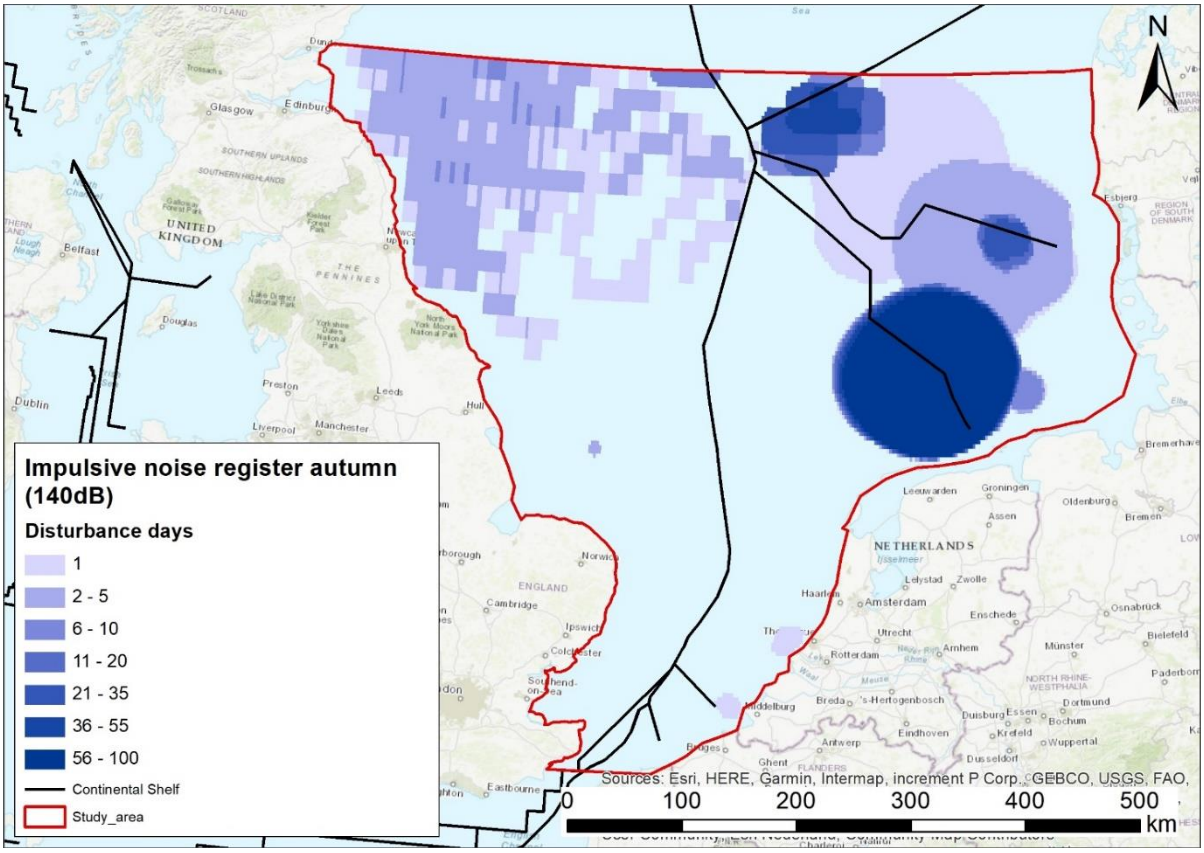

Figure 31: Disturbance days within the study area as a result of the total sum of noise sources for autumn based on the 140 dB disturbance threshold used for Harbour porpoises. Data shown are data from the Impulsive Noise Register.

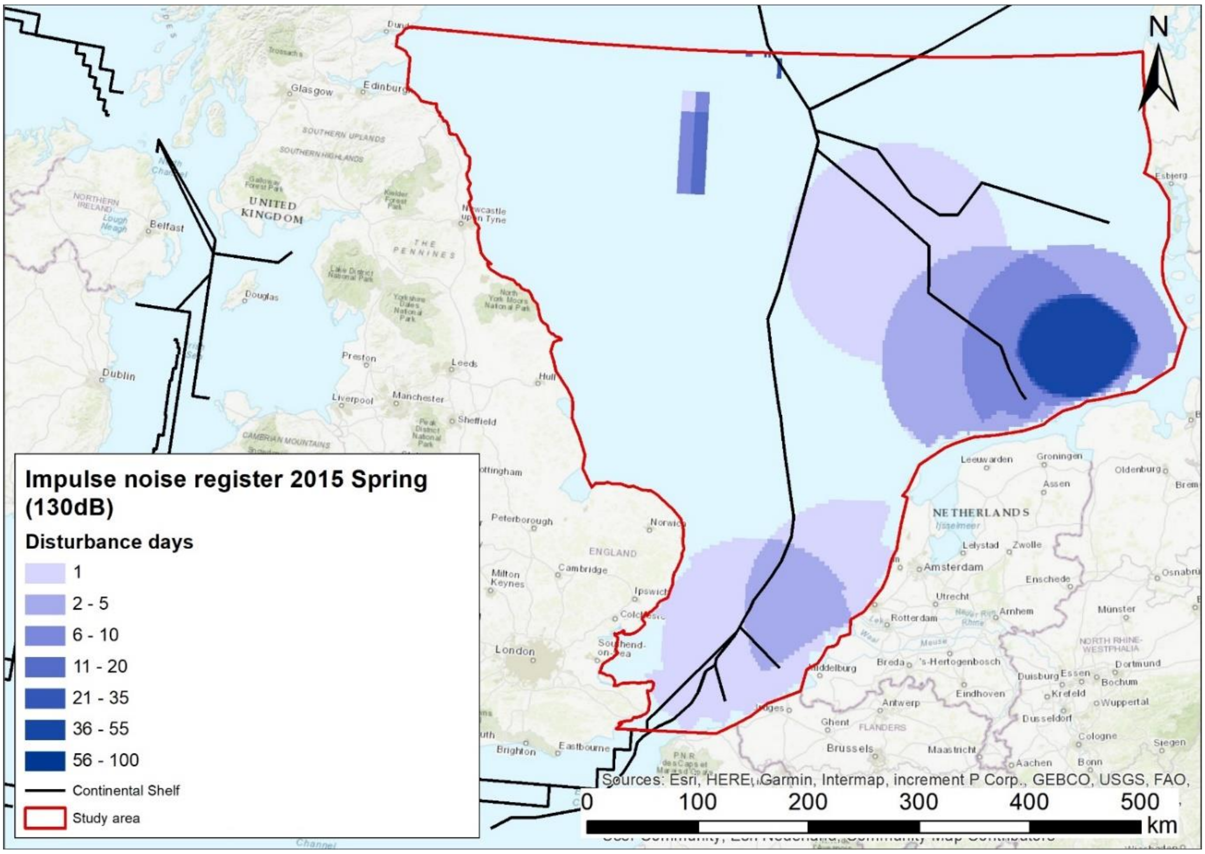

Figure 32: Disturbance days within the study area as a result of the total sum of noise sources for spring based on the 130 dB disturbance threshold used for Cod. Data shown are data from the Impulsive Noise Register.

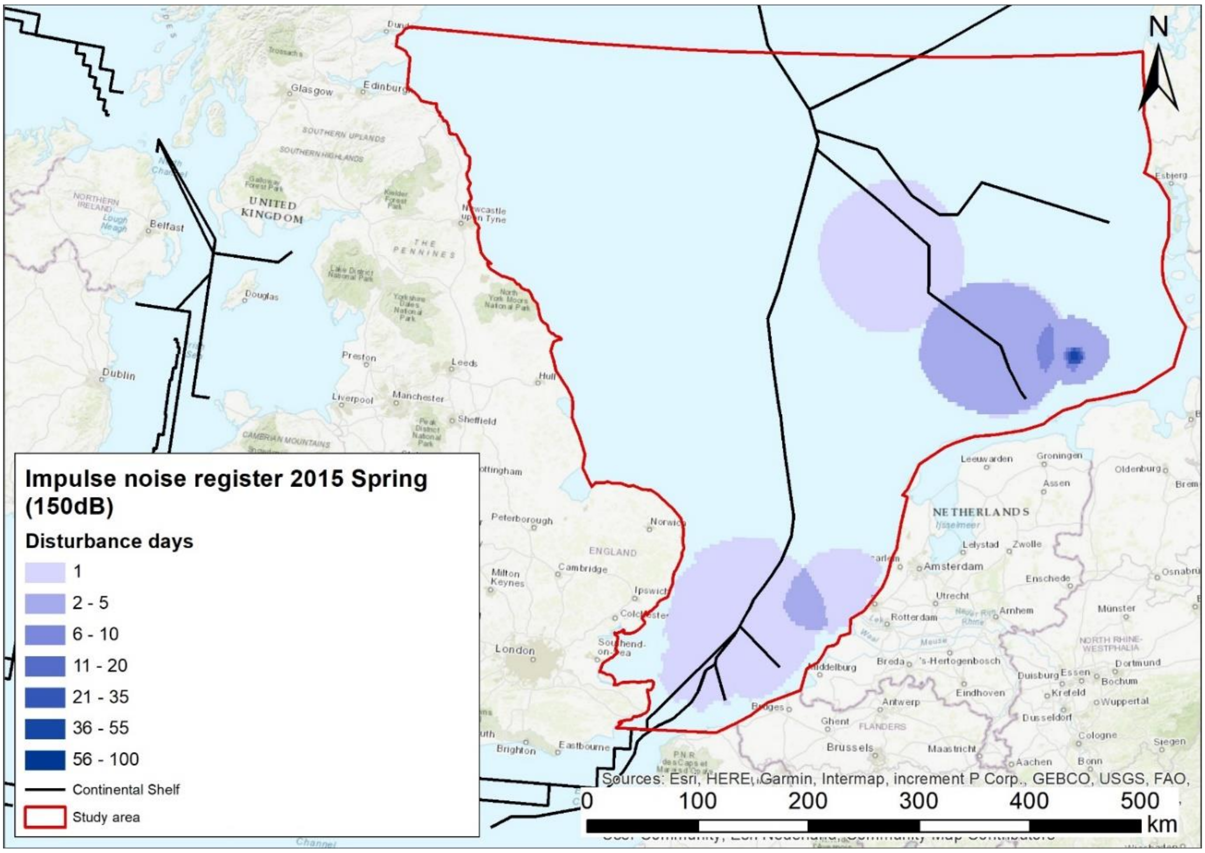

Figure 33: Disturbance days within the study area as a result of the total sum of noise sources for spring based on the 150 dB disturbance threshold used for Cod. Data shown are data from the Impulsive Noise Register.

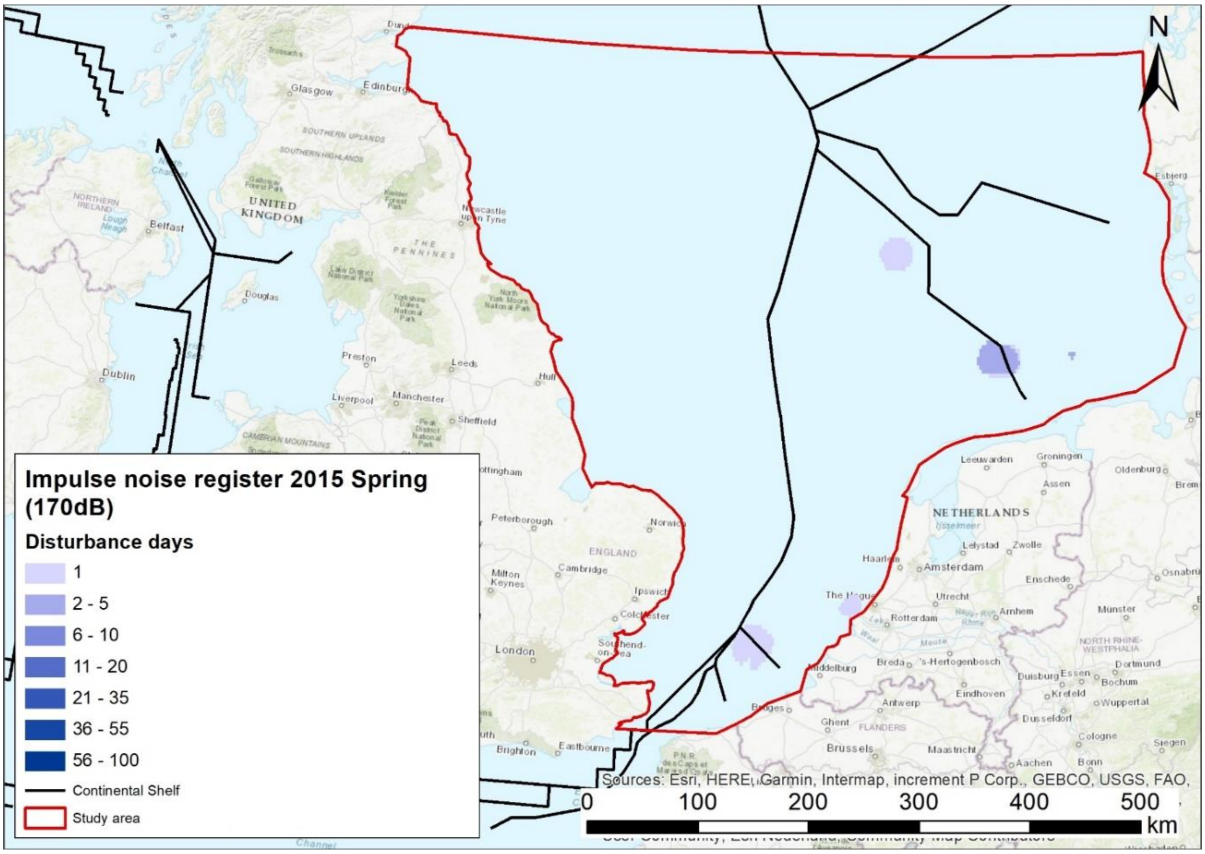

Figure 34: Disturbance days within the study area as a result of the total sum of noise sources for spring based on the 170 dB disturbance threshold used for Cod. Data shown are data from the Impulsive Noise Register.

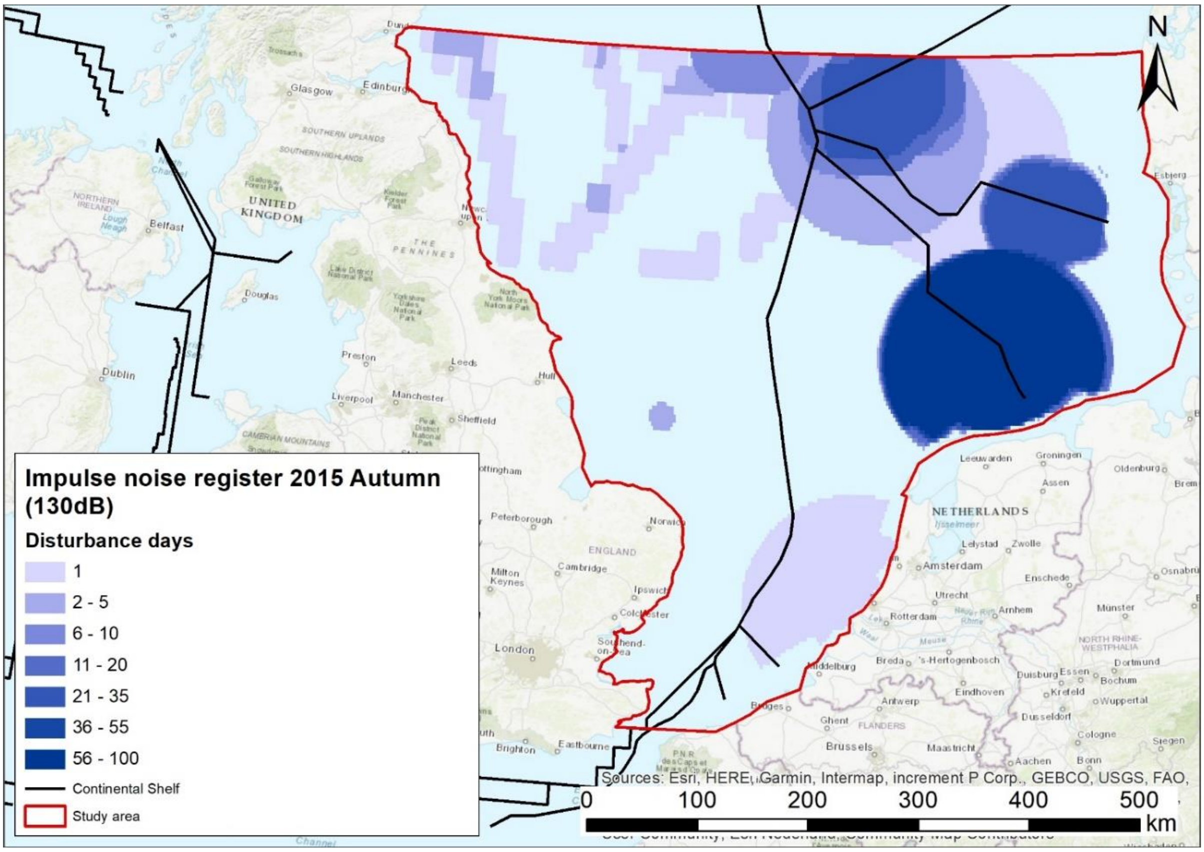

Figure 35: Disturbance days within the study area as a result of the total sum of noise sources for autumn based on the 130 dB disturbance threshold used for Cod. Data shown are data from the Impulsive Noise Register.

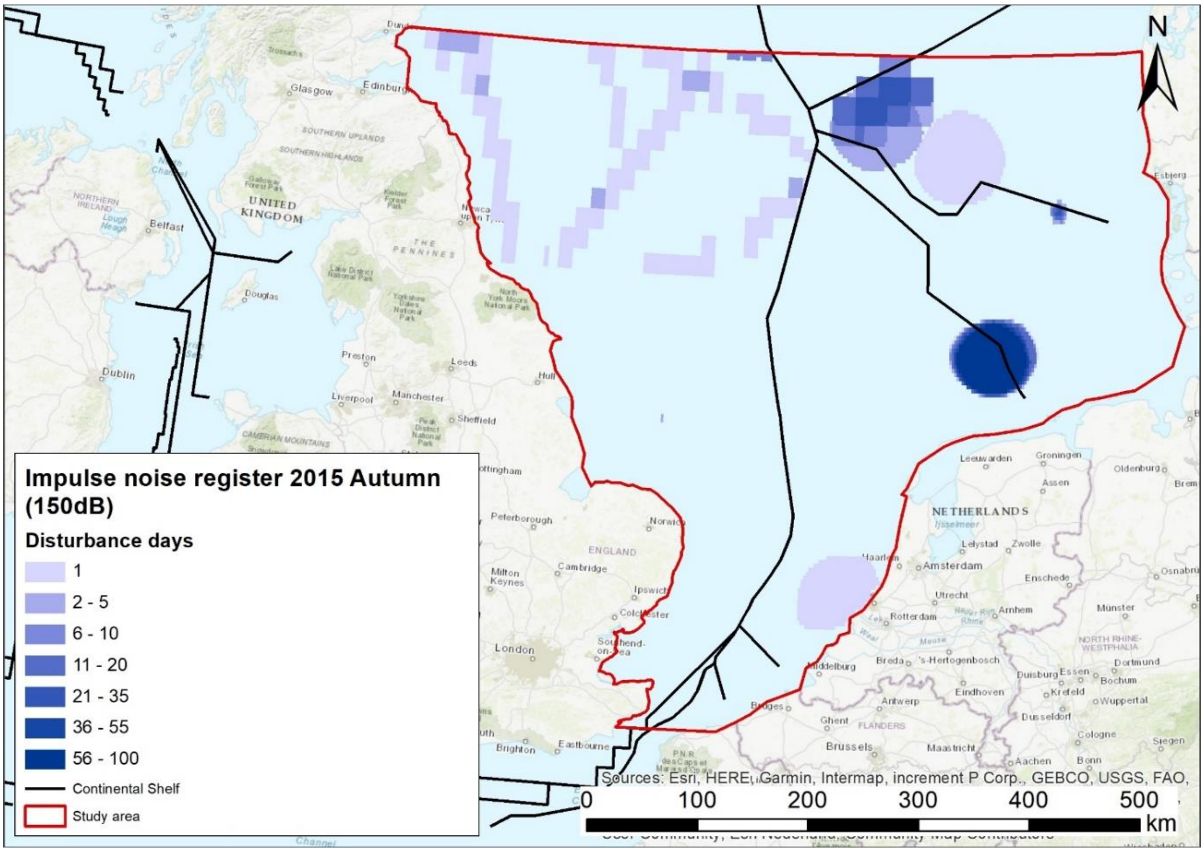

Figure 36: Disturbance days within the study area as a result of the total sum of noise sources for autumn based on the 150 dB disturbance threshold used for Cod. Data shown are data from the Impulsive Noise Register.

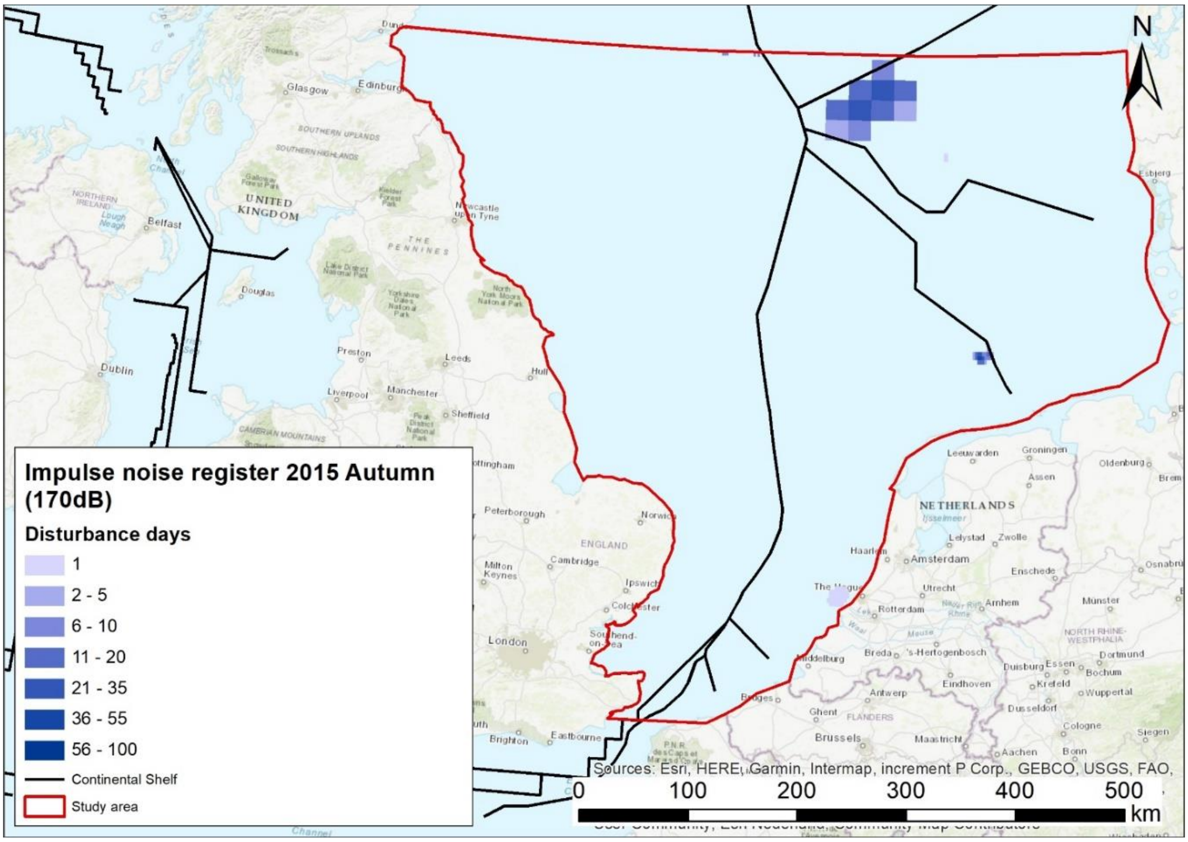

Figure 37: Disturbance days within the study area as a result of the total sum of noise sources for autumn based on the 170 dB disturbance threshold used for Cod. Data shown are data from the Impulsive Noise Register.

8.4.2 Propagation model selection

8.4.2.1 Model selection

Both generic uncertainties and uncertainties about animal location influence the model selection.

Generic uncertainties: Shallow-water propagation modelling is well-advanced, but the models in general suffer from uncertainties in environmental input parameters. This applies in particular to bottom sub-layering, and depth-dependence of geo-acoustic parameters leading to frequency dispersion, which strong affects propagation of low frequency sounds. This is especially relevant when predicting broadband SEL for broadband impulse sounds such as airgun arrays, pile driving and underwater explosions. Although recent approaches that estimate these parameters based on literature values (De Jong et al., 2019) provide important improvements in the prediction at large distances, in order to assess the generality of these assumptions further evaluation is required.

Animal location: The predicted sound levels need to be predicted at typical dive depths of the animal. Animal distribution in the water column depends on the species ecology (e.g. benthic, pelagic). Weighting of sound levels using dive distribution data for marine mammals and fish species can be used in principle to account for differences in depth distributions but have a limited effect on predicted effect distances for shallow water environments (SONIC, 2015). Given the shallow environment of the North Sea, the three indicator species can be considered to use the entire water column in the shallow. In the examples used here, maximum sound levels in the water column were considered.

8.4.2.2 Applicability of models

The applicability of acoustic models for predicting large-scale disturbance for a large number of sources is summarized in Table 7. In principle, acoustic model approaches are available for the source types contained in the Impulse Noise Register, however several models still require further development and validation in order to reliably predict disturbance at distances of 10 to 40 km. In addition, the lack of detail in the Impulse Noise Register limits the accuracy of predicted sound distribution produced by impulsive sound sources. More effort should be put into either improving the amount of detail or identifying representative sound sources for each category. Generic observed disturbance distances may provide a temporary solution (Merchant et al. 2018b), but due to the lack of observations in different environments and source categories it cannot be ascertained that the extrapolation is precautionary, or overly precautionary.

Table 5: Summary of availability and applicability of source and propagation models for sources contained in the Impulse Noise Register.

Source type	Impulse noise register	Source model	Sound propagation model	Modelled acoustic quantity	Model applicability
Pile driving	• Generic hammer model parameters need to be estimated; • Generic assumptions need to be made on mitigation effectiveness	available	Available	SEL	• Broadband SEL well predicted; • Uncertainties in geo-acoustic parameters; • Uncertainties in high frequency content; • Generic assumption mitigation measures; model validation foreseen in 2020/2021.
Explosions		available	Available	SEL	• Limited validation of SEL at large distance; • Uncertainties in geo-acoustic parameters
Airgun array	• Representative airgun array configurations need to be identified • Only coarse knowledge on source location.	available	Available	SEL	• Broadband SEL well predicted up to 5 km; • Significantly underpredict SEL at distances > 5 km; • Improve high frequency in source model; • Uncertainties in geo-acoustic parameters.
Sonar	• No information on source properties; • Only coarse knowledge on source location.	available	Available	SPL	Currently being evaluated by NL Ministry of Defence.
Generic source	Representative source need to be identified for each category.	available	Available	SEL or SPL	No validation with field data;

8.4.3 Maps per sound source

8.4.3.1 Sonar

Sonar sound sources are typically well characterized, and sound propagation models for shallow and deep water are readily available (e.g. Collins, 1993; Porter & Bucker, 1987; Porter & Reiss, 1984) and can be used to predict disturbance in marine mammals and fish (Sivle et al., 2012; Von Benda-Beckmann et al., 2019). However, due to classification issues, limited information on sonar source properties and location is available in the Impulse Noise Register. Only the number of days of sonar presence is reported by a few countries. Without further information generic assumptions need to be made. For instance, to be precautionary, one could assume only generic high sonar source level until further information becomes available, which likely leads to an overall overestimation of the predicted impact for the activities reported to the register. For sonar no maps are shown as no noise data are available for Sonar or acoustic deterrents for the summer of 2015.

8.4.3.2 Airgun arrays

Airgun arrays are categorized by their zero-to-peak source level in the Impulse Noise Register (Table 6). In the Von Benda-Beckmann et al. (2017) report, a simplified approach was proposed: convert the zero-to-peak source level (SL_{zp}) of the array in the downward beam into a dipole energy source level (SLE_{dp}) of the array, which is then used to estimate the sound exposure level using expressions for dipole source propagation. This method is not well suited for predicting frequency dependence of the sound propagation, which may be relevant when considering frequency-weighted risk criteria (Tougaard et al, 2015b; de Jong & von Benda-Beckmann, 20187). The 2018 report suggests that more advanced modelling would be possible when the INR would provide volumes, firing pressures and depth, and type of each airgun in the array. To use acoustic models, the array configuration needs to be specified. Some examples of typical airgun arrays sizes and their corresponding zero-to-peak source levels are provided in Table 5. To overcome the lack of detail in the Impulse Noise Register required to specify the array geometry, a practical solution would be to identify a typical array configuration for each category (very low, low, medium, high), which could be readily obtainable from publicly available source, such as DINO (https://www.dinoloket.nl/en). Whether representative configurations that cover can be found needs to be investigated.

Table 6: Airgun arrays: examples of representative array sizes for different source categories (expressed in total airgun volume of the array)

Level	Zero to peak source level / dB re 1 µPa	Comment
Very Low	209-233	
Low	234-243	160 in^3 ; 750 in^3
Medium	244-253	
High	> 253	4139 in^3

The validity of acoustic models for airgun arrays in shallow water, and predictions of the broadband levels have received relatively little attention (Ainslie et al. 2016). In Prior et al. 2020 a comparison was made of an airgun model which combines the AGORA (a publicly available airgun array source model; Sertlek & Ainslie, 2015) and a hybrid normal-modes/ energy flux model (Aquarius v3; Sertlek & Ainslie, 2015; de Jong et al. 2019; Whyte et al., 2020), to acoustic measurements of two 3D surveys in the North Sea. This comparison indicated that predicted levels of broadband SEL with this model are consistent with acoustic measurements at distances up to approximately 5 km (Figure 38). However, this model approach substantially underestimates the levels at larger distances (Prior et al. 2020).

There could be several possible explanations for this underestimation: limited calibration of the AGORA source model, uncertain geo-acoustic parameters, not modelled propagation through the seabed, or assumptions made in the source/propagation model coupling. Model predictions at higher frequencies (> 1000 Hz) appear to by systematically lower than high frequencies by 10 – 20 dB (Prior et al. 2020). As such, frequency weighted values (Tougaard et al. 2015; de Jong & von Benda-Beckmann, 2018) would be significantly underestimated. Therefore, acoustic airgun array models require more improvements in order to reliably predict disturbance at large distances. Figure 39 shows the available data on impulse noise for the summer of 2015 applying the 140dB exposure threshold relevant for harbour porpoises.

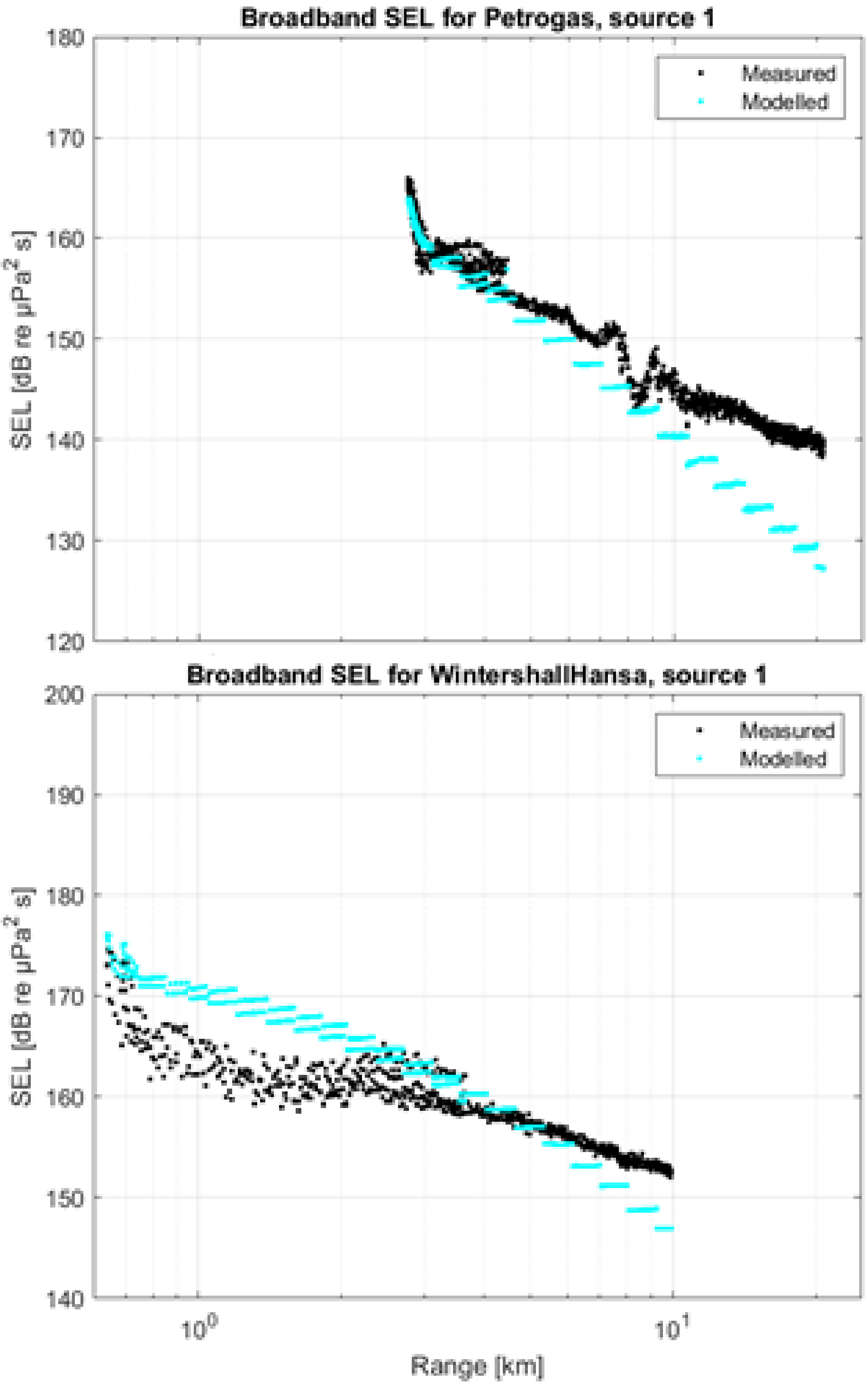

Figure 38: Comparison measured (black) and modelled (cyan) broadband SEL produced by two 3D airgun array surveys on the North Sea (of broadband SEL Prior et al. 2020). A reasonable match can be achieved within the first 5 km, however the models systematically underestimate the SEL at large distances.

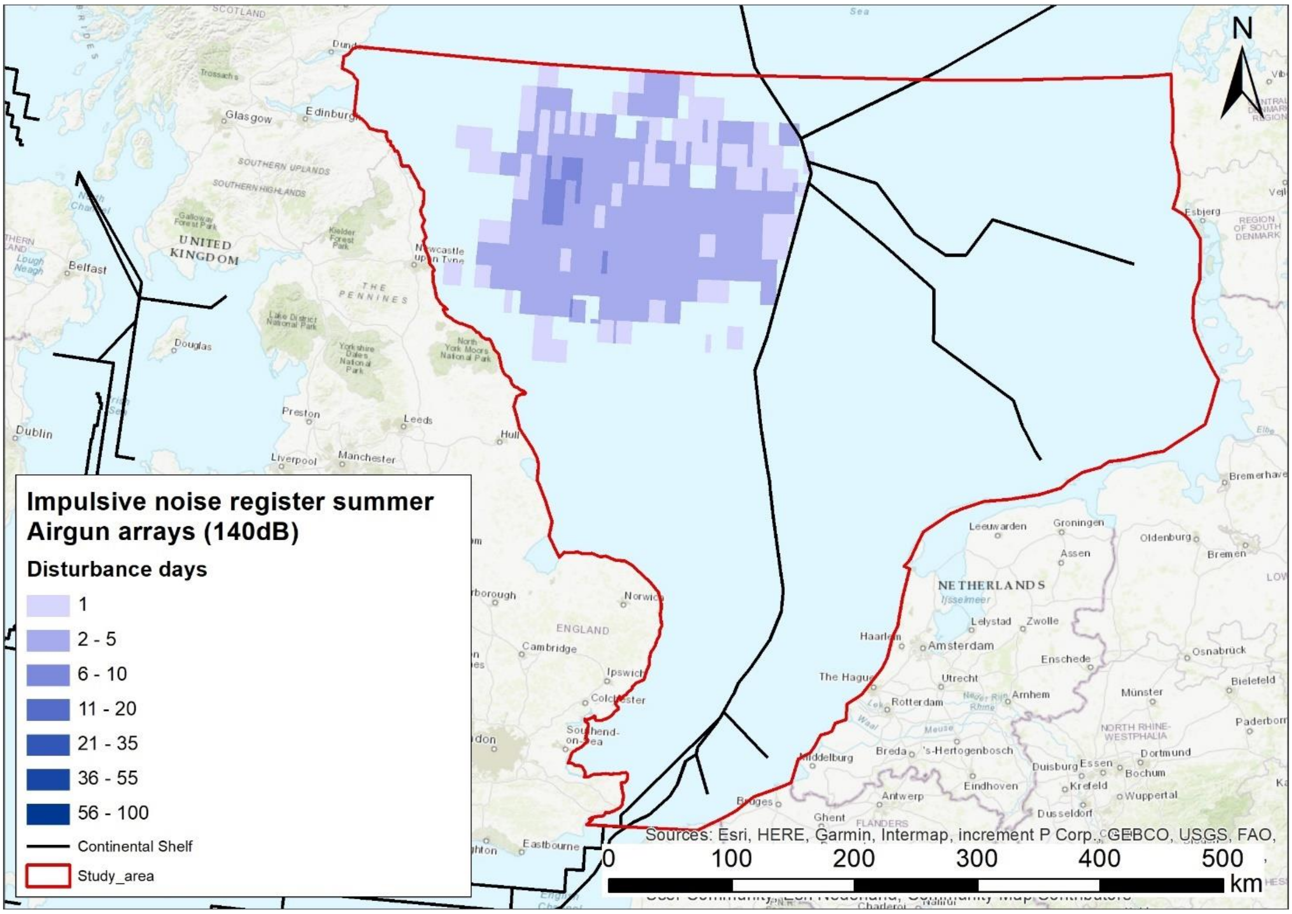

Figure 39: Disturbance days within the study area as a result of Airgun arrays for summer 2015 based on the 140 dB disturbance threshold. Data shown are data from the Impulsive Noise Register.

8.4.3.3 Generic impulsive sound sources

The generic impulse sound source category encompasses a broad range of sound source (e.g. sparkers, boomers, sub-bottom profilers). Although relatively little information is available about these different sources (only energy source level, but no frequency content), source parameters can be estimated based on literature values (Crocker et al. 2019). Examples of some systems used during offshore wind construction are provided as an example in Table 6. However, it is unclear what representative sources are, in combination with a wide range of sources and operating frequencies.

Sources contained in this category operate at a wide range of frequencies (up to 100 kHz), which has a major influence how sound propagates to large distances. In addition, this makes it necessary to consider the frequency dependence on animal sensitivity to these sound sources. Furthermore, field data to validate both model predictions are lacking. In the Netherlands the use of these instruments is not regulated and no data on their use is available and uploaded to the noise registry.

Table 7: Generic explicitly impulsive sound source

Level	ESL / dB re 1 µPa²s	Comment
Very low	186-210	Parametric sub-bottom profiler Multi-channel sparker
Low	211-220	
Medium	221-230	
High	> 230	

8.4.3.4 Underwater explosions

The shallow water model for underwater explosions used in von Benda-Beckmann et al. (2015b) can be directly applied to TNT equivalent charge mass reported in the register. Recent comparison to large distance (up to 12 km) explosions indicate that the model provides reasonable estimates of the broadband and high frequency weighted SEL, but that uncertainties in the geo-acoustic properties of the seafloor have a significantly influence low frequency contribution at large distances (TNO, unpublished). Figure 40 shows the available data on impulse noise from underwater explosions for the summer of 2015 applying the 140dB exposure threshold relevant for harbour porpoises.

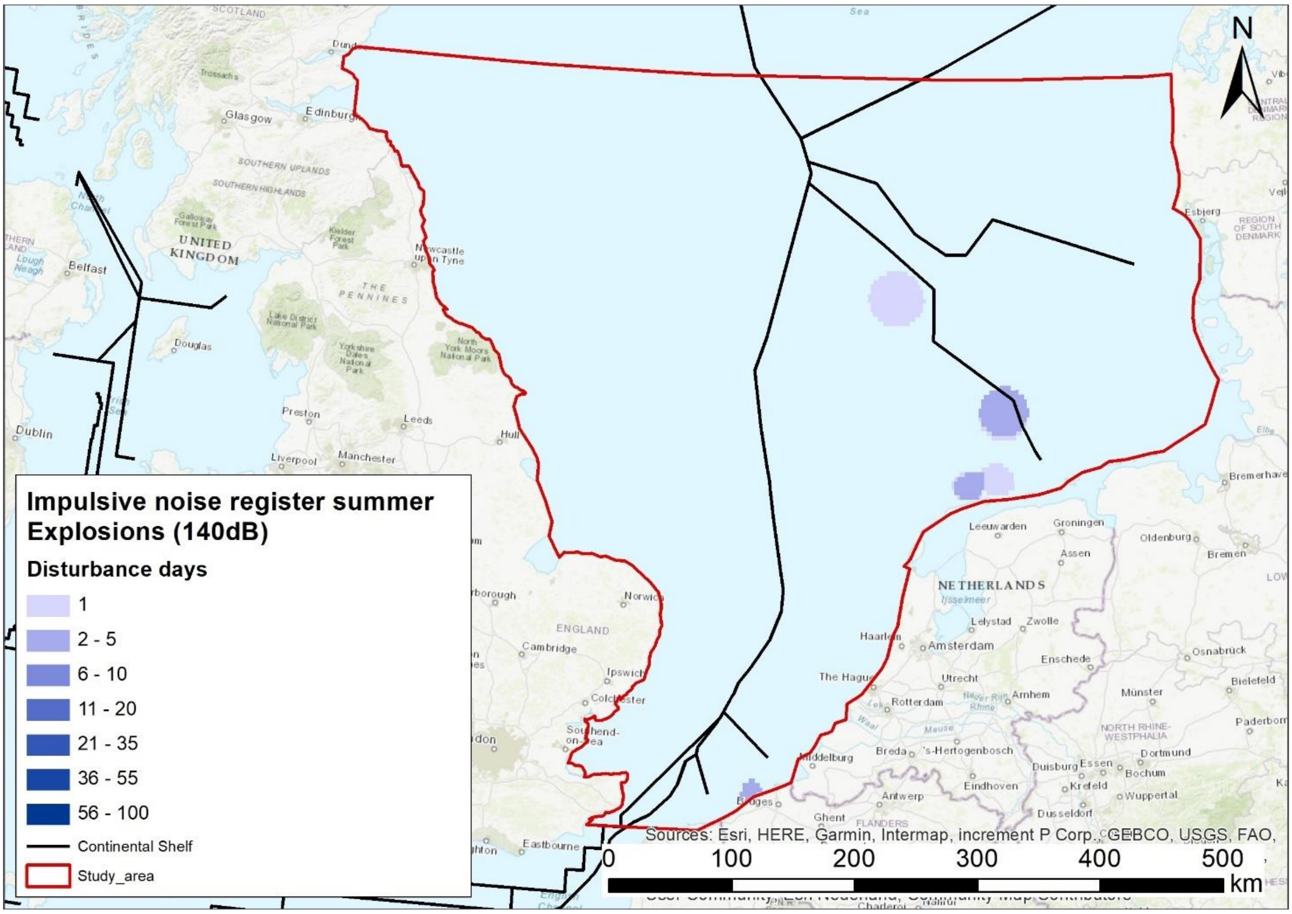

Figure 40: Disturbance days within the study area as a result of underwater explosions and acoustic deterrents for summer 2015 based on the 140 dB disturbance threshold. Data shown are data from the Impulsive Noise Register.

8.4.3.5 Pile driving

The Aquarius model has been recently updated to enable prediction of broadband SEL for unmitigated pile driving (de Jong et al. 2019). This model provides good agreement with measured broadband SEL of unmitigated pile driving sounds up to 70 km (Figure 41). The pile driving model relies on model parameters: hammer energy (available from the Impulse Noise Register), but also additional parameters such as pile geometry (length, diameter and wall thickness) and hammer mass, anvil mass and contact stiffness at the hammer-anvil interface. Since no specific information is available per project in the Impulse Noise Register, generic assumptions based on previous piling projects need to be used instead (Heinis et al. 2019). Model uncertainties in the hammer driving function lead to uncertainties in the SEL predicted at higher frequencies (Figure 41), which would lead to an underprediction of harbour porpoise frequency-weighted SEL. Modelling the mitigation effectiveness during pile driving is still a field of ongoing research. For the examples shown in von Benda-Beckmann et al. 2017, a 10 dB broadband reduction is assumed, which is a lower limit to the 10-20 dB that can typically be achieved with a combination of near-field mitigation measures, and double big bubble curtains. Figure 43 shows the available data on impulse noise for the summer of 2015 applying the 140dB exposure threshold relevant for harbour porpoises.

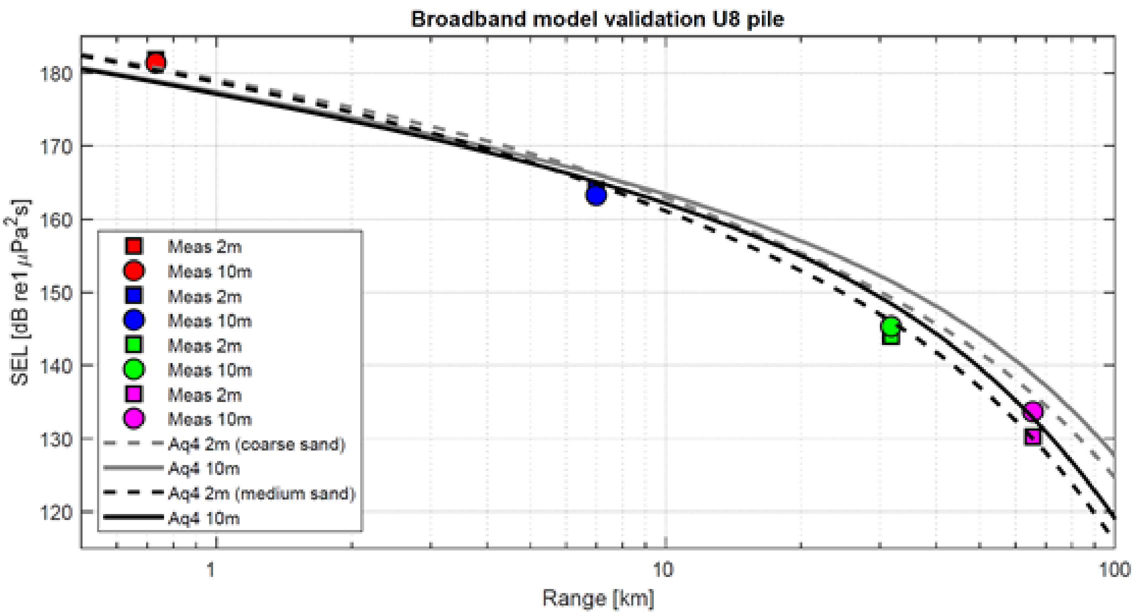

Figure 41: Model-data comparison of broadband SEL of the pile driving acoustic model with measurements of the Gemini wind farm construction without noise abatement technology, showing that good agreement is obtained up to large distances from the piling location (de Jong et al. 2019).

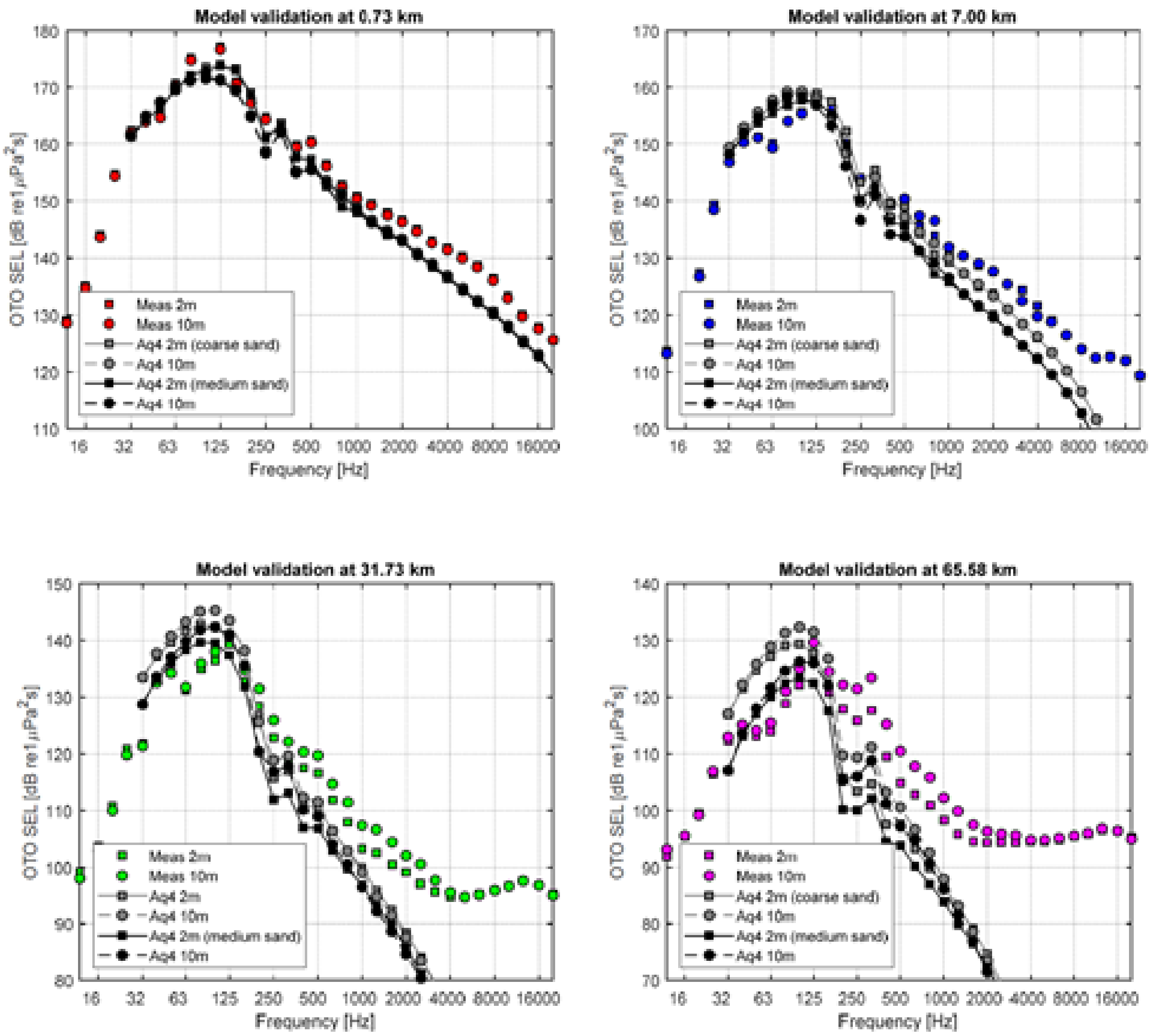

Figure 42: Comparison of third-octave-band (10-base) SEL measurements of the pile driving sound with model predictions. Measurements of the Gemini wind farm construction without noise abatement technology, showing that good agreement is obtained up to large distances from the piling location at lower frequencies that dominate the broadband SEL, but shows an underestimation at higher frequencies, due to uncertainties in the hammer model (de Jong et al. 2019).

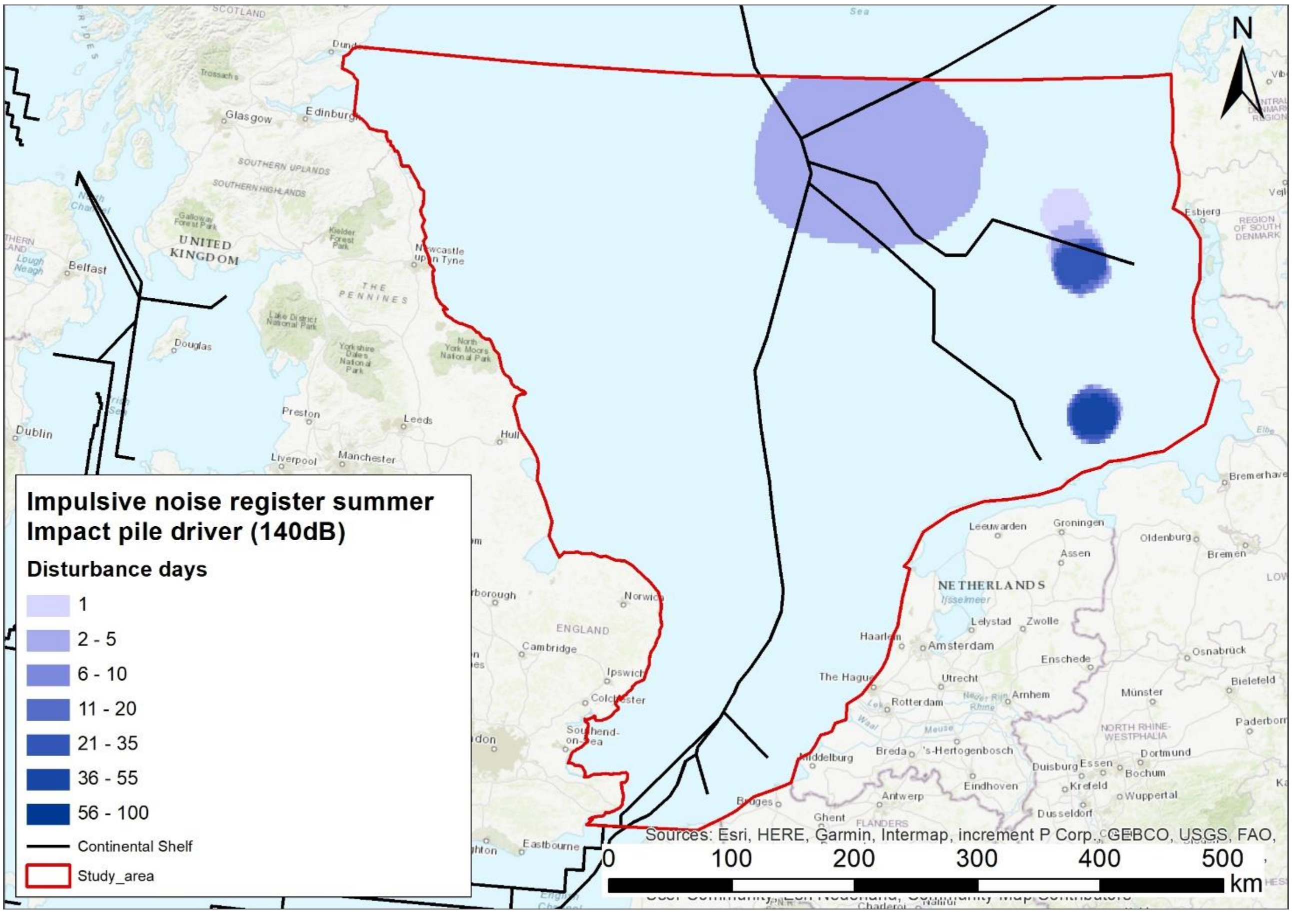

Figure 43: Disturbance days within the study area as a result of impact pile drivers for summer based on the 140 dB disturbance threshold.

9 STEP 6: EXPOSURE MAPS

In the previous chapters step 0: define stressor, step 1: select indicator species, step 2: define assessment area, step 3: define temporal resolution, step 4: Distribution or habitat map(s) and step 5: pressure map(s) were completed. In this chapter step 6: exposure/risk maps is described (Figure 44Figure 15). This step was taken simultaneously with step 7/8 that will be discussed in the next chapter.

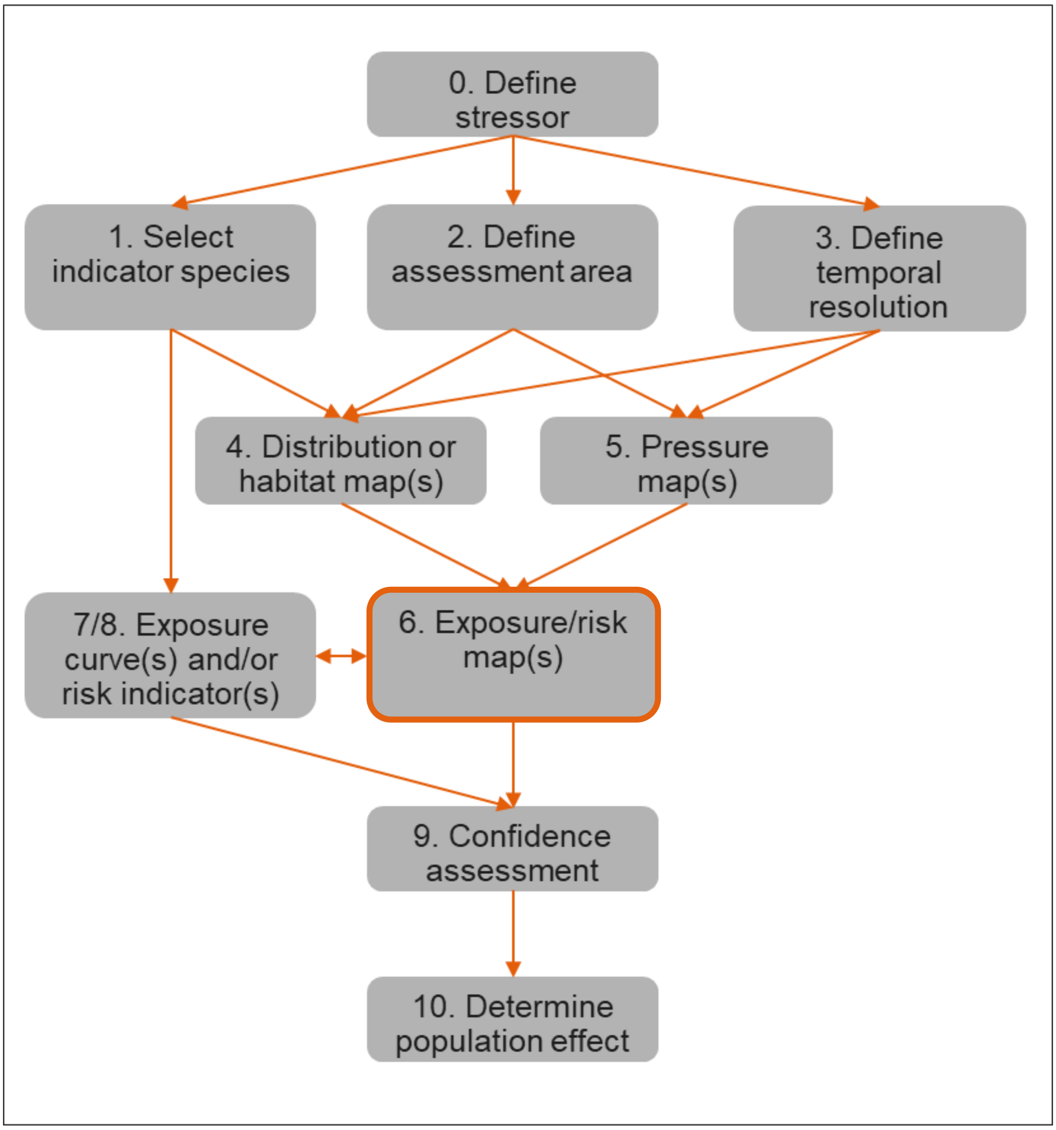

Figure 44: Stepwise approach, current step (highlighted orange).

To summarize once again: The identified stressor is impulsive noise. The selected indicator species are harbour porpoise, Atlantic cod and harbour seal, the assessment area is an extension of the Dutch North Sea and the temporal resolution is defined as 2015 to 2017.

In chapter 7 (step 4) the distribution of the three indicator species was discussed based on a combination of observed animal presence and habitat based models. Seal data proved unsuitable for further analyses. In chapter 8 (step 5) the pressure of impulse noise was discussed based on the information concerning noise produced inside of the study area.

Disturbance of the indicator species only occurs when:

- The species is present within the study area
- The species is sensitive to the specific disturbance factor
- The disturbance factor is present
- The disturbance factor exceeds the trigger threshold value

Therefore, to identify if and for how long a species is disturbed within a certain area, data needs to be available on the presence of a species, the trigger threshold value and the sounds presence and intensity over time. In practice it is impossible to know the number of animals present at a given time. This is especially so for highly mobile species like harbour porpoise and seal. Instead the probability of presence for a species (also referred to as habitat suitability) is used and the indicator resulting from this gives the risk of impact. As explained in chapter 8, the sound presence and sound level, together with the species sensitivity result in an amount of disturbance days for a certain location. When multiplying the disturbance days with the species density it results in the probability of disturbance days for that specific species present in that specific area.

This chapter will discuss the data availability and quality for the selected indicator species and discuss the steps undertaken to obtain exposure maps to the point that was attainable with the current data availability.

9.1 Cod

Data on the presence of cod in the North sea are collected via an international monitoring program called the North Sea International Bottom Trawl Survey (NS-IBTS). This monitoring program is a collaboration between eight countries (Denmark, France, Germany, Netherlands, Norway, UK Scotland, UK England and Sweden). Data from IBTS survey's (and comparable predecessors) have been collected since the first quarter of the 1960's.

Table 8: data requirements to be met to allow for exposure map creation.

Data required	Dataset	Unit	Remarks
Presence within study area	NS-IBTS (North Sea International Bottom Trawl Survey)	n/hour (CPUE, Catch Per Unit Effort)	
Species sensitivity threshold disturbance thresholds	No accurate disturbance thresholds available		SELss 130, 150 and 170 dB re 1 µPa^{2}s used
Noise presence and noise level within study area	Impulsive Noise Register	Disturbance days	

As there are no clear data on the disturbance thresholds for cod, three separate disturbance maps were created applying the three disturbance thresholds discussed in chapter 8 to evaluate the sensitivity of the risk maps to the adopted disturbance thresholds. Maps were created for both seasons in which data were available on the presence of cod.

It is apparent that accurate disturbance thresholds are essential to obtain representative data on sound exposure. When comparing and there is a large difference is disturbed area and severity of the disturbance. In chapter 10 this discrepancy is further visualized and quantified by creating exposure curves and calculating total disturbance.

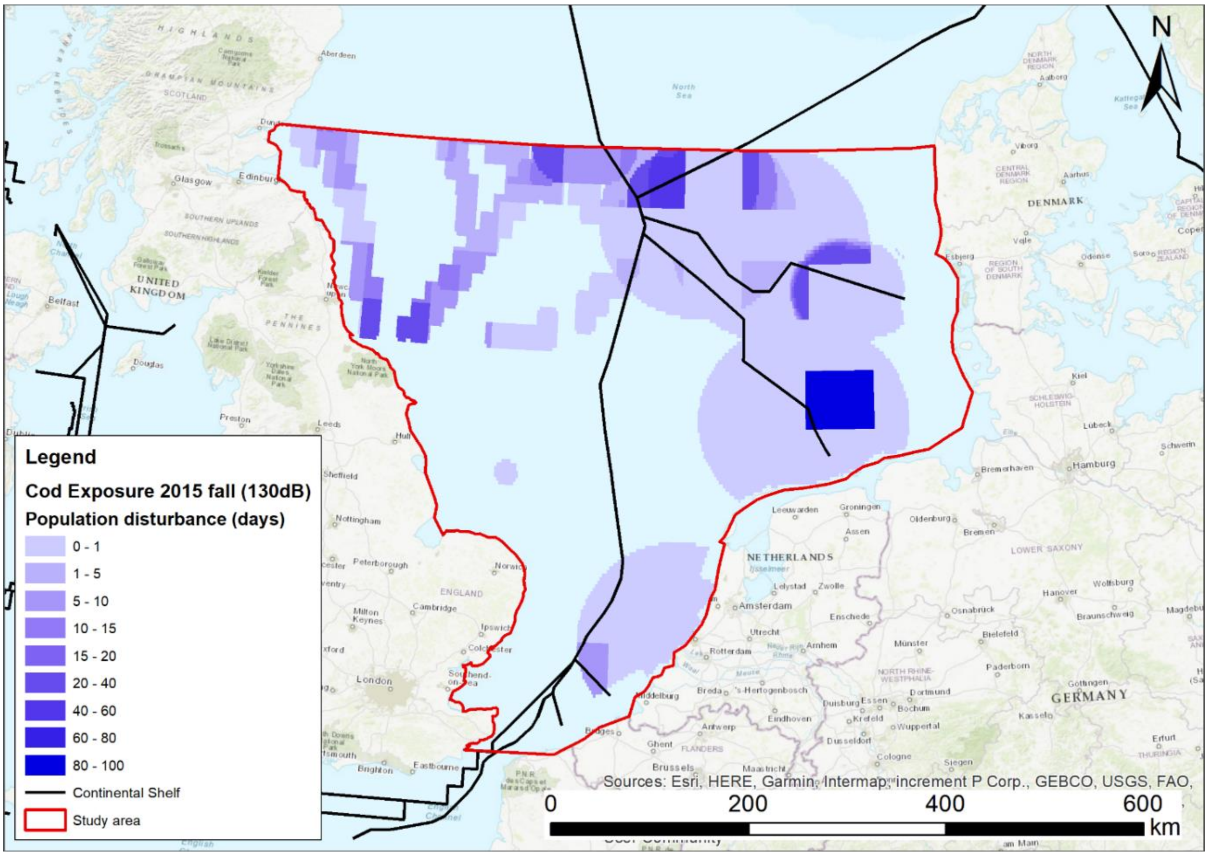

Figure 45: Cod exposure autumn. Data displayed in population disturbance days. Disturbance threshold 130dB.

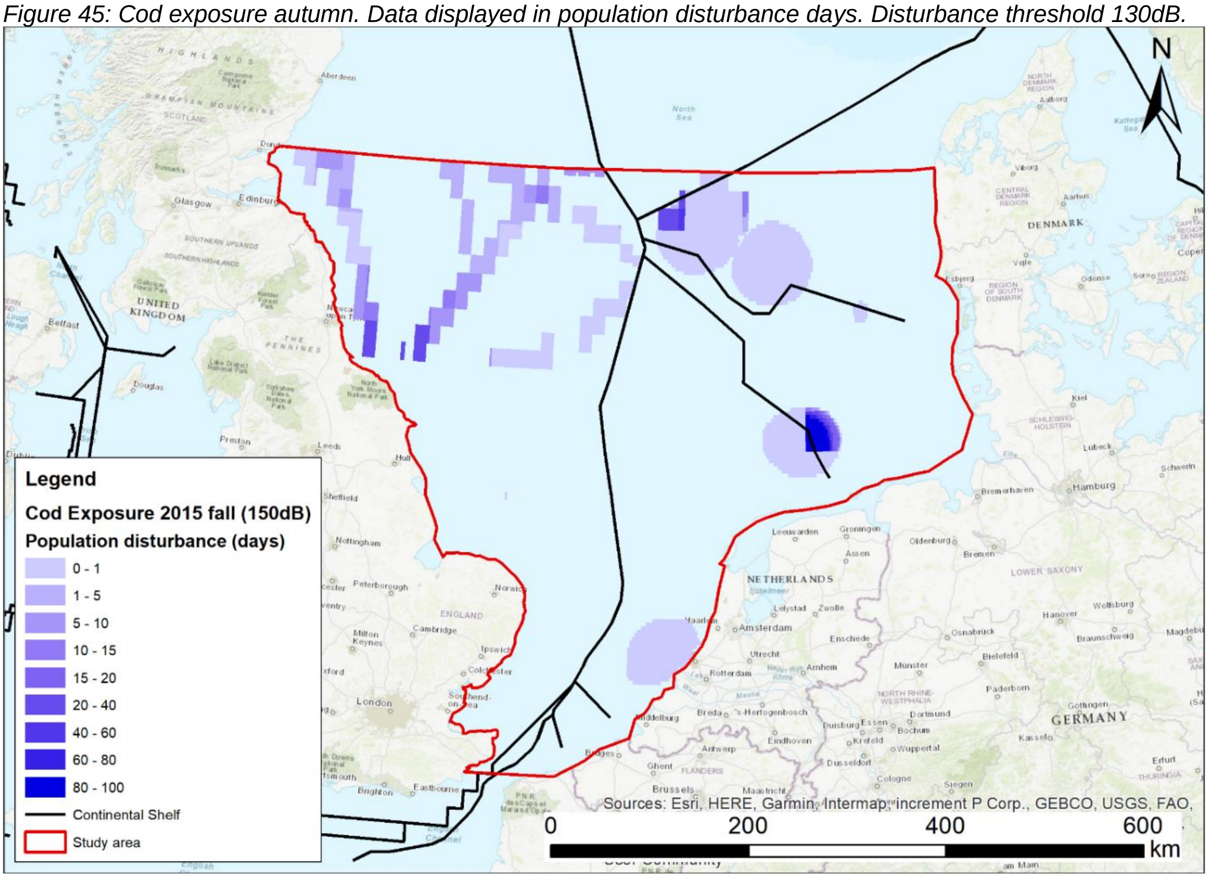

Figure 46: Cod exposure autumn. Data displayed in population disturbance days. Disturbance threshold 150dB.

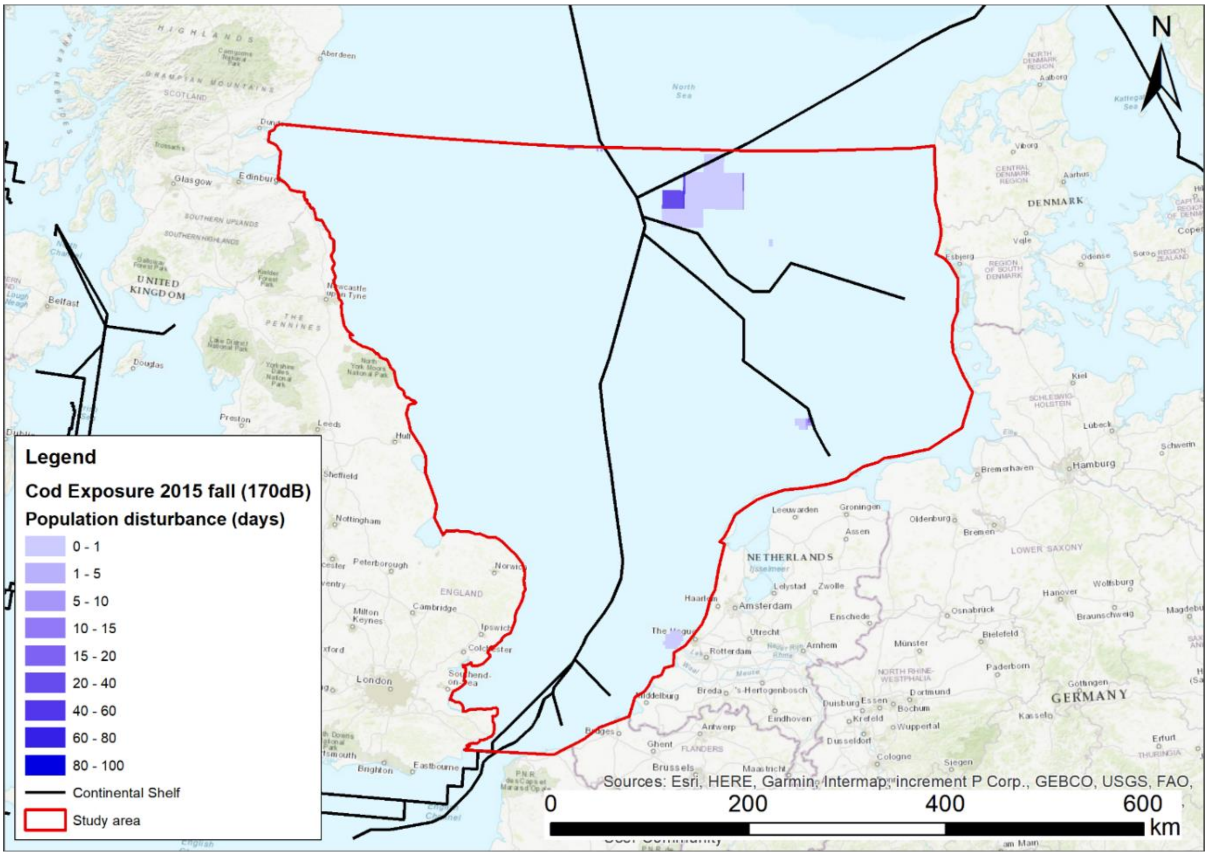

Figure 47: Cod exposure autumn. Data displayed in population disturbance days. Disturbance threshold 170dB.

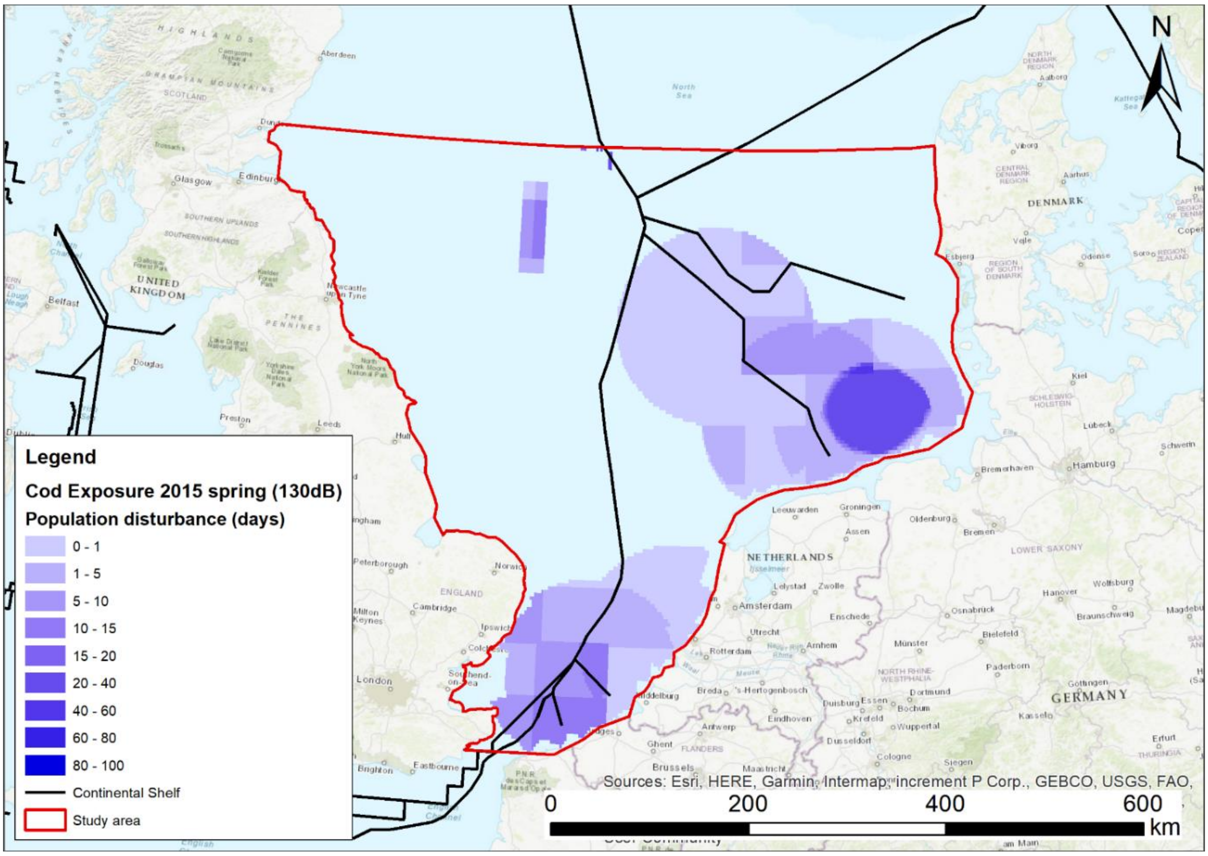

Figure 48: Cod exposure spring. Data displayed in population disturbance days. Disturbance threshold 130dB.

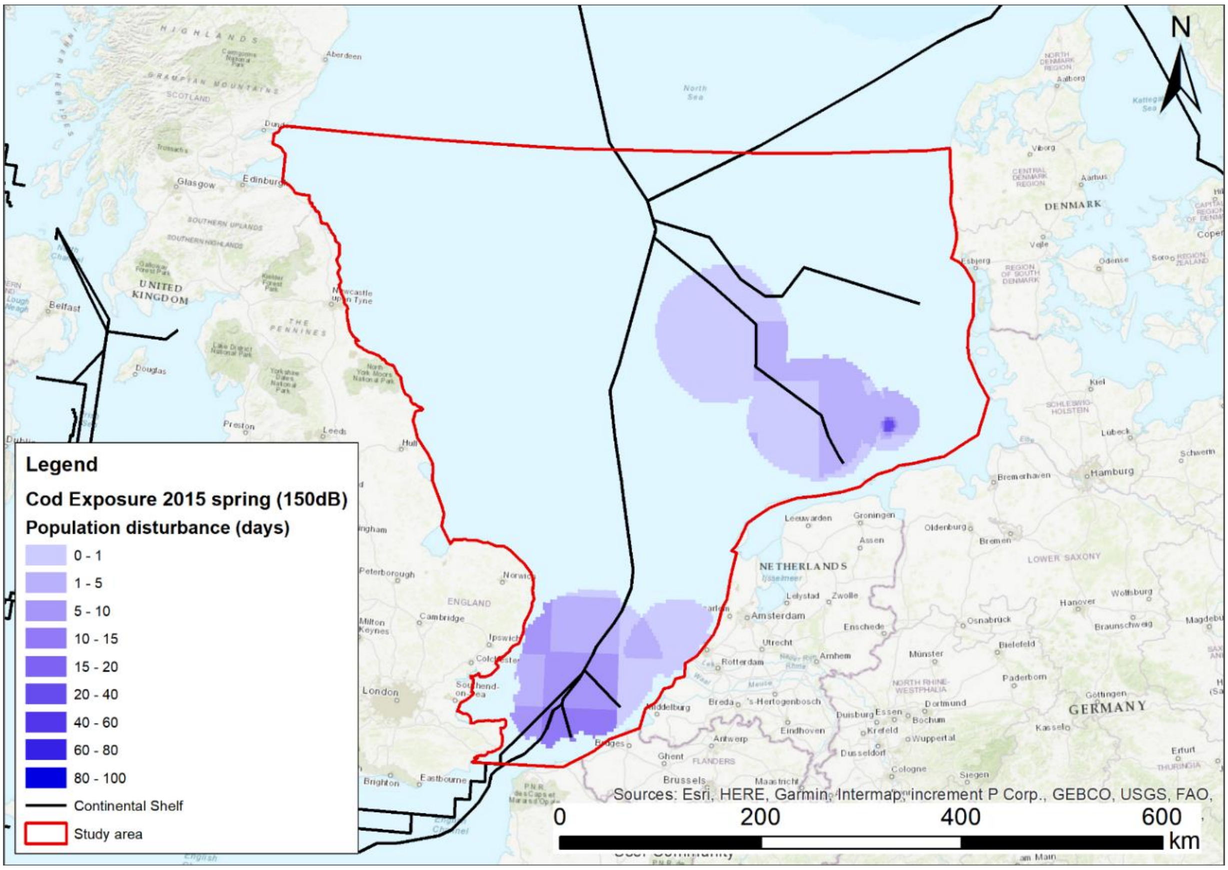

Figure 49: Cod exposure spring. Data displayed in population disturbance days. Disturbance threshold 150dB.

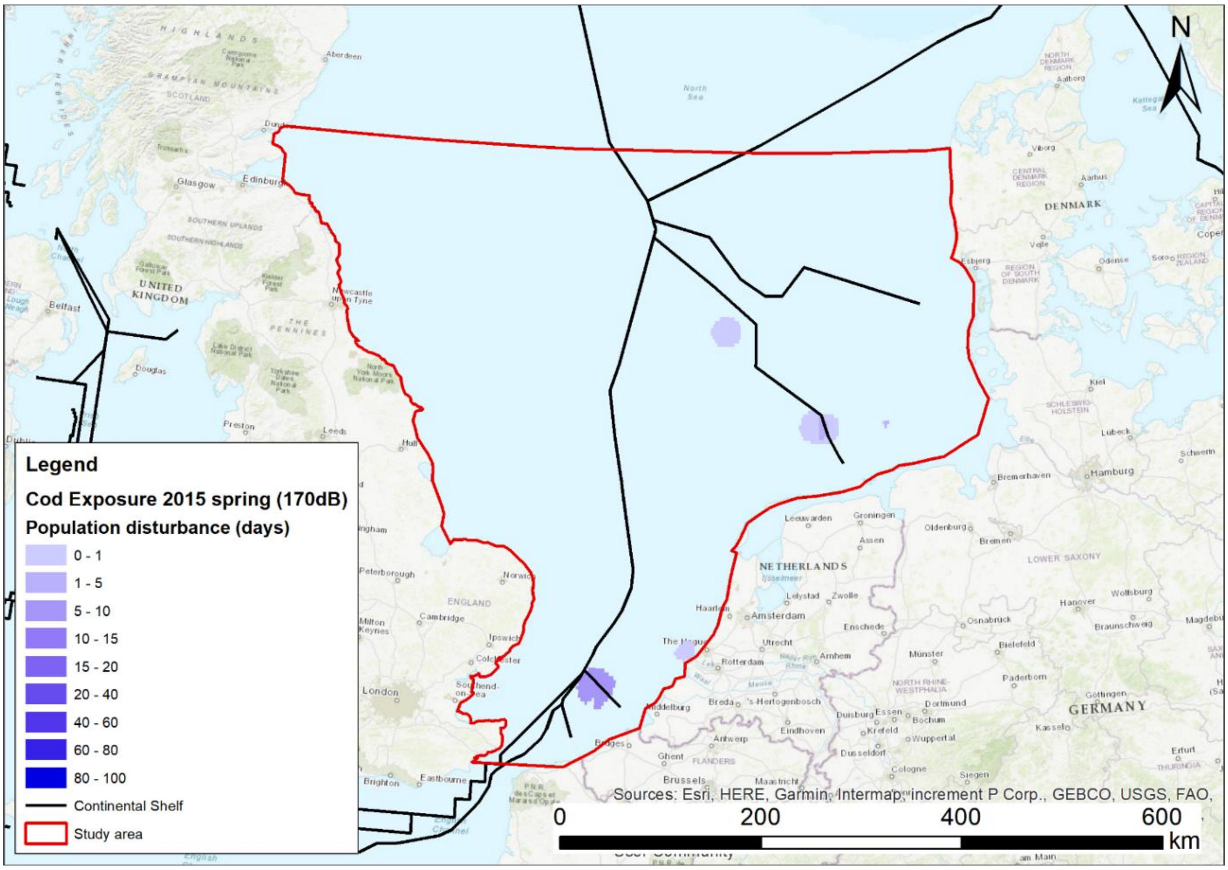

Figure 50: Cod exposure spring. Data displayed in population disturbance days. Disturbance threshold 170dB.

9.2 Harbour porpoise

Data on the presence of harbour porpoise were collected by dedicated aerial visual surveys in the national waters of Denmark, Germany, the Netherlands, Belgium and in adjacent UK areas between 2005–2019. Apart from these survey data, data from so-called SCANS-surveys covering the entire North Sea in 2005 and 2016 (Hammond et al. 2013b, 2017) were also used to model the abundance and distribution of harbour porpoises in the North Sea (Gilles et al., 2016). The model included location, sea surface temperature (SST), day length, distance to sandeel grounds, distance to coast, and average water depth, and predicted harbour porpoise presence in summer (2005-2013 and 2014-2019). Due to a lack of survey coverage in spring and autumn for both periods the predictions from 2005-2013 were used.

Table 9: data requirements to be met to allow for exposure map creation.

Data required	Dataset / data type	Remarks
Presence within study area	Dedicated aerial visual surveys	
Species sensitivity threshold disturbance thresholds	Multiple papers see 8.3.1	
Noise presence within study area	Impulsive Noise Register	Disturbance days

Figure 51, Figure 52 and Figure 53 show the species specific disturbance based on the combination of presence of harbour porpoise and presence of Impulsive underwater noise. These maps display the exposure to the sum of all underwater noise sources. Figure 51 trough Figure 53 display exposure data based on the distribution model output of 2005 up to 2013. Figure 54 contains the Harbour porpoise exposure based on the distribution model output from 2014 to 2019.

When looking at the exposure of harbour porpoises to impulsive sound sources from a legislative perspective it can be beneficial to obtain an understanding of the varying exposure intensities from the different noise sources. This can help create an understanding of which noise sources and activities are constituting the largest pressure on harbour porpoises. For instance, the exposure to impact pile drivers might be different from the exposure to airgun arrays. For this purpose the data from the impulsive noise register were separated by noise source (0) and subsequently combined with the harbour porpoise distribution data to obtain source specific exposure maps. These source specific exposure maps are shown in Figure 55, Figure 39 and Figure 57.

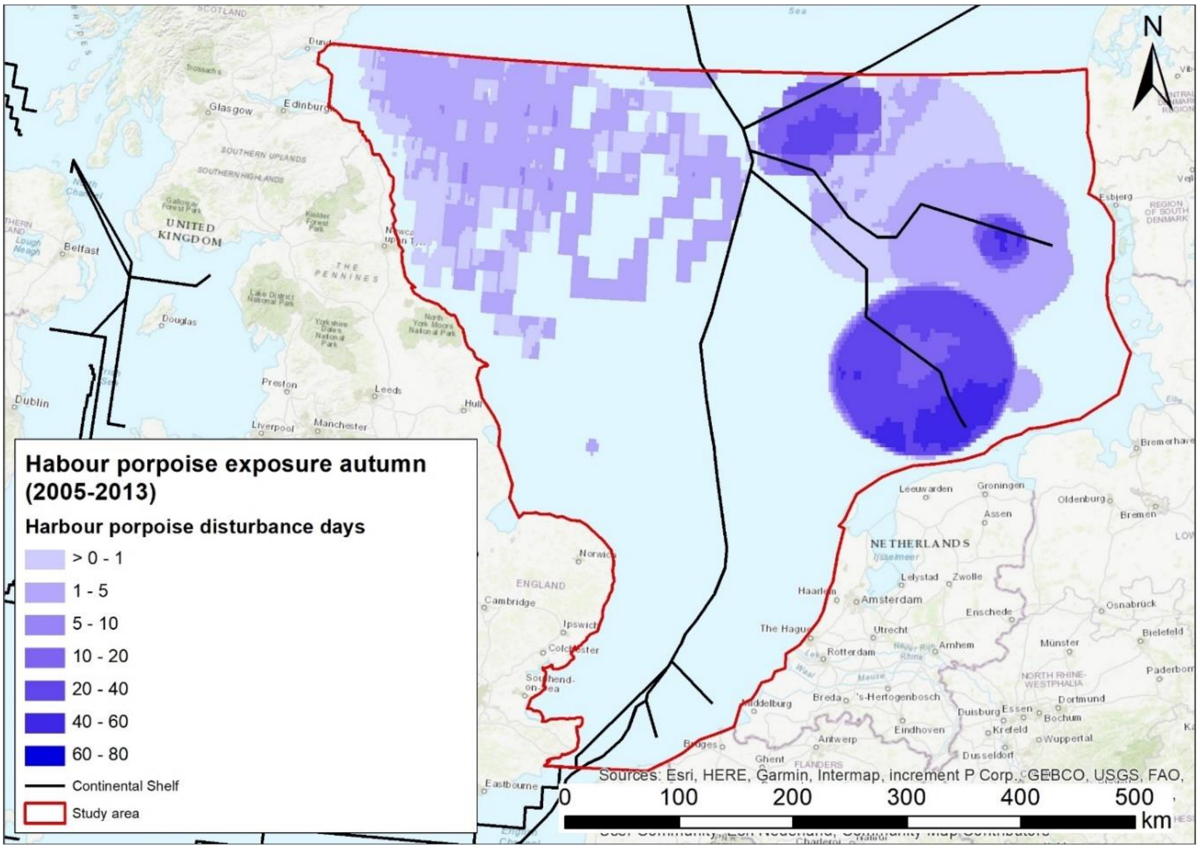

Figure 51: Harbour porpoise exposure autumn (2005 – 2013). Data displayed in harbour porpoise disturbance days.

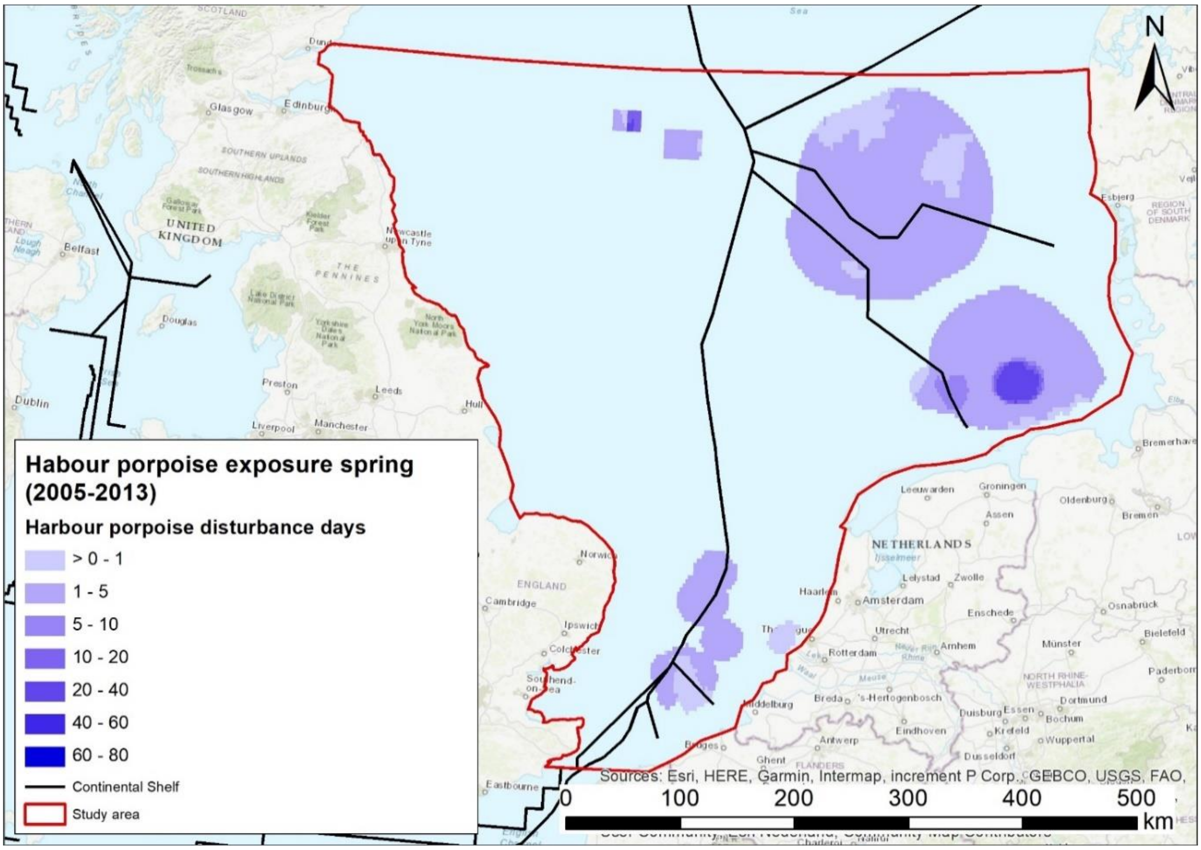

Figure 52: Harbour porpoise exposure spring (2005 – 2013). Data displayed in harbour porpoise disturbance days.

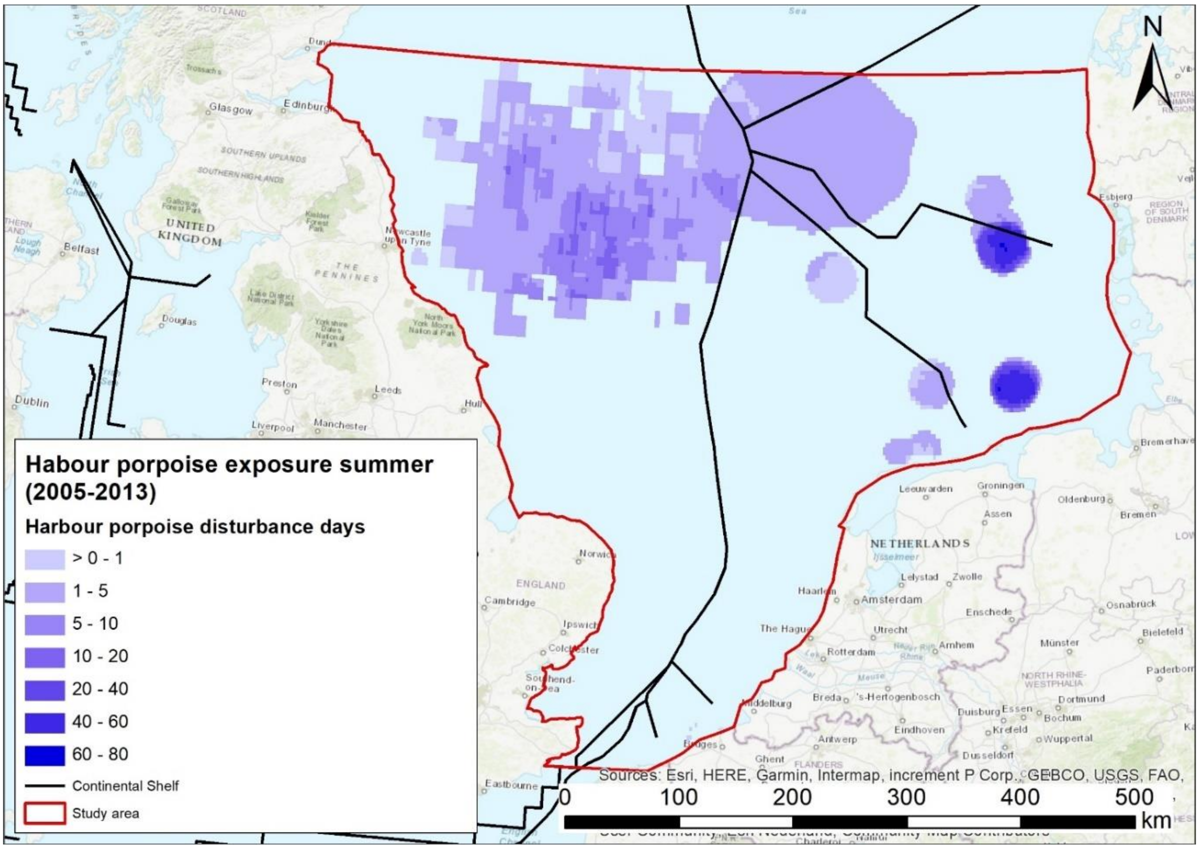

Figure 53: Harbour porpoise exposure summer (2005 – 2013). Data displayed in harbour porpoise disturbance days.

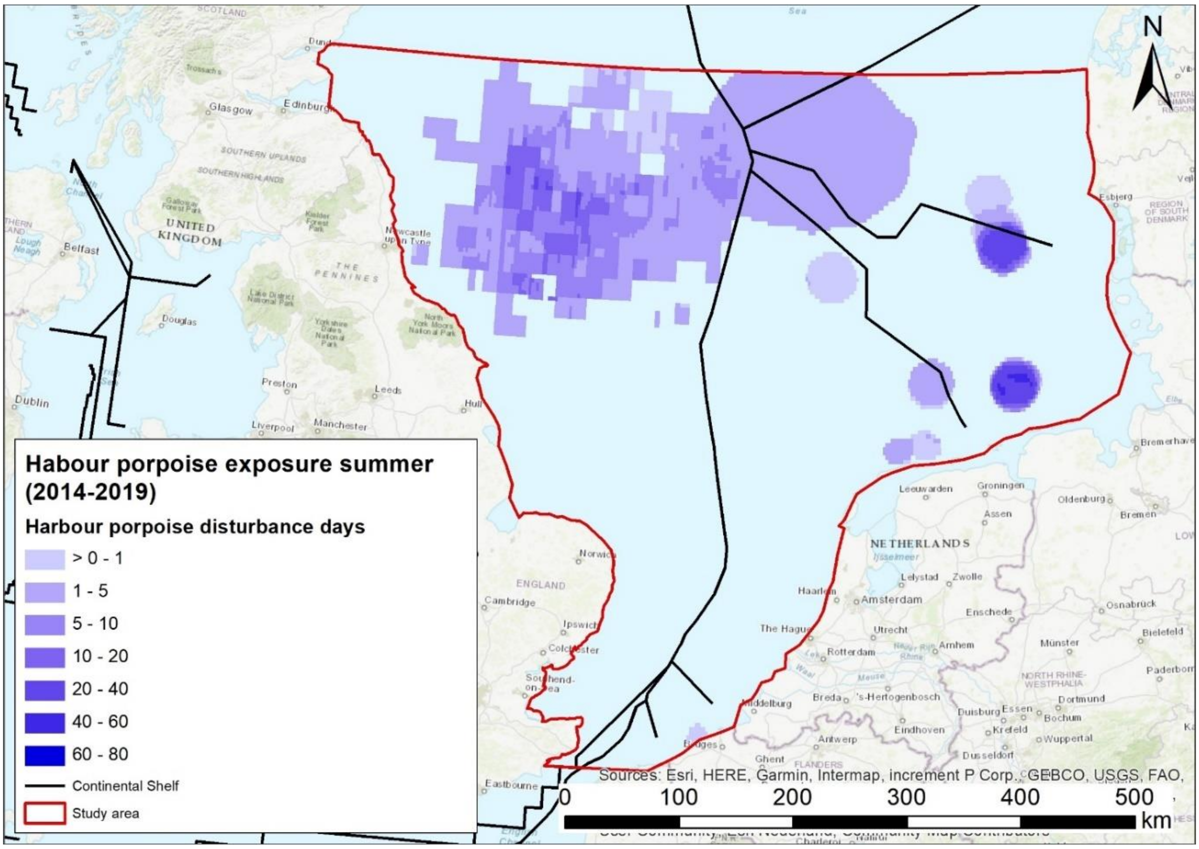

Figure 54: Harbour porpoise exposure summer (2014 – 2019). Data displayed in harbour porpoise disturbance days.

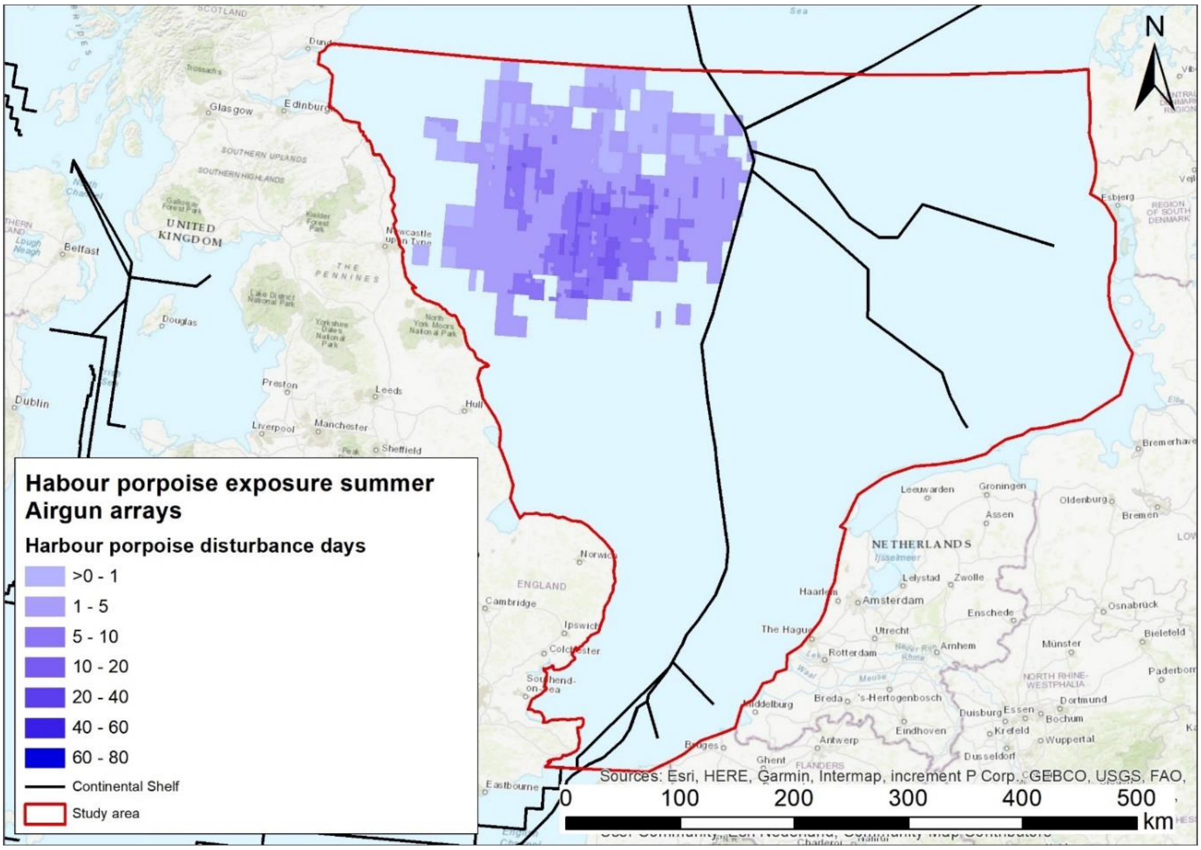

Figure 55: Harbour porpoise exposure to impulsive sound from Airgun arrays, summer (2014 – 2019). Data displayed in Harbour porpoise disturbance days

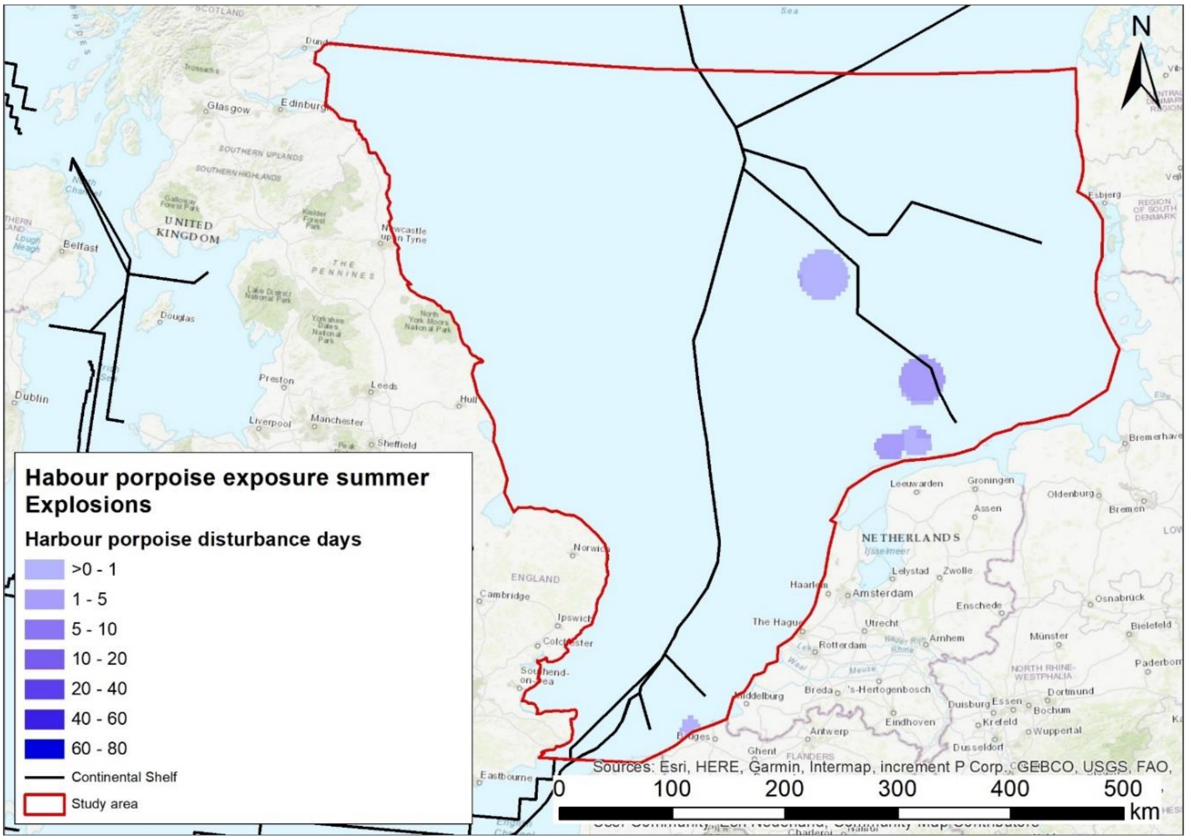

Figure 56: Harbour porpoise exposure to impulsive sound from underwater explosions summer (2014 – 2019). Data displayed in harbour porpoise disturbance days

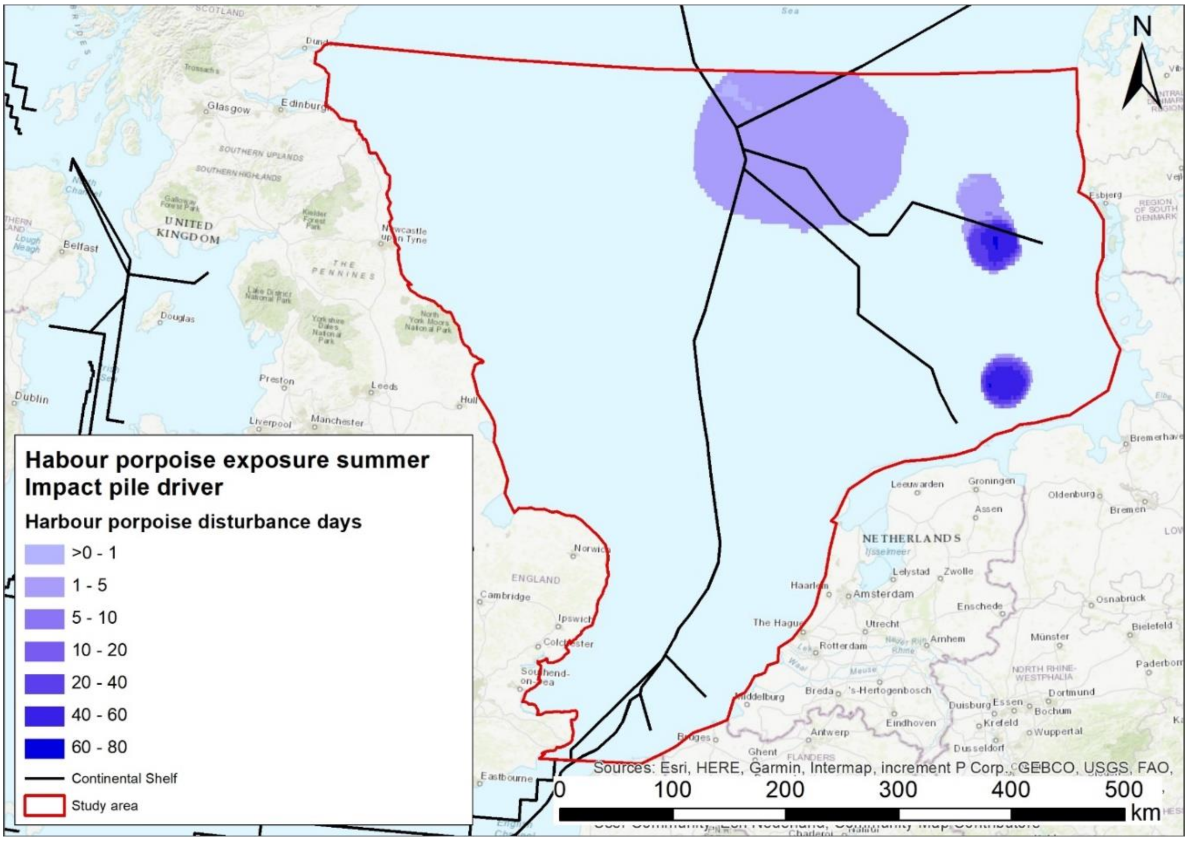

Figure 57: Harbour porpoise exposure to impulsive sound from impact pile driving, summer (2014 – 2019). Data displayed in Harbour porpoise disturbance days.

10 STEP 7/8: EXPOSURE CURVES AND/OR RISK INDICATORS

In the previous chapters step 0: define stressor, step 1: select indicator species, step 2: define assessment area, step 3: define temporal resolution, step 4: Distribution or habitat map(s), step 5: pressure map(s) and step 6: exposure maps were completed. The identified stressor is: impulsive noise. The selected indicator species are harbour porpoise, Atlantic cod and harbour seal. The assessment area is an extension of the Dutch North Sea and the temporal resolution is defined as 2015 to 2017. Distribution maps were created for cod and harbour porpoise. Data for harbour seals are not suitable for further analysis. Pressure maps were created overall and per sound source and merged with the distribution maps to form exposure maps. In this chapter step 7/8: Exposure curve(s) and /or risk indicator(s) is described (Figure 58Figure 26Figure 15). This step was taken simultaneously with step 6 that was discussed in the previous chapter.

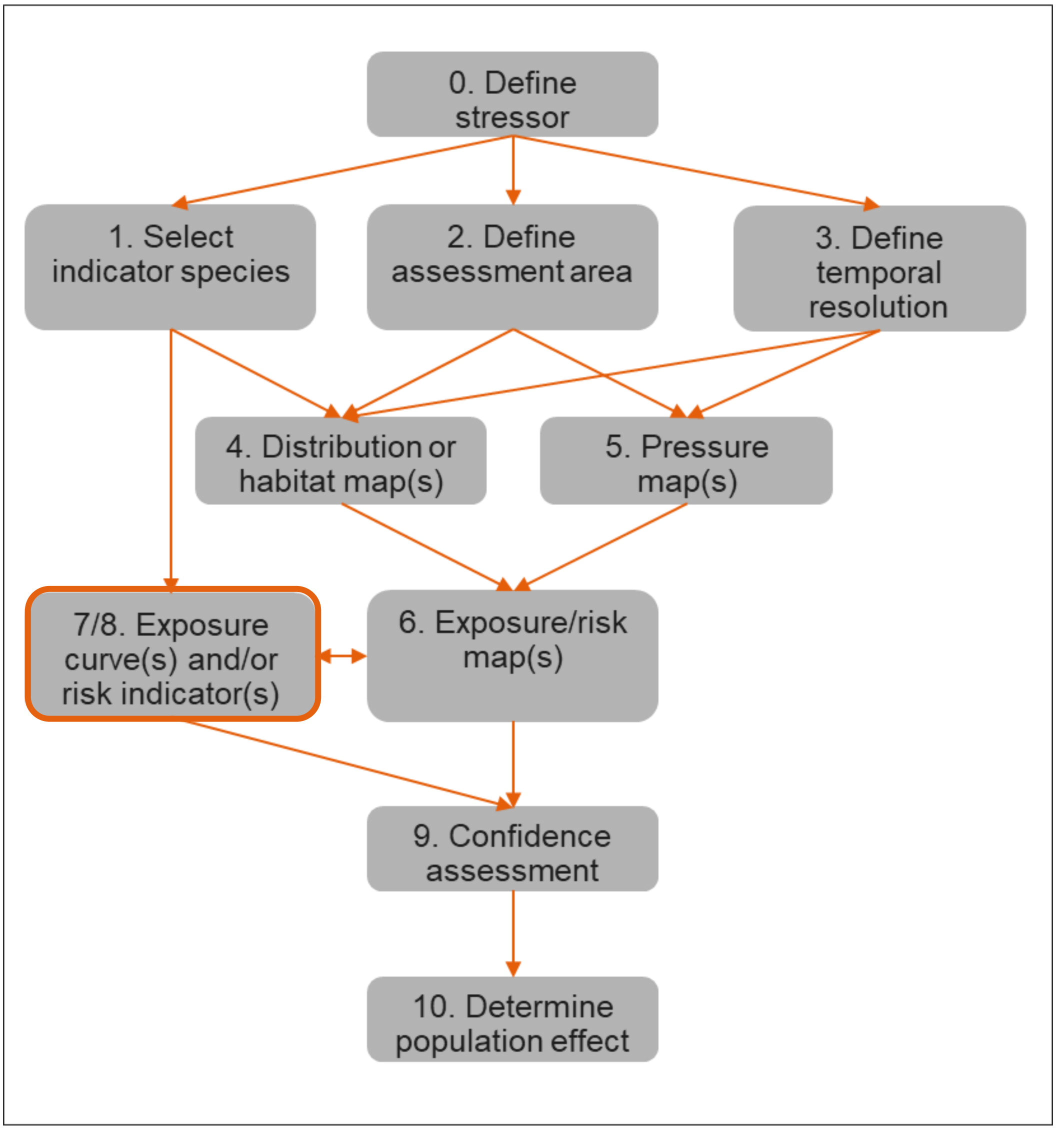

Figure 58: Stepwise approach, current step (highlighted orange).

In this chapter the concept of exposure maps and how to read them is explained in paragraph 10.1. In this paragraph a risk indicator is also introduced in the form of the exposure index (EI). Exposure curves and indexes are then presented for cod (10.2) and harbour porpoise (10.3).

10.1 Reading exposure curves

Exposure curves are curves that show the total area that is disturbed for a certain amount of time. These curves are created based on the exposure maps from the previous chapter and a method first presented by Merchant et al., (2018). When analysing the data at the basis of these maps, a dataset can be created that gives the number of disturbance days for the whole study area.

On the x axis of the graph the species-specific disturbance days are displayed and on the y axis the total area that is disturbed. The data are cumulatively displayed in de exposure graphs. The first value at the base of the x-axis gives the total area that is disturbed for more than 0 days in total. The further from the base of the axis the smaller the number becomes as there is a smaller area that is disturbed for instance 40 days than there is area disturbed for 2 days. As an example, the exposure map and corresponding exposure curve is shown for harbour porpoise (Figure 59). Looking at this figure it becomes clear that the largest area of the study area will have a total number of 0 harbour porpoise disturbance days, a significantly smaller area will have a least 1 disturbance day, a yet smaller area will have at least 5 disturbance days. Visually, the classification is limited by the class limits in the legend. However, the underlying data is more fluent. The exposure curve is an addition that shows this better than a map.

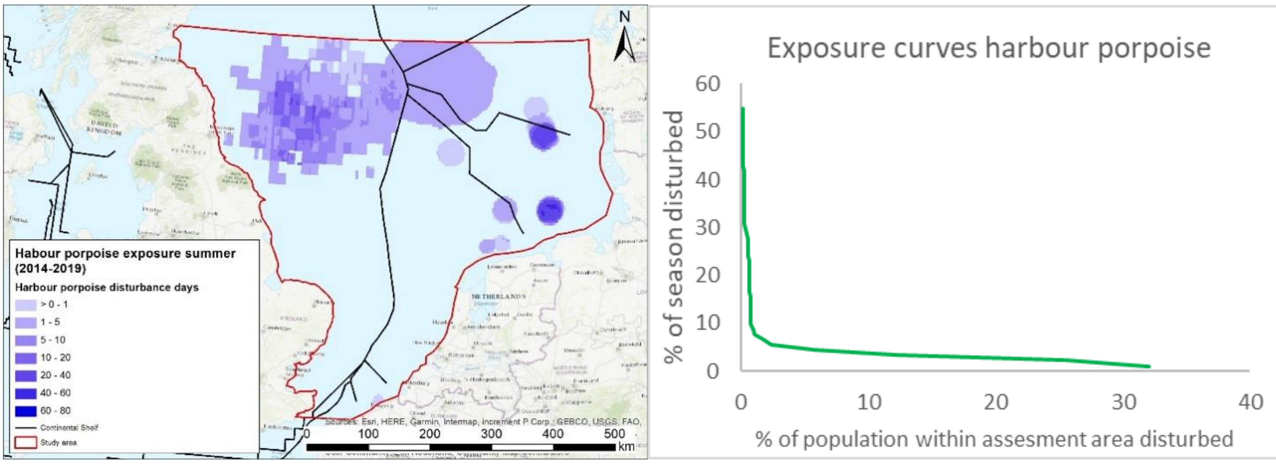

Figure 59: example of exposure maps resulting in exposure graphs. Data displayed are based on INR data from 2015 and harbour porpoise distribution data for the summer from 2014 to 2019 (WMR, in prep).

The exposure curves created in this example in Figure 59 are calculated based on the sum of all noise sources. Exposure to sound from pile driving for example can therefore not be distinguished from exposure to sounds from Airgun arrays. From both a scientific and a policy point of view it can be valuable to make a distinction in various disturbance sources. This can create an understanding of which sources of impulse noise disturbance are responsible for the possible effects.

When quantifying the disturbance there is an additional method available. The exposure of a species can be gathered in a single number, the Exposure Index (EI) (Merchant et al., 2018). The EI expresses the overall exposure of the population based on the area under the exposure curve. This area is log transformed and scaled from 0 to 10 as proposed by Merchant et al. (2018). Even though it can be a usefull additon to compare different exposure intensities between for instance seasons or noise sources, the use of an EI losses an important part of the information that is displayed by the exposure curves or maps. When using an EI it is no longer possible to disinguish between larger areas that are minimaly disturbed or smaller areas that are heavily disturbed. An EI could however be used in policy making to assess situations and establish thresholds once reference values have been commonly established.

10.2 Exposure curves for Cod

As previously touched upon, reliable data on the distribution of cod is available for two seasons (spring and autumn). Therefore, exposure curves can be created for these two different seasons. The computed risk curves for cod Figure 60 indicate that during Autumn, relatively more cod were disturbed for a longer period.

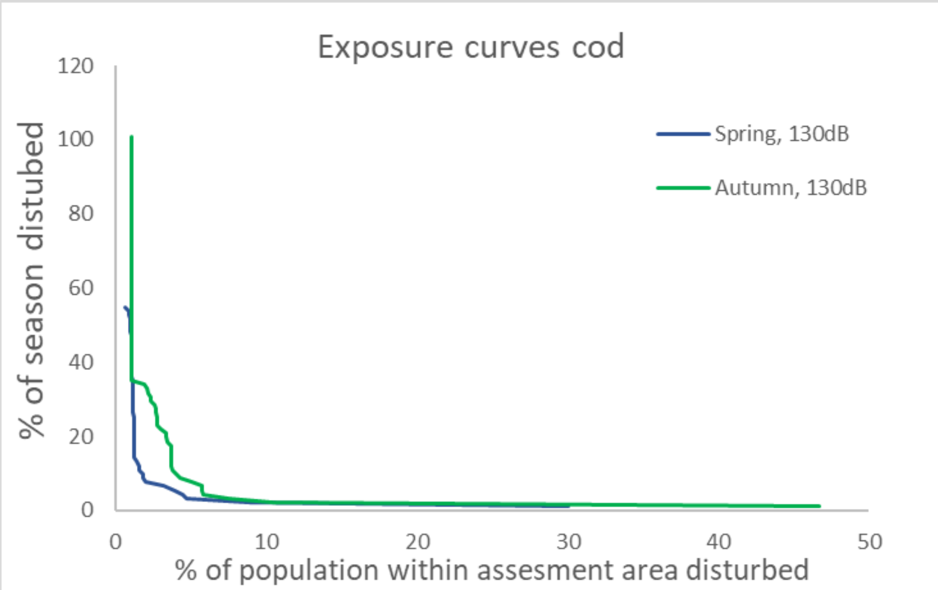

Figure 60: exposure curves for cod for spring and autumn for the worst lowest disturbance threshold 130dB

As discussed in paragraph 8.3.1 there are no reliable disturbance thresholds for impulse underwater noise for cod. The three different thresholds that were used therefore can also create three different exposure curves to compare. Figure 61 gives a comprehensive visualisation of the difference a disturbance threshold makes in determining the exposure of a species. The same becomes clear when calculating the EI for cod for the three different thresholds. When applying a 130dB threshold, the EI comes to 6.0 for autumn. With a 170dB threshold the EI is only 1.8.

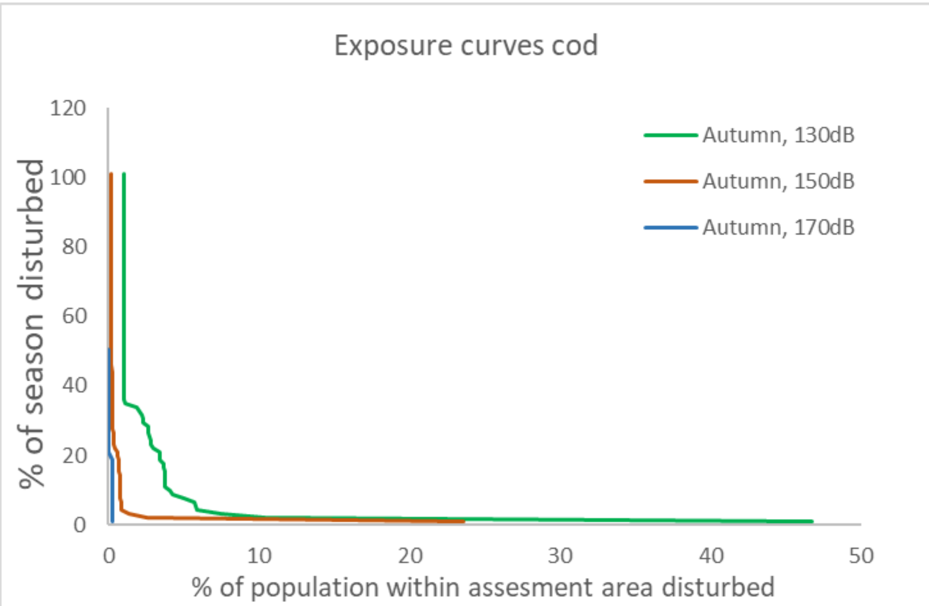

Figure 61: exposure curves for Cod for autumn for three disturbance thresholds (130, 150 and 170dB)

Table 10: Exposure index cod for three disturbance thresholds in spring and autumn. Impulsive noise register data are used from 2015. This is combined with cod distribution data from 2015.

Disturbance tigger value [dB re 1 μPa2s]	Season	Exposure index
130	Spring	5.2
	Autumn	6.0
150	Spring	3.6
	Autumn	4.5
170	Spring	0.8
	Autumn	1.8

10.3 Exposure curves for Harbour Porpoise

As became clear in the previous chapters, the harbour porpoise distribution data is available for spring, summer and autumn for the survey data from 2005 to 2013. Figure 62 shows the exposure curves for those three seasons. These curves display the same data as Figure 51 to Figure 53. From the exposure curves it becomes clear that the harbour porpoise exposure is highest in autumn and lowest in spring. Up to five percent of the population within the study area is disturbed for around 70% of the time. However the total harbour porpoise numbers within the study area are lower in autumn, than in spring and summer (Figure 18 through Figure 20). The difference in exposure between seasons is summarized in

Table 11 by calculating the (EI).

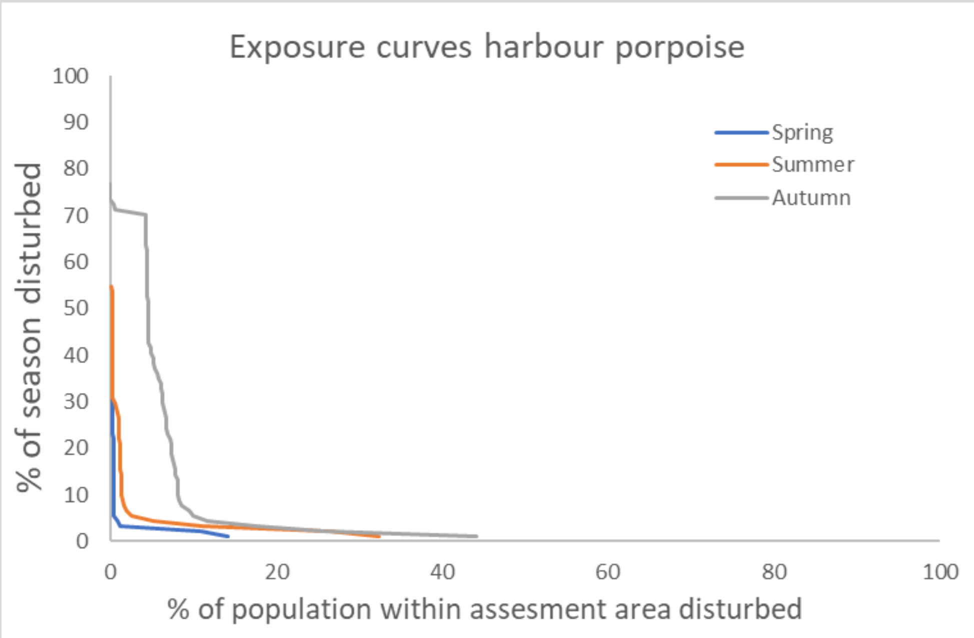

Figure 62: exposure curves Harbour porpoise (2005 – 2013) for three different seasons. disturbance threshold 140dB

Table 11: Exposure index for three different seasons. Based on distribution data from 2005 – 2013.

Disturbance trigger value [dB re 1 µPa2s]	Season	Exposure index
140	Spring	4.0
	Summer	5.2
	Autumn	6.7

For the survey period from 2014 to 2019, distribution data is only available for summer. As an example of how the data can be split to show the type of noise source with the biggest impact, the data from summer (2014 to 2019) was split by noise source type. The remaining source types for summer are airgun arrays, underwater explosions and impact pile drivers. Noise from sonar or generic impulsive sound were not present during the summer of 2015 (paragraph **Error! Reference source not found.**).

The exposure curves show an obvious difference in the exposure from airgun arrays and the exposure from impact pile driving. It becomes apparent the airgun arrays disturb a relatively higher percentage of the population but for a relatively short amount of time. For pile driving this is opposite with a fairly low percentage of the population being disturbed but for a larger period of time. Assuming the response of a species might vary based on these factors, this is valuable information. This information is lost when displaying the exposure as the EI (

Table 12). Here the exposure to noise from airgun arrays (4.6) seems relatively comparable to that of impact pile driving (4.1). It is important to note that the data in

Table 12 is log transformed.

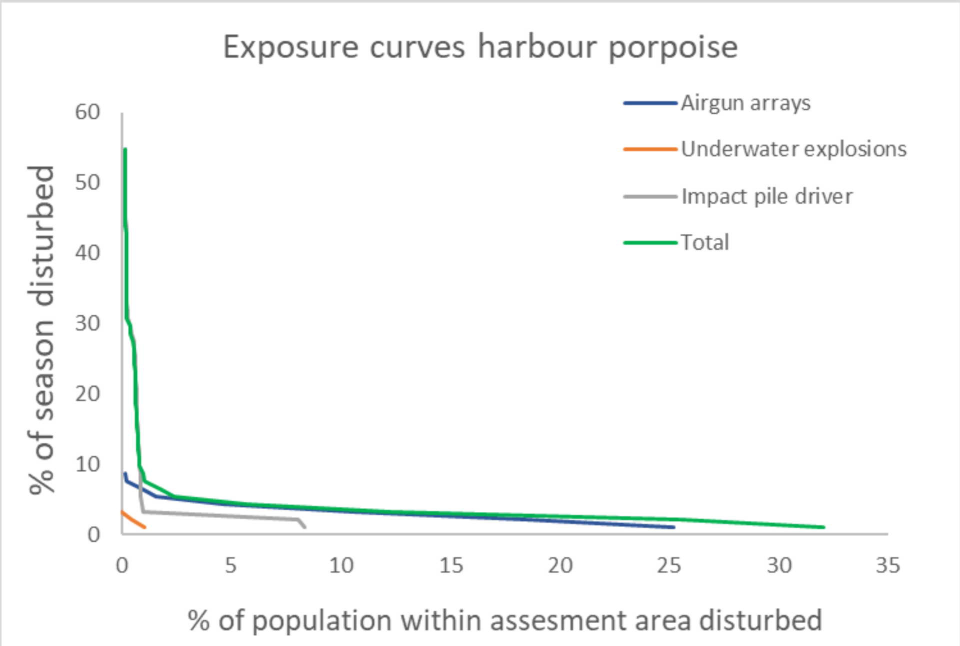

Figure 63: exposure curves Harbour porpoise (2014 – 2019) for three different noise sources. disturbance threshold 140dB

Table 12: Exposure index for three noise sources present in the summer of 2015. Combined with harbour porpoise distribution data from 2014 to 2019.

Disturbance trigger value [dB re 1 µPa2s]	Noise source	Exposure index
140	Airgun arrays	4.6
	Impact pile driver	4.1
	Underwater explosions	0.6
	Total	5.1

11 STEP 9: CONFIDENCE ASSESSMENT

11.1 Assessment method

In the previous chapters step 0 to 8 were completed. This resulted in exposure maps, curves and indicators for Atlantic cod and harbour porpoise. In this chapter step 9: a confidence assessment is executed, Figure 64.

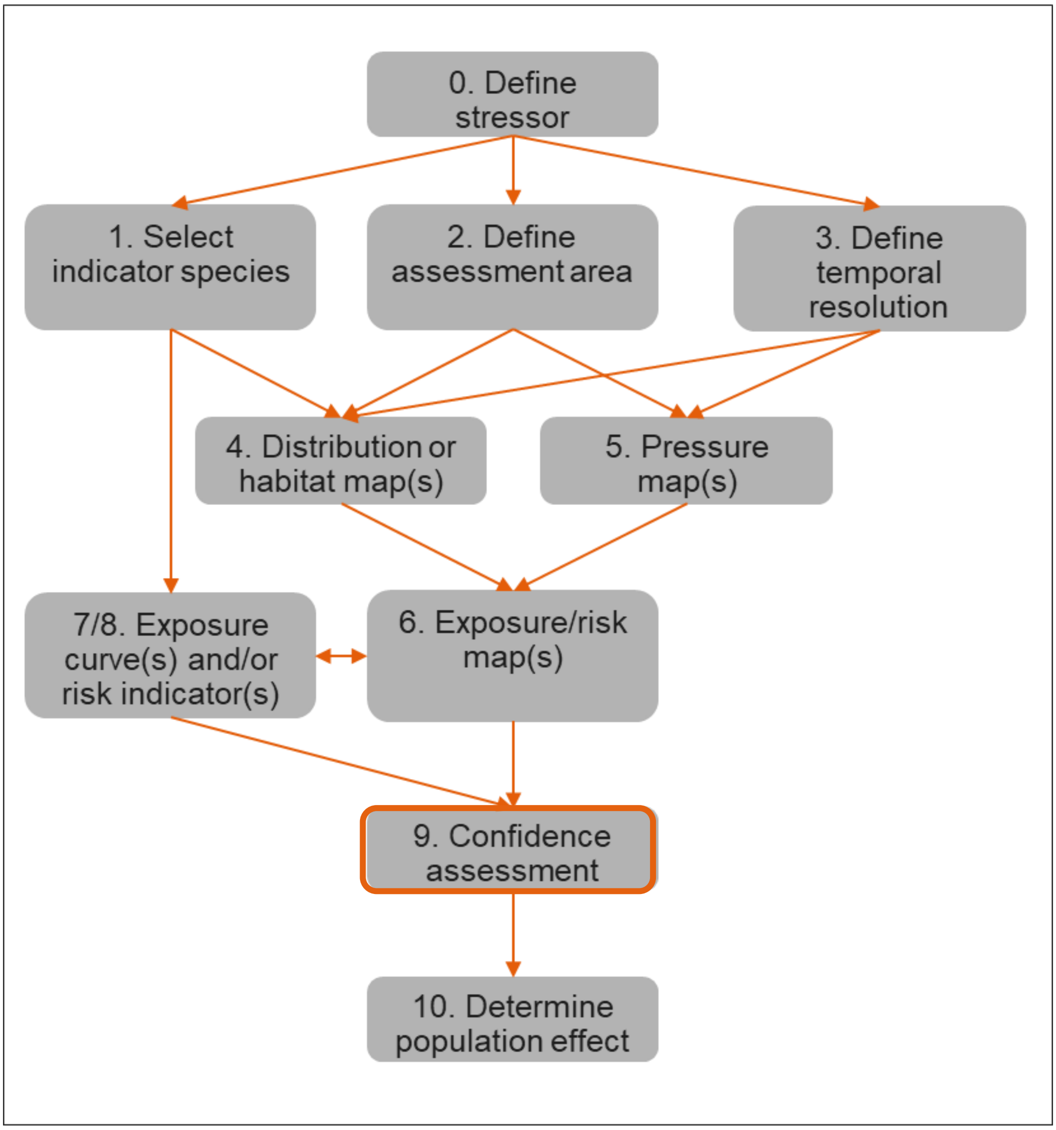

Figure 64: Stepwise approach, current step (highlighted orange).

11.2 Confidence assessment

It is important to discuss the confidence in the results from any assessment being made. At this moment it is impossible to determine quantitative confidence levels for the assessment results, therefore only a qualitative discussion on the confidence was made. Fundamental uncertainties of the method as a whole will be discussed in chapter 13. The following factors and uncertainties form a problem in the reliability of the results obtained:

- Impulsive noise register
  - The register is not complete. Some countries have not submitted data to the register and most countries have not submitted all possible source types to the register.
  - The source levels are approximated by categories. These categories are not consistent across different source types and only give a rough indication of the source levels.
- Effect distances
  - In many cases observed effect distances are not directly available, and can differ substantially within a single source category (e.g. unmitigated pile driving effect distance for harbour porpoises ranging between 3 and 25 km, Table 4). References in the literature are not always validated for the region for which the assessment is made.
  - Effect distances can be estimated by propagation modelling, but these models still have uncertainties in the source and sound propagation models, as well as in their input parameters. The disturbance thresholds used in this study were precautionary, but may not lead to precautionary effect distances, since model validation at large distances (several tens of kilometres) is required to obtain unbiased estimates of effect distances.

- Animal distribution.
  - For many species there are no or very poor distribution maps available.
  - Many species exhibit considerable seasonal variation in the distribution. These variations should be taken into account in the assessment, but often no numerical information is available on seasonal variations. Monitoring programmes do not account and especially monitoring in the winter is very difficult.
  - The data quality in distribution information varies greatly between species. For some species abundance with associated uncertainties can be calculated, for others only an index of occurrence is available. There are also great differences in spatial and temporal coverage.
  - A distribution map for a certain period cannot capture the spatio-temporal variation in a species' distribution within that period, especially for species that show seasonal movements.
  - The animal distribution may already be affected by the sound, which will not be picked up by this assessment approach. A habitat suitability method would be more appropriate for this method.

12 STEP 10: DETERMINE POPULATION EFFECT

In Figure 65 the assessment method is once again presented. Step 0 to 8 were completed in chapters 3 till 10. This resulted in exposure maps, curves and indicators for Atlantic cod and harbour porpoise. In chapter 11, step 9: a confidence assessment was executed. A large number of uncertainties were identified. Therefore in this chapter, step 10: determine population effect, no numerical attempt at determining a population effect is done. Limitation for this will be further discussed in chapter 12.1. Potential ways of determining population effects are then discussed in 12.2 thresholds and 12.3 population models.

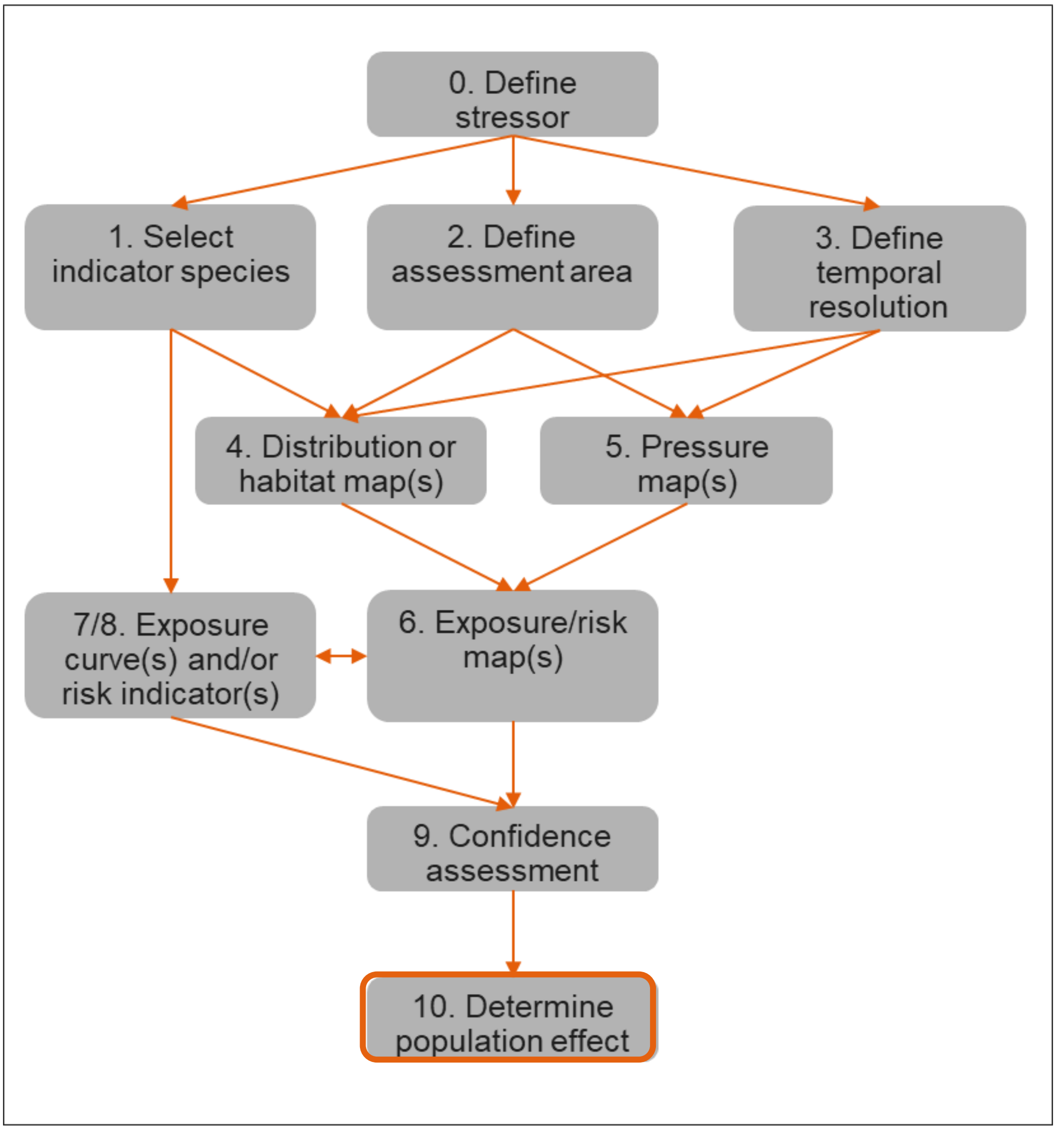

Figure 65: Stepwise approach, current step (highlighted orange).

12.1 Limitations

The aim of this assessment was to quantify the impact of impulsive noise on marine animal populations by calculating the overlap between indicator species distributions and species-specific weighted maps of multiple anthropogenic stressors (Maxwell et al. 2013). Although this can be a valuable tool to highlight regions where conflicts between human activities and indicator species are more likely to arise, the approach has some limitations (see also Johnston et al. (2019)):

1. The distribution of top-predators might already be influenced by human activities.
2. The effects of anthropogenic stressors on the different species are often poorly known.

3. These population level estimates do not capture individual movement processes, and hence cannot differentiate between multiple exposures to a small number of individuals, or a single exposure to many individuals (see e.g. Aarts et al (2016)).
4. The approach ignores indirect ecosystem effects. For example, marine top-predators rely on prey species that are often highly mobile as well. Therefore, changes in the distribution and abundance of prey well outside the predators foraging range might be carried over and influence top-predator population dynamics elsewhere.

Another major challenge is that that even for well-studied species considerable data gaps exist on behaviour, life-history and demography. In theory, the more complex approaches are based on more data and can thus provide the most accurate predictions of population impacts. Unfortunately, the results might be inadequate because they rely on input parameters that are simply not available or not reliable (Morris et al., 1999). Ultimately the method to use depends on the data availability and the goal of the analysis.

One more important point to raise is that most available frameworks to assess population impacts only consider one disturbance source. As mentioned before, marine animals are facing a large number of potentially harmful effects, caused amongst others by commercial fishing, incidental bycatch, habitat disturbance, pollution and climate change. In the future cumulative impacts need to be incorporated. To manage human impact threshold values are often used as management tool to determine what is and is not considered acceptable. These are explained in paragraph 12.2.

Our assessment for the Atlantic cod, the harbour seal and the harbour porpoise are all limited by different aspects of either the species-specific characteristics or the data availability and quality. Different analytical approaches to address population impacts of impulsive sound for these three species have been developed to address these challenges and in this section we present a brief overview in paragraph 12.3.

12.2 Developing thresholds

When assessing human impacts that cause direct mortality of animals most commonly effects are determined with thresholds. For example, in fishery management a ***catch limit*** is calculated for specific fish stocks, including Atlantic cod. Fisheries management aims to provide advice regarding the fishing mortality that results in the “maximum sustainable yield”. Fisheries management approaches often take into account that reproduction is maximised when populations are kept under their carrying capacity and catch limits are adapted accordingly.

Another method is the ***Revised Management Procedure*** (RMP) developed by the International Whaling Commission to provide sustainable catch limits for (large) whale populations (Cooke 1999). The RMP has been applied within the ASCOBANS framework, the regional agreement for the conservation of small cetaceans in the North Sea and adjacent waters. ASCOBANS has set a specific conservation aim for harbour porpoises in the North Sea, which is to reach 80% of their carrying capacity. There is some discussion on how to decide on the most adequate time line as well as the probability of this goal being reached. Currently the maximum removal limit has been set at 1.7% removal per year, in other words, each year maximally 1.7% of all animals present may be removed due to human activities.

Other approaches that are used in conservation actions are the ***Potential Biological Removal*** (PBR) (Wade 1998), which is central to the Marine Mammal Protection Act implemented in the USA.

One obvious disadvantage of this approach is that it only considers direct mortality for a specific population or sub-population. And while directed fishery catch can be monitored reasonably well per population, this is more challenging for incidental mortality caused by bycatch, ship-strike or impulsive sound. Moreover, when considering impulsive sound, direct mortality may not be the most harmful effect of disturbance. It can be assumed that most effects are sub-lethal, leading to changes in behaviour, with possible consequences for feeding and breeding success. To quantify these, as well as other sub-lethal impacts such as pollution, different approaches are needed than mortality thresholds.

12.3 Population models

Population impact can be modelled, which is a relatively new area of science. Therefore, there are still a lot of knowledge gaps regarding the application and use of models. Several population models are currently in use, which are discussed in the tree paragraphs below.

12.3.1 PCoD models and the interim PCoD model

The Interim (iPCoD) (Booth et al., 2016; Harwood et al., 2016; King et al., 2015) model has evolved from the Population Consequences of Disturbance (PCoD) approach (Pirotta et al. 2018).

PCoD approaches aim to quantify the physiological and behavioural changes of an individual animal due to a stressor, the effects of these changes on the individual's health and fitness and how this impacts the population (New et al. 2014, Pirotta et al. 2018). This framework needs information on what proportion of the population is subject to the stressor and the exposure level (intensity and duration) to determine the probability of exposure per individual. While this sounds straightforward, it can be challenging to obtain the needed spatial and temporal data of both animal distribution and abundance as well as for the stressor – in our case impulsive sound. In addition, as presented in chapter 4, adequate information on physiological and behavioural reactions to impulsive sound, and their impact on health and fitness is rare or non-existent for most taxa.

To address the challenge of the 'missing link' between exposure and vital rates the interim PCOD (iPCOD) has been developed (Booth et al., 2016; King et al., 2015). Where information is missing it depends on other sources of information, such as expert elicitation (Pirotta et al. 2018).This framework has been applied to predict the impact of pile driving on the harbour porpoise in the North Sea (Booth et al., 2020) as well as to other marine mammal species including harbour seals (Harwood et al., 2014).

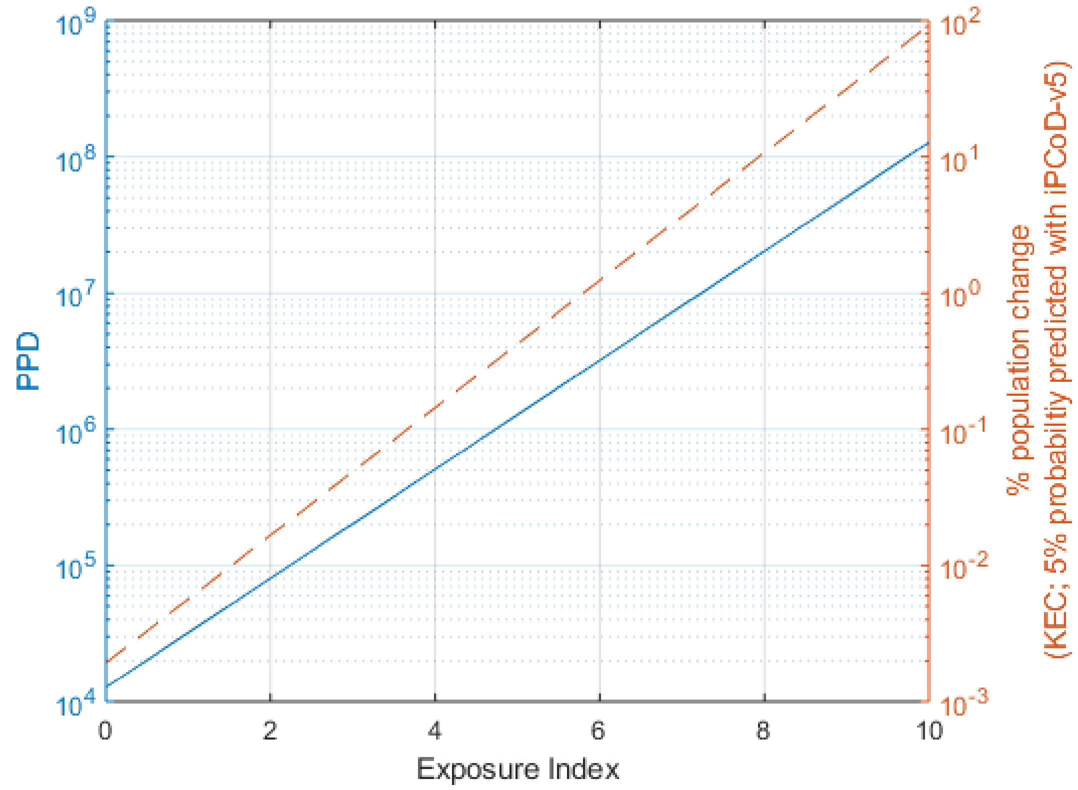

Figure 66: Relationship between the Exposure Index (as computed in chapter 10) to model predictions using the iPCoD model for offshore wind development (KEC, 2018). Note that due to the limited applicability of the iPCoD model, this figure serves to illustrate how the Exposure Index may facilitate future threshold setting using more quantitative PCoD models.

Based on the expert judgment of effect of disturbance on vital rates, the iPCOD model generates statistical predictions of how the marine mammal population develops. The iPCoD framework has been used in the process to establish Dutch noise budgets for offshore wind development (Heinis et al. 2019). In this framework, the model was used to predict the difference of the population trend with the presence of *new* sound producing activities (pile driving during offshore energy developments) compared to what was considered a baseline population development, for which any potential effect of other impulse sound activities that have been going on for many years (such as underwater explosions, seismic surveys, sonar, etc) had already been implicitly incorporated. Apart from the lack of validation of the expert judgments, the model has several other caveats: the model currently did not include a density dependence of the population growth, and it only did a coarse estimation of the animal movement to assess the probability of multiple exposure to individuals.

The iPCoD model predictions suggest a direct relationship between the number of porpoise disturbance days (result from multiplying the disturbance maps with animal density), Exposure Index (as derived in Chapter 10), and the predicted percentual reduction in harbour porpoise population size. This illustrates how the Exposure Index can be translated into more quantitative, once these type of models become more applicable to a wider range of impulse sound sources and species.

12.3.2 IBMs

In Individual (agent) Based Models (IBMs) the behaviour of individual animals in a population is simulated and differences between individuals are accounted for. Their individual actions are defined by a set of rules defined at the individual level, which control, for example, their movements across a spatial grid. Those can be very simple (e.g. McLane et al (2011)) and directly related to what the other ‘agents’ in their vicinity are doing, which might for example lead to a change in their direction of speed. They can be made more complex by integrating how the individual reacts to external stimuli based on their own internal state and also including a memory component which will affect how the simulated animals will react in the future. While, depending on their complexity, IBMs can require considerable data input they are also considered very flexible (Pirotta et al. 2018).

IBMs have also been applied as a part of PCOD frameworks (Pirotta et al., 2014; New et al., 2013). However, only few studies have used IBMS to predict how behavioural changes, for example changes in feeding, lead to population-level consequences. This has been done for the harbour porpoise considering the cumulative effect of underwater explosions on movement strategy (Aarts et al., 2016) and pile-driving noise on the harbour porpoise in the North Sea (Nabe-Nielsen et al., 2014; Van Beest, et al., 2015) in the DEPONS (Disturbance Effects of Noise on the Harbour Porpoise Population in the North Sea (DEPONS) project. A detailed comparison between the iPCOD and DEPONS models can be found in Nabe-Nielsen and Harwood (2016).

IBMs have also been applied to grey seals (Van Beest et al., 2019) and harbour seals (personal communication Geert Aarts).

12.3.3 Dynamic Energy Budget (DEB) Models

Just like IBMs, DEB models aim to understand population dynamics by looking at individuals in the population. The models resolve how feeding, growth and reproduction influences the amount of energy spent and gained throughout an individual’s life cycle. The effect of changes in the environment, such as impulsive sound, on these processes can be quantified. Using a DEB model, the population consequences of such changes can be calculated. Similar to the other examples presented here they depend on data availability for the species that is investigated. IBMs sometimes incorporate dynamic energy budgets in their agents (e.g. Martin et al. (2012)

Soudijn et al. (2020) applied such a model to analyse potential population-level effects of anthropogenic noise on Atlantic cod (*Gadus morhua*). They investigated the impact of increased energy expenditure, reduced food intake, increased mortality and reduced reproductive output. The results showed that an increased energy expenditure and a reduction in food intake impact population growth rate more than direct mortality or a direct decrease of reproductive output. This indicates to the importance of sub-lethal effects.

13 DISCUSSION AND CONCLUSIONS

In the previous chapters, three indicator species were used as case studies to apply the stepwise assessment framework to determine the effects of impulsive noise, as described in chapter 2. In this chapter the specific research questions as described in the project purpose in chapter 1 are discussed, as well as the uncertainties, choices made and lessons learned in this process. To enable this, halfway during the project an international expert symposium was organised to allow other experts in the field to comment on the case studies and the methods used in general. Their input was incorporated into the report, and particularly into this chapter.

13.1 Overall conclusion regarding the assessment method

The authors succeeded in completing step 1 to 8 for 2 out of three indicator species. After performing step 9 the authors concluded step 10 is currently unfeasible but this could be done in the future. Overall the assessment method is promising but could use fine tuning. Specific elements of the assessment are further discussed in the next paragraph.

13.2 Fundamental uncertainties in the method

Fundamental uncertainties in the method described in this report are:

- Vulnerability to impulsive noise: There are only very few species for which we have information on the effects of impulsive noise. Most taxa are either not represented at all or only with few species. Methods to assess responses are not standardised and results can vary even within different studies of one species.

- Indicator species selection: The major knowledge gaps on the effect of impulsive sound on marine animals dramatically limits the choice of potential indicator species. Data on distribution and density as well as seasonal changes thereof are of varying quality and resolution. This potentially leads to a pragmatic choice of indicator species based on the species for which most information is available.

- Despite these uncertainties the confidence limits of the assessment results should be small enough to be able to describe trends in the impact of impulsive noise on an indicator species. Considering the current status of data availability this is hard to achieve.

13.3 Selection species

Selection criteria were identified and chosen based on literature in order to select indicator species that could serve as case studies in the assessment of impulsive noise. The criteria focussed on sensitivity to underwater sound, the vulnerability of the species, the threat status, the commercial value of the species and the availability of data.

It proved difficult to fully apply all four criteria to the selected species. There is very limited to no a data available for many species, resulting in the choice of three species in this study for the North sea for which most data were available. This meant that invertebrates could not be included. Invertebrates, such as benthic organisms and zooplankton, form the basis of the North Sea food web and impacts of noise on those levels can have consequences on higher trophic levels and as such should also be considered where possible.

In our study the application of the criteria resulted in the choice for harbour porpoise, harbour seal and Atlantic cod as indicator species. The assessment showed that even for the harbour porpoise, the species for which there is the most available information, there was not enough information to complete step 10 in the assessment: determine population effects in the North Sea. The choice of the three indicator species in this assessment was based largely on information availability and known vulnerability. It could be that in the future other species turn out to be equally or more suitable for use as an indicator species. The availability of data and information for specific species is very region or location specific. In In other regions there may be sufficient information available for other species or populations. There is often more data available for smaller scale areas and locations and for local populations of well-studied species. This could possibly allow for a more detailed investigation of impacts for impulsive sound emissions in smaller assessment areas.

Another shortcoming in this study is that different stages of a life cycle of the different species were not included in the assessment. This is important because there are certain times in a life cycle of a species, for example during reproduction, when the vulnerability of a species to sound changes. Another important point that was not taken into account in this study is how populations can recover after an impact occurred. Some species tend to reproduce early and have a high number of offspring. Even though the survival rate of their offspring is low, as they have a low parental investment, they tend to be better at adapting to changes in the environment and are therefore likely to recover quicker form a disturbance. In contrast species that have a low number of offspring and a high parental investment are less adapted to change, including non-natural mortality. Species with shorter reproduction cycles are likely to recover more quicker from a disturbance such as sound than other species.

It is important that the choice of the indicator species is dependent on the region of interest and the result from a discussion of experts and that the choice is well documented. The choice of the three indicator species in this assessment was based largely on information availability and known vulnerability. It could be that in the future other species turn out to be equally or more suitable for use as an indicator species.

13.4 Noise mapping

The noise working groups of OSPAR and HELCOM have created an impulsive noise register which is hosted by ICES. This register is a major tool in the assessment of impulsive noise and a source of information for the mapping of noise and the geographical distribution of the noise pressure. The register is not complete. Not all countries have submitted data to the register and some source types are missing. Lack of detail in the source characteristics leads large uncertainties in effect distances. Some of this may be alleviated by using acoustic models, which can deal with environmental differences in sound propagation, however these also rely on more detailed knowledge of the sound sources. Proposals on how this lack of information can be dealt with are detailed in section 8.4.2.

Even for the well-studied species considered here, documented effect distances can be used, but often the reported effect distances are not available or validated the environment they are applied. Effect distances not only depend on the source type but also on environmental parameters like bathymetry, or other contextual factors. Contextual information in general is missing. Propagation modelling provides mechanisms to extrapolate between environments.

In using the register data to create disturbance maps the following challenges were encountered:

- To determine effect distances from modelling dose-response curves are needed. In many cases these curves are lacking.
- The sound propagation from unmitigated pile driving sources is known and well validated, but for seismic surveying the propagation lacks validation.
- Relevant new information has become available for harbour porpoises, harbour seals, and grey seals. However, it requires more work to derive dose-response relationships.

13.5 Exposure and maps

With the currently available data the production of exposure maps and exposure curves for cod and harbour porpoise succeeded. However, all previously mentioned uncertainties and data gaps apply to these maps. Improvements are needed on the quantitative aspects of the risk curves (metrics used).

The use of risk curves to evaluate the Good Environmental Status (GES) seems to be useful to combine both the spatial and temporal aspects of the exposure to underwater noise as required by the Marine Strategy Framework Directive (MSFD). However more experience is needed in the use of these curves and more examples need to be generated to understand the meaning. These examples should cover more species and more variation in noise exposure.

13.6 Population impact assessment

The MSFD requires that the status of the ecosystem be assessed and not just the exposure of individual animals. The requirement can be translated into the effect on the population of the indicator species or on the suitability of the habitat for the species under the exposure to noise. To determine GES for descriptor D11C1 therefore data on impact on different species should be combined.

Our current assessment methodology uses overlapping maps of populations and risks providing information on “hot spots” of potential impact. The obtained results and maps contribute to the set-up of management and mitigation measures to control impulse noise effects on (indicator) species.

The used approach does not always adequately consider the dynamic nature of the ecosystem. As described in section 12.3, there are many modelling frameworks available to translate the effect of underwater noise from individual animals to population consequences. All of them have their advantages and disadvantages and which one is most adequate to use will depend on the specific situation. What is common to all models is that they rely on input parameters that are often not sufficiently known. When data is not available assumptions have to be used, inadvertently leading to a reduction in precision of the outcome.

The ecological connections between predators and prey are complex and any disturbance within this system can have consequences that are difficult to predict. This is in particular true considering the lack of information on how the lower trophic levels are impacted by impulsive noise. In addition, other anthropogenic activities and their potential cumulative effects play a role that has not been given sufficient attention. In particular for species that are already experiencing a high mortality, e.g. through commercial fishing, or species that are recovering from historic overexploitation will have a different and unknown response.

Any approach will require more data on the biology of the relevant North Sea species and reliable information on how they are impacted by impulsive noise. This is a challenge that will not be solved over the short-term. In light of the current limitations successful mitigation and management of impulsive sound needs to follow a “best available practice”. This means following an approach in which it is taken into account that assessments need to be adapted as soon as new information is available, or when new methods are being developed.

14 LOOKING FORWARD

This project showed that the framework for assessing impulse noise has potential but can currently not be executed fully. This attempt at completing the assessment steps leaves a number of issues open for discussion and further research. We suggest the following actions and research topics for future work:

- The impulsive noise register needs a better coverage of all activities in the North Sea. This primarily concerns other countries, but The Netherlands can improve in the registration of high-frequency seismic noise (subbottom profiling) and ADDs.
- The concepts developed in this project should be incorporated into the framework developed by ICG Noise, which is led by United Kingdom.
- The MSFD requires that for determining GES, threshold values should be set. At this moment no numerical threshold values for impulse noise can be confidently determined based on available science. There are two routes of solving this issue:
  - More scientific research with a focus on translating impulse noise exposure to population damage. This means improving models that are under development and possibly expanding the number of relevant species under investigation. Improving models also creates a need to fill in data gaps of species distribution and behaviour.
  - Setting a threshold not based solely on scientific data. Thresholds could be set based on pragmatic policy choices and the expert opinions of leading scientists and policy makers in the field, with a precautionary principle.
- Currently, due to lack of information on particularly invertebrates, the indicator species choice is highly biased towards species at a higher trophic level. In selecting indicator species for these types of assessment caution should be applied: it is unlikely that an entire ecosystem can be assessed through one species with a sensitivity for impulse noise. Therefore, indicator species should preferably include more than one species with different ecosystem roles.
- The assessment methodology requires further validation for seismic surveying. In recent years most knowledge on impulse noise has been acquired through projects for offshore wind energy with pile driving. It is plausible that the same framework can also be applied to seismic surveying, but different components need validation, like propagation modelling, dose-effect curves, duration of disturbance and its effect on animal fitness.
- Currently, lack of information on species distribution, abundance, habitat use and their behavioural response to impulsive noise is one of the major problems identified in the impulse noise assessment. Dedicated research targeting identified data gaps should be carried out.
- Beside a general lack of information, there is a specific lack of spatial information and available GIS data. Therefore, internationally available data should be collated and analysed together in a joint effort.
- Ecological modelling approaches (chapter 12) show a lot of potential for impulse noise assessment. This relatively new area of research should be further developed.
- Alternative methods to the current stepwise assessment should also be developed: instead of overlapping maps, interaction between noise and habitat use should be modelled. This could include incorporating disturbance data into habitat suitability models, which can be used to explain the influence of impulse noise on habitat use.

15 REFERENCES

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APPENDIX A MINUTES OF THE IMPULSE NOISE WORKSHOP

SUBJECT
Minutes Impulse noise meeting 02-03-2020

PROJECT NUMBER
C05062.000533

DATE
12 March 2020

OUR REFERENCE
D10014710:28

FROM
Sarina Versteeg MSc.

TO
Niels Kinneging

1 ATTENDANTS

First	Last name	Institute
Julie	Cook	Offshore Petroleum Regulator for Environment and Decommissioning
Ross	Culloch	Marine Scotland
Christ	de Jong	TNO
Rene	Dekeling	Ministry of Defense
Aylin	Erkman	Rijkswaterstaat
Steve	Geelhoed	Wageningen Marine Research
Floor	Heinis	Heinis Waterbeheer en Ecologie
Gayle	Holland	Marine Scotland
Ron	Kastelein	Seamarco
Niels	Kinneging	Rijkswaterstaat WVL
Frans-Peter	Lam	TNO
Meike	Scheidat	Wageningen Marine Research
Jacob	Tougaard	Aarhus University
Bernd	van Kuijk	Arcadis
Sarina	Versteeg	Arcadis
Sander	von Benda Beckmann	TNO

2 PROGRAMME

10:00 Introduction on the assessment methodology for impulsive underwater noise by Niels Kinneging*
10;45 Coffee
11:00 Selection of indicator species by Meike Scheidat*
11:45 The impulse noise register and disturbance maps by Sander von Benda-Beckmann*
12:30 lunch
13:30 Generating, interpreting and processing expousure maps by Bernd van Kuijk*
14:15 Discussions
15:00 Coffee
15:15 continue discussions
16:30 summarizing discussion outcomes and closure by Niels Kinneging
17:00 Closing drinks
*The presentation slides are separately attached

3 REMARKS DURING PRESENTATIONS

During the introduction a question was asked on the probability of a long term PCOD model. According to Floor Heinis not much data is required to make one, but currently there is no funding. This is a work in progress.

During the indicator species presentation by Meike several points were raised:

- In laboratory studies sea bass proved more responsive to sound than cod. Why not choose sea bass?
- Could you try and explain the process of selecting species in the report? Pragmatic choices have to be made, explaining them will help people follow your logic.
- A discussion on selection criteria ensued: Which criterium is limiting? Isn't data availability always limiting?
- A discussion on spatio-temporal overlap, shouldn't that be in the selection of the area? Common agreement is it should be, so the overlap with impulsive sound should be out of the table and questions.
- Discussion on species distribution: should a species be distributed through the entire area? Harbour porpoise is a good one for the north sea because it is distributed evenly.
- A discussion on the differences between hearing capabilities of grey and harbour seals arises. Hearing is expected to be the same but grey seals are 'tougher' and may respond differently. In management both species are considered. Also in management, haul-out sites are protected while at sea sites aren't (or are less). This is partly because seals can avoid sound at haul-outs and don't rely on audio as much as harbour porpoise.
- An issue is raised on the impact of impulse noise on cod. Impact can never be fully measured as it can't be distinguished from other impacts such as fisheries.

During the impulse noise presentation by Sander several points were raised:
- Why is the impulse noise register divided into these categories (high-low)? The categories will be revised in the future.
- Do SEL and SPL differ per species? This is limited for practical reasons.

During the distribution maps presentation by Bernd several points were raised:
- What we see on the soundmaps is just sound? It is not weighed? No it is not.
- The problem still is: how to get from an exposure map to a population of moving animals? But, If one is i interested in habitat quality, this does give input.
- Remark. population modelling and impacts can't be calculated unless you know where the damage is.
- Effects could be predicted by mapping out sound beforehand.
- A discussion on the usefulness for other species occurs. If no cod were caught in one area, the value is 0. In that area other species might be present though. This than leads back to the discussion on choosing an indicator species. Do you choose one, or do you have to take several?
- A discussion ensues on just going back to the sound maps. A point is raised that that still says nothing about species impact and creating threshold values.

4 REMARKS DURING DISCUSSION

The discussion continues on thresholds:

- the main difference in doing the weighting by species distribution or habitats, is that species are more difficult but more tracible.
- the UK has installed threshold values for harbour porpoise (every day maximally 20% of an area is allowed to be disturbed), but not all regulators agree and not all areas are protected yearround
- in a SEC, you can look at the benthic community in the habitat and see how they change. It could be a good indicator but we've never researched this. A problem is that benthic communities vary from year to year and the effects of noise are hard to distinguish.
- perhaps everyone agrees these maps are limited, but we can perhaps use them to see where our knowledge gaps are.
- A discussion ensues on what if we just said an exposure of more than X days is wrong? This is too ridged from a management perspective. And no one would tell us if it worked. Also: it is better to have a lot of noise at one time, or a little that lasts longer?

Discussion after coffee:
What is needed to evaluate GES? Do we need to assess population reduction, or habitat suitability?

- we've discussed this a lot. There's been a lot of people, even co-autors against using the MERC data and the Waggot paper (2019). These people are currently preparing a review. We are not seeing systematic patterns and the data are not systematic. People feel uncomfortable drawing thresholds from this. But is Scans which is one day in summer, really better?
- These (the MERC) data are also from the 1980's while we know there has at least for the harbour porpoise been a distribution shift.
- but what's an average over 20 years? How much is the variation per year? If we build a park in Germany and all porpoises are in England, we have an error because the mean says they are hurt. Also, shouldn't ASCOBANS say which map is the best map? We already have to deal with the noise….
- More people agree the correct OSPAR and ASCOBANS groups should be contacted.

Should our efforts be on finding more maps, or getting a better understanding of the process?

- it comes down to: do we need a PCOD model for all the species we consider? When is it enough? It would be naïve to think this is a short term process.
- we are often not even happy with our expert opinion. We also need more information on the vulnerability of certain taxa.
- unless we start with dynamite fishing we will never affect the population of pelagic larvea. We can fish out all mussels and they will come back.
- what do we think about sessile versus mobile animals? Answer: depends on life stage, most important decision for these animals is when and where to settles. Which is mainly influenced by currents.
- if we have taxa which have the same hearing capacity we can put them together. unless they are distributed differently. Currently approach is driven by sensitivity and available in the area. If you want to address GES and these species aren't there… so either you let the species go, or the spatial distribution
- but you can use species as a proxy.
- we are not making maps of GES yet, we are making maps on which we can judge GES. So we should use the right metric.
- but for harbour porpoise we do have the population impacts. The question there is how to get to GES
- but how do you do ipcod, can you spatially specify this?
- we did, its based on local disturbance days. But it's a challenge to do this for the whole north sea and in projects where we don't know when, where and what is going to happen.
- A discussion ensues on the appropriateness of using harbour porpoise as an ecosystem indicator. There are many problems with determing if effects echo through a food web, and what than caused an effect. Harbour porpoised do however depend on echolocation and are impacted over a long range of distance. Argument arise that damaging their echolocation frequency with pile driving to get TSS is nearly impossible. In that sense seals rely more on their low frequency hearing.

The discussion is halted and Niels presents some final conclusions:

5 CONCLUSIONS

Selection species

- Mostly based on availability of information
- Usual a lack of information
- Top preditors can give information on whole food chain (exclusing other stressors?)
- Need for documenting the choice
- Pragmatic approach
- Discussion of experts needed for final choice
- Regional

Noise mapping

- Categories low/medium/high by TG Noise is unfortunate
  - No equivalence between source types
  - TG Noise will come up with ABCD
- Documented affect distance seems preferable
  - In most cases no evidence
- Propagation modelling second best
- Register valuable but has shortcomings
  - Incomplete
- Dosis-response curves are needed
  - Pile driving enough evidence
  - Seismic lack of evidence

Exposure and maps

- More experience in interpretation risk curves needed
- Clear metrics needed
  - axes unclear
- More species to evaluate
- Habitat approach slightly in favour of population approach
- UK guidance for assessment (Julie Cook)

Models

- No clear preference for either model
  - PCOD
  - DEPONS
- Now only for porpoise (and other marine mammals?)
- Time issue
  - Changing population distribution
  - Time resolution of information

COLOPHON

ASSESSMENT METHODOLOGY FOR IMPULSE NOISE
A CASE STUDY ON THREE SPECIES IN THE NORTH SEA

AUTHOR
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