

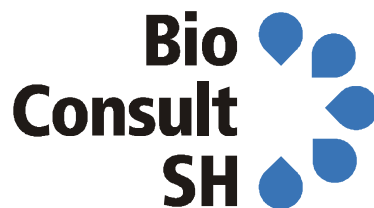
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Methodologies for measuring and assessing potential changes in marine mammal behaviour, abundance or distribution arising from the construction, operation and decommissioning of offshore windfarms

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Executive Summary

This reports reviews methodologies for measuring and assessing potential changes in marine mammal behaviour, abundance or distribution arising from the construction, operation and decommissioning of offshore windfarms. The report first describes impacts from offshore windfarms on marine mammals and defines the spatial and temporal scope of investigations in order to detect impacts on marine mammals. Impacts from offshore windfarming occur at very different spatial and temporal scales: Construction work, especially pile driving, may result in short-termed but long ranged impacts, whereas the operation of windfarms is expected to result in local but long-lasting impacts.

The report reviews the standard methods used in studies on marine mammals in relation to offshore windfarms. Statistical power of line transect surveys using aircrafts and ships and Static Acoustic Monitoring using T-PODs is analysed from datasets obtained in German studies.

Aerial surveys provide an efficient method for multi-species surveys covering rather large areas (2000 km²) per survey, which are considered to be sufficient to compare impact and reference areas. In areas where abundance of harbour porpoises are sufficiently high, which is expected to be the case in many UK waters, aerial surveys are useful to measure population size and population changes. Based on datasets in areas of high densities, it is concluded that 30 % changes can be detected by 1-3 surveys. Aerial surveys will provide additional data on other species, however, as either densities (dolphins, whales) or detection rates (seals) are low, no quantitative data can be obtained in restricted areas. Aerial surveys are often restricted by weather conditions and thus suffer from a low temporal resolution. The spatial resolution may not be sufficient to detect small-scaled impacts as expected during the operation of offshore windfarms.

Line transect counts from ships may also be used for marine mammal surveys. They have a lower spatial resolution and are even more restricted to suitable weather conditions. Based on datasets in areas of high densities, it is concluded that 30 % changes can also be detected by 1-3 surveys. The efficiency of ship transect surveys can probably be highly improved by towed hydrophones, which record echolocation clicks from porpoises and dolphins. The main advantage of towed hydrophones would be, that it can be used at night and during weather conditions, which do not allow visual surveys.

Static Acoustic Monitoring (SAM) using T-PODs or other devices provide excellent data on harbour porpoises and potentially also dolphin species at a high temporal but low spatial resolution. The spatial resolution can be improved by deploying several SAM devices in the area of interest. Statistical analysis from areas with low and high porpoise densities proved that a 30 % change in harbour porpoise presence can be proved with a sample size of 3-11.

Other methods, as telemetry tracking, photo-identification and haul-out counts for seals are described.

For impact studies in relation to offshore windfarms a BACI (Before-After/Control-Impact) design is generally recommended but recommendations are also given for cases, where no baseline data are available or a BACI may not be possible for other reasons.

It is recommended to combine line transect surveys using aircrafts or ships with Static Acoustic Monitoring. An impact study on offshore windfarms should ideally cover two years before construction, the construction period and at least two years of operation. If longer lasting effects are detected, the study during the operational phase should be extended. It is recommended to conduct line transect surveys in monthly intervals. In areas with a marked seasonal occurrence, surveys may be restricted to periods with high abundance, when sufficient data are more likely to be obtained. Continuous recordings of harbour porpoises with SAM are recommended for all areas, where these animals occur in relevant numbers. SAM will provide data which are needed to detect short-termed changes in behaviour and abundance as expected in response to pile driving, but also to detect changes on a much smaller spatial scale as can be detected by other methods as well as long-term changes in response to construction of operation.

For seals and dolphins severe problems in assessing the impacts remain, as their behaviour or low densities make it very difficult to obtain enough data for statistical analysis.

Glossary

- Acoustic tag:** Electronic tag records echolocation sounds.
- Amplitude:** Nonnegative scalar measure of a wave's magnitude of oscillation, that is, the magnitude of the maximum disturbance in the medium during one wave cycle.
- Broadband sound:** Sound consisting of a wide band of frequencies
- Data logger:** Small microchip unit with a memory capacity up to some dozens MB which records information on different sensors in defined intervals (e. g. 3-20 seconds), mostly put in a device of synthetic material.
- Detection threshold:** Lowest sound pressure level that can be detected
- Echolocation:** Active sensory system in certain animals: The emission of sound by an animal and perception of its echo to determine a bearing, range or the characteristics of an echoing object.
- Frequency:** Measure of the number of occurrences of a repeating event per unit time
- g(O):** Probability of detection on the transect line, usually assumed to be 1. In the case of marine mammals that spend substantial periods under water and thus avoid detection, this parameter must be estimated based on other type of information (Buckland et al. 2001).
- Haul-out location:** Areas on sand banks or on coasts that are used by seals for resting, pupping, or moulting.
- High density area:** Areas where the abundance of a marine mammal species exceed one animal per square kilometre. According to this, we use medium densities for areas with densities between 0.1 and 1 animal per square kilometre and low density areas when the density is below 0.1 animals per square kilometre.
- Hydrophone:** Submersible sound sensor. It is analogous to a microphone for listening to underwater sound
- Odontocetes:** Systematic name for toothed whales which form a suborder of the cetaceans characterized by having teeth.
- Pinnipeds:** Diverse group of semi-aquatic marine mammals comprising the families Odobenidae (walruses), Otariidae (eared seals, including sea lions and fur seals), and Phocidae (true seals).
- Satellite tag:** Electronic tag gives off repeating signals that are picked up by satellites.

Acronyms

- BACI:** *Before-After/Control-Impact*. Experimental design in order to assess the possible effect of an impacts.
- COWRIE:** *Collaborative Offshore Wind Research Into the Environment*
- ESW:** *Effective half-strip width*. Width of the area searched effectively on each side of the line transect (Buckland et al. 2001).
- PPD, PPM, PPH** *Porpoise Positive Day/Minute/Hour*: Parameter for analysing T-POD data: One Porpoise Positive Day/Minute/Hour is a monitored day/Minute/Hour with at least one porpoise registration.

PTS: *Permanent Threshold Shift.* Permanent hearing damage caused by very intensive noise or by prolonged exposure to noise

SAM: *Static Acoustic Monitoring.*

SCANS: Small Cetacean Abundance in the North Sea and Adjacent Waters

SEL: *Sound Exposure Level.* Sound level of a single sound event averaged in a way as if the event duration was 1 s. Used here for comparing sound levels of pile strokes, independent of the number of strokes per minute

T-POD: *Timing POrpoise Detector.* Static acoustic monitoring device, which logs high frequency clicks of odontocetes.

TTS: *Temporary Threshold Shift.* Temporary reduction of hearing capability caused by exposure to noise

UK: *United Kingdom.*

Units

dB: *Decibel.*

E: *East*

Hz: *Hertz.*

km: *Kilometre.*

km²: *Square Kilometre.*

m: *Metre.*

min: *Minute*

NW: *Northwest*

Pa: *Pascal.*

pph: *porpoise positive hour.*

ppm: *porpoise positive minute.*

1 Introduction

The high potential for offshore wind energy production has led to a fast development along European coasts including the UK. Offshore windfarms have been installed in Denmark, Sweden, The Netherlands, Ireland as well as in several parts of the UK, and there are plans for almost all European coastal areas. From a marine mammal point of view, the turbine itself is of little relevance as compared to the foundation and the construction processes of offshore windfarms. Offshore windfarms create new permanent sub sea structures and their construction, operation and decommissioning emits noise from various sources into the sea. As turbine sizes and distances to the coast are increasing, there is also a need for larger foundations suitable for carrying turbines with a hub height of above 80 m and a weight of several hundred tons. Especially the construction work has raised concerns about possible impacts on marine mammals as until now most foundations are fixed to the seafloor by steel piles, which are driven into the bottom by large hydraulic hammers. These noise emissions are strong enough to harm marine mammals in the vicinity and cause disturbances over large areas. On the other hand, permanent structures may create new habitats attracting fish and thus also marine mammals.

As offshore wind energy production is spreading, there is an increasing need to measure and assess the impacts on marine mammal behaviour, abundance and distribution. The harsh conditions of the offshore environment and the behaviour of these animals, however, pose quite some restrictions to study their presence and behaviour and make it difficult to obtain statistically robust data needed to assess responses to specific anthropogenic activities. Marine mammals are highly mobile animals, which generally occur in low densities. Of all marine mammals found in UK offshore waters, there is only one species, which regularly reaches densities of more than one animal per square kilometre, the harbour porpoise. All investigations into the responses of marine mammals to offshore windfarms are thus challenged by the demand to obtain sufficient data on abundance and behaviour of widespread animals spending most of their time below the sea surface. To describe the effects of human activities, which may be as short as a certain period of pile driving or locally restricted to a small fraction of the potential habitat of the animals, a wide range of methods are used and new methods need to be developed.

In order to assess the methodologies for measuring and assessing potential changes of marine mammal behaviour and abundance COWRIE has commissioned BioConsult SH and the German Oceanographic Museum to conduct a desk-based study that will aim to:

Describe in detail the methodologies available for assessing the impacts (or lack of impacts) on marine mammals during the construction, operation and decommissioning of offshore windfarms. The report should aim to describe a variety of methodologies in detail and indicate the benefits and difficulties of each methodology in order to be able to assess their relative benefit. An estimate of the relative costs of implementing methodologies should also be provided.

The objectives of the project are to:

Indicate a practical, statistically robust and cost effective methodology (or choice of methodologies) using evidence based reasoning that will provide evidence of the impacts (or lack of impacts) of construction, operation and decommissioning of offshore windfarms on marine mammals.

If applicable, identify areas where further research or development of methodologies is required due to a current lack of adequate information.

In this report we will first define the scope of monitoring activities on marine mammals based on expected impacts and responses of these animals to offshore windfarms. We will then review the available field methods and existing studies and available methods to analyse their statistical power. Finally, recommendations for monitoring strategies will be formulated taking practical considerations and costs into account. The focus of the report will be on harbour porpoises as this is the most abundant cetacean in UK and surrounding waters and because most available data concern this species. However, other species will also be considered.

2 Marine mammal responses to offshore windfarm construction, operation and decommissioning

2.1 Impacts from construction, operation and decommissioning

There are basically two factors from offshore windfarms which may lead to diverging responses of marine mammals. First, noise emissions from construction, operation and decommissioning, which may harm or disturb marine mammals, and second, the creation of artificial reefs, which might in turn attract marine mammals.

2.1.1 Construction

The construction of an offshore windfarm includes a variety of ship-based activities and for most sea areas it is a fair assumption that ship noise will be elevated during construction works. The main source of noise emissions will in many cases arise from the construction of the foundations of turbines and transformer platforms.

Most offshore turbines are placed on steel foundations, which are fixed by piles driven into the seabed. The foundation may consist of a monopile, which is a large steel tube of 2 to 5 m in diameter, or of more elaborated constructions such as tripod or jacket foundations. Tripod and jacket foundations are usually fixed to the seafloor by piles of 2 to 3 m in diameter. The piles are driven into the bottom by some thousand strokes of strong hydraulic hammers, causing very strong noise emissions, which may be audible for marine mammals over large distances of several 10 km (Thomsen et al. 2006, Nedwell et al. 2007, Nehls et al. 2007, 2008). The whole operation lasts a few hours per pile. As an alternative, some projects, as the Nysted offshore windfarm in Denmark, have used gravity foundations, which are ideally just placed on the sea bottom without piling operations and thus cause much lower noise emissions.

The strength of noise emissions are dependent on a variety of factors such as pile dimensions, seabed characteristics, water depth as well as impact strengths and duration. Pile diameter has proven to be a good predictor for source and near source levels (Nehls et al. 2007, 2008). Noise emissions increase at about 3.1 dB per metre pile diameter (Figure 2-1). Source levels of large piles (4 to 5 m) reach peak values of 235 to 240 dB re 1 μ Pa, corresponding to SEL values of about 200 to 215 dB re 1 μ Pa²s. Nedwell et al. (2007) refer to even higher source levels. However, as measurements are not made directly at the pile, source levels are calculated values and the results may differ, depending on various assumptions on sound propagation. Sound spectra of emissions from pile driving are characterized by low frequencies and most energy is emitted below 1000 Hz.

Sound waves travel very well in water, and noise emissions from offshore pile driving will be audible against background noise to marine mammals over large distances. However, in order to estimate impact radii around a source, transmission losses have to be considered. Transmission losses occur from geometrical spreading and sound absorption. Transmission losses are dependent on water depth, structure of the sea bottom and on the sound frequency. In shallow waters below 20 m sound transmission below a certain border frequency is strongly reduced (Urick 1983), which may have significant effects on noise from pile driving.

In planning the monitoring of marine mammals around offshore pile driving activities, noise emissions and sound propagations need to be estimated before the work starts.

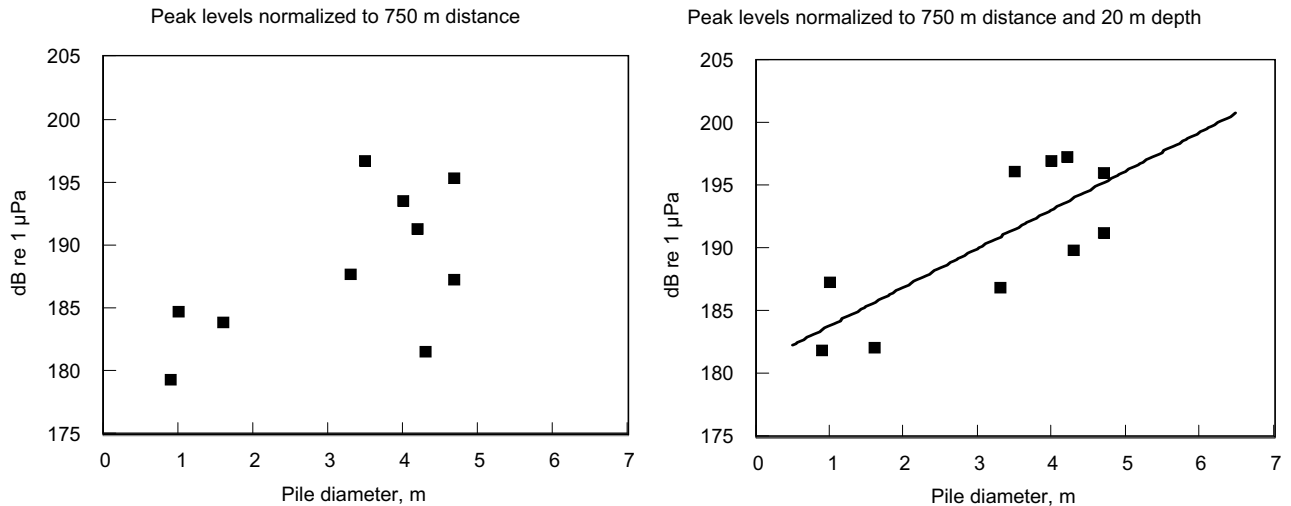


Figure 2-1: Peak levels of pile driving blows versus pile diameter. Left: Normalised to equal distance only. Right: With additional normalisation to 20 m water depth. The regression line in the right diagram has a slope of 3.1 D, where D is the pile diameter in metres (from Nehls et al. 2008).

2.1.2 Operation

During operation there will be noise emissions from the turbines and from ships servicing the windfarms. Whether or not ship traffic will be elevated compared to the pre-construction situation will depend on local conditions.

Noise emissions from operating wind turbines are of low frequencies and low intensity (Nedwell et al. 2007, Nehls et al. 2008, Figure 2-2) and probably not audible to marine mammals over distances greater than a few tens of metres, as the hearing abilities of these animals are best at higher frequencies.

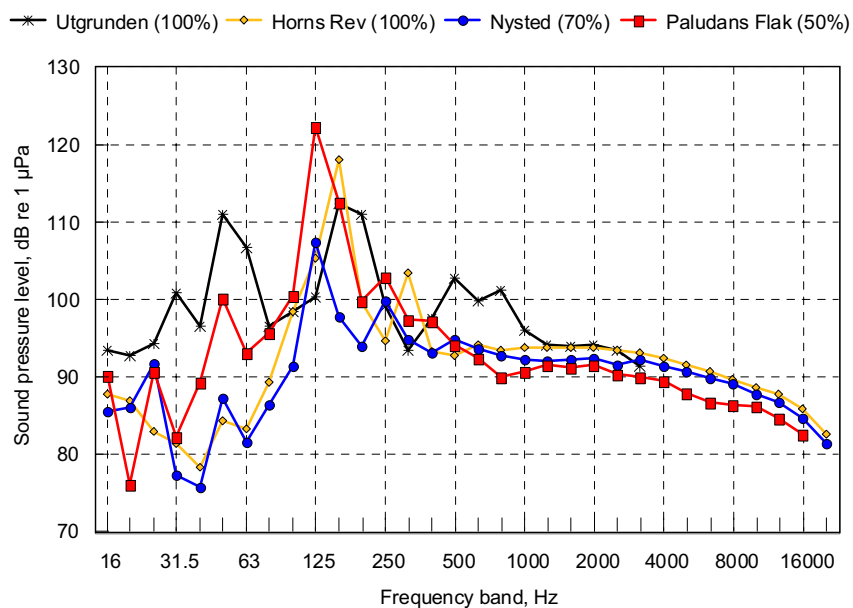


Figure 2-2: Third-octave spectra of noise radiated from offshore wind turbines. All measurements were made at 100 m distances. Values in brackets are approximate operating powers of the turbine during the measurement, with respect to its maximum power (ISD et al. 2007).

In addition to noise emissions, the function of the foundations as artificial reefs has to be considered. All underwater structures are soon overgrown by sessile benthic animals and algae which may significantly enrich the biomass locally and attract fish and in turn also marine mammals (Wilhelmsson et al. 2006).

2.1.3 Decommissioning

What activities will be exactly involved during decommissioning and how long it will take to remove a whole windfarm is not yet known (Nedwell & Howell 2004), however, elevated ship traffic is likely to occur. Pile foundation removal is likely to include abrasive jet cutting. Destruction of concrete foundations may eventually require blasting or the use of pneumatic hammers, if they cannot be lifted from the seafloor after dismounting the turbine. For abrasive jet cutting no sound measurements are available so far. Detonations would be a loud point source of underwater sound and consequently pose a serious risk of inducing tissue or hearing damage. Impact radii of explosions are mainly a function of charge weight (Urlick 1983).

2.2 Temporal and spatial aspects

Measuring the effects of windfarm construction, operation and decommissioning on marine mammals needs to involve data collection on very different temporal and spatial scales. Whereas pile driving operations are short-termed (hours per pile) and have potentially long-ranging impacts (tens of kilometers), the operation of wind turbines have long lasting (decades) but locally restricted effects. This has to be considered when selecting methods for the monitoring.

Data on spatial scales, e.g. measuring animal abundance in relation to the distance to an activity, allow to analyse differences between windfarm site and areas not or less affected by the windfarm whereas temporal data allow a comparison between different phases (before and during construction and during operation). With standard methods, data on either scale can be collected. E. g. in studies on cetaceans, static acoustic monitoring using T-PODs provides continuous data with a low spatial resolution (limited by the range and number of T-PODs used) whereas line transect surveys have a high spatial but low temporal resolution (limited by the number of surveys) (Tougaard et al. 2006b).

A species specific approach should take regional and seasonal variability in the occurrence of marine mammal species into account. This is of high importance as it may not always be possible to obtain sufficient data of a species within an area of interest in the relevant time. The collection of baseline data from a period before a possible impact as well as control data from a reference area outside the impact area offers the possibility to consider spatial and temporal aspects. If such data are available a BACI analysis (Before/After/Control/Impact) can be conducted (Chapter 5.2). In a BACI design, the seasonal fluctuations in abundance and distribution should be similar in both, impact and reference area. Thus, the baseline period must be long enough to account for seasonal variability in abundance and distribution, and the reference area should ideally be at a distance which warrants that it is not influenced by activities in the impact area.

Choosing the right variables and time frame of a study has implications on the ability to address specific questions. E. g., effects limited in time such as deterrence by piling noise as part of a longer "construction" period comprising many different activities may not be detected by a study comparing baseline and a general "construction" period due to averaging data over time. On the other hand, changes from period to period may be exaggerated by data from a short activity creating a pronounced reaction. Therefore it is helpful to know the extent of the response to any single activity. This could be achieved in additional controlled-exposure experiments (Tyack et al. 2004) or in an additional analysis of subsets of data, e. g. before and during construction (Tougaard et al. 2003).

From telemetry of seals (e.g. Tougaard et al. 2006d) and other marine mammals we may gain detailed information on the behaviour of a few animals. As behaviour may relate only to individuals, a general extrapolation to the whole population is problematic. Answering questions

about the behaviour inside versus outside the windfarm requires a spatial accuracy of the method, which cannot be achieved by ARGOS satellite transmitters (Tougaard et al. 2006d). For movements on a finer scale other methods like data loggers recording 3D diving tracks must be adopted.

2.3 Species differences

Although a higher number of marine mammals occurs in British marine waters, this report reviews methods which may be suitable for the more abundant species which are at first place harbour porpoise (*Phocoena phocoena*), followed by harbour seal (*Phoca vitulina*), grey seal (*Halichoerus grypus*) and bottlenose dolphin (*Tursiops truncatus*). Species may differ both in their sensitivities towards human activities like constructing offshore windfarms, as well as in their life styles and abundance, and thus require different monitoring approaches.

The methods chosen for impact assessments often suffer from difficulties in detecting marine mammals and exactly determining the species. For example, dedicated visual surveys in offshore areas are not well suited for seals. At the surface they are difficult to observe, except under very good conditions (sea state 0 to 2) and species determination at sea is difficult, even for trained observers. It may also be difficult to detect small cetaceans, and unless densities are high, numbers of sightings may be too low to apply statistical methods such as fitting of detection functions.

Acoustic surveys are well suited for the detection of small odontocetes which rely on their echolocation and therefore produce frequent clicks (Parvin et al. 2007). With T-PODs it is possible to distinguish between harbour porpoise and bottlenose dolphin vocalisations (see chapter 6.2.1). Whereas harbour porpoises use high-frequency narrow-band clicks, bottlenose dolphins produce relatively broad-band clicks at a lower frequency (Au 1993). Some authors thus refer to "high-frequency" and "mid-frequency" cetaceans (Southall et al. 2007). However, if different "mid-frequency" cetacean species (e. g., bottlenose dolphins and common dolphins) occur in the study area, conclusions from T-POD data are limited since to date it is not possible to distinguish between clicks from different species within these groups.

In contrast, pinnipeds require a different approach as they vocalise infrequently. They may be counted on haul-out sites where these are found in the vicinity to offshore windfarms (Teilmann et al. 2006) or followed individually by tagging (Tougaard et al. 2006d). However, conclusions from tagging studies are often limited because individual behaviour may differ substantially, and it is not clear if animals from haul-out sites further away do not use the area as well. As stated in chapter 2.2 the number of tagged animals is a critical factor in telemetry studies because a high variation in behaviour can be expected.

Marine mammal species differ in their sensitivities towards sounds with respect to hearing abilities, disturbance, masking, and hearing damage (National Research Council 2003; Thomsen et al. 2006; Southall et al. 2007; Nehls et al. 2008). For example, high-frequency cetaceans such as the harbour porpoise may be more susceptible to a threshold shift in hearing sensitivity than expected from the knowledge on mid-frequency cetaceans such as the bottlenose dolphin (Southall et al. 2007; Lucke et al. 2007a). The difference between hearing abilities, especially in the low-frequency range, is an important aspect to be considered especially with respect to the possible impact radius for odontocetes and pinnipeds (see chapter 2.4). For example, different thresholds eliciting behavioural response in harbour porpoises and harbour seals to the same kind of sound stimulus (underwater communication sound) is likely to reflect the different frequency-dependent hearing abilities, but also a different susceptibility between species or species groups (Kastelein et al. 2005; Kastelein et al. 2006).

It has to be noted, however, that differences in hearing abilities have not yet been documented to correspond to the animals' sensitivities and responses to noise emissions. There has been one study documenting marked responses of harbour seals to offshore pile driving (Tougaard et al. 2006), but responses of seals or bottlenose dolphins are not well documented. Speculations that seals may be more tolerant to loud impulsive sound such as piling or airgun impulses than harbour porpoises and other odontocetes may be due to limited data (National Research Council

2003). E. g., ringed seals (*Phoca hispida*) in Alaska, which most likely were habituated to industrial noise, did not respond dramatically to received impact pile-driving noise levels of 153 dB_{rms} re 1µPa (Blackwell et al. 2004). For bottlenose dolphins no data with regard to disturbance of piling noise exist so far. In other dolphins a significant response was shown. E. g., in the vicinity of piling activities indo-pacific humpback dolphin (*Sousa chinensis*) swam significantly faster during piling and abundance was lower immediately after piling compared to before activities (Würsig et al. 2000).

2.4 Estimating impact radii

The extent of the impact radii for noise emissions during windfarm construction, operation and decommissioning may have implications for the well-being of marine mammals because hearing impairment, masking or disturbance are based on the range-dependent received sound level.

Thus, impact radii are an important aspect in environmental impact assessments for offshore windfarms. An accurate estimation of impact radii can help to

- prevent damage to animals
- optimise mitigation measures
- plan monitoring activities and
- determine reference sites not influenced by activities.

Impact radii depend on the animals' specific sensitivity, the source level and properties of a sound (e. g., frequency band, continuous/impulsive), sound radiation and effect level.

Ranges at which animals may suffer damage can be estimated using measurements of source levels (peak, SEL), a suitable sound propagation model and information on sound exposure levels which are potentially dangerous for the different marine mammal species or species groups or elicit a significant response. Recently, Nehls et al. (2008) summarised the current knowledge on a number of underwater noise sources, different noise exposure criteria and their implications on marine mammals.

Existing and proposed regulations of noise emissions aim at different effect levels such as avoiding physical damage or death, masking of biologically significant sounds or preventing disturbance. This effect level forms a basis for an estimation of different impact radii.

Usually, four impact zones are differentiated (Richardson et al. 1995) corresponding to the different effect levels:

- the zone of hearing loss, discomfort, or injury,
- the zone of masking,
- the zone of responsiveness and
- the zone of audibility.

Physical damage

There is a broad consensus among scientists and the public that marine mammals may not be killed deliberately from any human activity. Similar demands are imposed by legal requirements from European and British legislation. In this respect, irreversible physical damage which does not directly kill a marine mammal but impairs its physiological functions and reduces its survival should be treated in a similar way. Concerning this matter, threshold levels for avoiding physical damage to marine mammals by intense sound are of particular importance. As a consequence, no marine mammal shall be exposed to noise levels putting it at a risk of physical damage. Thus, it should be assured, that no marine mammal occurs within an area where noise levels exceed a threshold at this effect level.

High-intensity sound can inflict injury or hearing damage on marine mammals. The degree of a physical damage depends on several properties of sound: the received energy content, peak

pressure, signal duration, spectral type, frequency (bandwidth), duty cycle, directionality, and signal rise time (Richardson et al. 1995). Generally speaking, continuous sounds, higher frequencies and narrowband signals are more dangerous for marine mammals than transient (impulsive) sounds, lower frequencies and broadband signals of the same strength (Ketten 2002). Therefore, each type of sound requires a separate assessment. E. g., during piling repeated high-intensity pulses occur over long periods. Each pulse adds to the overall sound energy received by the ear and increases the potential of hearing damage. Thus, in the case of multiple pulses, the cumulative impact must be considered (Southall et al. 2007).

A temporary hearing threshold shift (TTS) is the result of a metabolic exhaustion of sensory cells in the cochlea caused by intense sounds (Gordon et al. 1998; Ketten 2002; Marine Mammal Commission 2007). TTS may temporarily reduce the animal's ability to perceive biologically significant sounds and therefore has a similar effect as masking (see below) but persists for a period after sound exposure (Richardson et al. 1995). Within limits TTS is reversible (Nachtigall et al. 2003; Nachtigall et al. 2004). Repeated, prolonged or chronic exposure to high sound levels as well as brief exposure to extremely loud noise or short signal rise times (e. g., in underwater explosions) can inflict a structural damage of sensory cells resulting in a permanent threshold shift (PTS) or acoustic trauma (Richardson et al. 1995). Further, acoustically induced behavioural changes (such as rapid ascent of deep diving whale species such as beaked whales) can inflict non-auditory physical trauma (Nowacek et al. 2007).

In attempts to define threshold levels (and thus for the estimation of impact radii) it needs to be decided, whether TTS or PTS level are defined as the onset of physical damage. Although PTS will better fit to the definition of physical injury, TTS may also be considered as a relevant parameter in order to follow the precautionary principle and take remaining uncertainties into account. Southall et al. (2007) developed initial scientific recommendations of marine mammal noise-exposure criteria with respect to peak level and SEL based on PTS onset for seals and mid-frequency cetaceans such as bottlenose dolphins (Table 2-1). For high-frequency cetaceans such as harbour porpoises the suggested thresholds may be too high, as indicated by preliminary (not yet published) data which suggest a lower TTS onset than could be expected from mid-frequency cetacean data (Lucke et al. 2007a). Information on different approaches for defining noise thresholds are reviewed in Nehls et al. (2008).

Nedwell et al. (2007 and references therein) developed an approach to assess the effects of noise exposure to fish and marine mammals, relating the strength of a noise emission to the hearing threshold of a species ($dB_{ht(species)}$ approach). Exposures with more than 90 dB above the hearing threshold are assumed to cause hearing damage. The application of the $dB_{ht(species)}$ approach to hearing damage has recently been challenged by the Sea Mammal Research Unit (SMRU) of the University of St. Andrews (SMRU 2007) and it remains to be questionable, whether the $dB_{ht(species)}$ approach can be applied over the whole range of a species hearing abilities (see Nehls et al. 2008). As a result, noise emission injury radii is based on the data presented in Southall et al (2007).

Table 2-1: Injury(PTS-) criteria according to two different functional hearing types for different exposure types proposed by Southall et al. (2007).

<i>functional hearing type</i>	"High-frequency" cetaceans		Pinnipeds	
	peak pressure (unweighted) dB re 1µPa	total energy (SEL) ("M-weighted") dB re 1µPa ² s	peak pressure (unweighted) dB re 1µPa	total energy (SEL) ("M-weighted") dB re 1µPa ² s
<i>Single pulses</i>	230	198	218	186

Masking effects

The zone of masking is probably the most difficult to define on the basis of scientific evidence. It is generally accepted that man-made noise can interfere with the detection of sounds, which have a biological significance for marine animals. Masking occurs when both the masking noise and the signal have similar frequencies and overlap or occur very close together in time (National Research Council 2003). However, very intense sounds can mask signals even if the signal frequency lies outside the spectrum of masking noise (Richardson et al. 1995). In a noisy environment, structured signals such as echolocation click trains or complex songs may be better detected because their frequency content and temporal characteristics differ from those of the background noise.

Biologically significant sounds include communication calls, echolocation clicks, and sound produced by predators or prey. Especially for low frequency communication sound (review in: Richardson et al. 1995) there is a general potential for masking by anthropogenic noise, which often has its maximum energy at similar frequencies. High-frequency sound, e. g. of sonars, depth sounders and fish finders may interfere with echolocation and high-frequency communication sounds used by different odontocetes (cf. Koschinski et al. 2008).

Not many studies addressed the possible extent of masking. Masking in harbour porpoises has been shown by AEP measurements (Lucke et al. 2007b). In the presence of simulated small offshore wind turbine noise at a maximum received level of 128 dB re 1 μ Pa, perception of low-frequency acoustic signals (0.7, 1 and 2 kHz) was reduced by 4.8 to 7.3 dB in his study animal. At 115 dB no significant masking effect was measured. The authors describe the likely masking zone as extending several tens of metres. Indirect evidence for masking of marine mammal signals is the development of antimasking behaviour which may be lengthening or increasing source levels of calls or frequency adaptations. For example, killer whales (*Orcinus orca*) increase their call duration as a reaction to boat noise (Foote et al. 2004), humpback whales (*Megaptera novaeangliae*) lengthen their sound to compensate for acoustic interference by LFA sonar (Miller et al. 2000), indo-pacific bottlenose dolphins use whistles of lower frequencies with fewer frequency modulations (Morisaka et al. 2005), and beluga whales (*Delphinapterus leucas*) shift their echolocation clicks an octave higher as adaptations to high ambient noise (Au et al. 1985). Antimasking behaviour may compensate for a loss of information to a certain extent. However, it is unknown what consequences this change in behaviour may have for the viability of individuals and marine mammal populations (e. g., National Research Council 2005).

Masking is only an effect to be considered with continuous emissions as from shipping or operation noise of wind turbines and to a lesser extent with sporadic and short termed emissions as sonar or pile driving depending on their level and signal duration (Nehls et al. 2008).

Disturbance

Disturbance can be defined as a change in the natural behaviour of an animal which is induced e. g., by acoustic stimuli. Thus, any behavioural response to sound emissions may be regarded as disturbance. Behavioural responses to anthropogenic noise are highly variable with respect to the characteristics of noise sources and the species exposed to it. Behavioural responses to anthropogenic sounds may in many cases be subtle or not harmful to individuals or populations.

Behavioural responses of marine mammals to noise vary greatly from very subtle reactions (startle; small changes in swimming direction) to strong avoidance behaviour (swimming away from the noise source and avoiding a large area) (Thomsen et al. 2006). Responses to different noise sources have been reviewed by several authors (e. g., Richardson et al. 1995; Würsig & Richardson 2002; National Research Council 2003; Weilgart 2007; Nowacek et al. 2007). Madsen et al. (2006) give some specific information related to wind turbine underwater noise. In the Danish EIAs for Nysted and Horns Rev windfarms some responses to noise were also documented (see chapter 3).

Scientific studies have documented both, the presence and absence of reactions to sound stimuli. The absence of an adequate avoidance reaction to dangerous sound levels may be explained by different factors, including a high motivation to stay in a good foraging habitat or the existence of a complex sound field (multiple reflections create interference patterns)

preventing animals from escaping in the right direction. This may also have implications for hearing impairment (see above).

From an ecological point of view, the consequence of disturbance needs to be measured and assessed, rather than the disturbance itself (e.g. NRC 2005). Relevant questions to be asked in this respect are how does an animal respond, how long does a response last, how many individuals are affected, and do the animals actually avoid an area of sound emissions? Behavioural responses which result in strandings of marine mammals or abandoning of offspring or which significantly affect the conservation target of marine protected areas will have to be assessed as severe and measures should be taken in order to avoid such responses. The habitat directive prohibits deliberate disturbance of strictly protected species. Thus, significance of an impact has to be assessed on two levels: first the impact on individual animals (especially to the relevance a disturbance has for critical behaviour such as e. g. feeding, resting or nursing) and second in relation to the number and proportion of affected animals (Joint Nature Conservation Committee 2007; European Commission 2007).

Almost all data on disturbance of marine mammals, whether observational or experimental, have concerned short-term reactions. Due to lack of data, long-term consequences of short-term disturbance reactions for individuals and populations are not well understood (National Research Council 2005). Sounds resulting in one-time acute responses are less likely to have population-level effects than are sounds to which animals are exposed repeatedly or for a long time. Further, repeated disturbance of marine mammals in critical habitat or during vital times (feeding, reproduction, nursing, migration) leading to reduced foraging efficiency, habitat abandonment and confinement of animals to less favourable habitat may decrease their fitness and have the potential to induce long-term population effects. Such alterations are difficult to demonstrate (David 2002, National Research Council 2005, but see Bejder et al. 2006, Lusseau et al. 2006).

Assessing disturbance is a difficult task because for most marine mammals specific sound thresholds are not exactly known. Southall et al. (2007) proposed a general marine mammal disturbance threshold based on the onset of TTS for single pulses (Table 2-2). However, because there are many variables and uncertainties in the relation of noise source characteristics and responses of marine animals, a single value cannot adequately describe the impacts of disturbance. Behavioural effects have been measured in many occasions at considerably lower noise levels than the proposed ones, and TTS is usually rather discussed to be a subtle a form of physical damage. Therefore, a disturbance criterion for single pulses based on TTS onset is considered to be too high (Nehls et al. 2008).

Table 2-2: Single pulse behavioural response criteria (defined by onset of TTS) according to two different functional hearing types for different exposure types proposed by Southall et al. (2007)

<i>functional hearing type</i>	"High-frequency" cetaceans		Pinnipeds	
<i>dual injury criteria</i>	peak pressure (unweighted) dB re 1µPa	total energy (SEL) ("weighted") dB re 1µPa ² s	peak pressure (unweighted) dB re 1µPa	total energy (SEL) ("weighted") dB re 1µPa ² s
<i>Single pulses</i>	224	183	212	171

Richardson et al. (1995) report avoidance reactions of marine mammals exposed to continuous sounds above 120 dB and conclude that marine mammals would avoid areas with continuous levels above 140 dB. Other studies reported behavioural responses to higher continuous noise levels but so far little is known about the onset. The approach of the US National Marine Fisheries Services NMFS (2007) implies an onset of behavioural responses above 120 dB and 100 % aversive reactions above 180 dB SEL. At present it is difficult to judge at which point of this continuum a threshold could be set defining significant responses.

Based on the dB_{ht(species)} approach (see above) Nedwell et al. (2007) calculated behavioural impact ranges for specific UK offshore windfarm projects. Some of the estimated behavioural

impact ranges are as low as 2,5 km for harbour porpoise despite rather high calculated source levels. As our understanding of the relationship between signal strength and behavioural response is still very limited, it is recommended to take much larger impact areas into consideration.

3 Description of existing studies

As stated above (chapter 2.3) current methods to investigate the effects of offshore wind farm construction and operation are not well suited to investigate seals. Until now no studies have looked directly at effects on any cetacean species other than the harbour porpoise. Here we present mainly the findings of the relevant published reports from Danish wind farm sites with respect to harbour porpoises and harbour seals.

A BACI (Before-After Control-Impact) experimental design is desirable to assess impacts. Some care must be taken when interpreting BACI results. It must be ensured that monitoring covers the entire period, starting at least one annual cycle before any construction activities, the area monitored extends beyond likely impacts (ideally more than 20 km from the source of activity) and the study is maintained well into the operational phase, preferably at least to five years from the start, to assess possible long-term impacts (Evans 2008). In order to obtain a good understanding of the context in which the area is being used by the target species (e. g., for feeding, breeding or migration) a full year of baseline data in a control situation before any human activity has started, should be collected at the minimum.

3.1 Harbour porpoises

3.1.1 Nysted

The effect of Nysted Offshore Wind Farm on harbour porpoises was examined by Tougaard et al. (2006a). Due to low sighting rates of harbour porpoises during bird surveys, a line transect survey method was considered to be ineffective. Their study rather focussed on the deployment of T-PODs at 3 locations inside the wind farm and 3 reference locations, presumably affected by the same natural variations in the environment as the impact area (e. g., hydrography and food availability), at a distance of approximately 10 km to the windfarm site. With respect to piling (gravity foundations with some sheet piling), which creates very intense sound, a reference station should ideally be at a greater distance than 10 km from the impact area (a T-POD study during ramming of monopiles at the Horns Rev windfarm demonstrated pronounced behavioural reactions at a radius of 21 km from the construction site (Tougaard et al. 2006b, see below)).

The Nysted windfarm study uses a BACI-analysis comparing the reference and impact area during the baseline period and the impact periods (construction, first year of operation and second year of operation). The baseline period started in April 2002 when three T-PODs were deployed in the windfarm and three T-PODs at a reference station. The study ended in December 2005, 27 months after operation started (September 12, 2003). The construction lasted from mid-June 2002 to July 27, 2003. During construction, impact may have arisen from laying out gravel, sheet pile vibration, placement of concrete foundation, ballast filling, erosion protection, digging and cable laying.

One of the drawbacks of this study is that the baseline period did not cover a full season before impact and a full BACI design with T-PODs serving as a control was in place only five months before construction. Fortunately, the number of clicks logged during this short pre-construction period was high, considered by the authors to be sufficient to allow a BACI analysis.

Tougaard et al. (2006c) showed that porpoise detection via T-PODs may be density related. Therefore, suitable indicators for a change in habitat use by harbour porpoises are differences in daily echolocation activity described by the parameter *porpoise positive minutes per day (PPM)* and frequency of occurrence described by the parameter *waiting time* (further description of the parameter in chapter 6.2.1). In addition, two behaviour related parameters were used in the

Nysted wind farm study. These are *encounter duration* describing the time porpoises spent in the vicinity and *number of clicks per PPM* describing the intensity of echolocation within the detection range of T-PODs with a high number indicating foraging behaviour or investigation of objects close to the T-POD (including the T-POD itself). Since from the latter parameters no conclusion about the exact behaviour can be drawn (cf. Koschinski et al. 2008) conclusions are mainly based on *PPM* and *waiting time*.

Statistical analysis of parameters took the variation between areas (impact and reference), stations (within areas), period (pre-construction, construction, first and second year of operation), month (seasonal variation) and area x period (BACI-effect) into account.

Spatial variation between impact and reference area was significant for all four parameters providing evidence for systematic differences between those areas. Variation between periods was significant for all parameters except for the *number of clicks per PPM*. And seasonal variation was significant for all parameters except for *encounter duration*. Finally, changes in all parameters differed significantly among periods between impact and reference area (BACI effect).

During the 5 months **baseline period**, differences in *waiting time* and *PPM* were not significant between impact and reference area.

During **construction**, *waiting time* increased and *PPM* decreased in both areas. The BACI effect for the comparison between baseline and construction period was significant for both parameters characterised by a more pronounced decrease in *PPM* and increase in *waiting time* in the impact area compared to the reference area. In the windfarm area, the effect was dramatic. *PPM* decreased by a factor of 45 compared to the baseline data and *waiting time* increased from a median of 11 h to 53 h. In the reference area *PPM* declined only by a factor of 2.3 and waiting time increased from 16 to 18 h. This indicates a general effect of construction on porpoise abundance in the whole surrounding area and a strong deterrent effect close to the construction activities decreasing with distance.

Also during the **first year of operation** increased *waiting times* and decreased *PPM* compared to the baseline period were detected indicating that fewer porpoises were present in the windfarm area. The BACI effect was still significant for a comparison with the baseline period. *PPM* rose only slightly in both areas and *waiting time* declined marginally (from 53 to 52h in the impact area) compared to the construction period.

In the second year of operation there seemed to be a tendency towards a return to baseline levels with respect to *waiting time* and *PPM*. The BACI comparison was still significant compared to baseline data indicating a higher impact inside the windfarm compared to the reference area. *PPM* doubled in this period in both areas compared to the previous year. As a result, in the reference area outside the windfarm, activity was back to baseline levels whereas in the wind farm area the level was still reduced by a factor of 8. For *waiting time* there was a similar result. Whereas the value in the reference area became shorter than during the baseline period (10h vs. 16 h) the waiting time in the impact area was still three times larger than in the baseline period (34 vs. 11 h).

The echolocation behaviour related parameters *encounter duration* and *number of clicks per PPM* corroborate these results. The BACI contrast for *encounter duration* showed a strong relatively larger decrease in the impact area compared to the reference area from baseline to construction followed by a significant relatively larger increase from construction to operation. However, after two years of operation, the *encounter duration* was still reduced by 20 % within the wind farm. The BACI effect for the *number of clicks per PPM* was significant between the baseline period and construction and between baseline and first year of operation. In the second year the *number of clicks per PPM* was back to baseline as demonstrated by a non significant BACI effect.

3.1.2 Horns Rev

The effects of Horns Rev windfarm was investigated by Tougaard et al. (2006b). Based on the employment of up to 6 T-PODs and aerial and ship based line-transect surveys, the authors

investigated harbour porpoises before impact, during construction and during operation of the windfarm.

T-POD study

Data were collected via T-PODs between July 2001 and the end of 2005. Due to technical and logistical problems the time series of T-POD data contained larger and smaller gaps. Also lost T-PODs had to be replaced with other versions, making it difficult to compare the data between T-PODs. Data were partitioned into four periods: baseline period before construction from July 1, 2001 to March 3, 2002; construction period from March 4, 2002 to December 18, 2002; a semi-operational phase from December 18, 2002 to December 31, 2004; an operational phase from January 1 to December 31, 2005. Two T-PODs were deployed within the windfarm area and four T-PODs were deployed at different stations at distances of 7 km (1 NW, 1 E), 10 km (NW) and 21 km (NW). The stations outside the windfarm area served as reference positions.

For data analysis the same parameters as in the Nysted windfarm study (see above) were used. As in the Nysted windfarm study, statistical analysis of parameters took the variation between areas (impact and reference), stations (within areas), period (pre-construction, construction, semi-operational and operational phase), month (seasonal variation) and area x period (BACI-effect) into account. Further, T-POD version and transducer type were considered due to replacement with different T-POD types.

Although substantial losses of data led to a highly unbalanced dataset, the factors of the BACI analysis were not confounded. Significant effects were found for the following indicators: T-POD specific variation (significant in 3 of 4 parameters), transducer type (only significant for *waiting time*), area (only significant for *encounter duration*). Also a significant seasonal variation and variation between periods were found (for all parameters). A significant BACI effect was only found for the parameter *PPM*. This means that only for this parameter the period variation was different between areas. Also a significant correlation for successive observations was found (for all indicators).

During the 8 months **baseline period**, differences in *waiting time* and *PPM* were not significant between impact and reference area. *PPM* was only slightly higher in the impact area (29.9 min) compared to the reference area (25.3 min). The mean waiting time was 2.3 h for both areas combined.

During **construction**, the BACI effect for the comparison between baseline and construction period was not significant for all parameters. An effect of ramming activities demonstrated elsewhere (Tougaard et al. 2003) may not have been detected by the BACI analysis due to reference positions too close to the impact area and the definition of the "construction" period including many different activities, which may elicit different behaviour. During **semi-operation** a particular low *PPM* was observed in the windfarm (4.1 min vs. 29.9 min in the baseline period). *Waiting time* increased to 3.7 h (baseline 2.3 h). These differences were not significant in the BACI analysis, however.

During full operation waiting time decreased to 1.8 h. *PPM* increased to 86 min (21-fold increase relative to the semi-operation period) in the impact area and to 87 min (+87 % compared to semi-operation) in the reference area. This was a significant shift from the semi-operation period to full operation. However, there is no significant relative shift from baseline to operation. This can be explained by a particularly low *PPM* in the impact area during the semi-operation period.

The BACI contrast further revealed a significant relative increase of 31 % in number of clicks per *PPM* from semi-operation to operation phase. This was caused by a sharp decrease of this parameter in the reference area. The encounter duration was consistently higher in the reference area than in the impact area.

Pronounced short-term reactions to piling noise were also reported by Tougaard et al. (2006b) from an analysis of subsets of data. The first waiting times after pile-driving had ceased, were significantly longer ($p < 0.0001$ for three T-POD positions within the windfarm, at 7 km and at 21 km distance) than all other waiting times during the pile-driving period. Mean *waiting times*

increased from between 1.9 and 2.9 h to values between 5.6 and 8.1 h. Interestingly, the magnitude of response did not diminish with distance from the construction site. This reaction was not found in the BACI analysis because it was based on the whole construction period, which includes piling and a range of other activities.

Line transect study

A ship based survey covered an area of 800 km² and consisted of east-west orientated transect lines 500 km in length. During construction, the surveys only covered the core area, which could be completed within one day. Three observers on the same platform recorded observations with time/position, bearing, distance and number of animals. A spatial modelling analysis included the fitting of detection functions to the sightings data of each cruise and correlation with environmental factors such as surface temperature, salinity and tide. A BACI analysis of porpoise density was conducted from 20 out of 30 cruises (cruises with > 30 sightings) over the impact area (plus 280 m margin) and three reference areas arranged in shells around the impact area. Five surveys were conducted in the baseline period before impact, three during construction, and six each during semi-operation and operation.

There was a considerable variation in porpoise sightings between surveys and across the year. Most porpoises were seen during late summer with generally few sightings during the winter months. However, one winter survey had the highest number of sightings.

The BACI analysis of the density model revealed significant variations between periods and between cruises. Variation between areas and the overall interaction between area and period (combined) were not significant. Only during construction a significant change in porpoise density was observed when comparing the impact area with the outer reference area (distance from impact area > 10 km to W and E, > 8 km to N and S). A density gradient from the impact area to the outer reference area indicated an effect of construction activities. The main results were a decline in density by a factor of 8.8 from baseline to construction, an increase by a factor of 4.8 from construction to semi-operation and 4.4 to operation, respectively. Furthermore, during the course of the environmental impact study, behavioural observations revealed significant differences between the frequency of observed behaviours such as directional/non-directional swimming, logging and porpoising in ramming periods compared to periods without ramming (Tougaard et al. 2003). At a range of 15 km non-directional swimming (supposed to be associated with foraging) was clearly under-represented while directional movement was over-represented during ramming.

Aerial surveys for birds and marine mammals (one platform with two observers, altitude 76 m, speed 185 km/h) conducted along 30 north-south oriented track lines, covering a larger area (1800 km²) revealed no results. No spatial modelling could be applied to the data because only 7 out of 36 surveys had more than 30 porpoise sightings and hence most surveys were not suited for statistical analysis.

Other studies

At North Hoyle offshore windfarm site in the Irish Sea (NWP Offshore Ltd 2006) marine mammal sightings data was extracted from dedicated bird survey data. The low number of marine mammal sightings between 2003 and 2006 (46 sightings of mainly harbour porpoises and grey seals) and the lack of standardisation of survey effort across the area made the use of statistical methods (such as a BACI analysis) impossible.

T-POD study Horns Rev and Nysted on behalf on the German Federal Ministry for the Environment

The main objective of this study, which was carried out by the University of Hamburg, Germany in co-operation with BioConsult SH, is the review of potential effects of offshore windfarms in the areas surrounding windfarms and/or single turbines. Study areas were the two existing windfarms in Denmark, Nysted (Baltic Sea) and Horns Rev (North Sea). The project started in 2005 when the windfarms were already built and in operation for more than two years. The final

report will be finished in summer 2008. First results are published in Blew et al. (2006) and Diederichs et al. (2007a,b). The following questions were handled without a base line:

Do small-scale differences exist between the presence / absence of harbour porpoises between areas inside and outside?

Do small-scale differences exist in the behaviour of harbour porpoises inside and outside?

Can wind farms cause potential differences?

Which role do different factors, such as water depth, topography of the seabed, noise from ships, etc. play?

To answer these questions, a passive acoustic monitoring method was used, consisting of T-PODs. The data investigation lasted from June 2005 until November 2006. An important precondition for the comparison of data from different T-PODs is the same sensitivity of the devices. Therefore all T-PODs used were calibrated in a test tank as well as in the field. The study design was to install 10 T-PODs in each windfarm at the same time. Respectively, 5 devices were fastened in a row with a distance of approximately 600 m to each other (Figure 3-1). Two devices of each row were placed outside the wind farm up to a maximum distance of around 1,400 m to the next wind turbine. Two of the three devices within the wind farm were placed next to a wind turbine (below 200 m).

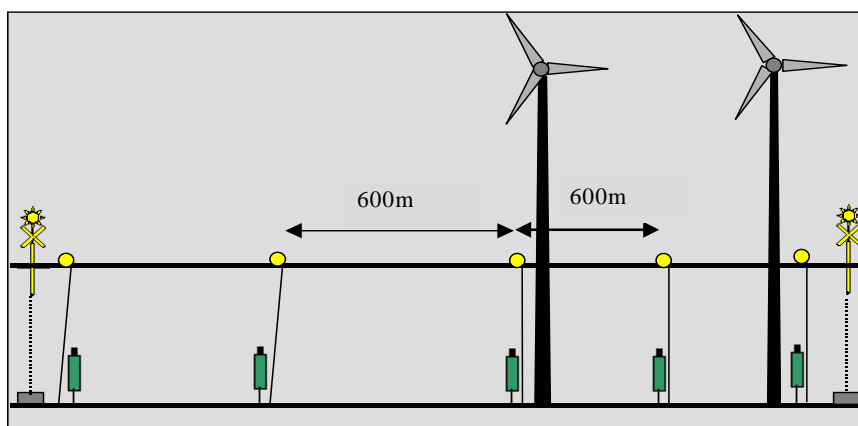


Figure 3-1: Linear array of T-PODs (transect) from outside (left) to inside the wind farm (right). T-PODs inside the wind farm have different distances to the wind turbines.

The results showed that harbour porpoises were recorded on the T-PODs nearly daily in the two areas of Horns Rev (North Sea) and Nysted (Baltic Sea). A significant difference between the marine areas regarding the number of registered harbour porpoises was detected with a high rate of harbour porpoise contacts in the North Sea. This result corresponds with aerial surveys that observed the density of harbour porpoises in both seas (Siebert et al. 2006).

In both areas a clear seasonal pattern was observed with a maximum in summer and minimum in winter. The differences in presence of harbour porpoises inside of the whole investigated area of the windfarm was higher between two areas that were spatially separated through some kilometres than the difference within one T-POD row between inside and outside areas of the windfarm. Referring to this, no consistent trend was observed so that no significant influence of the windfarm on the occurrence of harbour porpoises could be seen. Wind itself and thus the different performance of the turbines had no significant influence on the presence of harbour porpoises inside the windfarm. Even the shut down of all turbines in the offshore wind farm Nysted had no effect on the presence of harbour porpoises. Clear differences of activity of harbour porpoises were detected between day and night depending on the position of single T-PODs in relation to single wind turbines in both windfarms. Close to the turbines a higher activity during night could be measured (Figure 3-2). These findings are consistent with surveys of the occurrence of fish within the area of the Nysted windfarm (Leonhard et al. 2006). First

analyses of the clicktrain – structures show a different behaviour of harbour porpoises in Nysted and Horns Rev. The study shows that valuable results regarding the response of harbour porpoises to the operation of offshore windfarms can also be obtained without a solid database of data gathered before the windfarm was constructed. This does not include any results about a comparison of a potential BACI-effect on the abundance of harbour porpoises, but shows potential to investigate the responses of animals to a windfarm to find conclusions regarding their reactions.

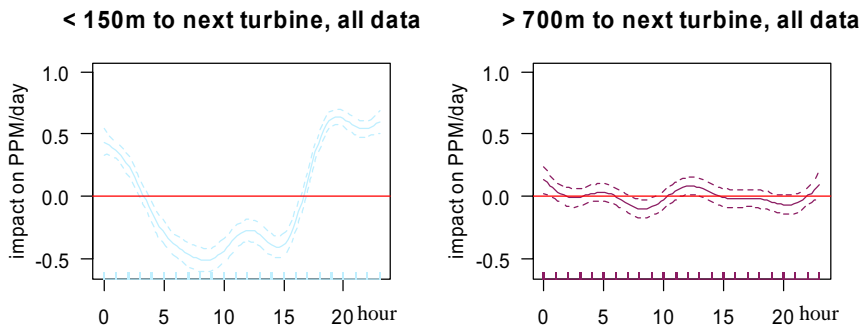


Figure 3-2: GAM smoothing curves fitted to the 24 hours of a day on the presence of porpoises (ppm/day) closer than 150 m to the next turbine (left) and more than 700 m away from the next turbine (right) in the Nysted windfarm.

Baseline data on the harbour porpoise in relation to the intended wind farm site NSW, in the Netherlands 2004

In order to provide a thorough description of the ecological reference situation this study, conducted by Brasseur et al. (2004), deals as a baseline study to evaluate the possible impacts of a planned near shore windfarm on harbour porpoises. In order to provide a robust baseline of porpoise activity within a proposed nearshore wind farm and suitable reference area, Brasseur et al. (2004) employed 3 data collection methods; firstly, during a whole year echolocation sounds of the animals were collected via T-PODs. Secondly, bi-monthly ship-surveys were conducted to obtain an estimate for density. Finally, hydrophones were towed behind the survey ship to corroborate the visual data. These studies proved that porpoises frequently occurred in the target area and also in the control sites. Intensity of the porpoise activity was clearly higher in winter months. Observations surpass the expectations with respect to the amount of animals and recordings. The study presents a solid baseline on which possible effects of the not yet built wind farm on harbour porpoises can be proved in future.

3.2 Seals

3.2.1 Nysted

At Nysted windfarm, a telemetry study was conducted to examine the relative importance of the windfarm area for grey and harbour seals from the nearby seal sanctuary Rødsand (Dietz et al. 2003). 95 % Kernel home ranges and track lines were determined for four harbour seals and six grey seals. Harbour seals remained within approximately 50 km of the sanctuary year-round whereas grey seals made extensive movements of up to 850 km to Sweden, Germany, Estonia and Latvia during the breeding season in winter. This implies that the Rødsand area may be more important to harbour seals than to grey seals. The windfarm site was included in all four 95 % Kernel home ranges of the harbour seals. But only three of the animals have actually been documented in the area. Presence in the windfarm site made up only 0.41 % of all recorded locations. Four of the six grey seal Kernel home ranges included the windfarm site. Due to the large dimensions of their home ranges, the windfarm site made up only 0.07 % of all recorded grey seal locations. This study did not examine the effects of construction and operation of the wind farm.

Effects of construction activities (mainly piling and boat traffic) was investigated on the seal haul-out site Rødsand at a distance of approximately 4 km to the windfarm using remote video monitoring (Edrén et al. 2004). During construction the number of seals on the haul-out increased compared to the baseline period (decline in April, increase in May, June and July, no difference in August). During the construction period (without ramming) the number of seals on the haul-out increased compared to the baseline period (decline in April, increase in May, June and July, no difference in August). During ramming periods in particular, a significant decline (31 to 61 %) in seal numbers on the haul-out was recorded.

A further study at Nysted deals with aerial surveys of seals resting on haul-out sites close to the windfarm (Teilmann et al. 2006). The study investigated if seals tend to avoid the disturbance from the windfarm, and use alternative seal sites further away from the windfarm than before the construction. Rødsand seal sanctuary, lies 4 km away from the windfarm, and is therefore the closest land site for seals in the area. Rødsand and five other seal haulout sites in the area are believed to hold a closed harbour seal population with little exchange to other harbour seal populations. Monthly aerial counts of harbour and grey seals were conducted from March 2002 to October 2005. Furthermore, aerial surveys from late August from 1990-2000 were included as part of the baseline data. The data provide information on the seasonal and interannual use of the different seal haul-out sites. The seal epidemic in 2002 killed about 20% of the harbour seals, but in August 2003 the number of harbour seals had almost recovered completely. During 2003-2005 the population increased by almost 17%. During the construction of the windfarm the relative importance of Rødsand seal sanctuary decreased slightly, but not significantly compared to the other five most important seal localities in the southwestern Baltic Sea area. During the operation of the windfarm in 2004 and 2005 the proportion of seals (harbour and grey seals combined) at Rødsand increased to 34 and 33%, respectively, and thereby again became the most important seal site in south-western Baltic. Except for an increasing importance of Rødsand during operation in May and June 2004-2005, no general shift in proportion of seals (harbour and grey seals combined) at Rødsand relative to the other localities was seen. Whether the increasing proportion of seals during operation in May and June 2004-2005 could be due to a positive effect from the windfarm is unknown. Rødsand remains less important to the harbour seals during October-March. There are no indications that the construction activities from late June 2002 to December 2003 and the first two years of operation of the windfarm in 2004-2005 affected the local Rødsand harbour and grey seal populations differently from the other populations in the western Baltic Sea. The Rødsand seal population has increased substantially in size in 2004 and 2005. Whether there are any positive effects from the windfarm, e.g. by creating an artificial reef that attracts more fishes, and hence more seals remains to be investigated.

3.2.2 Horns Rev

Tougaard et al. (2006d) investigated the movements of harbour seals before, during and after construction of Horns Rev windfarm using satellite telemetry and data loggers for parameters relevant for the movement of animals (depth, speed, pitch, roll and 3d compass orientation) and some environmental parameters temperature and light level). Although this method is not ideal (chapter 2.2), tracking of animals from nearby haul-out sites were the only option for a seal study. However, even in combination of satellite transmitters and data loggers the spatial accuracy proved to be too low for drawing conclusions on the scale of the windfarm.

The study considers the possible impact on habitat use by piling and other construction activities and operation compared to a base line period without any influence by an offshore windfarm. The method does not qualify for a BACI analysis due to the low spatial accuracy reached and a possible bias by individual seal behaviour. 36 harbour seals were caught between January 2002 and November 2005 at a nearby haul-out site. 13 seals were equipped with ARGOS satellite transmitters, 15 with sophisticated data loggers, and 8 with both. Data loggers recorded pressure (depth), speed, pitch, roll, 3D compass orientation, temperature and light level. From this information individual diving tracks were tried to be reconstructed. Only seven data loggers were retrieved.

7 animals were tagged before construction, 3 during construction and 11 during semi-operation or operation allowing for a separate treatment of data from baseline, construction and operation periods. Satellite data was filtered considering different precision categories given by ARGOS. Parameters examined were

- probability of being observed in a given 10x10 or 4x4 km rectangle
- likely number of positions in the rectangle
- mean time (min per day) spent in a 10x10 km grid cell as a measure of relative importance of that cell
- individual 3D diving tracks from the data loggers
- dive data from pressure recordings (e. g., u-shaped dives indicating foraging)
- visual observations during line-track surveys.

The central questions raised in the study were whether Horns Ref is an important foraging area, if seals enter the windfarm after construction and if their behaviour is affected inside the windfarm compared to outside. The results only partly provided the answers.

General results from satellite tracking data showed that during summer seals foraged outside the Wadden Sea up to 100 km from the coast. Individual seals seemed to have preferences for confined foraging areas. Presence data from satellite uplinks suggest a concentration in an area centred on Horns Reef and 50 to 100 km to the north, west and south. Horns Rev is part of this important foraging area which stretches basically from Holmsland Klit, which is about 50 km north to the German border, which is about 50 km south of the reef. The time of habitat use (derived from interpolated tracks between ARGOS positions in 10x10 km grid cells) averaged over all seals in baseline, construction and operation periods showed a high activity in the Horns Rev area indicating the importance of the reef.

During baseline and construction (2002 data) seals travelled longer distances offshore (3 out of 10 animals made foraging trips up to 300 km from the shore) compared to semi-operation and operation (2003 to 2005) when all animals remained within 100 km from shore. A similar picture is drawn by the presence in 4x4 km grid cells showing a scattered presence around the wind farm area during baseline and construction periods and a more consistent use of the windfarm area during semi-operation and operation. However, with respect to animal numbers, times and positions the data set was unbalanced (with 4 times as many days with data during the latter two phases). The analysis is vulnerable to individual bias: differences in areas used may reflect individual preferences rather than the influence of the windfarm.

The accuracy of the ARGOS positions was not sufficient for drawing firm specific conclusions on how construction activities affected the seals within the wind farm area. Data logger data was also insufficient for drawing conclusions on the scale of the windfarm. Out of 11 recorded foraging trips one animal seemed to have spent considerable time in the area around Horns Rev and have passed through the windfarm twice and nearby on 6 occasions performing many u-shaped dives indicating foraging activity. However, calculated position and recorded depth data did not correspond with the bathymetry of the windfarm area. A comparison of sightings data over the periods from baseline to operation were biased by a phocine distemper virus epidemic in autumn 2002 killing a large portion of the population.

During construction (spring and summer 2002) very few harbour seals were sighted during surveys compared to periods before and after.

During operation less seals were sighted in an area around the windfarm (approx. 8 km range) compared to baseline, semi-operation and operation periods. During operation sightings were more evenly distributed. Some sightings within the windfarm area were recorded during the construction phase, none during pile driving activity.

In conclusion, satellite track data does not indicate a deterring effect of construction activities on the scale of tens of kms (ARGOS accuracy) whereas sightings indicate an effect, especially during piling activities. A seal scarer used during the construction period may have had an effect on the distribution of seals. During operation no effects were observed. However, limitations of the methods used suggests that only large effects would have been detected.

3.2.3 Other studies

At North Hoyle offshore windfarm site in the Irish Sea monthly grey seal counts on a haul-out site on the West Hoyle Bank, approximately 10 km south east of the wind farm site, were interpreted as not being influenced by windfarm construction activities (NWP Offshore Ltd 2006). During pre-construction phase of the windfarm (before April 2003) and during construction (April-December 2003) similar numbers were recorded on the haul-out site. No details on the methodology or statistical methods used are given in the report. No post construction data was included in the study.

4 Scope of monitoring

In planning monitoring activities aiming to measure and assess possible changes in marine mammal behaviour, abundance or distribution arising from offshore windfarms, the following questions have to be answered:

1. Which area is affected by a certain activity?
2. How long does an effect last?
3. How many animals are to be expected in the impact area?
4. Which method is best suited to provide statistically robust data of key species in the relevant area and the relevant time period?

The information presented in the first chapters of this report can be used to answer the first two questions and define the spatial and temporal scope of monitoring activities.

The available data lead to contrasting requirements to monitor marine mammal responses during construction and operation of the windfarm. Impacts during construction are rather short but affect large areas. The duration of a single piling operation lasts a few hours and a first study in the Danish offshore windfarm Horns Rev indicates, that responses of harbour porpoises are in the same order of magnitude. However, as the construction of a whole windfarm leads to a frequent repetition of the pile driving processes, longer responses still have to be taken into account and should be considered for the monitoring.

As a first conclusion, data from offshore pile driving noise emissions and the responses of marine mammals can be used to estimate response radii which are needed to define the spatial extent of monitoring activities. Figure 4-1 presents calculated sound pressure levels of a large pile assuming a source level of 240 dB re 1 μ Pa. Two simple functions of transmission loss are applied which have been derived during measurements in German piling operations (ISD 2007). The data indicate an area where injury may occur of less than 100 m, however, if TTS would be defined as onset of physical injury, which would be more sensible, this area would be larger and might reach a few kilometres for pinnipeds (compare Table 2-2). The range where disturbance must be considered would reach at least 20 to 30 km and the range beyond which an effect would be excluded would be >30 km. It has to be stressed that these zones should be calculated for each specific project and more detailed models of transmission losses should be used taking the local conditions, especially water depth, into account.

Table 4-1 summarises expected impact ranges and duration. From these data, first recommendations for the scope of monitoring activities can be concluded:

1. Construction – injury.

Marine mammals may be injured in the vicinity of the piling operations and as a first proxy, a radius of 500 m is expected to define the area where such effects may occur. As marine mammals must not be injured by the construction work, pingers and seal scarers could be used to scare them out of this zone, pending further investigations (SMRU 2007). The aim of a monitoring would be, to assure that no marine mammal is present in a danger zone. The scope would thus be, to survey this area completely during the time of the operation (hours). Data should be obtained immediately and any method should allow a complete coverage of the area.

2. Construction – disturbance

Available data indicate that responses of harbour porpoises – and thus possibly also of other marine mammals – may range over 20 to 30 km. Behavioural responses at Horns Rev lasted over a few hours, but longer responses cannot be excluded yet. The aim of the monitoring would thus be twofold: First, to detect possibly short-termed behavioural responses over large distances in order to define the extent of the impact area, and second, to detect changes in abundance and distribution in the impact area over a longer period. Depending on the duration of the construction work in an area, which may last several months and extend to more than one year in large windfarms, monitoring activities may need to be

conducted over several years. It has to be noted, that reference areas must be situated in a large enough distance where no behavioural effects can be expected.

3. Operation – disturbance or attraction

The operation of offshore windfarms appears to have little effects on harbour porpoises and harbour seals, recent data even indicate an attraction of harbour porpoises which might reflect a response to locally enriched fish abundance (reef effect). As data are only available so far for one species and two windfarms, more studies in other areas and on other species are recommended. The aim should be, to describe the utilisation of the windfarm areas by marine mammals. The spatial scale should be the windfarms and the near surroundings and studies should cover at least two years during full operation.

4. Decommissioning – disturbance

The spatial scales of marine mammal studies in relation to decommissioning of windfarms should not be defined before decommissioning methods and their associated noise emissions are known.

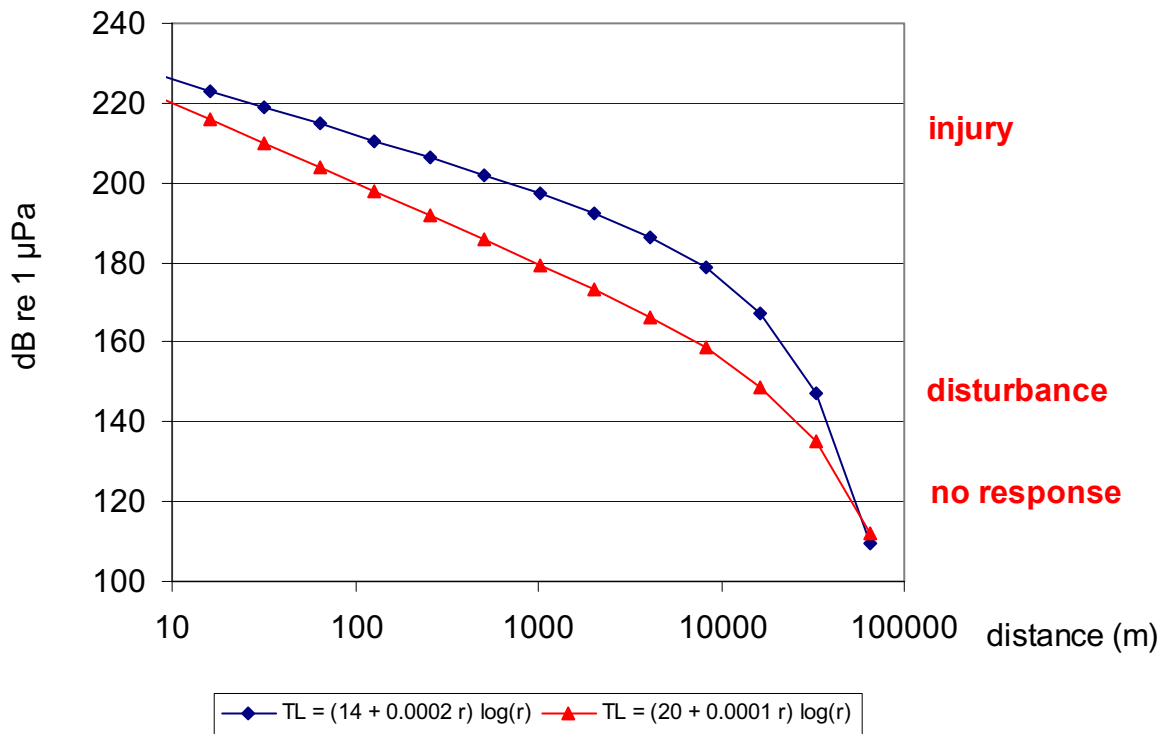


Figure 4-1: Calculated peak sound pressure levels of offshore pile driving assuming a source level of 240 dB re 1 µPa for two functions of transmission losses (TL). On the right side the corresponding impact areas are indicated (see text).

Table 4-1: Range and duration of effects caused by offshore windfarms in different periods.

	Range of effects	Duration of effect
Construction - injury	< 500 m	hours
Construction - disturbance	20 – 30 km	hours – years
Operation - disturbance	< 100 m	years
Decommissioning - disturbance	< 1 km	months

5 Basic statistical considerations

The purpose of impact assessments is to evaluate if a stress factor has a significant effect on the environment and to estimate the magnitude of this potential effect. The evaluation is often difficult, because several factors may influence the environment additionally to the impact. While the type of impact and the environments are very different, the design of analysing the effects of the impacts stays similar, involving comparison of impact areas with control areas. When data are available prior to impacting the environment this design is called Before-After Control-Impact (BACI).

5.1 Before-After Design

The simplest way to detect changes in environmental conditions is to compare data prior and after human activity (before-after design (BA)) (e.g., Green 1979). The typical approach to analyse these data is to treat them as independent samples using a two sample test (Smith 2002). Differences found are attributed to the impact. A problem with this design is that it is assumed, that the only effect on the environment is the impact and that all other environmental conditions are fixed. However, changes in environments are always present, e.g. following seasonal changes, weather conditions or natural movements of animals or nutrients. To avoid the influence of other environmental parameters than the impact, the BA design should cover only a short time period. Otherwise, with BA design, only a change in variance is possible to detect rather than in the mean (Figure 5-1 (a)).

A statistical model for the BA design can be formulated as follows (Smith 2002):

$$X_{ik} = \mu + \alpha_i + \tau_{k(i)} \quad (1)$$

with X_{ik} as response (e.g. abundance of a species), i = samples before ($i = 0$) or after ($i = 1$) the impact, k number of replications, μ is the overall mean, α_i the effect of the impact and $\tau_{k(i)}$ is the time within the periods ($k(\text{before})=1,2,\dots,t_B$, $k(\text{after})=1, 2, \dots, t_A$, t time measured before (B) the treatment, after (A) the treatment). This can be solved as a standard two sample t-test or as linear model, testing the significance of the single parameters. For detecting changes in abundance cause by a human impact the number of samples is important. If the population mean is small in comparison to the variance of the data, the sample size must be large to detect changes before and after the impact. Under such conditions small changes are hard to detect. A power analysis can be used here to assess the number of required controls after an impact, based on the mean and standard deviation of the data found before the impact (e.g. Crawley 2002). If only a few samples are available with a large standard deviation, bootstrap methods (Efron 1979) can give practical solutions. The number of samples needed is raised by generating many random samples based on the real sites.

A possibility to make this design more robust is to sample a number (M) of sites rather than a single site. For analysis here it is relevant whether these sites are independent replications or subsets of the same site. Ideally these sites are independent replications rather than subsets of the same site.

5.2 Before-After/Control-Impact Design

To assess the effect of an impact avoiding the disadvantages of the BA design, the BACI design is an appropriate solution (e.g. Smith 2002, Green 1979). Using this method distinguishing between the effect of the treatment and the effects of environmental parameters becomes possible. Usually impact and control areas are not randomly chosen and most times only one treatment area is available so that statistical analysis can not be done at the basis of a two-way

ANOVA (Hurlbert 1984). A paired analysis between control and impact area refers to this problem and was suggested by Eberhart (1976) as the control-treatment pairing (CTP) (see also Bernstein and Zalenski, 1983, (Figure 5-1(d)). Stewart-Oaten et al. (1986) described this paired design as BACI design, although it might be better to refer to it as a paired BACI design (Schmidt 2002). The sample layout is designed as followed:

The impact area is sampled t_B times before the treatment and t_A times after. As the variable of interest is X (e.g., abundance of a taxon) X_{ijk} represents the data of the locality (k), j number of samples and i the sampling period (before or after) (see Table 5-1). A single site is measured multiple times, thus each site pair-by-time combination is treated as a unit, where different sites are interpreted as independent, comparable with a repeated measured design.

The model formula is given by

$$X_{ijk} = \mu + \alpha_i + \tau_{k(i)} + \beta_j + (\alpha\beta)_{ij} + \varepsilon_{ijk} \quad (2)$$

where μ is the overall mean, α_i the effect of period (I before or after), $\tau_{k(i)}$ is the time within the periods ($k(\text{before})=1,2,\dots,t_B$, $k(\text{after})=1, 2, \dots, t_A$, β_j is the effect of location, $(\alpha\beta)_{ij}$ the interaction between period and location, and ε_{ijk} is the remaining error. This linear model can be solved as a type of ANOVA (e.g. Smith 2002) or in the interpretation of a Generalized Linear Model (McCullagh and Nelder 1989), where the distribution of the response is not fixed to the Gaussian distribution (e.g. Carstensen et al. 2006).

Table 5-1: Table from Smith (2002) Data for a paired BACI design

Period	Sampling occasion	Control	Impact
Before	1	X_{111}	X_{121}
	2	X_{121}	X_{122}
	.		
	.		
	t_A	X_{11t_A}	X_{12t_A}
After	$t_A + 1$	$X_{21(t_A+1)}$	$X_{22(t_A+1)}$
	$t_A + 2$		
	.		
	.		
	.		
	.		
	$N = t_B + t_A$	$X_{21(t_A+t_B)}$	$X_{22(t_A+t_B)}$

As far as the locations are selected at random they should be treated as a random effect within the model (the variance is not of interest). Underwood (1991) suggests to involve alternative tests based on contrasts. This leads to a Generalized Linear Mixed Effect Model (Faraway 2004, Pinheiro and Bates 2000). A good case study related to the influence of pile driving of offshore wind farms on harbour porpoises is given by Carstensen et al. 2006. Again, to assure that existing differences between before and after treatment can be detected, the sample size has to be assessed carefully. In this case a paired power test is recommended to calculate the number of required repeated measurements, based on a given mean and standard deviation.

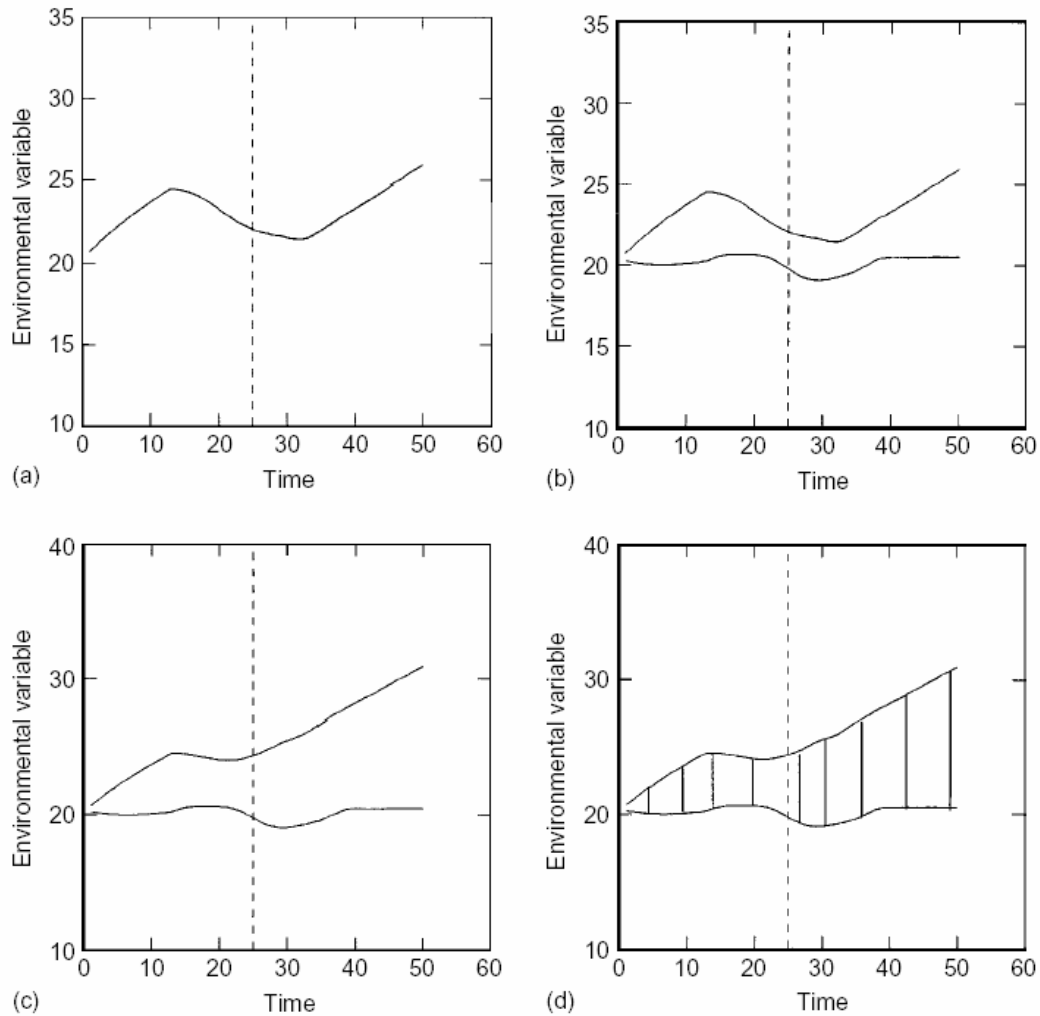


Figure 5-1: Figure from Smith (2002): Plots of patterns at site(s) describing different situations for impact assessment. (a) Example data profile for before–after analysis. Dashed line indicates time of activity. (b) Example data profile for BACI analysis, assuming no impact. (c) Example data profile for BACI analysis given an impact. (d) Example data profile for BACI assuming impact. Lines indicate pairs of samples.

5.3 Lack of baseline data

If there are no data available for the period before the impact, differences between control and impact area might overlay the effect of the impact itself: Regional effects can result in the same or even greater differences than the effect of the impact. Here the sample size must be higher (because variance is not known) and additionally more environmental parameters are required to be measured. Only using multivariate regression analysis it can be addressed whether the difference between areas is due to the impact or other factors.

Without baseline data, the power of any investigation may be considerably reduced and only be useful to detect strong effects. However, at present, there is still a fundamental lack of data for several species and any study may provide new and important insights. For example, any observation of dolphin species in relation to offshore windfarms would be of high interest. In addition, there are two good possibilities to overcome problems of lack of baseline data:

1. Effects on a small scale, as expected for the operation of offshore windfarms, can be investigated using reference sites in a rather close distance where equal conditions can

be assumed. BioConsult SH used such an approach in the studies in the Danish windfarms Horns Rev and Nysted (chapter 3.1.2).

2. Short-termed effects, as expected behavioural responses to pile driving, can be investigated by a continuous study using passive acoustic monitoring. Intervals between piling operations may thus serve as reference periods and although it will be difficult to assess moderate impacts on species abundance, magnitude and duration of behavioural responses may well be described.

5.4 Spatial modelling

An alternative way of comparing the influence of environmental factors to a biotic response is to compare the spatial distribution of the biotic response e.g., before an impact and after an impact. Here in general two methods are available, kriging (e.g., Diggle and Ribeiro 2007) and Generalised Additive Models (GAM) (Wood 2006). Both methods provide good results in modelling surface / spatial distribution, as kriging requires accurate point data whereas GAM's do not need this precision (e.g., Fahrmeir et al. 2007). However, when establishing such models, border effects, autocorrelation and covariance must be considered carefully. Short term effects can have large influence, when the data basis covers a longer research phase, tests whether data can be pooled are required. A good and practical solution is given with the library mgcv (Wood 2006) for the free software R 2.7.0 (R Development Core Team, 2008) using GAM and Generalized Additive Mixed Models (Wood 2006) in combination with Markov Chain Processes (Fahrmeir and Lang 2001). Two different surfaces (before after impact) can be compared and tested for significant differences (e.g. Wood 2006). As far as such models interpolate between two measurement points, the number of sampling units depends strongly on given regional conditions and cannot easily assessed using other data sets. Though spatial modelling is considered as a powerful tool in analysing marine mammal distribution, a detailed presentation of it is beyond the scope of this report.

6 Marine mammal survey methods (harbour porpoise, bottlenose dolphin, seals and other species)

6.1 Line transect distance sampling

Objectives:

In order to assess the potential impact of the construction and operation of an offshore wind farm the ideal objectives of line transect counts are to achieve the mapping of cetacean density distributions

1. at the highest possible spatial precision,
2. over the greatest possible area,
3. in the shortest possible time,
4. without causing disturbance to the underlying pattern of distribution,
5. using analytical techniques that derive workable precision estimates for density estimates.

The overall principle of line transect surveys is to count animals from a platform which is moving along predefined transect lines. For each animal the observers record the distance from the track line, hence "Distance Sampling" (Buckland et al. 2001). Provided that the observers detect all animals on or very near the track line, the method allows for a proportion of the animals present at the time the observer passes, to be missed within a given distance w . The encounter rate along the track line can be thought of in terms of an effective half-strip width of μ , which represents the distance from the track line at which as many animals are detected beyond μ as are missed within μ of the track line. Thus μ refers to the effective detection probability. In this case, density D is estimated by:

$$D = \frac{n}{2\mu \cdot L}$$

where n is the number of counted animals and L the total transect length.

To estimate the effective half-strip width μ , the determination of the detection function $g(y)$ is required, i.e. the probability of detecting an animal given its distance y from the track line (Buckland et al. 2001). This is derived from the population of animal observations generated from the survey results, which will show a reduction in detected animals with increasing distance from the track line. In order to generate robust estimates of overall densities, however, the method is based on the assumption that all animals on the track line are always detected (i.e. that $g(0) = 1$). In addition, the method also requires that (i) objects are detected at their initial position, prior to any movement in response to the observer and that (ii) distances to objects are accurately and consistently determined (Buckland et al 2001, Thomas et al. 2004). As long as the underlying assumptions are met such model-fitting provides the most statistically robust method of estimating cetacean densities over extensive bodies of open sea, even when very small percentages of the total number of animals are detected away from the track line.

However, for most cetacean surveys, the assumption that $g(0)=1$ is questionable or known to be false. In these surveys, $g(0)$ is usually <1 , for two reasons:

1. they may be unavailable for detection because they are underwater (bias from this source is called "availability bias"), or
2. observers may fail to detect them even though they are available (bias from this source is often called "perception bias").

It is difficult and challenging to estimate $g(0)$ precisely and the availability of animals depends not only on their behaviour, but also on the detection process, the weather and sea state conditions and the platform used. The aim of line transect distance sampling is to assess the most suitable method for deriving marine mammal density surfaces describing extensive areas of open sea. The following review assesses the suitability of the two different sampling platforms ship and aircraft, sampling protocols, sampling methods and data collection techniques by comparing the advantages, shortcomings and limitations. Some assessment of data handling issues will be touched upon in relation to providing a suitable basis for analysis. Finally, since the objective is to generate marine mammal data as a basis for assessment of the influence of offshore windfarming, the suitability of application of different statistical and modelling approaches will be reviewed by means of own datasets.

6.1.1 Aerial surveys

Methodology

As no standard protocol for aerial surveys on marine mammals exists, the following description is an introduction to the common surveying practice. In principal the methodology follows the standards for the counting of birds by aerial surveys described by several authors (Diederichs et al. 2002, COWRIE 2003). A combination of surveys for birds and marine mammals is possible, especially in areas where no high bird concentrations are expected (like at wintering grounds for sea ducks) and where the density of the target mammal species is fairly high so that enough sightings for the density calculation using distance sampling procedure can be obtained (> 60 sightings per survey).

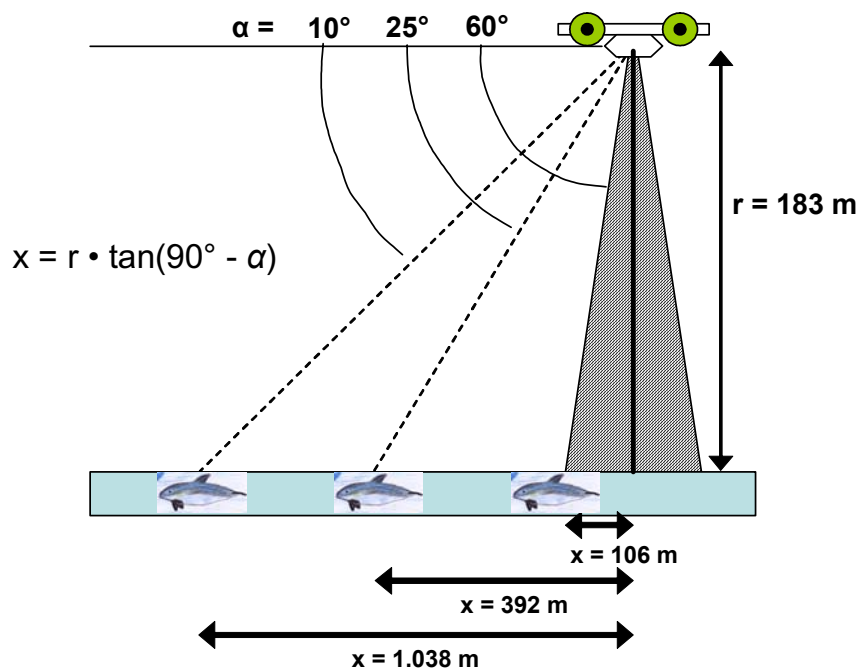


Figure 6-1: Standardised aerial survey method for counting marine mammals (after Diederichs et al. 2002).

For safety reasons it is recommended to use a twin-engine, high-winged aircraft (for example Partenavia 68) for aerial surveys. This should ideally be equipped with bubble windows on the rear seats. The aircraft surveys at a defined altitude. Dependent on size and density of the target species the recommended altitude ranges from 250 feet (76 m, often in combination with seabird counts) to 600 feet (183m). Recommended speed is around 100 knots (185km/h, Palka 2004, Scheidat et al. 2004, Grünkorn et al. 2005). Bubble windows allow two observers on the

aircraft to search the area from the abeam line to the track line. Whereas some groups use a third observer in the co-pilot seat recording all sightings as well as changes in sighting conditions during on-effort periods (Scheidat et al. 2004), we recommend that a third observer on the rear seats records sightings on the side with the best sighting conditions. This results in one double counted site on each transect (Grünkorn et al. 2005). For cetacean sightings, estimated pod size and also the declination angle to the pod as it came abeam is recorded using a hand-held inclinometer. From the declination angle and the aircraft altitude the perpendicular distance to the sighting can be calculated (Figure 6-1).

Additionally, behaviour, swimming direction and pod composition (presence of calves) is noted. Every observation is recorded to the second using a hand-held tape recorder. The environmental conditions are recorded at the beginning of each transect and whenever they change. These conditions include sea state, cloud cover, angle obscured by glare, severity of glare, turbidity and an assessment of overall sighting conditions as "good", "moderate" or "poor". A GPS-logger, which is synchronised with the observers' watches registers the position of the aircraft every two to five seconds.

Due to an assumed correlation of marine mammal occurrence with water depth it is recommended to choose transects, so that they cross the depth lines perpendicularly. Thus transects are usually also perpendicular to the coastline.

Species identification

Under good conditions the entire bodies of cetaceans are well observable, such that the aerial survey method will enable multi-species data to be collected. Due to the high speed of the observer platform and the relatively short time when the animals are detectable at the surface, only well experienced observers are required to be able to distinguish between similar species like white beaked and white sided dolphins.

Determination of seal species is often difficult at sea, and especially young grey seals can easily be mistaken for harbour seals. Furthermore seals often swim with their head out of the water and with a perpendicular bearing in offshore waters. This makes it difficult to determine seals on a species level especially during aerial surveys. Satellite tracking of harbour seals showed that these animals spend most of their time in offshore waters swimming along the seabed, only coming up to the surface for breathing every few minutes (Adelung et al. 2004).

Spatial scale

Due to the high speed of an aircraft it is possible to survey a distance of more than 800 km per day by aerial surveys. With a distance of 3 km between single transects, it is therefore possible to cover a sea area of approximately 2.400 km² per day. This means a very good coverage of large areas. The spatial resolution within the covered area is dependent on the distance between transects. A minimum distance of 2 km between transects is recommended in order to avoid double counting of animals. Under very good conditions a long effective strip width can be reached (Buckland et al. 2001), and thus a relative high spatial resolution can be achieved.

Temporal scale

One aerial survey provides a picture of the distribution of marine mammals at that specific day and is therefore comparable to a photograph. Depending on the size of the covered area it is often only possible to manage one survey per day. The temporal resolution can therefore be one day at minimum and requires suitable weather conditions during two following days. In summary, aerial surveys provide data on a high spatial scale but on a low temporal scale.

Methodological variability of the data

The probability to detect marine mammals during aerial surveys, and thus the data quality, strongly depends on a wide range of different methodological variables (e.g. type of platform, sighting conditions like glare or sea state, water turbidity, observation skills). By using distance sampling (Buckland et al. 2001) a flight specific detection function dependent on the perpendicular distance of the sightings can be calculated provided that a sufficient number of animals was counted (Figure 6-2). This detection function ideally includes any number of variables, which might affect detection probability in addition to distance. It is usually impossible to include all variables causing heterogeneity of the data. A dataset of BioConsult SH on harbour porpoises counted by aerial surveys in the area west of the island of Sylt shows that even in a high density area with more than 3 animals per square kilometre during the months with highest abundance only in 40 % of all flights a sufficient number of sightings was reached to calculate a flight specific detection function. Therefore the data of several flights have to be pooled in order to get the minimum number of 60 sightings (Buckland et al. 2001).

Another crucial assumption is that $g(0)$ is 1, which does not usually apply to cetaceans due to both, availability bias and perception bias. It is difficult and challenging to estimate $g(0)$ precisely. The two most promising approaches involve the analysis of data obtained by double platforms.

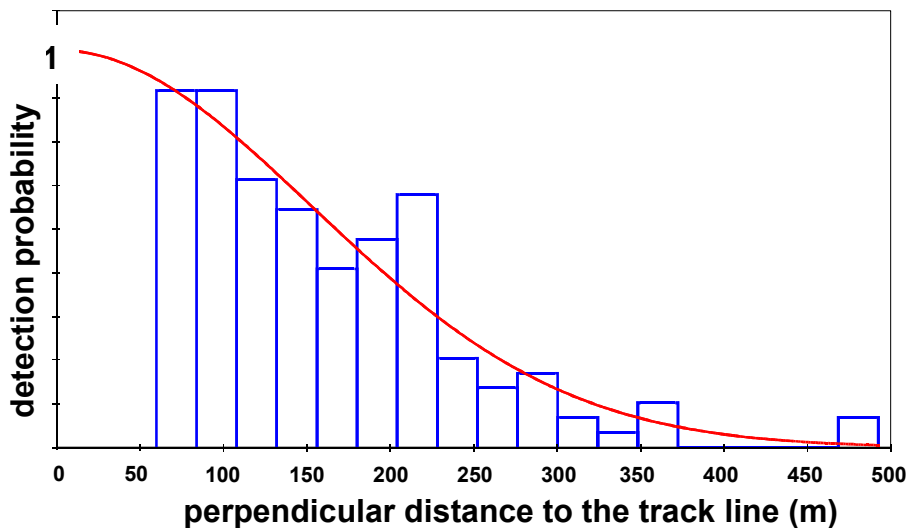


Figure 6-2: Example of a detection function of harbour porpoises from aerial surveys (half-normal key function with a cosine expansion; left truncation = 44 m (=60°)).

1. Circle back method (Hiby und Lovell 1998, Hiby 2004)

Scheidat et al. (2004) applied a method, that was developed by Hiby und Lovell (1998) and that was also used for the large scaled project "Small Cetaceans in the European Atlantic and North Sea" (SCANS II, Hammond 2007). This method integrates both sources of error (perception/availability) when calculating a $g(0)$ -correction term and does not distinguish between them. The method involves flying a circle 30 seconds after a harbour porpoise sighting under good or moderate conditions, and coming back to the track line 1500 metres before the sighting location. If the animal is not seen again, it either dived or is overseen. A correction term can then be calculated over several such circle flights. One potential problem with this method is, that in areas with high porpoise densities, it can not be excluded that the re-sighting was of a different animal. Another problem occurs in low density areas when not enough sightings can be recorded during one survey so data over several flights have to be pooled. Different conditions as well as different observers at different surveys can cause a bias in $g(0)$ calculations.

2. Independent double counts from one platform (Grünkorn et al. 2005, Thomsen et al. 2006, 2007)

Grünkorn et al. (2005) determined a $g(0)$ -correction term, using a combination of the double platform approach and available information on the average use of the upper two metres of the water column by harbour porpoises. With the pre-condition that the two observer sitting behind each other count completely independent from each other, the perception bias can be corrected by determining the re-sighting rate of harbour porpoises. Our data show that the re-sighting rate varies between single flights from 44 % to 91 % depending on weather conditions and observer skills (Table 6-1).

Table 6-1: Estimation of $g(0)$ for aerial surveys between January 2001 and November 2004. m = "marked" sightings of the control observer; perception bias: ratio of double sightings by the principal observer; surface time: after Teilmann 2000. **) Average after Teilmann 2000; ***) esw calculated from pooled data.

Date of survey	Altitude [m]	m	Perception bias	Surface time	$g(0)$	esw [m]
20.03.01	76	26	0.88	0.56**	0.49	92***
26.05.01	183	102	0.55	0.57	0.31	164
26.06.01	183	98	0.65	0.52	0.34	195
23.07.01	76	23	0.44	0.56	0.25	92***
21.08.01	76	36	0.58	0.51	0.30	136
24.11.01	76	18	0.50	0.56**	0.28	92***
12.03.02	76	18	0.47	0.56**	0.26	92***
24.03.02	76	23	0.80	0.56**	0.45	129
21.04.02	76	32	0.64	0.64	0.41	92***
02.06.02	76	73	0.68	0.52	0.35	113
05.07.02	76	12	0.82	0.56	0.46	92***
16.08.02	76	76	0.84	0.51	0.43	151
20.03.03	76	21	0.44	0.56**	0.25	92***
13.04.03	76	45	0.60	0.64	0.38	92***
17.06.03	76	75	0.79	0.52	0.41	78
14.07.03	76	31	0.52	0.56	0.29	92***
10.09.03	76	20	0.65	0.56**	0.36	92***
25.03.04	76	35	0.63	0.56**	0.35	92***
16.04.03	76	70	0.76	0.64	0.48	77
29.05.04	76	118	0.65	0.57	0.37	72
22.07.04	76	65	0.84	0.56	0.47	113
09.10.04	76	11	0.91	0.56**	0.51	92***

For correcting the availability bias, literature data on the dive behaviour of harbour porpoises were used. The ratio of animals being deep under water and thus undetectable, can be assessed by data collected with data loggers measuring diving depth and diving duration of individual animals (Barlow et al. 1988, Reed et al. 2000, Teilmann 2000). In terms of availability, Barlow et al. (1988) determined that harbour porpoise are at or near the surface only 23.9% of the time; Reed et al. (2000) calculated the percentage time spent submerged as 89%. In combination with satellite telemetry, Teilmann (2000) could show that the use of the upper two meters of the water column by harbour porpoises varied both with time of day and season. Animals, which were fitted with such data loggers in Danish waters, spent an average of 56 % of their time in the upper two metres of the water column (0.51 % in August and 0.64 % in April) and were thus detectable for observer in an aircraft.

An advantage of this approach is to get a proper flight specific $g(0)$ estimation. A disadvantage is given by the different information on the time the animals spend undetectable in deep waters. Up until now only a rough estimation exists based on data from tagged animals. As the

greatest data pool comes from Danish waters we recommend referring to this data set for analysis in UK waters also. The bias caused by differences in dive duration remains unknown.

Biological variability of the data

The distribution pattern of marine mammals in offshore waters is highly variable both in time and space. Seasonal changes as well as patchy distribution patterns caused by the patchy distribution of prey result in high standard deviations, which makes it difficult to receive statistically robust result for possible changes in abundance of these animals caused by the construction of offshore wind farms. A good example for natural data variability over time is given by the results from the two SCANS surveys in 1994 and 2005 (Figure 6-3). High density areas in the distribution of harbour porpoises along the British east coast completely changed within the 10 years from north (east of Scotland and North-England) to south (east of England).

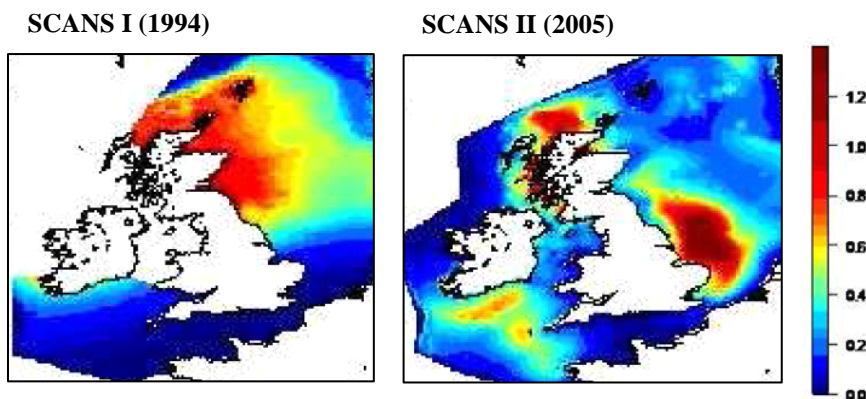


Figure 6-3: Harbour porpoise estimated density surface (animals per km²) in 1994 and 2005 (from Hammond 2007).

Data variability and thus standard deviation is a key factor to account for changes in abundance. When comparing two situations (like pre-construction period with post-construction period) using means with given standard deviations, the sample size has a huge influence on the results. If the sample size of data with high variance is too low, differences might not be calculated as significant due to a low statistical power, but not due to the lack of differences in the tested area or time. Another problem dealing with biological data is to distinguish between diverse effects of environmental factors. When comparing impact areas to control areas it is usually not possible to hold all the influencing factors constant. The method of Mixed Effect Models (Pinheiro and Bates 2002) or more general Generalized Additive Mixed Effect Models (Wood 2006, Zuur 2007) provide a good tool to distinguish between effects of interest and random factors.

Power test – estimating sample size

In order to get a rough estimation on the needed sample size using aerial surveys, we run power tests for different levels of divergence using aerial survey data sets from harbour porpoises counted between 2001 and 2004 in the German Bight west of the island of Sylt, Germany. The data were collected by BioConsult SH. The observed area was divided into six 270 km² equal sized subsets each crossed by five transect lines (Figure 6-4). The mean numbers of harbour porpoises per transect kilometres inclusive standard deviation were calculated for different months in different years. In order to make these means more general a bootstrap algorithm (e.g. Efron 1979, Veables and Ripley 2002, Rizzo 2008) was assessed. This procedure minimised the standard deviation given by the transect samples. The size of most wind farm areas is much smaller than 270 km², however, reducing the size of the area

covered would lead even to higher variability of the data and thus diminish the explanatory power of the power analysis.

Assuming an independent random distribution of harbour porpoises, the estimated meanboot (xboot) and the resulting standard deviation (sdboot) were used within a power analysis. A power analysis calculates the sample size (n) of a test set, which is required to test for significant differences to a given (known) mean. Using the open software R 2.6.1 (R Development Core Team 2007), the function power.t.test was assessed to the meanboot and sdboot of each single subset using a power of 0.8. The required number of samples for the test set (here: number of transects) was calculated for different deviations from the given meanboot (deviation of 10 %, 20 %, 30 %, 40 %, 50 %). High variation and small sample sizes as it is often found for the distribution of marine mammals, the standard deviation is almost very high and close to the mean.

It should be kept in mind, that the number of surveys needed to detect an assumed percentage of change is only relevant to the site where the data were obtained. Therefore, the results presented here provide a rough estimation and may also be applicable to a similar sized site with a similar density of animals that fluctuate in a similar manner. However, you would have to know that information before hand. For any site only apriori studies would give you some idea of the natural variability and then the power analysis tells you the size and frequency of the samples you would need to obtain to be able to detect a particular level of change.

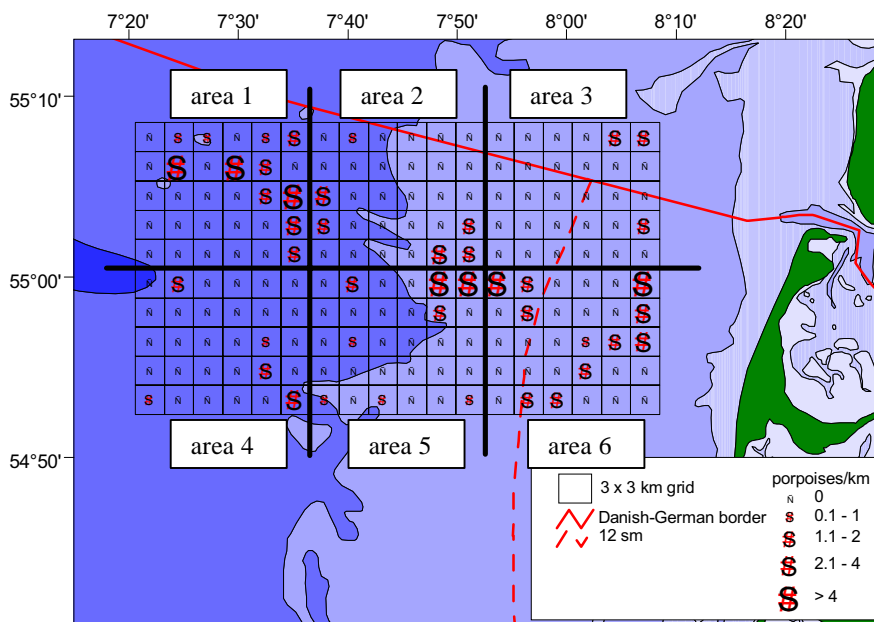


Figure 6-4: Subsets of 6 equal sized areas and the distribution of harbour porpoises at June, 17th 2003 as an example.

GAMM – distinguish between impact and seasonal effects.

Generalized Additive Models (GAM) are suitable models when dealing with non linear relationships (Hastie and Tibshirani 1990, Zuur 2007). In a GAM individual smoothing functions are used for single sections and result in a fitted curve with a high variance explained. Single environmental factors are treated additive, that means smoothing functions are calculated independently. Within a Mixed Effect Model it is possible to partition the variance into a random part and into a fixed part. When comparing the density of a marine mammal species between an impact area and a control area seasonal changes in density can take place through out the year (Verfuß et al. 2007). The variance within the data caused by this seasonal effect is not of interest and should be masked out. The use of a GAMM (e.g. using the R software package

mgcv (Wood 2004)) is strongly recommended. While the impact is entered as a fixed factor and the density of the species as response, season should be entered as a random factor.

Short term effects:

In order to test for short term effects like pile driving aerial surveys can be used by counting animals just before the impact takes place and comparing this distribution to the time when pile driving activities are running. Additional information can be obtained by analysing the swimming direction of the animals. During the base line period swimming direction should be of no special direction whereas the swimming direction during pile driving might show a direction away from the piling site. An example for the results of the power analysis on the basis of the data set, derived from the area west of Sylt, is shown in Figure 6-5.

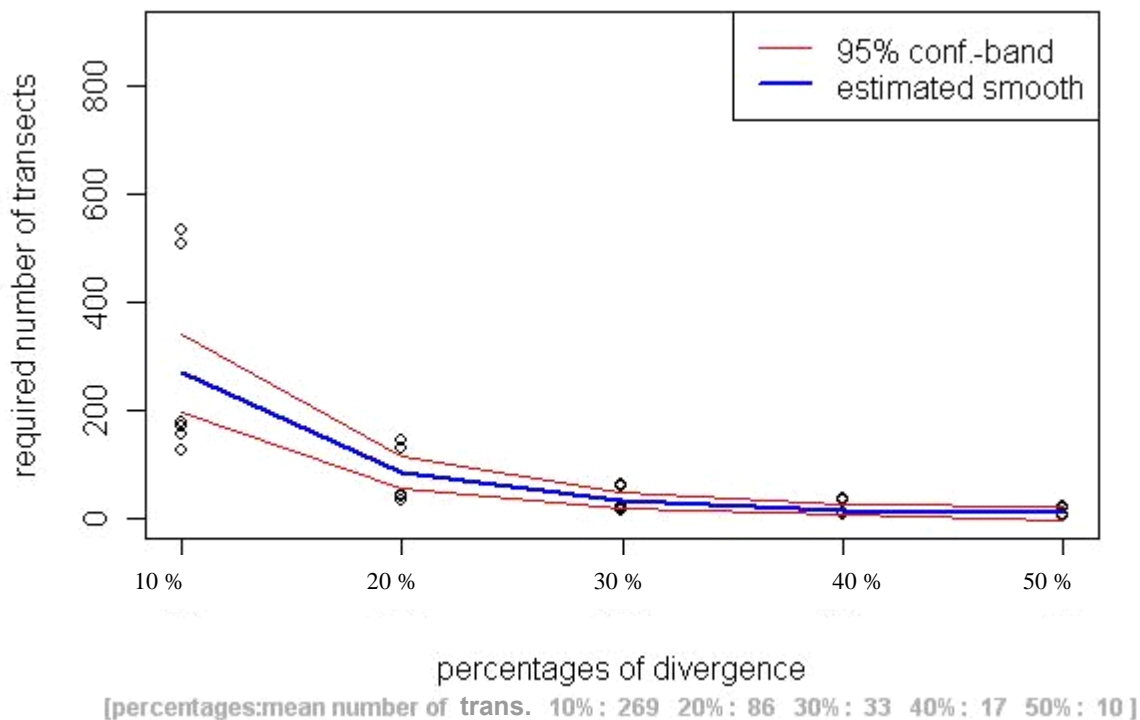


Figure 6-5: Required number of transects for aerial surveys to prove a difference in number of harbour porpoises calculated by bootstrapping the original data set from data of one flight in June 2003 in the North Sea, west of Sylt. Data source: BioConsult SH.

Here, the baseline period consisted only of one survey (June 2003), which was chosen to simulate the situation just before ramming takes place. Percentages of divergence means the change in the relative abundance of harbour porpoises compared to the baseline period. For example, the required number of transects, which have to be flown to prove a statistical significant difference of 20 % to the baseline, is 86 transects. Because one survey consist of 5 transects, the number of 18 surveys has to be conducted. The larger the difference in abundance between baseline and impact period, the fewer transects (surveys) have to be surveyed. Even though the data derive from a high density area, it is shown that any calculation, aiming to detect a significant difference is highly dependent on the database. Table 6-2 summarises the results when the data is given only by one survey. All data used for calculation were conducted during summer when the highest densities of harbour porpoises occurred in that area.

Table 6-2: Required number of aerial surveys (with 5 transects each) to prove a difference in the number of recorded harbour porpoises at different levels of changes (percentages of divergence) when the database is n = 1 survey (original data are bootstrapped). Area size is 270 km²; Data from 2001 – 2004.

percentages of divergence	required number of surveys (5 transects)				
	April 2003 n = 1	April 2004, n = 1	May 2003, n = 1	Jun 2003, n = 1	Jul 2004, n = 1
10%	183	40	54	54	39
20%	53	12	21	18	12
30%	20	5	9	7	5
40%	11	3	4	4	3
50%	7	2	2	2	2

The high variability in the distribution and in the low number of animals present in the study area requires a solid database in order to detect changes which are clearly related to the investigated impact. The lower the local density of the particular species the higher the effort needed to prove possible impacts of the investigated interference.

Long term effects

The impact of offshore wind farms on the presence of marine mammals during the operational phase is expected to be much weaker both, in time and space, than the influence of pile driving activities. To statistically prove a change in the number of animals using the impact area with the results of aerial surveys is therefore only possible when data from a baseline period are available. Figure 6-6 gives an example of the required number of transects, which have to be flown when the baseline dataset consists of 11 surveys conducted between April and June of 4 consecutive years (in an area with peak densities during May/June of 2-4 animals/km²).

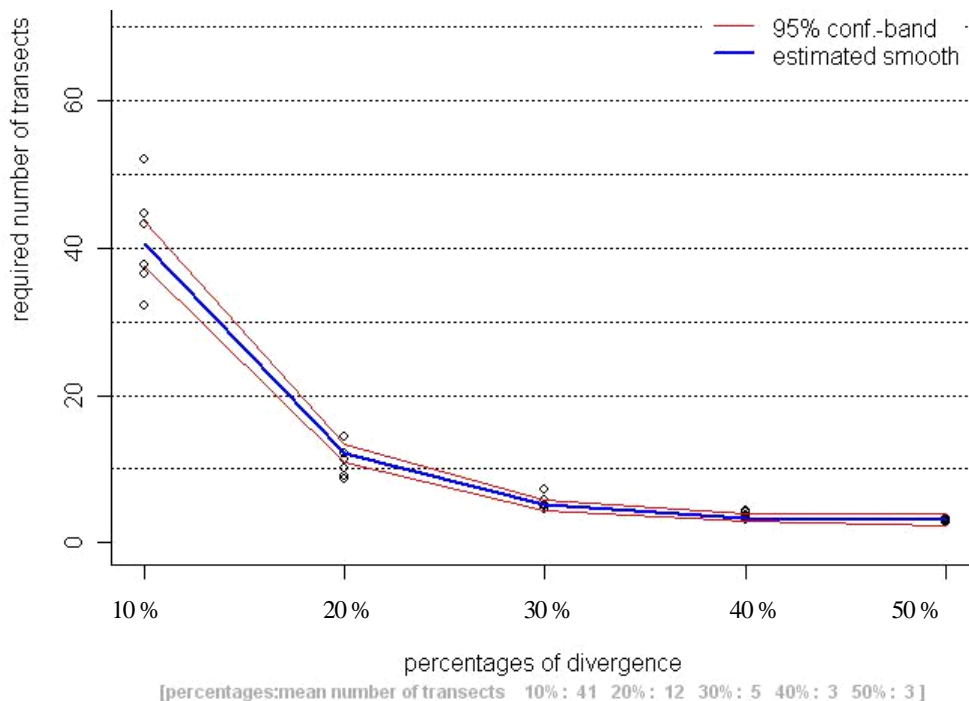


Figure 6-6: Required number of transects for aerial surveys to prove a difference in number of harbour porpoises calculated by bootstrapping the original data set from data of four consecutive years from April to June in the North Sea, west of Sylt. Data source: BioConsult SH.

In this case 8 surveys with 5 transects each have to be flown within the time span April to June in order to prove a 10 % change in abundance compared to the baseline. This results in 2-3 surveys per month, a requirement that can only be achieved during the summer months when the weather conditions are more suitable.

Table 6-3 shows that for other time spans with a smaller data base the required number of surveys for a 10 % change in abundance expands to even 23 surveys, a value which cannot be achieved within three months. Provided that the density is sufficiently high and the data variability relatively low, the results of the power analysis show that aerial surveys are a powerful tool in order to prove statistical significant differences between a given impact area and one or more different reference areas. The dataset was collected in a high density area with more than 1 animal per km². Similar densities in UK waters for harbour porpoises were achieved during the SCANS II survey in 2005 in two separated areas: North of Scotland and east of England (Figure 6-3). Because data variability is the key issue from a statistical point of view, a baseline period is urgently recommended to run power analyses in order to decide if line transect methods should be chosen to prove any changes in marine mammal abundance.

Table 6-3: Required number of aerial surveys (with 5 transects each) to prove a difference in the number of recorded harbour porpoises at different levels of changes (percentages of divergence) when the database emanate from 4 consecutive years (original data are bootstrapped). Area size is 270 km²; Data from 2001 – 2004.

percentages of divergence	required number of surveys (5 transects)			
	Apr,May,Jun n = 11	May,Jun,Jul, n = 10	Jun,Jul,Aug n = 9	Jul,Aug,Sep, n = 6
10%	9	9	13	23
20%	3	3	4	7
30%	1	2	2	3
40%	1	1	1	2
50%	1	1	1	1

Requirements for aerial surveys:

Aircraft: twin engine high winged

Speed: 90-100 knots

Altitude: 600 feet or 250 feet

Number of observer: 3 experienced, ideally a constant team for the time of monitoring

Conditions: Sea state <= 2, sight > 5 km

G(0) correction: circle back method (Hiby 2004) or double platform in combination with mean dive duration derived from data logger (Grünkorn et al. 2005).

Co-variables: Ship traffic with exact positions, gill nets,

Classifiable behaviour: swimming (without and with direction), diving, resting at the surface, fast swimming at the surface (flying)

6.1.2 Ship surveys

In principal ship surveys follow the same theory as aerial surveys. Also line transect distance sampling is applied on the data gathered by ship surveys. An ideal survey vessel would provide a stable viewing platform at a sufficient height above the surface (Figure 6-7). Stability will improve sighting conditions and precision of distance and angle measurements. A team of three experienced observers operate from the flying bridge at a minimum height of 5 m above the sea surface. They continuously scan the sea surface on both sides and ahead of the ship for marine mammals. Two observer search in a narrow strip of 800 m width each in a 90 degree sector relative to the ships midline. The third observer looks forward along the centreline in order to reduce the likelihood that animals are missed on the track line. All observer use binoculars but also, at least once a minute, observe the surface ahead of the ship by naked eyes for animals in the vicinity of the ship. Whenever animals are observed, the bearing angle and the direct distance are used in order to calculate the perpendicular distance from the ship to the animals.

Direct distance can be measured by a vertical angle measuring device, specially constructed for this purpose (Pihl and Frikke 1992). Each observation is recorded with time, distance and bearing, number of animals observed and behaviour classified into directional swimming (with direction), non-directional swimming, rapid swimming near the surface and resting at the surface. The observer who first sees the animals is also recorded, as well as whether they are first seen with or without binoculars. This information allows for subsequent analysis of differences between observers and evaluation of search strategies. The position of the ship is recorded by GPS every 10 seconds. The GPS clock is synchronised with the clocks of the observer. Ideal speed of the vessel is about 10 knots.

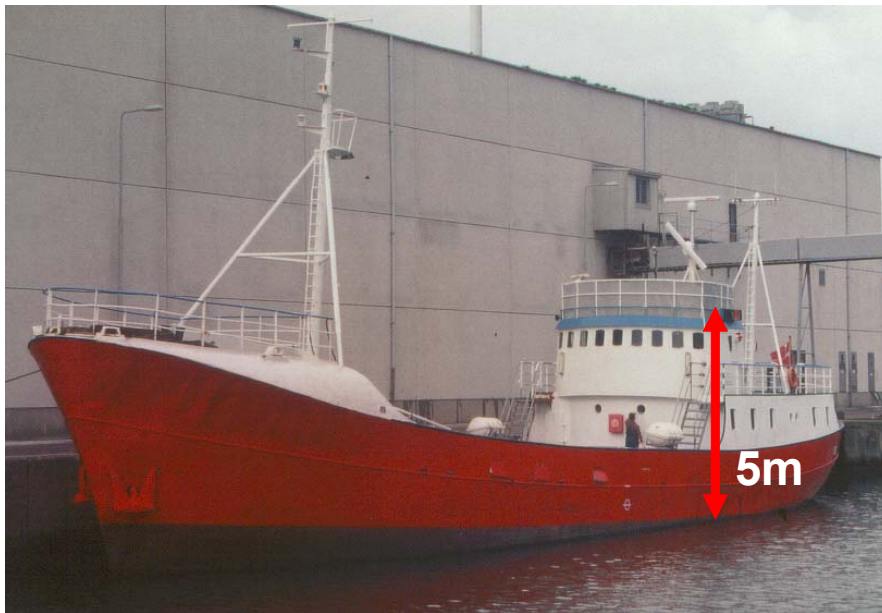


Figure 6-7: Typical vessel for offshore surveys of marine mammals.

Using double platforms on ships can be useful to increase the preciseness of the calculated density values. However, a double platform requires more observers and equipment, and it is not possible to install two observation platforms, which are completely separated from each other, on every ship. In case of monitoring studies in relative small areas (compared to the area utilised by the animals) like windfarm related study areas, this approach still suffers from too few sightings. This results in still using relative abundance data or in pooling data over several surveys to get estimations on absolute abundance. Due to unrelated high costs in comparison with the expected advantage, we recommend double platform approaches only in areas where more than 60 sightings per survey are expected in order to calculate survey specific detection functions with a precise $g(0)$ value. The effectiveness of marine ship surveys depends on the possibility to record a lot of environmental variables during the survey, which could play a major role in the explanation of the distribution pattern of marine mammals. Salinity, sea surface temperature, mixed flocks with sea birds are some of these variables.

Species identification

An advantage of ship surveys is that multi-species data can be collected. Due to the slower speed of the observation platform, animals, which stay longer at the sea surface or come more to the surface can be seen for a longer time, and therefore species identification can be easier than from an aircraft. Furthermore marine mammals are often associated with sea birds, which point the observer to the animals.

Spatial scale

Due to the slow speed of a vessel of 10 knots it is not possible to survey a distance of more than approximately 340 km per day. Thus, with a distance of three kilometres between single transects, it is possible to cover a sea area of approximately 340 km² per day. Taking into account that the mean 50 % home range of 50 satellite tagged animals in Danish waters was 5000 km² (Teilmann et al. 2004) this means a rather poor coverage. Regarding the area covered by a windfarm (e. g. with 80 turbines) the area is sufficiently large to compare the impact area with a reference area. The spatial resolution within the covered area is dependent on the distance between the transects. A minimum distance of 2 km between the transects is recommended in order to avoid double counting of animals. Under very good conditions a long effective strip width can be reached (Buckland et al. 2001), and thus a relative high resolution within the covered area can be achieved.

Temporal scale

One ship survey provides a picture of the distribution of marine mammals at that specific day and is therefore comparable to a photo. Only one survey per day is possible to manage. Thus, the temporal resolution can be one day at minimum and requires suitable weather conditions during two or more following days. Due to the relative slow speed of a vessel a considerable time is often required to reach the study area. This means that a sudden reaction to short time windows, where the weather is suitable for a survey of seals and small cetaceans, is often not possible. In summary, ship surveys provide data on a relative small temporal scale.

Methodical variability of the data

The same principles that apply to aerial surveys also apply to ship surveys: The probability to detect marine mammals and thus the data quality, strongly depends on a wide range of different methodological variables. Platforms, sighting conditions like glare or sea state, water turbidity and observation skills are some of these variables. Especially for small cetaceans like harbour porpoises, sea state has a strong effect on the sightability because the only visible part of the animals is the dorsal fin during breathing at the sea surface. Teilmann et al. (2003) stated that every ship based abundance estimation will be biased downwards if any effort in sea state greater than 0 (on the Beaufort scale, Clark 1982) is included. Especially for harbour porpoises the detection of the dorsal fin is much reduced as soon as small white-capped waves occur. Hammond (2007) concludes that only data of sea state less than 2 are analysable. On the other hand, marine mammals travel slowly hence they are more likely to be seen at the surface when the observer platform moves slowly. Therefore, under very good conditions (sea state 0), the chance to see a marine mammal is much higher compared to the aircraft.

Due to the normally low abundance of cetaceans and seals in offshore waters a sufficient number of sightings cannot normally be counted during single survey days to calculate survey specific detection functions by the means of line transect distance sampling (Buckland et al. 2001). For this purpose data of several surveys have to be pooled. Ship survey data from BioConsult SH from the same area, where also aerial surveys were conducted, show that only two out of 18 surveys during the summer with highest densities yielded sufficient sighting rates for calculating a detection function. One important concern for ship surveys is the assumption that the animals are not influenced by the observer platform. Both, dolphins and porpoises move in response to an approaching ship. Whereas porpoises are known to avoid the approaching ship (Teilmann et al. 2003) it is known from dolphins that they are being attracted by boats and move towards them (Buckland et al. 2001, Dawson et al. 2004). This makes it difficult to calculate densities of marine mammals even when enough sightings were achieved. A g(0)-correction can be calculated by using double platforms on the same ship. However, additional costs and more observers have to be scheduled.

Ship surveys provide more valuable data in another context than the assessments of offshore windfarms. Projects with a focus on a larger scale like SCANS or other marine mammal distribution atlases are built on marine mammal data collected by ship surveys. From an ecological perspective density estimates in very small areas like offshore windfarm planning

sites, are not very informative when data are collected by a method which is strongly limited on the temporal and spatial scale like ship surveys.

Biological variability of the data

As the marine environment is an open space without clear boundaries separating different habitats, the natural ranges of marine mammal species are much larger than the area that can be surveyed by one ship day. Most cetacean species can move rapidly over large distances, and density estimated in a small area is likely to change over short periods of time. Following prey, the distribution pattern of marine mammals is often very patchy, and concentrations of animals are changing from one day to the other (Gomez de Segura et al. 2007).

Short term effects:

Short term effects which are in focus of investigations on pile driving activities and decommissioning of windfarms are aiming at a larger spatial scale of more than 20 km away from the ramming site and a smaller temporal scale of only a few hours around pile driving activities (Tougaard et al. 2006b). The low speed of the observer platform makes it nearly impossible to get a sufficient number of observations in different distances to the ramming site. In this case more than one ship has to be used in the area and weather conditions have to be very calm in order to collect sufficient data.

Long term effects

Due to the slow speed of the ship and the high sensitivity to sea state in combination with a patchy distribution pattern of marine mammals in offshore waters, a high effort for ship based surveys is required.

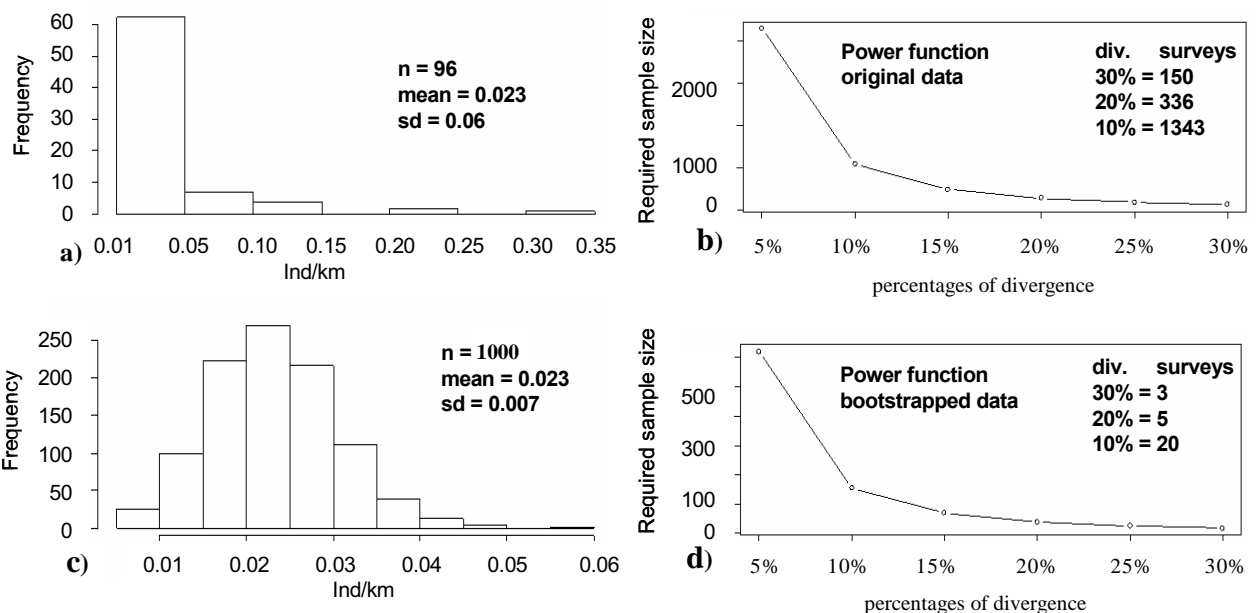


Figure 6-8: Example dataset from 12 ship surveys (with 8 transects each) in the fourth quarter of the years 2001 and 2002, a) frequency distribution of the original data including number of transects (n), mean and standard deviation (sd), b) power analysis: required sample size (transects) for a given mean and standard deviation (original data) c) frequency distribution of the bootstrapped data including number of random samples (n), mean and standard deviation, d) power analysis: required sample size (transects) for a given mean and standard deviation (bootstrapped data). One survey consists of 8 transects.

Counting data of harbour porpoises from 42 ship surveys conducted in the North Sea in the area 2 in Figure 6-4 were used in order to get an estimation of the required number of surveys to prove a significant difference to a baseline period (Data source: BioConsult SH). Due to the high variability of the data set the area of approximately 340 km² was not divided into smaller sub samples. The data were obtained in two consecutive years and because of a distinct seasonally occurring pattern the dataset was divided into 4 quarters of the year. Each survey consisted of 8 transects.

Table 6-4: Required number of ship surveys for different percentages of divergence when the database emanate from 2 consecutive years (data are bootstrapped). Area size is 270 km²; Data from 2001 – 2004.

percentages of divergence	required number of ship surveys (8 transects)			
	1. Quarter n = 8	2. Quarter, n = 10	3. Quarter n = 12	4. Quarter n = 12
10%	12	24	8	20
20%	3	6	3	5
30%	2	3	1	3

Figure 6-8 shows the required number of surveys for the fourth quarter of the year. Similar to aerial surveys in a second step a bootstrap method was applied to the original data in order to minimise the required sample size. Thus, a similar result to the aerial surveys could be achieved with 3 surveys in order to prove a difference of more than 30 % to the baseline. Table 6-4 shows the results for the bootstrapped data of all quarters of a year. To prove a divergence of less than 10 % to the baseline, up to 24 surveys are needed. This effort cannot be reached by ship surveys due to costs and weather conditions.

Requirements for ship surveys:

Ship: Observer height 5 m above sea surface at minimum, stability is an advantage in offshore waters.

Speed: best is 10 knots, but variations are possible.

Number of observer: 3-4 experienced, ideally a constant team for the time of monitoring.

Conditions: Sea state <= 2, better <= 1, sight > 5 km.

g(0) correction: double platform on the same ship possible with two observer teams at separated platforms, but low sighting rates require a high survey effort.

Co-variables: Ship traffic with exact positions, gill nets, salinity and sea surface temperature measurements are possible to record.

Classifiable behaviour: swimming (without and with direction), resting at the surface, fast swimming at the surface (flying).

6.1.3 Towed hydrophones

Since harbour porpoises are relying on their echolocation system for foraging (Beedholm and Miller, 2007) and navigation (Verfuß et al. 2005) it is assumed, that they frequently use their echolocation abilities. This has partly been shown by Akamatsu et al. (2007) by attaching an acoustic tag to harbour porpoises in Danish waters. The tagged individual used intense sonar clicks on average every ten meters, but in 1 % of the recorded time remained silent for approximately 100 meters. This indicates that passive acoustic devices are feasible to assess the occurrence of harbour porpoises. An automated system for the detection of harbour porpoise echolocation clicks has been proposed by Gillespie and Chappell (2002) and Chappell et al. (1996).

There are also indications, that towed hydrophone arrays can be used for other species as well. It has been shown for instance for right whales (Gillespie et al. 2004, Moscrop et al. 2004), sperm whales (Gillespie 1997) and beaked whales (Gillespie et al. 2007). There are also some

indications, that towed hydrophones can be used to assess minke whales through their unique 'boing'-vocalisation (Rankin and Barlow, 2005).

Major work for the development of towed hydrophone systems has been done by the International Fund for Animal Welfare (IFAW) who supplied the first SCANS Survey (Hammond et al. 1995) with their technology. Recent work of Akamatsu et al. (2007b) shows that A-Tags, originally developed to gain knowledge about the acoustic behaviour of finless porpoises (Akamatsu et al. 2007a), can be used as a towed hydrophone system to detect finless porpoises. As the A-Tag has been used with success on harbour porpoises (Akamatsu et al. 2007a) it would be feasible to use A-Tags as towed hydrophone systems as well.

Methodology

Towed hydrophone surveys work along standard line transect distance sampling survey guidelines (Buckland et al. 2001, Gillespie et al. 2005). The methodology can thus be read in chapter 6.1.

It is possible to estimate group sizes of passing animals comparing the bearings of their vocalisations, when they pass from ahead to astern of the survey vessel (Gillespie et al. 2005, Akamatsu et al. 2007b).

Species identification is based on the knowledge about differing vocalisations concerning communication and echolocation sounds. Thus different frequency bands have to be observed for different species. Oswald et al. 2007 report about a real-time acoustic identification system for delphinid whistles, which can be used in areas where a number of delphinids occur.

For harbour porpoises Gillespie et al. (2005) used a semi automated click classification algorithm using a minimum amplitude of 105 dB re 1 μ Pa and a ratio of plus 25 dB in the band between 115 and 145 kHz compared to two lower control frequencies. The bearing to the vessel was observed and used as a criterion in a later subjective judgement process to eliminate non-porpoise clicks. The possible detections can then be used to calculate relative densities.

Spatial Scale

The same limitation to the spatial coverage and resolution as mentioned in chapter 6.1.2 apply to towed hydrophone surveys. The system is less affected by the sea state and can be used during night time. Thus the cost efficiency of towed hydrophone surveys can be much better than for observer surveys. In addition the ship size can be much smaller, reducing the rental price. Synergetic effects might be usable during the construction of a wind farm, when ships are frequently cruising the area of concern.

In comparison with visual ship surveys the time period that can be covered is longer due to the ability to work at night, however, limitations to the spatial resolution remain unchanged.

Temporal Scale

Chapter 6.1.2 describes the limited temporal scale of ship surveys.

Methodological variation

The bias of weather conditions is largely reduced when using towed hydrophone arrays compared to visual ship board methods, as a serious interference is reached at much higher wind speeds and sea states. It is still possible to work with towed systems up to sea state five or even higher.

The assumption, that animals are not influenced by the vessel will always be violated except for very large cetaceans which might not show a behavioural reaction to a smaller vessel. For small odontocetes like harbour porpoise avoidance reactions have been documented (Teilmann et al. 2003) while for delphinid species approaching reaction could be shown (Buckland et al. 2001,

Dawson et al. 2004). This is even more difficult to handle for towed hydrophone systems compared to visual surveys, as a far ranged aversive reaction will reduce the possibility to detect animals on the track line greatly. It is even more puzzling, that towed hydrophone systems still increase the possibility to detect animals on the track line, which could be a hint, that the actual $g(0)$ for ship surveys is much lower than previously assumed. Akamatsu et al. (2007b) stated, that acoustic detections were highly increased compared to visual observations of single finless porpoises in the Yangtze River on the same survey. The detection functions for higher group sizes were then comparable. This indicates, that a combined passive acoustic and visual approach can be used to inter calibrate the methods. It also implies, that passive acoustics methods might be more feasible for species occurring in small group sizes.

Acoustic detections are greatly influenced by the platform used. This makes it necessary to calculate detection functions for different ships and inter calibrate the acoustic detections with visual surveys.

Biological variation, detection of short and long-term effects on cetaceans

There are not enough data available yet to perform power analysis calculations. The soon expected SCANS II survey report might give some insight into this field, as there were several surveys conducted using both visual and passive acoustic methods.

There are some relevant theoretical conclusions about the use of passive acoustic systems and their statistical relevance:

1. If passive acoustic methods increase the number of detections and have similar detection functions and distances, then their statistical power will be higher compared to visual ship surveys.
2. It might be very difficult to estimate a sensible $g(0)$ for acoustic detections, as the (post-)processing will be important in the estimation too. If an automated algorithm has a high detection rate, then the false detection rate might be increased as well. This is very important in low density areas, as false detections will bias the findings.
3. Ambient noise and anthropogenic ship noise will also have an effect on $g(0)$. Therefore Gillespie et al. (2005) suggest performing measurements of background noise in areas where no porpoises occur and thus find the false detection rate at different ambient noise levels.

For other odontocete species it seems to be sensible to record not only their echolocation sounds, but also their communication signals to substantiate the findings.

For both short- as well as long-term effects chapter 5.1.1 and 5.1.2 give information about the survey design to assess different criteria.

Requirements for towed hydrophones:

Actually different types of hydrophones are used:

For example the IFAW-system, produced by Seiche Measurements Ltd. (UK), the A-Tags, used by Akamatsu et al. (2007) or towed hydrophones used by Gardline Environmental Ltd. (UK).

A cable of minimum 200 m length is required to be sure that the hydrophone is as far away as possible from the noise produced by the ship.

A quiet ship is urgently recommended in order to avoid noise masking echolocation clicks.

6.2 Point transect distance sampling

Point Transect Sampling can be divided into visual observations and stationary acoustical surveys. Both have in common, that a stationary observer or hydrophone is monitoring the area of interest for a period of time or regular time intervals. As there is no platform for observers in the circumference of a planned wind farm site it is not feasible to use an observer based visual observation.

Acoustical methods like T-PODs, click detectors to register the echolocation sound of odontocetes, have proven to be useful to detect harbour porpoises (Thomsen et al. 2005, Verfuß et al. 2004). For static acoustic monitoring Tougaard et al. (2006c) showed, that, assuming a link between a change in the amount of registrations and porpoise abundance, as well as click train parameters and behaviour, a valid detection function similar to line transect surveys can be found and employed to calculate absolute densities of harbour porpoises. While this study showed a high potential for static acoustic monitoring (SAM), it still had some uncertainties regarding the generality of the found detection functions, comparability of data gained by different or differently set measuring equipment and the role of group size estimates. The above-mentioned studies show the high usability of static acoustic monitoring.

6.2.1 Static acoustic monitoring (SAM)

Methodology

At the time being there are only two types of measuring devices used for static acoustic monitoring of small odontocete species.

The commonly used instrument is the T-POD (Figure 6-9a). With the T-POD, originally developed to gain knowledge about the echolocation behaviour of harbour porpoises and other delphinid species close to fishery nets (Tregenza et al. 1998), a wide variety of studies have been conducted (e.g. Blew et al. 2006, Verfuß et al. 2007, Carstensen et al. 2006).

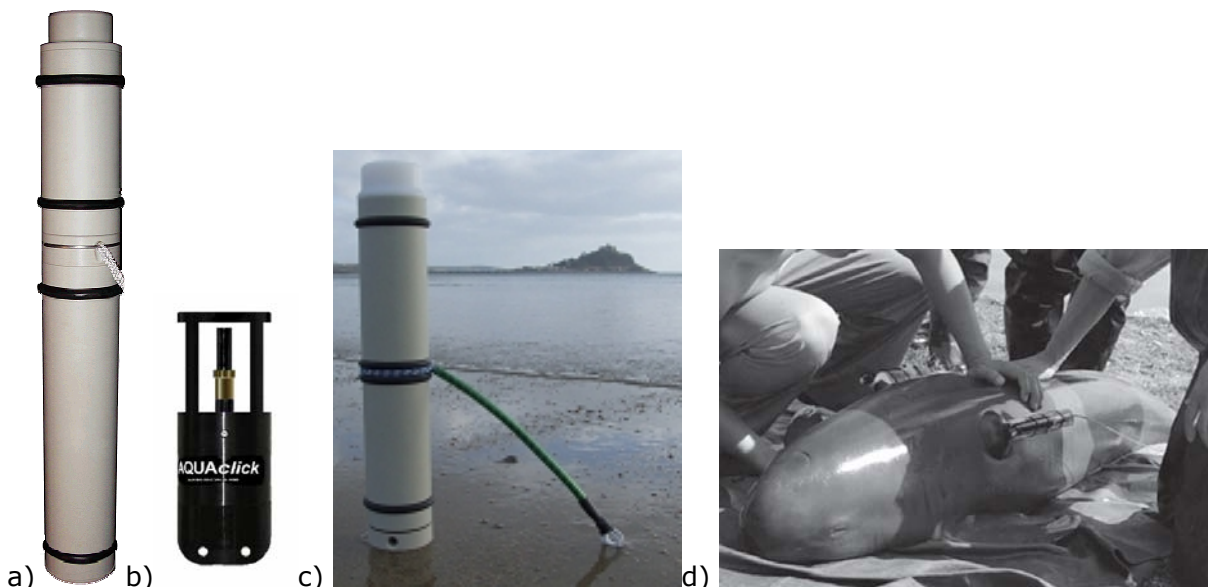


Figure 6-9: Different passive acoustic devices: a) T-POD¹, b) Aquaclick 100 (PCL)² c) C-POD d) A-Tag, attached with suction cups to a finless porpoise (from Akamatsu et al. 2005)

¹ <http://www.chelonia.co.uk/>

² <http://www.aquatecsubsea.com/>

T-PODs utilise the relatively small bandwidth of harbour porpoise echolocation clicks centred at 130 kHz (Goodson et al. 1995, Kamminga et al. 1999) by comparing the energy content of the center frequency and a reference frequency. The reference frequency should ideally be outside of the typical harbour porpoise frequency spectrum and is usually set to 90 kHz (Verfuß et al. 2004). When a sound fulfils the above-mentioned criteria, it is registered with the exact time of occurrence and duration of the sound.

Another suitable instrument is the PCL offered by the Aquatec Group, UK (Figure 6-9b). This device was also originally planned to be used by fishermen to investigate the echolocation behaviour of harbour porpoises in the surroundings of gill nets and other fishery equipment (Amundin et al. 2007). It has, at present, limited memory capacity (8 MB) and battery life (up to two weeks) compared to the T-POD (128 MB Memory, up to 10 weeks of deployment), manufactured by Chelonia Ltd, UK.

For future projects the C-POD, also produced by Chelonia Ltd., will be available in June 2008 (Figure 6-9c). As there have been no studies utilising this instrument, it will only be included in the comparison of measuring devices (Table 6-5), but not for the analysis. The same applies to the PCL, as there is no field data available yet.

Another system, the A-Tag (Figure 6-9d), is currently evaluated for finless porpoises (Akamatsu et al. in press).

Depending on the locality where static acoustic monitoring should be used, it is necessary to evaluate, which kind of mooring system should be used. Two examples are given in (Figure 6-10a,b). Light anchoring systems can be used in protected areas, where no fishing takes place and only low tidal influences occur. Heavy anchoring gear needs to be used in areas of high fishing effort and traffic. Brasseur et al. (2004) describe a very heavy anchoring gear with a cardinal buoy, a smaller buoy and three anchors ranging in weight from 380 kg to four tons. This system provides enough protection for the measuring device in the North Sea.

Different types of analysis have been used for differing research aims. For example Carstensen et al. (2006) used the unit waiting time, as being the time between two consecutive registrations, to describe short-term effects during pile-driving at a windfarm site. This unit seems to be appropriate for the assessment of effects where a direct allocation of the time of the impact and the registration of animals is needed.

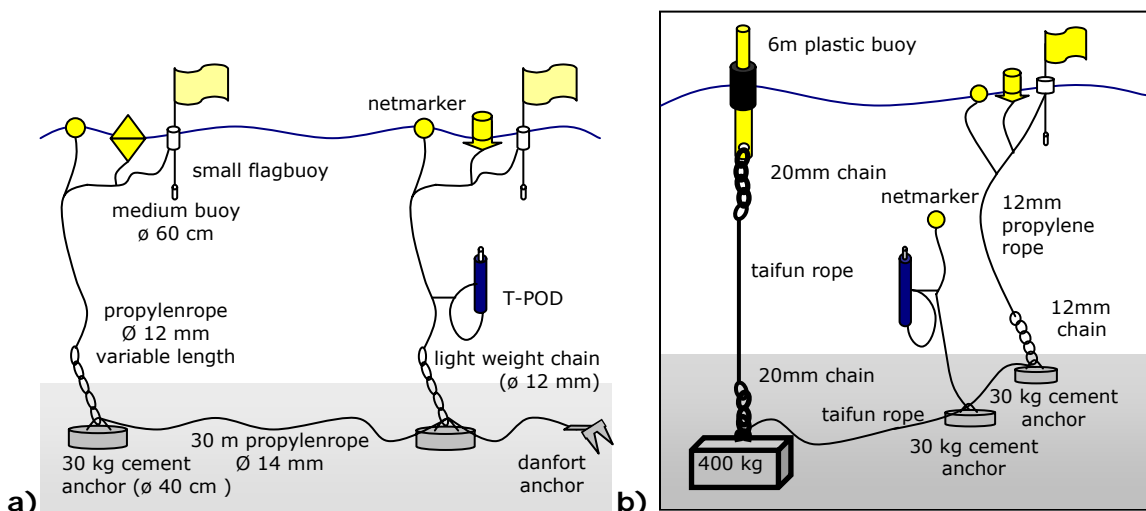


Figure 6-10: a) Light anchoring system for T-PODs in the Baltic Sea as used by Verfuß et al. (2007c), b) heavy anchoring gear used by Verfuß et al. (2007c)

Table 6-5: Differences between available measuring devices for static acoustic monitoring.

	PCL Aquatec Group	T-POD Chelonia Ltd.	C-POD Chelonia Ltd.
Hardware			
Memory	8 MB	128 MB	16 - 32 GB
<i>Battery capacity</i>	<i>appr. 2 weeks</i>	<i>6-12 weeks</i>	<i>variable, at least 4-5 month with 6 D-Cells, even longer with 15 D-Cells</i>
Battery Type	rechargeble NiMH Accumulator, 2 hrs. fast charge	6-12 D-Cells	6-15 D Cells
<i>hardware based noise suppression³</i>	<i>4th order band pass filter</i>	<i>noise adaptation (for version 4 and 5)</i>	<i>similar as for T-PODs</i>
Software			
Software for deployment and data download	Aquataalk, proprietary	T-POD.exe, proprietary	unknown
<i>View logged files</i>	<i>ClickView, proprietary</i>	<i>T-POD.exe, proprietary</i>	<i>unknown</i>
Postprocessing	manual, durations of two registered bands are compared taking other factors into consideration	preconfigured train detection algorithm	unknown
Species differentiation			
Harbour porpoise and Bottlenose Dolphin	only designed for harbor porpoises	possible	possible
Other odontocete species	no	device can be set to species specific criteria but cannot log all species simultaneously	device may log different species simultaneously

Other units commonly used are the percentages of porpoise positive days (%ppd), hours (%pph), ten minutes (%pp10min) and minutes (%ppmin) (Verfuß et al. 2007a, Dähne et al. 2008, Blew et al. 2006, Verfuß et al. 2008), defined as the fraction of the observed time intervals with porpoise registrations. These units can be valuable to determine long-term effects (Diederichs et al. 2007a,b) and describe preferred habitats as well as seasonal migrations of harbour porpoises (Verfuß et al. 2007a).

Other units previously used to analyse T-POD data have involved the discrimination of encounters and the number of recorded clicks per defined time interval. Encounters are separated by at least ten minutes of silence. Porpoises and especially other delphinid species have a range of echolocation behaviours as for instance fish catch and orientation towards an

³ as stated by the manufacturers

object, which involve different source levels of the signals and different repetition rates. As the units encounters will be influenced heavily by a wide variety of echolocation signals with different source levels and the number of recorded clicks per defined time interval will be biased by varying repetition rates, it is not recommendable to use these units as proxies for the density of animals in an area. The term 'click activity' for the number of clicks per defined time interval should not be used any longer, as it has been used for other units as well and might be misleading. Most people might understand, that the click activity for a day will be a proxy for how long a porpoise was using his sonar during that particular day, while the real meaning is how intense echolocation was used. The benefit might be a very coarse behavioural analysis, but in the authors opinion a detailed description of different behaviour should be used to develop algorithms for an automated behavioural analysis. The T-POD software already has an automated 'approach-sequence' finder. The characteristics of approach-sequences can be found in Verfuß et al. 2005.

The units porpoise positive time intervals and waiting times are less biased by behavioural changes, as porpoise positive time intervals represent artificial time segments where each segment has the same chance, that a special behaviour will occur or not and waiting times are usually long compared to the time when animals are recorded and could have different behaviours.

Siebert and Rye (2008) correlated T-POD data from the Baltic Sea with aerial survey data (from September 2002 to September 2005). The resulting GAM showed a high explained deviance of 73 % and stated for this dataset, that:

a percentage of less than 35 % porpoise positive days (ppd) indicates densities of less than 0.1 animals per km²

35 to 80 % ppd indicate 0.1 to 0.4 animals per km²

>80 % indicate more than 0.4 animals per km².

These calculations show clearly, that porpoise positive time units can be used as a proxy for relative densities. But it is necessary to repeat such studies to find better fitted models. It seems to be also relevant to use different porpoise positive time units, as days become unreliable, when they get close to 100 %. That the above mentioned study cannot go above 0.4 animals per km² using ppd shows, that a correlation with porpoise positive hours is needed, but definitely not, that T-PODs are not usable in high density areas.

Species identification

While harbour porpoises use only a small frequency band for echolocation, dolphins have different frequency spectrums with wider ranges of variation (Richardson et al. 1995). The bottlenose dolphin, being one of the more common species on the coasts of the UK, produces clicks with center frequencies around 110 to 130 kHz which also contain quite high energy contents at lower frequencies (Richardson et al. 1995, Au et al. 1989). This enables to discriminate between harbour porpoise and bottlenose dolphin echolocation clicks. T-PODs can divide each minute of logging into 6 separate intervals (scans). Each scan can be preconfigured to different centre frequencies as well as reference frequencies. Thus, a scan preconfigured for harbour porpoises may also register bottlenose dolphins, but scans set to lower centre frequencies will only register Tursiops. By alternating the settings for the scans, it is possible to register and differentiate between the species. Nevertheless, a differentiation between bottlenose dolphins and other delphinid species commonly occurring in the United Kingdom is not possible.

If bottlenose dolphins and harbour porpoises occur in the same area of interest, it is one possibility to use alternating scans. The other possibility is to use specifically set individual T-PODs in dedicated approaches to both species separately.

The C-POD will log the intensity, centre frequency, bandwidth, duration and envelope of individual clicks (Tregenza, pers. comment⁴), making it possible to address single clicks to their origin. Registering all of the above values simultaneously will give powerful species discrimination tools. As there will be no predefined settings appropriated to single species detection it will be possible to detect different species with the same device making solutions like alternating scans dispensable.

Spatial Scale

For small odontocetes static acoustic methods have a very small spatial coverage, which is delimited by the effective detection/hearing range which is less than approximately 200 m (Tougaard et al. 2006c). The spatial resolution depends on the number of measuring devices used.

Static acoustic methods have an extremely high time resolution and coverage. The explanatory power at the population level, however will usually be small, as the detection distance also limits the number of animals detected by each unit.

Temporal Scale

Static acoustic methods have proven to be reliable and enduring (Tougaard et al. 2006c, Verfuß et al. 2008). This makes it possible to get a temporal coverage of 100 %, if no devices are lost. T-PODs can log up to 1,000,000 clicks per second, which gives an extremely high temporal resolution.

Methodological variation

While line-transect surveys have been standardised and can be compared beyond borders and studies, this has not happened for static acoustic monitoring yet. There is a pressing need for all scientists and consulting agencies to commit themselves to use a standardised approach. This study can only make recommendations gained by operating experience at the German Oceanographic Museum and BioConsult SH. We will also discuss different approaches and their pros and cons.

Line-transect surveys have a high methodological variation due to environmental variables like sea state, glare, water turbidity, observation skills and cloud cover. All of these factors have no or at least only little influence on the data acquisition with T-PODs.

One seriously confounding variable is the number of recorded ambient noise clicks, due to high sea states and maybe boat noise, as too many clicks lead to a masking effect that will result in diminishing percentages of odontocete registrations. An important standardised rule should be the exclusion of periods with high background noise levels. In previous studies, time periods with more than 4000 clicks per hour registered during a 10 minute interval were excluded to prevent such a bias (Blew et al. 2006).

Other methodological variation arises from differences in T-POD sensitivities (Verfuß et al. 2004a, Dähne et al. 2006a, Kyhn et al. 2006). T-PODs with lower thresholds register clicks of the same amplitude at farther distances, thus enlarging the actual detection area or better volume of the water column. Kyhn et al. (2006) showed that there is a relationship, not only between the threshold of individual T-PODs and the number of registered clicks, but also between the threshold and the number of registered clicks classified as being from porpoise origin. There also seems to be also a relation between threshold and the encounter duration, as well as number of clicks in classified porpoise positive minutes per day. The sensitivity of T-PODs tested in the study of Kyhn et al. 2006 spanned over 9 dB, reaching from 114 to 123 dB RMS re 1 µPa. Verfuß et al. (2007b) showed, that even small changes in sensitivity (3 dB) can

⁴ Dr. Nick Tregenza, 5 Beach Terrace, Long Rock, Penzance, Cornwall, TR20 8JE, UK

lead to different % ppmin, hence biasing the data acquisition. Presumably this relationship will cease by using longer porpoise positive time intervals.

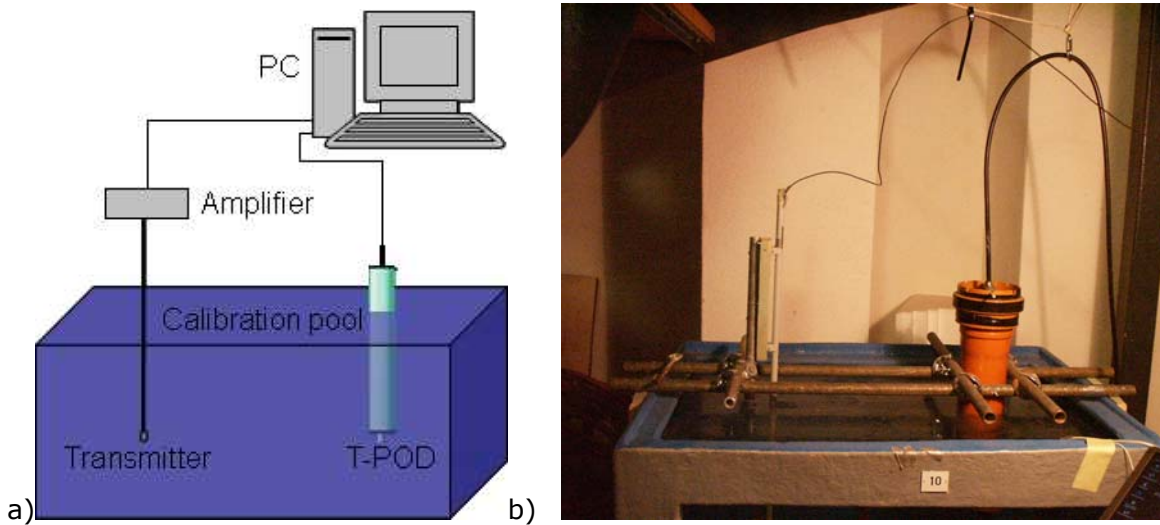


Figure 6-11: Test tank calibration of T-PODs, from Verfuß et al. 2007c, a) schematic drawing with the T-POD as receiving instrument, b) photograph of the actual pool with the reference hydrophone as the receiver.

Older T-POD versions (version 1 to 3) had much stronger variations in receiving thresholds than the newer version 4 and 5 T-PODs (Dähne et al. 2006). There are also differences in the number of registered porpoise positive ten minutes using the different T-POD versions V3 and V4 set to the same thresholds (Dähne et al. 2006).

To prevent biases due to different T-POD sensitivities it is necessary to calibrate all detectors. There are two possible solutions:

1. An absolute measurement of the minimum receiving thresholds in a closed environment under test conditions (test tank calibration, Figure 6-11), as described by Verfuß et al. (2004a), Dähne et al. (2006) and Tougaard et al. (2006b), is preferable to determine absolute thresholds, thus enabling the researcher or consulting agency to directly compare their results in the field. There are different opinions on how to use these results. While Kyhn et al. (2006) and Carstensen et al. (2006) used the measured thresholds as a covariate in a later modelling approach, Dähne et al. (2006) and Verfuß et al. (2007a) use their measurements to adjust the thresholds to the same level. The authors of this report prefer the latter approach, as this will give you the opportunity to use the T-PODs in a rotation where every device can be used at each position. Otherwise the loss of equipment would ruin the data acquisition since a replacement T-POD might have a different sensitivity. The T-POD hydrophones should be tested from at least eight angles to find their radial directivity (Dähne et al. 2006). Results can be used to compare data from different studies and locations if the equipment was calibrated under the same conditions.
2. A comparative measurement (intra calibration, Figure 6-12) of the T-PODs at sea is easier to do, but needs certain deployment times to give comparable results (Verfuß et al. 2007, Diederichs et al. in prep.). So far there are no publications available, which give recommendations on how long the deployment should last to receive comparable results in different porpoise positive time intervals. A decent approach to these issues should involve a separate power analysis using single measurement units as samples (porpoise positive intervals like minutes, ten minutes, hours or days, waiting times and encounters) assuming, that they are independent among each other. If they are not independent, then other measures should be taken into consideration. The results of intra calibrations are only comparable to other studies if at least some of the T-PODs

used are absolutely calibrated as well. This does not affect the BACI Design but will make comparisons like 'waiting times in EIA one were longer after impact as in EIA two after impact' impossible.



Figure 6-12: Intracalibration of T-PODs close to the wind farm Nysted, from Verfuß et al. (2007c), the devices are separated by at least 30 cm to prevent a bias from interferences and echoes.

There is also some concern about different types of behaviour of the animals influencing the data acquisition. For instance, harbour porpoises use adjusted source levels to different distances towards an object (Beedholm and Miller 2007). This would imply, that harbour porpoises are better detectable if they are orientating towards a distant object. This should be topic of future research.

Biological variation, detection of short and long-term effects on cetaceans

SAM data are treated differently depending on whether short or long-term effects are of interest. When studying short-term effects, Carstensen et al. (2006) used the monitoring unit 'waiting times' as being the time between two encounters of harbour porpoises lasting at least ten minutes. When studying long-term effects porpoise positive time intervals are suggested to account for double-counts (e.g. registering of the same harbour porpoises repeatedly during one hour) and different group sizes.

T-PODs have been successfully used in high (e.g. Tougaard et al. 2006b) as well as medium and low density areas (e.g. Tougaard et al. 2006a, Verfuß et al. 2007a, Dähne et al. 2006b).

Short-term effects

There are certain differences in the design for detecting short-term effects opposed to long-term effects. Carstensen et al. (2006) and Tougaard et al. (2006b) showed, that there are seriously lengthened waiting times after the direct impact of pile driving. Taken into consideration, that they could measure an effect up to a distance of 21 km (Tougaard et al. 2006b) and that they could not find the distance, where the effect ceased, raises the question, which distances should be measured during an EIA. The authors suggest that the effective distance should be stretched to at least 25 km for the first studies conducted. If an impact can be shown, even at such a distance, then the range needs be reconsidered and eventually the distance should be raised again for the next EIA.

The German Standarduntersuchungskonzept (standard research code of conduct during wind farm work, StUK, BSH 2007) proposes to use at least six devices inside the windfarm area and

at least three outside. The authors suggest increasing the latter number, as large distances have to be covered to show a ceased effect.

Long-term effects

To show how long-term effects can be assessed using static acoustic monitoring the methods already described in Dähne et al. (submitted) are given.

Dähne and colleagues used a dataset of five measuring positions employing T-PODs around the isle of Fehmarn and five positions around the Kadet Trench in the German Baltic Sea (Figure 6-13), to find the necessary sample sizes needed to monitor these two marine protected areas (MPAs). As the monitoring of long-term effects is very similar to finding inter annual differences between consecutive years, the same statistical assumptions must be fulfilled. For long-term monitoring, positions should stay stable over time to reduce biases due to geographical differences.

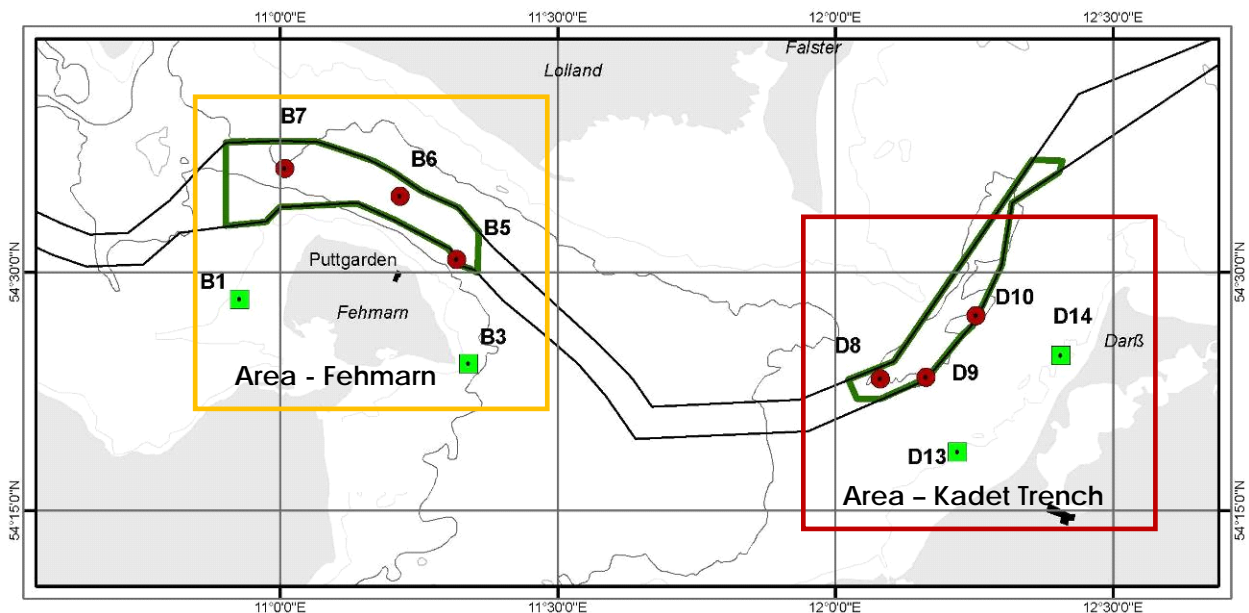


Figure 6-13: Measuring positions used to determine necessary samples sizes by Dähne et al. submitted for the areas Fehmarn (orange square) and Kadet Trench (red square) as examples for a future design of environmental impact assessments. Positions inside the MPAs (green thick line) are marked by a red circle, while positions outside are marked by a green square.

If positions stay stable, then a paired samples t-Test or a General Linear Model (GLM, McCullagh and Nelder 1989) can be performed to determine the possible impact. As it is very difficult to find the necessary sample size for GLMs the hereby cited study (Dähne et al. submitted) focuses on the paired samples t-Test. The data acquisition period lasted from August 2002 to March 2007, giving a dataset of five complete years to describe the area. All data obtained were scanned for trains of clicks using the algorithm 3.0 of the T-POD Software and all classified click trains (Cetacean high and low probability, doubtful and very doubtful classes) excluding boat sonars were exported. These sequences were then visually reviewed to decide if the trains were likely to be from porpoise origin (Dähne et al. submitted).

The percentage of porpoise positive hours, defined as the percentage of hours with at least one porpoise registration during the monitored time was calculated on a monthly basis.

Given, that porpoises are very mobile animals, which cover large areas in daily trips (Teilmann et al. 2004, Sveegaard et al. 2006, Teilmann et al. 2008), porpoise positive hours seem to be a sensitive unit to detect large and medium scaled trends without expressing too much locally

differing phenomena. At higher densities and higher contact frequencies, porpoise positive ten minutes may be a more appropriate unit. As large and medium scaled long-term effects on the population density might only have a small amplitude, it is necessary to evaluate longer time periods. Dähne et al. (submitted) chose to consider each month at each position as a repeated measurement within each quarter (January to March, April to June, July to September and October to December) of the year. This results in up to 15 measured values in each quarter of the year and up to 75 samples when the five years are pooled.

We will give some examples in Figure 6-14 to Figure 6-18. The results of the original data sets give us a mean value of 11 % porpoise positive hours for the Kadet Trench in the third quarter (Figure 6-14). The standard deviation of 7.2 % is in the same order of magnitude as the mean values. If means and standard deviations are in close vicinity, then a power analysis will always result in very high sample sizes to show low percentages of divergence. For the Kadet Trench this results in a sample size of 214 samples being necessary to show a divergence of ten percent porpoise positive hours.

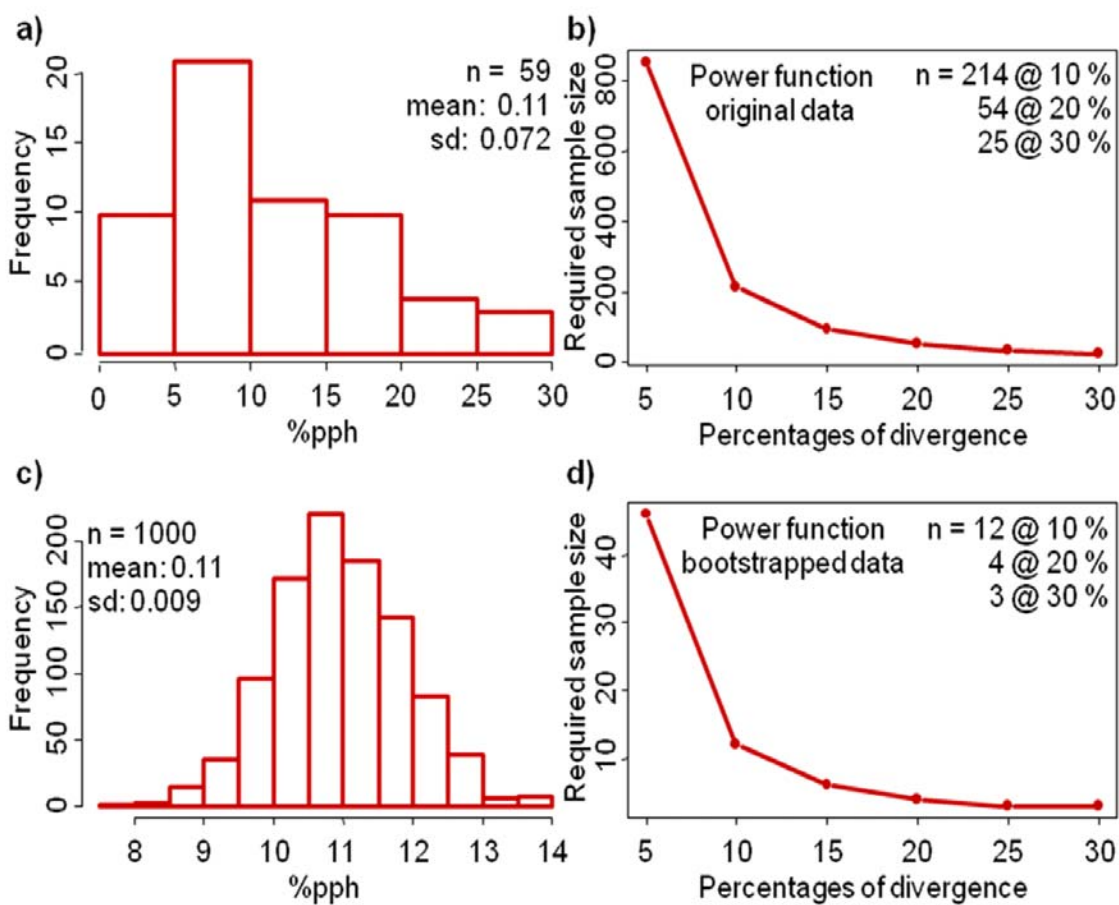


Figure 6-14: Example dataset from the five measuring positions around the Kadet Trench in the third quarter of the years 2002 till 2006 (Dähne et al. 2008), a) frequency distribution of the original data including number of samples (n), mean and standard deviation (sd), b) power analysis: required sample size (positions per month) for a given mean and standard deviation (original data) c) frequency distribution of the bootstrapped data including number of random samples (n), mean and standard deviation, d) power analysis: required sample size (positions per month) for a given mean and standard deviation (bootstrapped data).

As these sample sizes are not obtainable with static acoustic methods, it is necessary to perform a bootstrap procedure (Efron 1979) to resample the dataset by randomly choosing samples out of the original dataset with repetition allowed. This leaves the mean value more or

less unchanged (original: 0.11, bootstrapped 0.11) but seriously reduces the standard deviation (original: 0.072, bootstrapped: 0.009). Thus the necessary sample size is also reduced significantly. To show a ten percent deviation it would only be necessary to use a sample size of 12, which would result in four positions over a period of three months.

The same analysis was conducted with the dataset from around the isle of Fehmarn in the fourth quarter from October to November. In Figure 6-15 the frequency distribution and the results for the power analysis each for the original as well as the bootstrapped dataset are given. The mean percentage of porpoise positive hours is higher than in the Kadet Trench (28.4 % opposed to 11 %), but the standard deviation is also quite high (14.2 %) resulting again in a high necessary sample size of 393 samples to show a ten percent divergence in % pph. The application of a bootstrapping procedure is reducing the standard deviation, thus the necessary sample size again decreases to approximately twelve samples, expressed as a three month period with four measuring positions.

A regular seasonality in porpoise density is mirrored in a change of the mean percentage of porpoise positive hours per month as well as per quarter of the year.

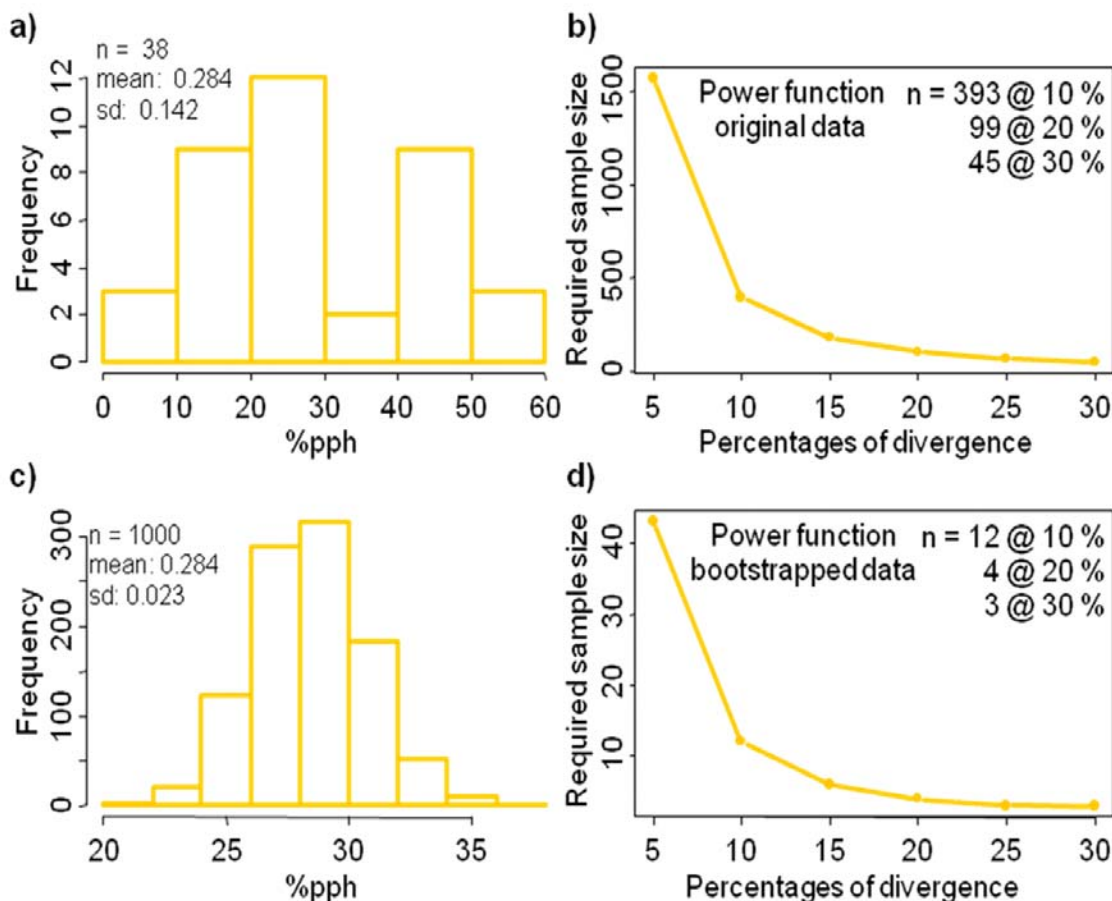


Figure 6-15: Example dataset from the five measuring positions around the Fehmarn in the fourth quarter of the years 2002 till 2006 (Dähne et al. 2008), a) frequency distribution of the original data including number of samples (n), mean and standard deviation (sd), b) power analysis: required sample size (positions per month) for a given mean and standard deviation (original data) c) frequency distribution of the bootstrapped data including number of random samples (n), mean and standard deviation, d) power analysis: required sample size (positions per month) for a given mean and standard deviation (bootstrapped data).

There is a dependency between the seasonal variation and the resulting standard deviation in this dataset (Figure 6-16). For Fehmarn the highest standard deviation is in the fourth quarter.

This correlates with the mean percentage of porpoise positive hours per year, which is also highest in the fourth quarter (Dähne et al. submitted). The same can be seen in the Kadet Trench data. Honnef et al. 2006 reported the highest % pph from the third and fourth quarter in the Kadet Trench and Figure 6-16 shows, that the standard deviation in these quarters of the year is also the highest. Still for the maxima of % pph in the years a fair number of samples can be used to detect even small differences in %pph. But what happens, when the % pph is lowest?

Figure 6-17 shows an example derived from the first quarter at the island Fehmarn. Because of hard winters, the loss of equipment was quite high in these periods. This leads not only to a limited number of samples in the original data set, but also to a mean of 14.1 % pph, which is actually lower than the standard deviation of 15.5 % pph. Even bootstrapping these data does not lead to acceptable sample sizes to show small differences (<20 %). To show a difference of more than 20 % reasonable numbers of measuring devices may be used.

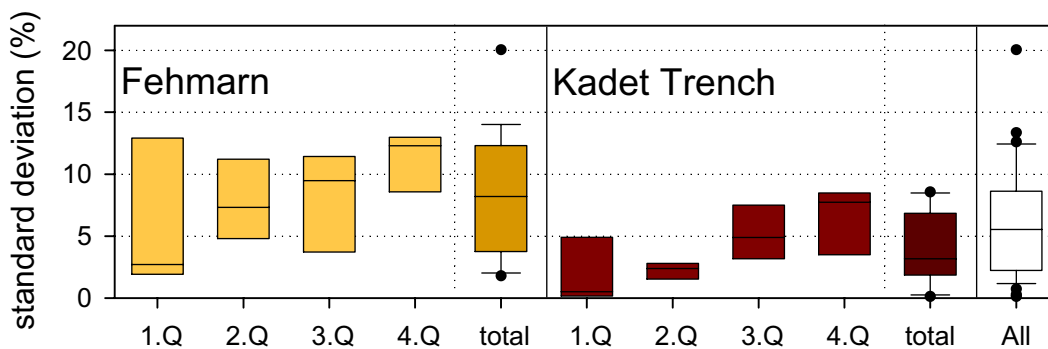


Figure 6-16: Boxplot of the standard deviation of the percentage of porpoise positive hours per month, grouped in quarters, of the years 2003-2007 at the area Fehmarn and Kadet Trench. Shown are the median, 25 % and 75 % percentile as well as the 10 % and 90 % percentiles and outliers, taken from Dähne et al. submitted.

This is a very important fact in areas where residual stocks of harbour porpoises or bottlenose dolphins occur. In areas where these remnant populations need special protection, the necessary sample size to detect small differences is increased and thus the effect, that can be shown, might not be sustainable at stock level at all.

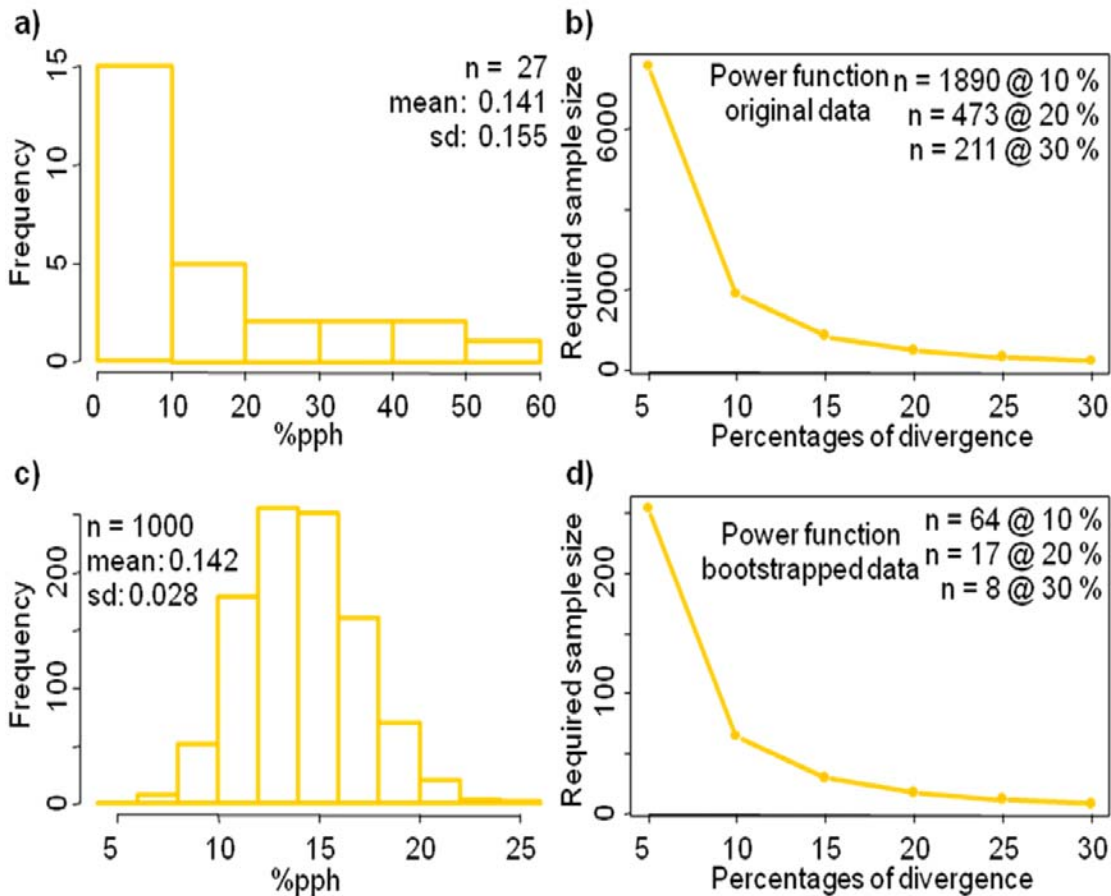


Figure 6-17: Example dataset from the five measuring positions around Fehmarn in the first quarter of the years 2003 till 2007 (Dähne et al. 2008), a) frequency distribution of the original data including number of samples (n), mean and standard deviation (sd), b) power analysis: required sample size (positions per month) for a given mean and standard deviation (original data) c) frequency distribution of the bootstrapped data including number of random samples (n), mean and standard deviation, d) power analysis: required sample size (positions per month) for a given mean and standard deviation (bootstrapped data).

To see what kind of outcome can be expected from a high density area, compared to the low and medium density areas Kadet Trench and Fehmarn, we conducted a power analysis with a dataset from the North Sea west of the island of Sylt (area 2 in 6.1.1, data source: BioConsult SH). This dataset consists of only two closely situated measuring positions (approximately 500 m apart) from 2002 to 2004, and thus the results should be treated carefully as sample size is low and no geographical gradient involved.

The mean percentage of % pph of 62.7 % is much higher than the maxima in the Baltic Sea. As we would expect from two close positions, the standard deviation is low compared to the mean percentage. Thus, it would be possible to show a 20 percent difference without bootstrapping the dataset with 13 measured data points resulting in approximately five measuring positions over a three month period. Bootstrapping the dataset further decreases the standard deviation, but not as much as for the Baltic Sea areas, as the sample size of the original dataset is smaller compared to the Kadet Trench and Fehmarn. The power analysis of the bootstrapped data results in a very small sample size of five measurements being necessary to show a ten percent variation in the %pph. This value is actually the lowest limit at which a bootstrap algorithm can be performed to decrease the standard deviation. As measuring equipment may be lost at sea, it must be assured that more T-PODs are deployed as the result of the power analysis suggests.

The power analysis for Sylt suggests five measurements translating into two positions per quarter of the year.

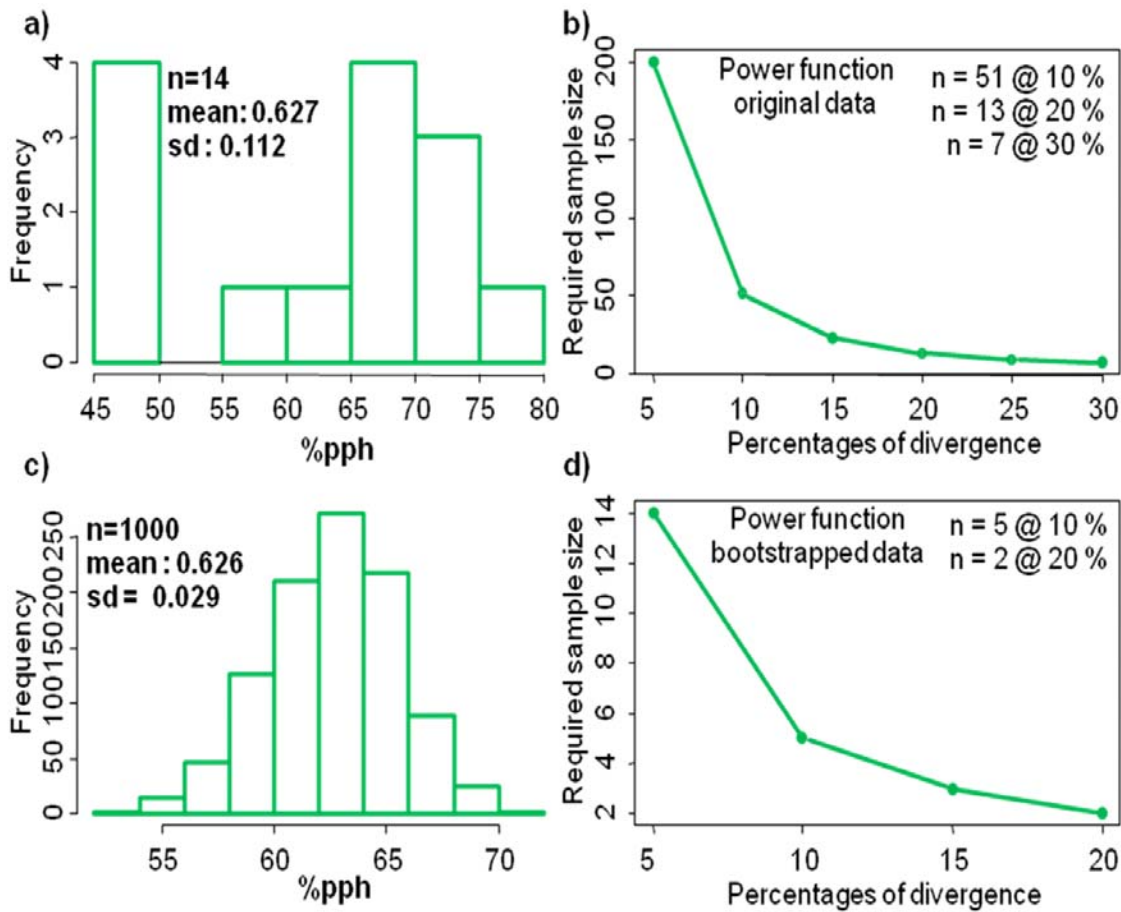


Figure 6-18: Example dataset from the two positions west of Sylt in the third quarter of the years 2002 till 2004, a) frequency distribution of the original data including number of samples (n), mean and standard deviation (sd), b) power analysis: required sample size (positions per month) for a given mean and standard deviation (original data) c) frequency distribution of the bootstrapped data including number of random samples (n), mean and standard deviation, d) power analysis: required sample size (positions per month) for a given mean and standard deviation (bootstrapped data).

Thus it would be sensible to consider two plus one T-PODs as the minimum number of units used in any approach. The hereby shown data are strong evidence that the necessary sample size for static acoustic monitoring depends on the geographical location and the seasonal variation at designated windfarm areas. This underlines the necessity to perform baseline studies prior to windfarm construction to find the necessary sample size required at the individual location, and to detect areas where protection measures are more important than the construction of a windfarm.

Requirements for SAM:

In conclusion, there are some points, which should be fixed or preassigned during the course of an environmental impact study using static acoustic methods:

There should not be a mixture of different types of equipment, this concerns

- o Manufacturer

- Version of acoustic devices.

Devices should be set to a standard threshold (Verfuß et al. 2007c) measured in absolute tank calibrations or

Devices should be intra calibrated and differences in receiving characteristics should then be used in a factorial manner in the statistical design, the use of a standard device which is always included in the calibrations is preferable, repeated measurement should be conducted.

Background noise should be eliminated as a confounding variable by introducing a limit to the maximum click number registered (during data acquisition) and analysed (post processing).

Background noise can also be suppressed, by choosing the standard threshold carefully. Verfuß and colleagues (2008) found, that a threshold of 127 dB P-P re 1 μ Pa was usable for T-PODs in the German Baltic Sea, thus seriously diminishing the side effects caused by high noise levels.

The time unit %ppd, %pph, %pp10min or %ppmin on which a calculation will be based upon, should be (re-)considered after the base line data acquisition. Chosen units should depend on whether the area is of high or low porpoise densities. Smaller time intervals will be more suited for high density areas, while longer time intervals will work well for low densities.

Tregenza and Pierpoint (2007) noted that there is a link between all of these units and that theoretical consideration can lead to comparability. They also noted that the unit most suited for a dedicated approach is the one that gives you up to 30 % positive time intervals. This unit will be only marginally affected by differing devices and the so called ceiling effect, where the variation in the data strongly increases with only small increases in the actual number.

T-POD settings, except for minimum intensity or sensitivity should be constant within the individual devices and during the course of the study, while minimum intensity or sensitivity can be used to set the T-PODs to a standard sensitivity. In Table 6-6 standard settings for T-PODs to distinguish between the two species harbour porpoise and bottlenose dolphin are given.

A power analysis should be conducted with the base line data to find the necessary sample size for the control-impact assessment.

The minimum number of T-PODs used should not be less than three devices and higher than given by the power analysis results to account for data losses.

The devices should be rotated between the positions to reduce the bias caused by different sensitivities or individual T-PODs.

Table 6-6: Recommended T-POD settings for different species⁵

Setting	Applicable for	Harbour Porpoise		Bottlenose Dolphin		Both	
		V2 & V3	V4 & V5	V2 & V3	V4 & V5	V2 & V3	V4 & V5
Target (A) Filter Frequency	V2-V5	130 kHz	132 kHz	50 kHz	50 kHz	Alternating 50/130 kHz	Alternating 50/130 kHz
Reference (B) Filter Frequency	V2-V5	90 kHz	90 kHz	70 kHz	70 kHz	90 kHz	Alternating 70/92 kHz
Selectivity (Ratio A/B)	V2,V3	6		2		Alternating 3/4	
Click Bandwidth	V4,V5		4		5		Alternating 5/4
'A' integration period	V2,V3	Short (10)		Short (10)		Short (10)	
'B' integration period	V2,V3	Long (18)		Long (18)		Long (18)	
Noise adaptation	V4,V5		++		++		++
Minimum Intensity/Sensitivity	V2-V5	Variable (6)	Variable (12(V4)/16(V5))	Variable or 6	Variable (9 (V5))	Variable	Alternating 9/16 (V5)
Scan limit of N of clicks logged	V2-V5	240	240	160	240	160	240

6.3 Telemetry and Dataloggers

Telemetry is the use of technology to transmit measured data from a sensor to a spatially separated receiver. Telemetry has been widely used in zoology to transmit positions of animals over large areas in order to study their habitat usage or their seasonal migrations. In these cases, the term telemetry is referred to as the combination of a position measurement by a tag attached to an animal and its transfer to the researcher. New sensory technological abilities have been incorporated into recent tags. This enabled scientists to measure a variety of different values to assess all aspects of animal life cycles.

There are several tracking methods available for different approaches to either different species or different spatial scales.

Methodology

All telemetry studies require that the animals are either tagged in their natural habitat or that they are intentionally or unintentionally caught and tagged on land, boat or in the water. For big enough marine mammals, like sperm whales, it is possible to attach tags by using suction cups and long poles or crossbows to deploy the tag (Madsen et al. 2002). The tag will then stay on the animal for a short time, separate, float onto the surface and can be retrieved using VHF-antennas to get a bearing to the tag. This technique is only negligible invasive.

⁵ <http://www.chelonia.co.uk>

For smaller cetaceans and pinnipeds a tagging without capture is not possible. Catching the animals is the only applicable method to attach a tag, which does not seriously hurt the animals in the process. Tagging of harbour seals in Germany involves two ships towing a net towards a sand bank (Adelung et al. 2004). The seals lying on the bank are caught in the net by two people jumping off the two ships and dragging the net towards and onto the beach. Seals are immediately separated in ring nets with their heads facing the beach to reduce the urge to get to the water. The animals are under surveillance by a veterinarian and cooled with sea water. The tags (Figure 6-19a) are attached to the animal's fur using epoxy glue (Figure 6-19b) (Wilson et al. 2007, Liebsch et al. 2007, Tougaard et al. 2006d). The tags will fall off the animals with the fur during moulting. This tagging-procedure provides plenty of stress to the individual and should thus be used only on a small fraction of the population.

To tag harbour porpoises is even more difficult. Researchers in Denmark have succeeded by co-operating with fishermen using pound nets in which harbour porpoises will not drown when being captured in them (Teilmann et al. 2004). If harbour porpoises are caught by coincidence in these nets, the fishermen will call the scientist who will then equip the porpoise with a satellite tag and release it afterwards (Figure 6-20). The tags will be attached to the dorsal fin with a screw that corrodes within a certain amount of time, eventually leading to the release of the tag. For finless porpoises Akamatsu et al. (2007a) used tight fitted straps to attach the tag to the animal.

The oldest method using VHF-Transmitters and beam antennas to track animals in close vicinity is still the most accurate next to an approach with GPS-modules in combination with satellite tags⁶, which register the position via GPS, but send the gathered data via satellite to the researcher. Single satellite tags⁷ can deduct the position using the Doppler-effect and send them via satellite either to an e-mail account or to mobile phone via SMS.

Newer systems also include sensors for depth, temperature, light level as well as wet/dry sensors which can give a good idea of the animals' behaviour. This is especially important for seals, as none of the other previously described methods can assess the effect of windfarms on seals directly.

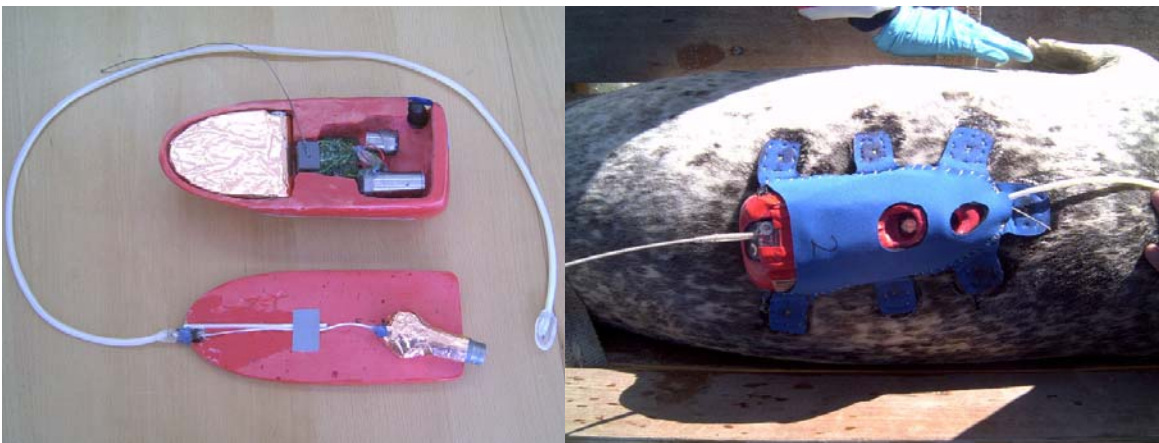


Figure 6-19: a) Close up of the tag used for seals in Germany (from Adelung et al. 2004) b) Harbour seal equipped with a satellite tag, datalogger and mandibular sensor (from Adelung et al. 2004). Note that the floating device is shaped to minimise vortices and drag. The floating device is released by a timer and can be retrieved after being washed ashore.

⁶ <http://www.wildlifecomputers.com>

⁷ <https://www.argos-system.org>



Figure 6-20: Harbour porpoise equipped with a satellite tag⁸.

Teilmann et al. (2008) show that satellite tagging can be used to monitor the movements of harbour porpoises in Danish waters for long time periods (up to 349 days) thus gaining information about seasonal variation, geographic distribution and diving behaviour.

The tags currently in use can be roughly divided into

1. VHF-Transmitters

Good location accuracy

Not possible to use for marine mammals in a wide range area, limitation to range: approximately twenty kilometres, fifty kilometres could be possibly reached

Good resolution in time (every time a position fix can be obtained)

Low battery consumption

2. Satellite Tags without GPS-Sensors

Very low accuracy (dependent on Argos transmission class (1-3) between 1500 to 150 m in the 68 % percentile)

Feasible for large area surveys

Very low resolution in time (only a few transmissions per day in high accuracy classes)

Low battery consumption (up to a year durability)

3. Satellite Tags equipped with GPS-Sensors

Very good accuracy (appr. 20m precision)

Feasible for large area surveys

High resolution in time (GPS-Position fixes can be stored and later be transmitted via satellite)

Very high battery consumption

4. Data loggers

⁸ [http://www.dmu.dk/International/Animals+and+plants/Satellite+tracking/Harbour+ Porpoise](http://www.dmu.dk/International/Animals+and+plants/Satellite+tracking/Harbour+Porpoise), 26.03.2008

All of the tags mentioned above can be equipped with data loggers recording for instance depth, temperature, salinity, pitch, roll and light level. Liebsch et al. 2007 used an inter-mandibular angle sensor (IMASEN) to detect feeding events for seals.

The A-Tag (Akamatsu et al. 2007a), an acoustic data logger, records the high frequency echolocation sound events of harbour porpoises via a stereo hydrophone system (Akamatsu et al. 2005). This gives information about the behaviour of the animal during dives, travelling, prey catch including its daily variation.

The D-Tag (Madsen et al. 2002), another acoustic data logger to be attached with suction cups, records echolocation as well as communication vocalisations of cetaceans in a lower frequency range (0,6 to 45 kHz). It has been used only on large cetacean species, for instance sperm whales, to reduce the influence that this large tag is having on the animal. The tag also includes a three-axis celerometer, a three-axis magnetometer and a pressure sensor (Jones et al. 2008).

The resolution of the recorded data is very fine scaled caused by a high sampling rate, therefore the observation time is limited to the memory capabilities.

5. Acoustic Telemetry

These are small acoustic devices transmitting coded signals. They will be implanted into the skin of the target animal or should be eaten by it. Animals targeted so far are different fish species, and acoustic telemetry is currently in use for identifying their key habitats (e.g. Lindholm et al. 2005, Lindholm et al. 2006)

Several receivers can be operated from different stationary or mobile platforms (ship or other).

Spatial and temporal Scale

If enough animals can be tagged from different locations, it is possible to identify their preferred habitats (Teilmann et al. 2008). Hence the spatial coverage is very large for satellite telemetry.

The spatial resolution is different for all kinds of tags. Concerning the accuracy of satellite tags without GPS-Sensors, taking into account, that not many animals can be tagged, no reasonable number of accurate positions can be obtained inside and outside the windfarm. There are several reasons for that. Firstly, if a seal or harbour porpoise is travelling through the wind farm, it would still be necessary that an ARGOS satellite is in a favourable orbit to give a position. This is not happening for most of the time. Furthermore the windfarm area is very small compared to the surrounding oceanic environment and also compared to the estimated home ranges of harbour porpoises (Teilmann et al. 2008). If a decent estimate should be made about an avoidance reaction, it would be necessary to tag a lot more animals than has been done so far.

Tougaard et al. (2008) report a method to post process the satellite positions using a regular grid and the relative location of a received transmission in the grid cells to determine the probability of the detection being in the grid cell. The simple assumption is that a location in the centre of the cell has a higher probability to originate from inside the cell, than a position from the edge. The probability functions can be calculated and used to classify the locations. This should allow for a more precise calculation regarding how much time animals spend inside and outside of the windfarm area.

The drawback of the more accurate positioning of GPS-sensors is, that the battery life of such tags is highly reduced compared to simple satellite tags. Because pinnipeds or cetaceans may only spend a small fraction of their time in the windfarm area durable tags are preferable.

Dataloggers used by Liebsch et al. (2007), Wilson et al. (2007) and Tougaard et al. (2006d) are an excellent solution to find behavioural changes inside and outside of the windfarm area, because they allow for the detection of different types of dives, as well as feeding behaviour and swimming speed. Usually dataloggers are equipped either with releasing mechanisms or they will fall off in the molting period. Whenever that happens the tag has to be recovered using a

positioning or relying on incidental findings when the battery is drained. Short-term tags can be equipped with VHF-transmitters to get a bearing to the device

If acoustic telemetry, like already used for fish species (Lindholm et al. 2006), will be introduced into marine mammal science, then it should be treated like a stationary point transect survey, as the receivers will then be stationary placed inside and outside the windfarm in a BACI design. For the currently used acoustic transmitters it has to be tested, whether they have an effect on the animal. If the signal is audible to the animal, behavioural changes might emerge.

There are also ambitious efforts to develop tags using combined GPS-positioning and GSM (Global System for Mobile Communication)-Transmission. Verfuß et al. (2008) used such a system to monitor the positions of their T-PODs with some success. This proves the practicability of the hardware to detect accurate positions. But the herein used system is not feasible as a tag because the size is too large and the battery consumption is way too high. New communication systems should solve such problems.

Methodological variation

There are some general remarks on the method of tagging animals in the wild:

Tagging is always an invasive method and should be used carefully. If other non-invasive methods are available to receive the desired results, they should be preferred.

There are reports about animals tagged and released, which had serious problems with growth, infections or even open lesions (Hazekamp et al. 2007). These animals can not be used for an extrapolating research, as their behaviour will be differing from healthy individuals.

The behaviour in the first hours or days might be influenced by the tagging process. Thus these data have to be discarded.

As all processes to localise animals are technical solutions the methodological variation will be related to the accuracy of the measuring system used.

Biological variation, detection of short and long-term effects on cetaceans

As previously explained the rather small achievable sample sizes limit the statistical explanatory power greatly. Nevertheless, this method is the only available tool especially for seals or larger whales in order to get windfarm related data. We therefore strongly recommend to do a lot more work on the usability of telemetry for windfarm related applications, not on a project specific way but on a cross-project way.

The variation will be strongly dependent on the sample size, regarding the number of animals tagged, as each animal might have different preferences for feeding areas or different home ranges. Thus it is very important to tag animals at different sites to reduce this bias.

In the case of satellite telemetry it very difficult to state anything about short-term effects on seals in the vicinity of a windfarm under construction, as it would be necessary, that a previously tagged animal is in close range and hopefully satellites are in a position to get a transmission from the tag. The chances are very small that this is going to happen. It seems more reasonable to actually equip seals with satellite tags for baseline and long-term studies, as the sample size can be increased over years and can give a decent insight into long-term avoidance reactions and trends as well as giving the base knowledge about preferred habitats.

To monitor short-term effects, different methods need to be used and developed. One possibility would be to equip a large enough number of seals with GPS/GSM-systems or dataloggers some time before construction work begins in the impact area and a reference area. It would be preferable to also have some individuals tagged in between those two areas. Tags should be equipped with a releasing mechanism to be able to retrieve them after the ramming under controlled conditions. This would be a possibility even in cases where no baseline data

exists. It would also be feasible to equip seals with long-term satellite tags as well as short-term dataloggers to cover for both types of effects.

Telemetry should only be used, if there are no other choices to detect effects of wind farms on animals. For harbour porpoises for instance, acoustic surveys using T-PODs or visual and acoustic line-transect surveys will give better results regarding their interpretability, because the analysis is not being done for single individuals. For pinnipeds or minke whales other choices for monitoring wind farm effects are very limited, thus leaving telemetry as the only choice to monitor large and small scale, short- and long-term effects. The statistical power will always be very low regarding small and short-term effects, due to the limitations in the number of animals tagged and their unpredictable position in relation to the wind farm during construction. For large scale and long-term effects the methods is much more reliable but requires a continuous work over years. Nevertheless, telemetry shows a tremendous potential for further research.

6.4 Haul-out site counting

Seals (Harbour and Grey seals) are usually counted by aerial surveys when they come ashore at haul-out sites. Haul-out sites are places where seals rest on land.

Methodology

Two observers are counting the seals by vision and in addition pictures are taken for verification of the counting for groups larger than 10 individuals. This procedure minimises the risk of some seals being overlooked. Seals on land and in the water next to the haul-out site should be included (Reijnders et al. 2006, Teilmann et al. 2006, Cunningham 2007). In the UK and the Wadden Sea, aerial surveys of harbour seals are usually conducted during the first three weeks of August, when seals are moulting. Additional counts are performed during the whelping period in June in order to get an estimation of the maximum number of pups. Animals which are counted at land during their rest are not directly connected to planned offshore windfarm sites. Using haul-out counting results is due to investigate if seals tend to avoid the disturbance from windfarm areas which are close by haul sites, and use alternative seal sites further away from the windfarm than before the construction. On the other hand such a country wide monitoring on a population level gives the chance of getting an estimation of the influence of several windfarms on the population of harbour and grey seals.

Methodological variability of the data

Two types of error may occur: 1) Some seals may move between sites during a single survey. 2) The number of seals hauled out may vary during the day, due to the weather, time of day, or because of disturbances.

Surveying all sites within a short time period (app. 2 hrs) eliminates the first error. The second type of error is reduced by conducting the surveys at the same time of day and during similar weather conditions every time (e.g. avoid rain and strong wind). Investigations from Cunningham (2007) showed that seal counts in the UK have a coefficient of variation of around 15 % and the relationship between counts and total population size is likely to vary spatially and temporally. This variation should be included in the estimates of the CV of correction factors. Cunningham (2007) could show that a 5% annual change in harbour seal population size was predicted to take around 14 years to detect based on annual surveys and a CV = 0.15. This detection period increases when monitoring methods with lower precision are used, or surveys are made less frequently.

6.5 Mark-recapture and photo-identification

To identify individuals in a population is a powerful tool to assess a number of different research aims (Wells, 2002):

1. Finding site-fidelity patterns and the range of individuals or groups can be used to assess the habitat use of the species observed.
2. Observing the behaviour of individuals in groups may lead to a better understanding of the group composition. Gathering information about behaviour as well as other individually differing facts like gender, reproductive condition, age and genetic relationships is needed to assess the reproductive cycle and life history patterns of animals, thus giving information about the population state if the mortality can be evaluated as well. To approach these issues focal animal behavioural observations are needed (Altman, 1974)
3. Mark-recapture techniques can be used to calculate population abundances under certain assumptions about geographical and demographical closure, permanence of markings and equal catchability (Hammond, 1986, Hammond et al. 1990).
4. Resighting of animals in other areas may provide valuable information about migration routes or distribution shifts.

Methodology

Identification of animals can be done using natural markings (e.g. Hammond et al. 1986, Yochem et al. 1990) or by applying temporary (Wilson et al. 1999, Erickson et al. 1993) and permanent markings (Irvine et al. 1982, Erickson et al. 1993) or by attaching tags (Scott et al. 1990, Irvine and Scott, 1984, Wright et al. 1998). Considerable work has also been done on 'voiceprinting' to identify individuals (e.g. Clark, 1989) and on biopsy sampling to receive genetic fingerprints (Jeffrey et al. 1985). Using of the above mentioned techniques was in most cases intended to gain knowledge about the habitat use and population parameters. Some examples for photo-id as well as marking are given in Figure 6-21.

Today the most commonly used technique is photo-identification for mark-recapture studies. Especially for the Cardigan Bay and the Moray Firth extensive work with photo-identification has been done on bottlenose dolphins (Evans and Hammond 2004, Baines et al. 2002). For both study areas good estimates for the population size can be given.

Photo-identification is based on natural markings of animals. Humpback whales (*Megaptera Novaeangliae*) for instance have very distinct patterns on the ventral side of their tail flukes (Evans and Hammond 2004, Wells 2002). Bottlenose dolphins (Evans and Hammond 2004, Würsig and Würsig, 1977) can be easily identified photographing the dorsal fin with its usual notches on the trailing side. Other species where photo-identification has been used successfully are minke whales (*Balaenoptera acutorostrata*, Tschertter and Morris 2007), Long-finned pilot whales (*Globicephala melas*, Ottensmeyer and Whitehead, 2003), fin or finback whale (*Balaenoptera physalus*, Mussi et al. 1999) and many others.

Using photo-identification on short-beaked common dolphins (*Dephinus deplhis*) might be possible in some regions, but may be very difficult in others (Stockin and Vella, 2004) and seems to be impossible for the difficult to photograph harbour porpoises.

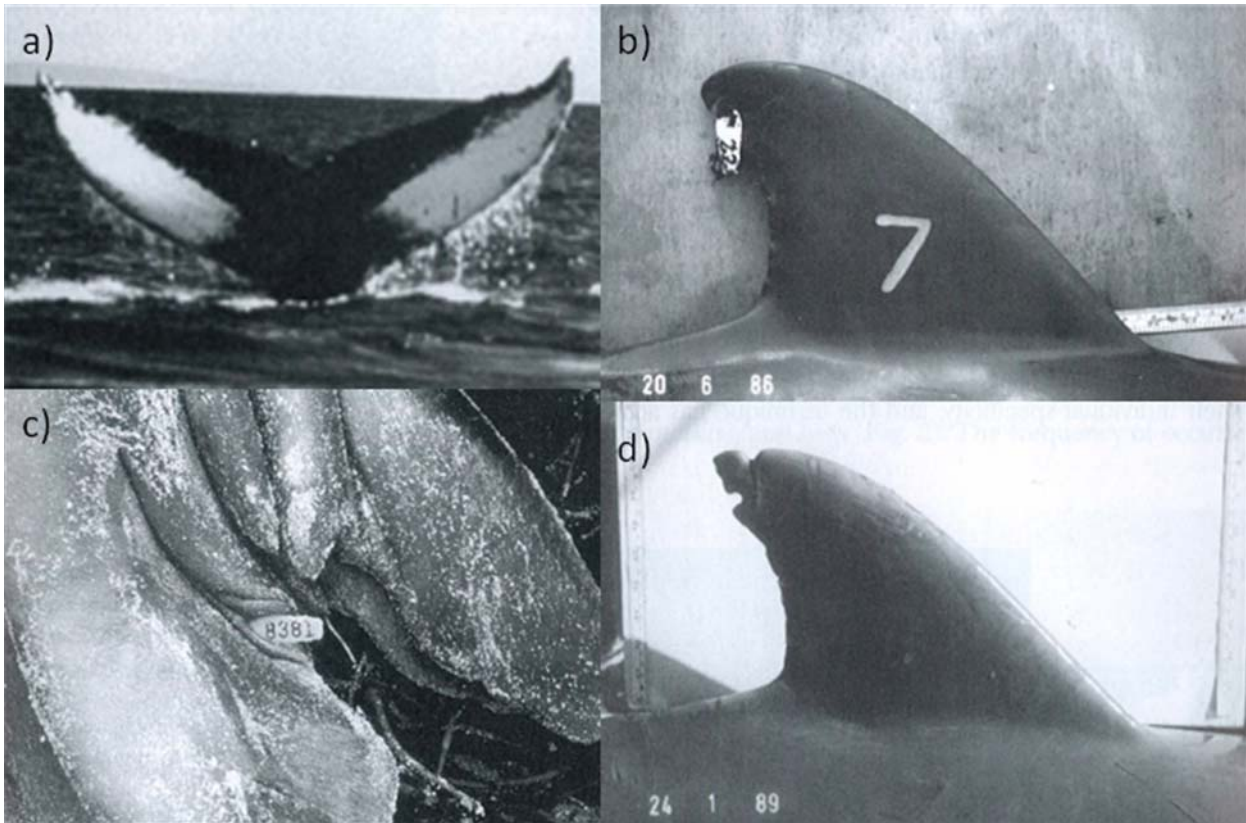


Figure 6-21: Examples for natural and anthropogenic marking from Wells (2002), a) humpback whale's fluke with distinctive natural patterns, b) bottlenose dolphin's dorsal fin tagged with a one year old marker and a freeze brand, c) northern elephant seal's flipper equipped with a tag, d) same bottlenose dolphin as under b) with a naturally acquired notch on top of the dorsal fin, a notch received by loss of the tag, and the freeze brand.

A very big advantage of photo-id over similar marking studies is that the animal must not be caught nor hurt by tagging it (Evans and Hammond, 2004). Natural marks are also more stable than tags, as they cannot be lost or stripped off, but tend to change slightly over time (Evans and Hammond, 2004). As long-term recognition is essential for mark-recapture trials, it is necessary to choose features carefully being aware that additional marks may occur in the lifetime of an individual (Würsig and Jefferson, 1990).

Photographs are usually taken with fast shuttering reflex-cameras (SLR) either equipped with high-speed slide or negative films for bigger animals, low-speed films for small animals as a good resolution of the picture is needed to identify the features (Würsig and Jefferson, 1990). High resolution digital cameras will substitute film cameras in the near future completely. All pictures should be made from a perpendicular angle to the photographed feature and both sides of the animal should be recorded. A mounting of the cameras and usage of telephoto lenses is recommended for most species (Würsig and Jefferson, 1990). The later analysis is done with the help of pattern-recognition or shape-recognition software after all usable picture are rated for their quality and catalogued.

Methodological variation of the data

If the analysis is based on only two sampling occasions, then a closed population model must be used to estimate a population size, assuming that the population does not change over time, thus experiencing no change due to births, deaths, immigration and emigration (Evans and Hammond, 2004). If more sampling occasions are available other open population models can be used (Evans and Hammond, 2004, Hammond, 1986).

A large influence on the processed data stems from photographs with different grades of quality – with decreasing quality the number of animals falsely classified as new individuals will increase, as markings may not be visible or identifiable anymore (Evans and Hammond, 2004). Markings will actually change over time (McCann, 1974), but this might not change the ability to match a new picture to a previously photographed individual (Evans and Hammond, 2004).

There will be a bias if not all animals of a population have highly detectable features and thus cannot be matched. But the percentage of well marked individuals can be assessed by using the number of identifiable individuals in a group or school to predict the percentage of identifiable animals for the entire population (Wilson et al. 1999, Evans and Hammond 2004).

Another assumption is that a marked individual must be recognised again if it is available for detection (Evans and Hammond, 2004). A major influence might come from the representativeness of the used sample for the whole population (Evans and Hammond, 2004). If animals occur in stable groups and these groups do not intermix then an estimate will be biased through the fact, that on the next sampling occasion the same or another group will be sampled, but rematches are either complete or none. This can be assessed by employing the methods for a large number of occasions.

Biological variation, detection of short and long-term effects on cetaceans

For small and resident populations like for instance the Tursiops populations in the Moray Firth (Evans and Hammond, 2004) or in the Cardigan Bay (Baines et al. 2002) good population estimates can be derived using photographic methods. This is of special importance, as both areas are partly Special Protected Areas (SACs) and thus bottlenose dolphins have to be protected under the habitat directive. The already available data is valuable and can be used to appoint future frames for environmental impact assessment outside the SACs.

It is very difficult or maybe not possible to assess short-term effects using mark-recapture procedures, as a statistical approach is not available yet. Nevertheless, if it is observed, that animals flee the wind farm site during construction, it is strong evidence. Identification of the observed subject can lead to conclusions about intra species variation of behavioural reactions.

For long-term studies it is necessary to provide a large dataset with a number of years recorded before and after the wind farm construction.

Requirements for photo id

For a number of species photo-id catalogues have been established over the last decades. For these species a mark-recapture analysis may be feasible. For all other species, or where photo-catalogues are not open to the public, it is not manageable to perform photo-id studies over the time frame of base-line, building, working and decommissioning phase of a wind farm.

Nevertheless, it would be a good idea to fund this basic research as a fundament for future strategies to where wind farms should be build.

Some species occurring on the coasts of the United Kingdom cannot be easily assessed using other methods. These would be the fin whales, long-finned pilot whales and killer whales for the cetaceans and grey seals for pinnipeds. If there are no other solutions available, then at least the long-term effect should be monitored through photo-id studies.

6.6 Conclusions

Table 6-7 summarises the main relevant characteristics of the methods described in chapter 6. This leads to the following conclusions towards the applicability of the methods for the scope of marine mammal monitoring in relation to offshore windfarming:

Aerial surveys:

Aerial surveys are the only method allowing multi-species surveys sufficiently large to cover impact and reference areas during the construction phase of a windfarm. Aerial surveys also have the highest probability in obtaining sufficient data to quantify species abundances as sightings rates are higher than in other methods. Data from different surveys and areas are highly comparable. A disadvantage of aerial surveys is the low spatial resolution, and the data may not allow to detect small scale impacts. For example, sightings rates within the windfarm area are usually too low to draw conclusions, and short-termed effects may be missed due to restriction from weather conditions. Aerial surveys are strongly restricted to suitable weather conditions and in many areas may not be reliably conducted at a frequency of more than one survey per month. Aerial surveys have also been proven to be a cost effective method (Hammond 2006). In German offshore windfarm studies aerial surveys is the most commonly used method for baseline and impact studies (BSH 2006).

Ship surveys and towed hydrophones:

Ship surveys for visual observations are even more restricted by weather conditions than aerial surveys and only allow to cover much smaller areas. Although ship surveys are a standard method for marine mammal studies, their applicability for offshore windfarm studies is strongly restricted, and before the start of a project it should be assessed, whether species abundance and weather conditions of the area may allow sufficiently high sighting rates.

The efficiency of ship surveys may be highly improved by towed hydrophones to record porpoises and dolphins. This may enable to conduct ship surveys at night and during unfavourable weather conditions and thus greatly increase the potential to survey relevant areas in relevant times. Costs of ship surveys are usually much higher than aerial surveys (see Hammond 2006). However, as towed hydrophones may enable the observers to use smaller ships, costs may be reduced.

Ship surveys may also be used to record the behaviour of marine mammals in relation to certain activities at the windfarm sites, e.g. pile driving, however, as behavioural observations are only possible at excellent weather conditions, this is not concluded to be a reliable method.

Static Acoustic Monitoring (T-PODs)

Static Acoustic Monitoring (SAM) using T-PODs or other devices has proven to be an efficient method to collect data on harbour porpoises and bottlenose dolphins with a high temporal resolution. At present, this appears to be the best method to detect short-term effects in relation to pile driving. SAM has also proven to be efficient during the monitoring in the operational phase to detect long-term effects and has thus been the most important method for studies in the Danish offshore windfarms. The low spatial resolution can be compensated by using a higher number of these devices.

Static Acoustic Monitoring devices are less susceptible to harsh weather conditions, although recordings may suffer from disturbance through wave action and increased background noise.

A disadvantage of the method is the fact, that it does not yet provide abundance data. The applicability of SAM may be restricted in some areas through fishing activities, which may lead to losses of equipment or greatly increase the effort for anchoring systems (see Brasseur et al. 2004).

Telemetry and Dataloggers

The use of telemetry and data loggers to track movements and behaviour of marine mammals might provide supplementary data to line transect surveys and SAM data. In areas, where it is possible to catch marine mammals and equip them with these devices, highly useful data may be obtained. However, as these data cannot be reliably related to specific projects, this method is recommended for studies on a higher level and not as a standard for single projects. Studies

using telemetry and dataloggers can provide the necessary basic knowledge about preferred habitats as well as seasonal migrations to assist environmental impact assessments in their dedication and to find cost effective methods to assess the needed environmental aspects.

Haul-out site counting

Counting seals on their haul-out sites by aerial surveys or other methods is highly efficient to obtain reliable data on seal abundance and abundance changes. A problem in relation to the assessment of offshore windfarms is that haul-out sites will only occasionally be found in the vicinity. There will therefore only be a few occasions where haul—out site counting can provide useful data on a project specific point of view. From a cross-project point of view valuable information on population level and maybe cumulative effects of offshore windfarms can be obtained by this method.

Mark-recapture and photo-identification

These are special and cost-intensive methods, which could provide very valuable information on the individual responses of animals to offshore windfarms. Thus, these methods are recommended to be applied in cross-specific studies.

Table 6-7: Summary of all presented methods for monitoring marine mammals (na = data not available).

	Line Transect			Point Transect		Tracking				Other	
	Aerial Surveys	Ship Surveys		Static Monitoring		Telemetry				Haul-out site	Photo ID
methodology	aircraft	visual	acoustic	visual (platform)	acoustic	Satellite	Satellite + GPS	Satellite + Datalogger + GPS	VHF	Haul-out site	Photo ID
platform	high winged, double engine aircraft (e.g. Partenavia P68)	ship (minimum 30m long, observer height >5m)	ship (also smaller ships)	observer height > 30m (platform/coast/...)	T-POD / PCL						
manpower											
data collection	3 observer	4 observer	1 observer	1-2 observer	1 technician + 1 assistant per maintenance a 1-2 days	well-rehearsed team for telemetry	well-rehearsed team for telemetry	well-rehearsed team for telemetry	well-rehearsed team for telemetry	2 observer	1-2 observer, 1 photographer/technician
data analysis	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist	1 scientist
dependency on weather and daylight	seastate 0-3, only during day time	seastate 0-2, only during day time	seastate 0-5 independent of daylight	seastate 0-2, only during day time	seastate 5-7, dependent to substrat & shiptrafic, independent of daylight	low	low	low	low	seastate 0-5, low tide	seastate 0-2 for small ceateacan
spatial scale	aircraft	visual	acoustic	visual (platform)	acoustic	Satellite	Satellite + GPS	Satellite + Datalogger + GPS	VHF	Haul-out site	Photo ID
unit	density/animals per transect	density/animals per transect	animals per transect (density)	animals per time	presence/absence ("detection positive time")						
coverage	high	moderate	moderate-high	low	very low	complete homerange	complete homerange	complete homerange	up to 50 km	high	na
	500-1000 transect kilometres per day	80-120 transect kilometres per day	160-240 transect kilometres per day	sight radius (~1000m)	effective detection range <400m per device	without limitation	without limitation	without limitation	dependent on antenna and transmitter range	from a population point of view	-
resolution	low	very low	very low	very low	very low	very low	high	high	high	high	na
	(>2 km, dependent on transect length and interval)	(>2 km, dependent on transect length and interval)	(>2 km, dependent on transect length and interval)	dependent on no. of observer/platforms	dependent on no. of devices in an specific area	Argos-positions				only for resting animals	
temporal scale	aircraft	visual	acoustic	visual (platform)	acoustic	Satellite	Satellite + GPS	Satellite + Datalogger + GPS	VHF	Haul-out site	Photo ID
coverage	low	low	low	low-moderate	100%	high	high	high	moderate	low	high
	= observation time/ total time of the year	= observation time/ total time of the year	= observation time/ total time of the year	= observation time/ total time of the year	= observation time/ total time of the year	limited by batterylife	limited by batterylife	limited by batterylife	limited by batterylife	= observation time/ total time of the year	whole lifetime
resolution	low	very low	low	very low	very high	moderate	moderate	high	high	low	very low
between single surveys (>1day)	time between two surveys (>1day)	time between two surveys (>1day)	time between two surveys (>0.5 day)	time between two surveys (>1day)	time between consecutive clicks = μs ; time between encounters > 10min	limited by surface rate and sattelite coverage	limited by surface rate and sattelite coverage	limited by surface rate and sattelite coverage	limited by surface rate	limited by no. of surveys	na
within single surveys	na	low-moderate	low-moderate	high	high	na	na	na	na	high	na
explanatory power on population level	very high	moderate	moderate	very low	low-moderate (dependent to no. of devices)	dependent on no. of animals und lenght of recorded time (normally low)	dependent on no. of animals und lenght of recorded time (normally low)	dependent on no. of animals und lenght of recorded time (normally low)	dependent on no. of animals und lenght of recorded time (normally low)	very high	moderate-high
behaviour	aircraft	visual	acoustic	visual (platform)	acoustic	Satellite	Satellite + GPS	Satellite + Datalogger + GPS	VHF	Haul-out site	Photo ID
classifiable behaviour	swimming (without and with direction), diving, resting at the surface, fast swimming at the surface (flying)	swimming (without and with direction), diving, resting at the surface, fast swimming at the surface (flying)	maybe prey catching	swimming (without and with direction), diving, resting at the surface, fast swimming at the surface (flying), spy hopping	feeding, travel orientation, social behaviour, (sea bed orientation)	nothing	nothing	dive duration, dive depth, resting behaviour, pitch- and rol angle, swim speed, dive speed, echolocation	nothing	nothing	swimming (without and with direction), diving, resting at the surface, fast swimming at the surface (flying), spy hopping
Quality of behavioural observations	low	moderate	future research needed	high	moderate-high	na	na	very high	na	na	high

7 Recommendations for monitoring

On the basis of the previous chapters we formulate recommendations for monitoring strategies for the most common marine mammal species in UK waters taking practical considerations, statistically robustness and costs into account. The chapter is divided into the four different periods baseline, construction, operation and decommissioning of offshore windfarms.

7.1 Baseline

In any area, at first an overview on species abundance must be obtained either from available data or from dedicated surveys. Visual line-transect surveys using either planes (a higher spatial coverage can be obtained!) or ships as observer platforms are thus a standard requirement for a baseline study, unless sufficient data are already available. As the harbour porpoise is a species occurring regularly in all UK waters the use of SAM (T-PODs) is recommended for collecting baseline data in a very high temporal resolution and independent from weather conditions.

In order to describe the seasonal abundance of marine mammals in an area of interest, monthly surveys over one or two years are recommended. Accordingly one or two years of deployment of SAM devices is recommended. This is usually considered as sufficient for a description of species occurrences. Based on these data, decisions on more detailed investigations can be made, depending on species composition and abundance in the area of interest. Baseline data should also be used to assess the statistical power of the data in order to decide on the effort needed for the coming construction and operation period. If harbour porpoises or dolphin species occur in significant numbers in an area of interest, decisions on the use of SAM and other methods should be made during the baseline study, and a BACI design should be set up by deploying these devices in impact and reference areas (at a distance <2 km to the impact area for comparisons within the operational period, and at a distance of 20-30 km to the impact area for comparisons with construction period, see below).

It is possible to use a reference area's data as the baseline dataset when a former reference area will turn into an impact area after the complete BACI-design of the first assessment is finished. This still calls for another reference area, where the baseline data for the second assessment will be gathered.

7.2 Construction period

7.2.1 Harbour porpoise

Aerial surveys and possibly ship surveys are likely to provide sufficient data on abundance and large-scaled and long-term effects, but not on short-termed responses to pile driving. At present, only one study on pile driving impacts is available. There is a strong need to cover both short-term and long-term effects on a large scale and it is recommended, not only to rely on line-transect surveys but also to use Static Acoustic Monitoring devices to study porpoise abundance and behaviour.

Thus the following study design is recommended:

Aerial surveys should cover a large area (2000 km²) and need to be conducted at least on a monthly base during times when abundance are high enough to obtain sufficient data. If knowledge on seasonal patterns in porpoise abundance is sufficient, the surveys may be restricted to relevant times of the year.

Static Acoustic Monitoring devices should be deployed on several locations with increasing distance to the windfarm sites. Available data indicate that distance of reference areas should be at least 20 to 30 km to the impact area, but decisions should be made based on expected ranges of noise emissions. It is recommended to deploy SAM devices at 5 to 10 locations in the windfarm and 5 to 10 locations outside the windfarm. A power analysis should be conducted

after the first year of baseline acquisition to take account of necessary in- or decreases in the number of measuring sites in relation to the occurrence of harbour porpoises in that specific area.

The duration of a study should be about 5 years: 2 years baseline, 1 year construction and 2 years post-construction. If longer lasting effects are detected, the post-construction period should be extended.

7.2.2 Bottlenose dolphin and Common dolphin

In areas where these species occur, SAM devices in addition to line-transect surveys should be deployed in order to obtain data with high temporal resolution. As changes in abundance will in most cases be difficult to obtain due to low abundance, any information on species responses are considered to be relevant. Especially for bottlenose dolphins already existing photo-id catalogues are of enormous value, as a lot of species and population specific parameters can be deducted from these. If possible, ongoing studies should be taken into account and funded to continue their work.

7.2.3 Minke whale

Minke whale abundance will in almost all locations be too low for systematic surveys, however, aerial surveys and to a lesser extent ship surveys may provide some data. Photo-id catalogues as well as satellite telemetry may provide additional information about this species.

7.2.4 Common seal/Grey seal

Although both species are considered to occur in relevant numbers in many UK offshore windfarm sites, it will be difficult to monitor and assess their responses to offshore construction work. Visual surveys will obtain some data. However, these are not expected to be sufficient for statistical analysis as sighting rates are too low. If haul-out sites are located close to planned offshore windfarm areas, the monitoring data of seals at these haul-out sites derived from aerial haul-out-countings should be considered. Unfortunately, no other methods can be recommended for investigations on a project level. On a higher level, case studies using telemetry or other methods may provide general data on seal responses to offshore windfarm constructions.

7.3 Operation period

7.3.1 Harbour porpoise

Static acoustic monitoring is considered to be the most efficient method to detect long-lasting effects on a small spatial scale. Sighting rates from line transect surveys are considered to be too low to detect changes in abundance within a windfarm area. For the operation phase it is thus recommended to deploy SAM devices within the windfarm if sufficient baseline data are available, or to add some locations in the close vicinity (< 2 km), which might serve as reference areas.

The duration of a study should be at least three years of full operation of the windfarm, however, a longer period is recommended as it is yet unknown, whether habituation and long-term changes in the fauna colonising the underwater constructions may lead to changing porpoise abundance in the windfarms.

7.3.2 Bottlenose dolphin and Common dolphin

In areas where these species occur, SAM devices should be deployed in order to obtain data with high temporal resolution. As changes in abundance will in most cases be difficult to obtain due to low abundance, any information on species responses are considered to be relevant, and

decisions on additional studies should be based on the specific situation of the project. Especially for bottlenose dolphins already existing photo-id catalogues are of enormous value, as a lot of species and population specific parameters can be deducted from these. If possible, ongoing studies should be taken into account and be funded to continue their work.

7.3.3 Minke whale

Minke whale abundance will in almost all locations be too low for systematic surveys, however, aerial surveys ship surveys may provide some data. Photo-id catalogues as well as satellite telemetry may provide additional information about these species.

7.3.4 Common seal/Grey seal

Visual surveys will obtain some data. However, these are not expected to be sufficient for statistical analysis as sighting rates are too low, and therefore they will only provide some information on whether seals use the windfarm area or not. If haul-out sites are located close to planned offshore windfarm areas, the monitoring data of seals at these haul-out sites deriving from aerial haul-out-countings should be considered. Unfortunately, no other methods can be recommended for investigations on a project level. On a higher level, case studies using telemetry or other methods may provide general data on seal responses to offshore windfarm constructions.

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