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A methodological update of the Framework for the Appropriate Assessment of the ecological effects of Offshore Windfarms at the Dutch Continental Shelf

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dr. A.R. Boon
S. Dirksen (Bureau Waardenburg)
M.F. Leopold (IMARES)
A. Brenninkmeier (Altenburg & Wymenga)

1205107-000

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Client	**Project**	**Reference**	**Pages**
Rijkswaterstaat Dienst Noordzee	1205107-000	1205107-000-ZKS-0018	38

Keywords
Appropriate Assessments, Offshore Windfarms

Summary
This report gives an update on the methodology for assessing ecological effects of the construction and operation of offshore windfarms at the Dutch Continental Shelf. It updates the report by Prins et al. (2008), but it can not be viewed as its replacement. For proper guidance of setting up appropriate assessments for offshore windfarms, both documents should be used.

Version	Date	Author	Initials	Review	Initials	Approval	Initials
	jul. 2012	dr. A.R. Boon		dr. T.C. Prins		T. Schilperoort	

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final

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1 Introduction

1.1 Background

In 2008, Deltares wrote a report called ‘Development of a framework for Appropriate Assessments of Dutch offshore wind farms’ (Prins et al. 2008) as a basis or guideline for the wind farm-specific Appropriate Assessments for nineteen initiatives for offshore wind farms (OWFs) on the Dutch continental shelf (DCS). This report is further referred to as the "Framework". This Framework was an important step in the process of coming to a standardised Environmental Impact Assessment (EIA) and Appropriate Assessment (AA).

Below, a short history is given of the process of environmental impact assessment of OWFs in the Netherlands:

- Two first Dutch OWFs were build in 2006 and 2007, “Offshore Windpark Egmond aan Zee” (*OWEZ*) and Q7, later on called “Princess Amalia Wind Park” (*PAWP*). These were called the first-round OWFs. EIAs, but no AAs were written for these OWFs. A monitoring and research plan was set up and carried out on the environmental effects of the construction and operation of these OWFs, of which the results so far were compiled in Lindeboom et al. (2011).
- For the so-called second sound OWFs, the licensing process for 19 windfarm initiatives for was started, for which EIAs and AAs were written. The process of writing the EIAs was complex and long and led to the conclusion that significant negative effects on natural values protected in N2000 areas could not be excluded. Therefore, AAs were needed. The unsatisfactory EIA process led to the overall remark from both the industry and the governmental bodies that it would be sensible to standardise the methodology for writing AAs. This methodology was described in the 2008 Framework (short for *Development of a framework for Appropriate Assessments of Dutch offshore wind farms*, Prins et al 2008).
- After completion of the second-round AAs (e.g. Arends et al. 2008), it became clear that on many aspects of the possible environmental impacts, knowledge was lacking to come to a well-founded conclusion. It was decided that a research plan was needed to solve major knowledge gaps, which led to the so-called Masterplan (Boon et al. 2012).
- The Masterplan led to the set up and execution of the “Shortlist” monitoring and research plan, which was carried out in 2010 and 2011. This research led to filling in some of the major knowledge gaps, but in most cases it was also acknowledged that this Shortlist program was an important but first step (Lindeboom et al. 2011).
- The results of the Shortlist monitoring and research were reviewed and knowledge advancements and possible follow-up was described in Boon (2012a). Based on these results, a follow was given to the Shortlist program, called VUM (Vervolg Uitvoering Masterplan, Follow up Execution Masterplan) or Shortlist II. This VUM is currently being carried out.
- The results of the Shortlist program also led to the current report and another report which is being written concurrently (Boon 2012b). The latter report describes the relevance of the Shortlist results for the inclusion of preventive measures for any environmental effects

in the future licenses for OWFs, and the possibilities for improving the spatial planning possibilities for OWFs at the Dutch Continental Shelf (DCS).
- The current report is also a spin off of the Shortlist results: it describes the improvements in the methods for environmental impact assessments done in the 2008 Framework (Prins et al. 2008) made possible by the Shortlist results. It is called the Update to the 2008 Framework report.

To facilitate the use of this increased knowledge base, Rijkswaterstaat asked for an update of the 2008 Appropriate Assessment Framework (Prins et al. 2008). The results from the shortlist research, and from the relevant and publicly available studies carried out in recent years at the two already existing offshore windfarms *Offshore Windpark Egmond aan Zee* (*OWEZ*) and *Prinses Amalia Windpark* (*PAWP*), were used to update the Framework. Adding to the results of these studies, results from relevant international studies were used to compare to the Dutch results.

Earlier versions of this report were read and commented on by the commissioner of this report, specifically M. Graafland, J. Bakker, I. Klein-Hendriks (Rijkswaterstaat, Ministry of Infrastructure and Environment) and R. Dekeling, H. Merkus, (DGRW, Ministry of Infrastructure and Environment) and T. Verboom (Ministry of Economic affairs, Agriculture and Innovation). A second version was read and commented on by the former people, and additionally by V. Gales (Ministry of Defence), W. Zevenboom, N. Kinneging, P. Westerbeek, M. Roos (Rijkswaterstaat, Ministry of Infrastructure and Environment), R. Dekeling (DG Water, Ministry of Infrastructure and Environment), H. Boomsma, S. van Sluis (Ministry of Economic Affairs, Agriculture and Innovation) and S. van den Akker (North Sea Foundation). Their contributions significantly added to the structure, readability and quality of this report.

1.2 Delimitation of this update

It is important to note that this update report is not a stand-alone product. It should be viewed as an addendum to the existing 2008 Framework for Appropriate Assessments by Prins et al. (2008). Both the 2008 Framework and this update report should be viewed as the basic guidelines for any future AAs for OWFs. It should be stressed that any knowledge advancements on possible environmental effects of OWFs made after completion of this update report should be taken into account when writing new AAs. Also, other approaches and interpretations of methodologies or dose-response relationships are not excluded from the process. On the contrary, when the arguments are well described and valid, they are likely an improvement on what is proposed in this report.
Another important issue is that this report discusses the methods to assess effects, not to judge effects. The difference is subtle but essential. In this report there is no reference to valuating the extent of the effects, only to the extent itself.

This report only contains the methodological update of the tools described in the earlier Framework from 2008 (Prins et al. 2008). What this report does not do is reiterating the calculations that were made in the original Framework and repeating the judicial context. Applying the earlier Framework in actual AAs made it clear that such calculations are only

relevant in OWF plans that are quite specific in location and configuration. Generic calculations, such as have been done in the 2008 Framework are therefore not repeated.
The methodological update of the Framework is a logical first step in updating the 'tools' needed to describe the possible ecological effects of offshore wind farms with the best available knowledge. It contains the specified, updated description of the input data, calculation models and the generic knowledge behind or motivation of the models. As an adaptation to the original Framework, the Construction Phase and Operational Phase will be described separately, because the piling activities during construction are a major source of possible effects, which do not manifest during the operational phase.

1.3 Set up of the Framework update

The update is subdivided in three main parts. The first part discusses the generic knowledge advancements on the distribution of fish larvae, marine mammals and birds that were the result of surveys conducted in the shortlist context. The second part treats the update regarding the possible ecological effects of the construction phase, i.e. piling the foundation for OWFs. Although other foundation constructions are possible, monopiles still are the main foundations for the Dutch OWFs as planned. The ecological effects of the construction phase are mainly related to the effects of underwater noise. Underwater noise may affect fish larvae, fish and marine mammals to various degrees. The third part deals with the potential ecological effects of the operation phase of OWFs. During the operation phase, the most important effects are the assumed collisions of birds from breeding colonies and during the migration season in spring and autumn.

Per effect, a description will be given of what the methodology was in the 2008 Framework, what has been studied and monitored so far in the so-called shortlist studies and the monitoring programs for the existing OWFs *OWEZ* and *PAWP*. Next, the results from these studies and those available from international studies will be shortly presented and a discussion will be given of the adaptations to the methodology of the Framework.

For this report, a choice has been made to keep the presentation short and concise. Regarding the effects of underwater noise on fish and marine mammals only concise information has been added to this report, which has been added with information on international results for as far this was not already done in the background document itself. Regarding the effects on birds, the authors of the original background document (Van Belle et al. 2008) for the 2008 Framework have written an updated text, which has been fully integrated into this document.

This update report was partly written and edited by A.R. Boon. He also wrote the chapter on the mortality effects of noise on fish larvae, based on the report that was written by Bolle et al. (2011) on their experimental work on this subject. L. Bolle did a quality control on this chapter. Written contributions on the other subjects were made by the researchers that were also involved in the 2008 Framework and the studies of the shortlist and *OWEZ* and *PAWP* wind farms. The paragraphs on birds in chapter 5 was largely written by S. Dirksen (Bureau Waardenburg), M.F. Leopold (IMARES) and A. Brenninkmeijer (Altenburg & Wymenga) and edited by A.R. Boon to fit the report format. The report paragraphs on the effects of underwater noise on marine mammals were written by A.R. Boon.

2 Current status on knowledge of distribution of fish larvae, marine mammals and birds

As part of the shortlist research, three studies have been conducted into the spatio-temporal distribution of relevant potentially impacted species: fish larvae, marine mammals and birds. In this chapter, the results from these three studies will be described, as well as the implications for the update of the methods for assessing potential OWF effects.

2.1 Fish larvae

2.1.1 2008 Framework

In the 2008 Framework, data on the distribution and transport of fish larvae were used from a description in a background report (Ter Hofstede et al. 2008). In essence, generic knowledge is available on the spawning grounds, overall transport routes and nursery areas. This knowledge mainly originates from four types of surveys that were conducted over the last decades since the 1960s. However, the spatial and temporal resolution of these surveys is directed towards sustaining stock assessments, and not towards an understanding of fish biology and ecology.

In the 2008 Framework, this knowledge was used as input to a modelling exercise for mortality of three species of fish larvae due to piling of OWFs: herring, plaice and sole. Sole was not part of the study in the background report (Ter Hofstede et al. 2008). The model itself and the input parameters have been described more extensively in Bolle et al. (2005).
Spawning areas were assigned to specific locations in the modelled Southern Bight, spawning was assumed to occur throughout the first half of the year, but with different periods for the different species. Transport was species specific, but also dependent on the developmental phase of the larvae. Transport of the larvae was therefore mostly hydrodynamically forced, but mediated by tidal-specific vertical migration of the larvae.

A clear lack of knowledge (see Boon et al. 2010) was a higher resolution of the spatial distribution of fish larvae, especially in the planned OWF areas and the intra-annual variability in concentration in the southern North Sea. A survey to fill this gap was conducted as part of the Shortlist work in 2010 and 2011, of which the results are discussed below.

2.1.2 Results shortlist survey

Monthly ichthyoplankton surveys elucidated the distribution of fish eggs and larvae in the southern North Sea between April (2010) and March (2011) (Van Damme et al. 2011). Highest densities of species important as staple food (clupeids, sand eels, flatfish, gadoids) occur in the first half of the year particularly in the coastal areas. Overall, the results are in line with what was known from earlier surveys, but they added much more detail in spatial resolution and temporal variability and trends throughout the year.

Species-specific results

Herring larvae were found in high numbers on our coast from February to April, especially in their early stage, but metamorphosing larvae only in low numbers, and mostly north of the Wadden Sea islands. Plaice eggs and larvae were found in high numbers in the eastern Southern Bight, from January to March mostly. Metamorphosing larvae were found mostly north and northwest of the Wadden islands in April. Sole was found in the whole of the Southern Bight from April to July, but somewhat confined to coastal areas.

2.1.3 Updated framework methodology

Already before the survey in 2010 and 2011, it was known that the study would not be able to pinpoint with more certainty the spawning areas of the fish species. So, no additional knowledge on the exact locations of the spawning areas has become available. What we do know better, is the spatio-temporal variability of fish eggs and larvae, the speed of development of the eggs to larvae and the various stages of the larvae. Such data could be used to improve the description of the behaviour of the larvae and to calibrate the larval transport models, which have been used in the 2008 Framework and the AAs. This means that the current model can perform better regarding the simulation of larval dispersal and transport towards N2000 areas. As a result, the proportion of larvae transported into N2000 areas can be better modelled. Using the enhanced data on mortality due to piling, a better estimate of the decrease of larval and juvenile fish transport into the N2000 areas can be modelled. Knowledge on the level of predation of juvenile fish by birds and mammals has not been improved; the calculations needed to estimate the trophic effect of the reduced availability of juvenile fish is still best described in the AAs for the second round OWFs (e.g. paragraph 6.2 in Arends et al. 2008).
In conclusion, the quality of the results of the modelling exercises will be improved when the models are rerun with the new data on mortality (for sole, see paragraph 3.3.2) and when the larval distribution of the model is calibrated with the data on larval distribution from the surveys. Additional surveys will further improve this model.

2.2 Marine mammals

In the 2008 Framework four species of marine mammals have been taken into account with regard to the possible effects of piling and presence of OWFs in the Dutch North Sea: harbour seal, grey seal, bottlenose dolphin and harbour porpoise. In the second round AAs, the potential effects of the construction and operation of OWFs have been estimated for the seals and the porpoise only. The other dolphin and whale species (such as the bottlenose dolphin) can be regular visitors in our waters, their presence and density is not considered to be such that the potential harm of the construction and presence of OWFs will negatively affect their populations. The discussion below will therefore only refer to the two seal species and the harbour porpoise.

2.2.1 2008 Framework

Harbour seal and grey seal

In paragraph 7.2 of the 2008 Framework, descriptions are given of the distribution and numbers of these seal species on the Dutch Continental Shelf. In the 2008 Framework, no specific densities of seals in coastal waters were mentioned; a figure was adopted from Brasseur et al. (2008, the background report) that showed the relative densities from low to high in our coastal waters, but there were no absolute densities mentioned. In the second round AAs, a figure from Lindeboom et al. (2005) has been used, depicting the chance of a harbour seal being present in the coastal waters. Combining data from different reports gave the estimated numbers of seals being present in a given area being influenced by underwater noise.
In an additional note to the EIA commission, Boon & Heinis (2009) used new data on the distribution and densities of harbour seals and grey seals for the calculation of possible effects of piling. The basis for this calculation was the data presented in an interim report by Brasseur et al. in 2008 on the presence of seals in relation to the OWF *OWEZ*. These

densities were calculated with a modelling exercise (for habitat preferences based on tagged seals) in combination with survey data. In this note, methods were given on how to calculate the numbers of seals affected by underwater noise from piling.

Regarding the grey seal, data about distribution are even rarer than those for the harbour seal. In the 2008 Framework, a distribution map based on two tagged grey seals was used from Brasseur et al. (2008).

Harbour porpoise

In the 2008 Framework, data on the distribution and densities of harbour porpoise are presented from the two SCANS studies (Reid et al. 2003), showing the now well-known southward shift in distribution of the harbour porpoise in the North Sea. This shift also appeared in the counts of the porpoise along the Dutch coast. In the 2008 Framework, no data on densities were presented. Other data on the distribution of porpoise in Dutch waters come from the spring MWTL (National Dutch Monitoring Program/Meetnet Waterstaatskundige Toestand des Lands) surveys and the assembled ship surveys (Brasseur et al. 2008). No reliable map on densities could be produced from these data. In the second round AAs, average densities for the Dutch part of the North Sea of 0.4 animals/km^2 were mentioned (Osinga et al. 2007). This does not apply to the higher densities seasonal or geographical densities.

Moreover, the general migration pattern of porpoises was assumed to be inshore in autumn and winter, and to be offshore in spring and summer. The existence of two sub-populations of porpoises in the southern North Sea is hypothesised, but there are no data to confirm this.

2.2.2 Results surveys

Harbour seal and grey seal

In the shortlist studies, no surveys were conducted specifically on seals. Seal surveys are being carried out under the umbrella of the MWTL monitoring program, which focus at counting the number of hauled-out seals and not at counting swimming seals in the coastal waters. Since 2008, numbers of harbour seals in the Wadden Sea area (including those in Germany and Denmark) have increased to an estimate of 32,000 individuals, of which over 6,000 individuals in the Dutch part of the Wadden Sea (Geelhoed & Polanen Petel 2011).

The results of the shortlist surveys on the harbour porpoise and birds do include observations on seals, which are presented in Geelhoed et al. (2011), Poot et al. (2011) and Van Bemmelen et al. (2011). The highest number of seals was observed during the aerial surveys (Poot et al. 2011). Most seals were observed at and close to their haul out sites, and also further offshore. In Poot et al. (2011), a comparison is made with the counts from the ship-based survey (Van Bemmelen et al. 2011) and the MWTL surveys. From these results, it appears that offshore most seals were observed to the north and northwest of the Wadden Sea and fewer seals were seen to the west of the Dutch coast. In general, this relative distribution was predicted by a study in which modelling of habitat preferences (such as distance to haul-out sites) was combined with the results from tagged seals (Brasseur et al. 2012).

Harbour porpoise

Aerial surveys specifically for porpoises were carried out in the Dutch part of the North Sea in July 2010, October/November 2010 and March 2011 and reported in Geelhoed et al. (2011). The data sampling during these surveys have a much higher spatial resolution than the usual MWTL data. On the basis of these data, a spatial model was set up to extrapolate the data to a contour grid, which describes the seasonal distribution and density data of harbour porpoises in the Dutch part of the North Sea. Densities were calculated and ranged for the DCS (see their table 4, overall densities) from 0.44 animals/km² in July, 0.51 animals/km² in October/November and 1.44 animals/km² in March. Locally and occasionally, densities can be higher, with maximum average densities of ca. 3 animals/km², found in March 2011. The highest density found at any location was 5.8 animals/km² (C.I. 95%). These relatively high densities were found in the area northwest of the Wadden Sea islands. The densities on the OWF areas (such as the area west of the Holland coast) were lower, on average 0.4 animals/km² in October/November 2010 to 1.1 animals/km² in March 2011.

When compared to the data from the two SCANS surveys, the average densities appear quite comparable. MWTL densities appear to be lower. Nevertheless, patterns in space and time are quite comparable. It should be noted here that the average monthly densities mentioned above show considerable interannual variation, in the order of two to threefold.

Additional information is provided about mother-calf pairs. These were occasionally observed in July around and west of the wind farm area west of the Holland coast.

2.2.3 Updated framework methodology

Harbour seal and grey seal

As a result of the shortlist surveys and the OWEZ results, an experimental relative density map of seals has been produced in Brasseur et al. (2012). This map can not yet be used for estimates of the assessment of effects of piling (e.g. when estimating proportions of the total amount of seals affected by piling); currently there are no reliable data on seal densities in Dutch coastal waters.

Harbour porpoise

Aerial surveys specifically for porpoises have delivered new data that should be used in future assessments of impacts of construction activities such as piling on harbour porpoises. Two pieces of information are relevant here: location-specific porpoise densities and the occasional presence of porpoise calves west of the Holland coast.
Densities of porpoises in the Dutch part of the North Sea vary considerably in time and space, but on average, they appear to be around the (corrected) values from earlier surveys: 0.44 to 0.51 animals/km^2. In March, average densities are higher, 1.44 animals/km^2, which corresponds well to the spring average of 1.12 animals/km^2 mentioned in Camphuysen & Siemensma (2011, referring Scheidat & Verdaat 2009) for the southern half of the DCS. In addition, local densities may deviate considerably from this average, with values up to 5.8 animals/km^2. However, this high value was observed in one occasion only. Impact assessments should consider such values, when planned work is short-term. For construction works such as piling for OWFs, which take place for ca. half a year, local values are more relevant (although currently, licenses exclude piling during the first half of the year). In addition, the interannual variation can be considerable, so observed numbers and the extrapolated densities should not be taken as the absolute truth for any given location or any given moment. A realistic approach should consider the average to higher range of the extrapolated averages for a specific location and its surroundings as found in Geelhoed et al. (2011), taking into account the values as they have been observed in other studies (see table 8 in Geelhoed et al. 2011) and considering the interannual and seasonal variation as described in the porpoise conservation plan (Camphuysen & Siemensma, 2011). The most likely effect scenario will be based on the average values, but an AA demands a worst-case approach and therefore insight in the more severe effects based on the higher range of values.

Next, during the latest survey, mother-calf pairs have been observed in July, and in areas that are relevant for the construction of OWFs. Although this is a first observation of calves in Dutch coastal waters, the relevance of our waters for porpoise calves should be taken into consideration in future impact assessment in combination with the assumed higher sensitivity of calves (or mother-calf pairs) for disturbance (due to the need for foraging). A few assumptions can be made about susceptibility to disturbance. Calves will have a lower swimming speed than the adults and a higher body surface to body mass ratio than the adults. On the other hand, they have a higher relative blubber mass (Read 1990). They have a good insulation to the cold waters of the North Sea, but they will need more time to escape an area with an uncomfortable underwater noise level. Such assumptions may lead to the conclusion that calves are more susceptible to disturbance of their feeding habits than adults are. However, as for adults, no concrete figures can be given about the effect of disturbance by underwater noise on the fitness of the animals.

2.3 Birds

2.3.1 2008 Framework

In the 2008 Framework, the discussion of possible negative impacts of the construction and operation of OWFs on birds focussed on methodology mostly, and less on bird ecology. One simple reason for this is the high number of species that are potentially affected. The estimated number of species being protected under the Bird and Habitat Directives in the UK and the Netherlands was almost 100, present at over 240 sites (Prins et al. 2008). The approach in the Framework therefore was different: a method of reduction for species and sites was given on a deterministic basis: if it is unlikely that a species will be flying over or remain in the vicinity of any (planned) OWF, then it is allowed to be excluded from the impact assessment. Next, nine bird species were discussed as example species. In a separate document (Troost 2008), the methodology of calculating the potential amount of collision victims was presented.

The nineteen AAs for the second-round OWFs roughly followed the same approach, with small modifications. A distinction was made between breeding colony birds, and migrating birds. Seabirds occurring locally and non-breeding colony seabirds (“floaters”) were discussed, but excluded from the actual impact assessment in the AAs for various reasons. Only species that face a chance of colliding with a turbine, such as foraging coastal breeding birds and migrating species were considered to be potential victims of operating wind farms. Earlier Environmental Impact Assessments showed that bird species that are present at sea, but not migrating or related to a (Natura 2000) breeding colony will not be affected at the population level.

Birds from breeding colonies were assumed to forage in a half to three-quarter circle seawards from their colony. Birds migrating were only considered when flying from the continent to the mainland of the UK and back. North to south moving birds were not considered, since they were not assumed to pass over any planned OWF area.

2.3.2 Results surveys for seabirds at sea

Two types of studies were carried out in the context of the Shortlist research. On the one hand, there were aerial and ship-based surveys for birds at sea, which have been reported in Poot et al. (2011) and Van Bemmelen et al. (2011). The ship-based survey was dedicated to sampling fish larvae, and not optimised for bird surveys. Next, lesser black-backed gulls from two different breeding colonies were GPS tagged. These colonies were chosen because of their protected status (according to the Dutch Nature Protection Act). Although other large near-coastal colonies exist (such as the one the Maasvlakte), these colonies are not legally protected and thus of no (direct) relevance for AAs. The results of these tagging studies have been reported in Camphuysen (2011) and Gyimesi et al. (2011).

Survey birds at sea

In the aerial survey, about 20 seabird species have been identified, the ship survey observed 25 seabird species (but 90 bird species in total). The difference is caused by those species confined to the part of the wider Dutch North Sea that was not covered by the aerial survey: the north to western Dutch continental shelf, and parts of the British continental shelf. Little auk, Atlantic puffin, white-billed diver, black-throated diver, Manx shearwater, Balearic shearwater, sooty shearwater, Arctic skua and European storm petrel were uniquely observed in the ship-based survey, while the long-tailed skua was only observed in the aerial survey. An advantage of the ship survey is that many birds can be identified more easily because they can be viewed in profile. Note that also many non-seabird species (i.e. species that do not spend their non-breeding life at sea) were found, such as geese, ducks, waders, and songbirds. These were not taken up in the further discussion, although the data may be of interest when more knowledge on migrating birds is required. This was however not part of the shortlist project

Most observed seabird species from the ship were the larger gulls, guillemot, northern Gannet, and northern fulmar (all in the thousands) closely followed by the terns, the razorbill, smaller gulls, great cormorant and the common scoter (all in the hundreds). Other birds were observed in the tens or less.

The aerial survey showed more or less the same pattern, but because of the different spatial coverage (more observations relatively nearshore) the nearshore species showed up in higher numbers in comparison to the offshore birds than in the ship survey. The aerial survey in general showed much higher numbers, among other things because of the larger area covered by the plane in half of the survey time of the ship.

GPS-tagged lesser black-backed gulls

Regarding the distribution of flights, results showed that the tagged birds from the Volkerak did not use the North Sea at all. Only two individuals flew once towards the North Sea, probably after breeding failure. According to the report (Gyimesi et al. 2011) this may imply that non-breeders (so-called “floaters”) could forage at sea more often than breeding birds do. Breeding individuals flew inland, visiting terrestrial sources, bringing in terrestrial food items from a distance up to 25 km mostly. The birds flew regularly to specific areas, and were not homogeneously distributed.

At Texel, the situation was quite different (Camphuysen 2011). The majority of the males foraged at the North Sea (time spent). The majority of the females foraged at the Wadden Sea, Texel or the mainland. Prey items were predominantly marine species. This sexual difference in foraging areas was not apparent in the Volkerak colony. The distribution of gulls foraging on the North Sea was not homogeneous, but mostly restricted to an area west to southwest of Texel, at a distance of 40 to 45 km (3rd quartile). Birds that failed their breeding had a different foraging behaviour (area, frequency etc.) than the breeding birds.

2.3.3 Update framework methodology

In the AAs, focus was only on the possible collisions of birds from breeding colonies protected under Natura 2000 (Bird and Habitat Directives). The 2008 Framework suggested to do

calculations for loss of habitat by disturbance from operating wind farms. This update will adhere to the latter. In a next round of planning OWFs, the accumulation of effects of many windfarms on a relatively small area such as the Southern Bight may cause a large negative loss of foraging habitat. The report by Poot et al (2011) is the most recent update on bird distributions and densities. Their chapter 4 contains a comparison with long-term monitoring data (MWTL) on the Dutch continental shelf, which is subsequently discussed in chapter 5 of Poot et al. (2011) Although the authors conclude that no definite answers are to be expected from one year of monitoring, these data are the most recent and with the highest coverage on the DCS so far. Therefore, these data constitute an important basic data set for the next round of AAs. Of course, future AAs will profit importantly from this data set, but need to consider the results in the context of the earlier surveys on the DCS, such as those from the MWTL program.
Depending on the bird species, any possible effect of an operating OWF may influence the bird's distribution. Moreover, if this leads to long-term displacement away from the OWF, the bird might lose preferable foraging, resting or moulting area. Whether that results in a negative effect depends on the quality of the remaining area, possible density-dependent effects and the bird's ecology.
The best approach currently seems to check for each species if it is present in an OWF plan area, to check the densities during the year and its variations through several years when data are present, and set up scenarios that calculate the possible habitat loss with one annual average density and one annual high density (e.g. 95% confidence limit). Each case needs to be discussed separately, since there is no blueprint for how to assess population effects.

Regarding the breeding birds, the approach in the second round AAs was based on assumptions and observations on distribution, flux, flying height et cetera. For the studied lesser black-backed gulls, the distribution over the area adjacent to their colonies seems to be very different from what was previously thought. Birds from the southern colony (in Volkerak) did not forage at sea, except for two flights in a whole breeding season (Gyimesi et al. 2011). The new spatial distribution of foraging birds from the Texel colony (Camphuysen 2011) seems sufficiently consistent to be used in the new calculations of possible bird collisions (other parts of the methodology are discussed in chapter 5 of this report). Other birds that were considered in the AA were the northern gannet and the storm petrel. Since these species do not breed on Dutch territory, no studies could be undertaken to tag them also with GPS transmitters. For these species, the assumptions used in the second-round AAs still seem appropriate.

3 Effects of underwater noise from OWF piling

This chapter describes the updates for the methods of assessing the possible effects of piling the OWF foundations on fish larvae and marine mammals. The first two paragraphs concern the acoustic aspects of underwater noise from piling. The first paragraph discusses the results from the studies on the methodological standardisation of units and measurements used in underwater acoustic experiments and field studies. The second paragraph treats the source level aspects and propagation modelling of underwater sound.
The reason for treating these aspects here is that here have been considerable advances in knowledge of the standardisation and measurement methods of underwater noise. Although in the context of the second-round AAs no measurements of underwater noise have been carried out, it is part of the terms of reference for monitoring in the licences for OWFs. In addition, for comparability, an AA set up for such obtaining the licence should follow the advised standards for monitoring underwater noise from piling.

Three aspects are treated here. First, the methodology of describing and measuring underwater sound has been the subject of studies by TNO and a proposal for standardisation for units and measurements has been made. Also, a first set up of a propagation model has been published and will form the basis of further propagation model development. Last, the advances on describing the effects of underwater noise on marine animals have been described. Especially experimental work on fish and mammals has been carried out that drive these knowledge advancements. These advancements have shed more light on the complexity of the causality of underwater noise effects on marine animals (mostly mammals). When it comes to translating the experimental results to practical guidance on how to deal with evaluation of these effects with the context of an Appropriate Assessment (basically avoidance distances), only slight improvements were achieved. This will be further addressed in paragraph 3.4.

3.1 Underwater noise: methodological standardisation

The studies on the acoustic aspects of underwater noise were carried out by TNO. They focused in general at “the development of standards for the measurement and reporting of underwater sound, with a primary focus on acoustic monitoring in relation to the environmental impact of offshore wind farms” (Ainslie 2011). Reporting of the project was done in two different parts; one report treats the generic properties of underwater sound and the standardisation of units. The practical implementation of these definitions, the procedures for measuring underwater sound in connection with offshore wind farm licensing, is addressed in an accompanying report (De Jong et al. 2011).

3.1.1 2008 Framework

In the 2008 Framework, chapter 6, a generic description is given of underwater noise especially in relation to the effects on marine fauna. In general, the descriptions of definitions, metrics and units and propagation of sound are still valid. It is at the level of specific information that information has been updated. The amount of information from the first report (Ainslie 2011) is difficult to compare with the descriptions of underwater noise in the 2008 Framework. In the second report (De Jong et al. 2011), some practical suggestions are done for the use of metrics and units in the context of OWFs.

3.1.2 Results of the studies

De Jong et al. (2011) start describing the relevant “types” of sound: single pulse, multiple pulse and non-pulses. De Jong et al. propose a classification (based on Southall et al. 2007) in three types of sound: continuous, transient and repeated (or multiple) transient sound. Each can be subdivided into three characters: incoherent broadband, narrow band and coherent broadband. Next, sound indicators are described in relation to the environmental effects (i.e. on mammals and fish). Each type of sound is “assigned” a specific metric, an SPL (Sound Pressure Level) or an SEL (Sound Exposure Level). These metrics mostly relate to the studies in which effects were shown on mammals and fish. Frequency weighting of the sound is sometimes used in the process of assessing possible risks of the sound to a specific species or group of animals.

Chapter 4 in De Jong et al. (2011) is an important conclusive chapter: it gives a proposal for a measuring and reporting procedure, specifically focused at the licensing of OWFs in The Netherlands. It distinguishes four phases: a pre-construction phase (T0), the construction phase (Tc), the operation phase (Top) and the deconstruction phase (Tdec). It advises on monitoring of the background noises during all phases, and of the specific sounds during phases Tc, Top and Tdec. First, they describe the requirements for the terms and definitions for underwater sound, such as the metrics for use of SPL (Sound Pressure Level), Peak pressures, SEL (Sound Exposure Level), and cumulative SEL. Next, they describe the requirements for the measuring equipment: hydrophones, amplifiers, filtering, conversion and recording, and how to calibrate the instruments.

3.1.3 Update of the framework

When it comes to using the concepts, terminology, metrics and standards in describing underwater noise from piling and operation of OWFs in future AAs, the advice given in De Jong et al. (2011) and Ainslie et al. (2011) should be followed (see their chapter 4). This is essential for comparison among different (future) studies but especially for the proper assessments of the ecological effects of underwater noise from OWFs.

3.2 Underwater noise: source level and propagation modelling

Next to the studies on methodological standardisation, also two piling experiments were done in Kinderdijk (Jansen et al. 2011) to describe the sound of underwater noise of piling for wind farms at the source level, i.e. at a standard level that is not dependent on habitat characteristics such as depth, salinity, temperature, et cetera. Together with propagation modelling and effect levels, this would lead to an improved assessment of the risk of underwater noise for marine life. In addition, a model for noise propagation was set up (Zampolli et al. 2011). This model uses the data from the Kinderdijk experiments and, when finished, will be able to describe short-distance sound wave propagation and give insights into the characterisation of the pile as an acoustic source.

Although the experiments at Kinderdijk were successful, the model development is not yet finished. The report on the model development contains no concrete results yet of any pragmatic value for the update of the 2008 Framework. Current research is aiming for the further development of this model. Moreover, it needs to be validated under realistic circumstances, i.e. the predictions from the model need to be verified by independent field measurements during piling operations.

3.3 Effects on fish larvae and trophic effects on birds and mammals

3.3.1 2008 Framework

In the 2008 Framework and the second round AAs, possible effects of underwater noise from piling on fish larvae and the reduction in larval transport to Natura 2000 areas was modelled. Model results were only available for three species (plaice, sole and herring). These results were extrapolated to other species and to juvenile life stages based on expert judgement. There were large uncertainties, especially on the assumed mortality of the larvae by piling underwater noise. The assumption in the 2008 Framework (Prins et al. 2008) was 100% mortality at a range of 1000 meter. This assumption has been tested in an experimental study in the context of the Shortlist program. All other assumptions, incorporated in the larval transport model and applied in the expert judgement extrapolations, have not been tested.

3.3.2 Results shortlist experiment

The results from the experimental shortlist study on the mortality of larvae by piling underwater noise refer to sole larvae only. Other species were not tested. Sole larvae mortality at the highest exposure level applied (cumulative SEL = 206 dB) was not significantly higher than at silence (α error[1] 5%, Bolle et al. 2011). Analysis of the variance indicated an insignificant probability (<5%) of an exposure effect >14%. Cumulative SEL is dependent on the distance from the sound source and the number of pulses. Based on estimates of pile-driving frequency and mean drift velocity, an exposure of 206 dB cumulative SEL is not expected to occur beyond 400 m from the source. Based on these results, the worst-case scenario for sole larvae is 100% larval mortality up to a distance of 400 m and 14% mortality at a distance of 400-1000 m from a ‘typical’ North Sea piling site. This is a lower mortality rate then assumed in the earlier second round environmental impact assessment (100% mortality over 1000 m).These results cannot be extrapolated to other species.

3.3.3 Updated framework methodology

Additional experiments on presumably more sensitive species (such as herring, sea bass, due to possessing an air-filled swim bladder) are planned for Shortlist II (2012-2013). It can be argued that other flatfish species such as plaice or dab will experience the same mortality as sole since these species have a larval development comparable to sole (i.e. limited or no development of a swim bladder). Nevertheless, there is no certainty on the mechanism of noise impact on larvae. Therefore, for other species than sole it is advised to maintain the 1000 m / 100% mortality criterion until new results may indicate other ranges.
For sole, the worst-case assumption would be 100% larval mortality up to a distance of 400 m and 14% mortality at a distance of 400-1000 m from a ‘typical’ North Sea piling site. This adapted worst-case approach would lead to a reduction of ±50% of the direct mortality effects estimated in the 2008 Framework for common sole. This concerns the effect of the piling on the mortality directly around the piling location. Whether the larval supply to the N2000 areas will increase proportionally, depends on other factors, such as transport, distance to spawning and N2000 areas, etc. This will be different for each OWF.

Additionally, to improve the modelling exercise of mortality and transport of larvae, it might be worthwhile to further develop the model (e.g. include natural mortality and include more species) and to validate/calibrate the model using the larval distribution data from the larval survey carried out in the Shortlist program (Van Damme et al. 2011).

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[1]An alpha or Type I error is committed when we fail to believe a truth. In this experiment, the alpha error means that there is 5% chance of accepting there is no effect, but in reality, there is an effect.

Regarding the trophic transfer of the reduced availability of prey fish for birds and mammals in Natura 2000 areas, there has been no methodological improvement compared to what has been done in the second-round AAs. For new AAs, it is advised to follow the procedure described in these AAs (e.g. Arends et al. 2008).

3.4 Effects on marine mammals

This paragraph deals with the non-lethal effects of underwater noise on marine mammals. This subject is of great importance, since the common assumption is that there are potentially large effects of piling for OWFs on marine mammals and fish.

Various reports and other results have been delivered that describe:

1. an update on the distribution of harbour seals and field observations of the effect of OWF piling thereon (Brasseur et al. 2012),
2. the effects of piling on marine mammals based on experimental work (e.g. Kastelein et al. 2011), and
3. field studies on the observations of marine mammals or their communication activity before, during and after OWF construction in the North Sea (Lucke 2010, Brandt et al. 2011).

Below, the assumptions and findings from the 2008 Framework (Prins et al. 2008) are described and updated to a limited extent. It was decided that a full update of the 2008 Framework regarding the effects of underwater noise on marine mammals was not possible at this stage. The main reason for this is that the causality chain going from underwater noise to effects on (individuals or even populations of) fish and mammals is unclear. Although relevant to furthering the understanding of this causality chain, these relationships will only be discussed very shortly in this report. The advice in this report on how to deal with assessing the possible effects in near future AAs is ultimately a very pragmatic choice, based on what we know to be reliable information (such as the studies from Kastelein et al. 2011 and Brandt et al. 2011) and on the already used descriptions of underwater noise and its effects from the earlier 2008 Framework. Although the methodologies to come to the noise effect thresholds and distances given in the 2008 Framework are not supported by the results and interpretations described in Kastelein et al. (2011), we adopted part of them due to the lack of a better alternative. This creates tension between what we advice here as a pragmatic solution, and what is suggested by the results from Kastelein et al. (2011). However, there currently is no practical alternative to our advice for the short term. The experimental work started by Kastelein and co-workers in the Shortlist context was only the beginning of a larger research plan that still needs to be finished. This plan is expected to importantly increase our knowledge on the causality chain between underwater noise, the occurrence of TTS and possible physiological changes leading to deviant behaviour of porpoises and seals.

3.4.1 2008 Framework

The effects of underwater noise on marine mammals in the 2008 Framework were treated in chapter 7, especially in paragraph 7.2. This paragraph was based on the background document by Kastelein et al. from 2008. In paragraph 7.2 of Prins et al (2008), the sensitivity of common seals and harbour seals were discussed on the basis of foreign studies but noise levels were mostly derived from the studies by Kastelein and colleagues. Especially their dose-response relationships were of importance for the AA Framework.

In Prins et al. (2008), the effects of underwater noise on biota are described in Figure 3.1 below:

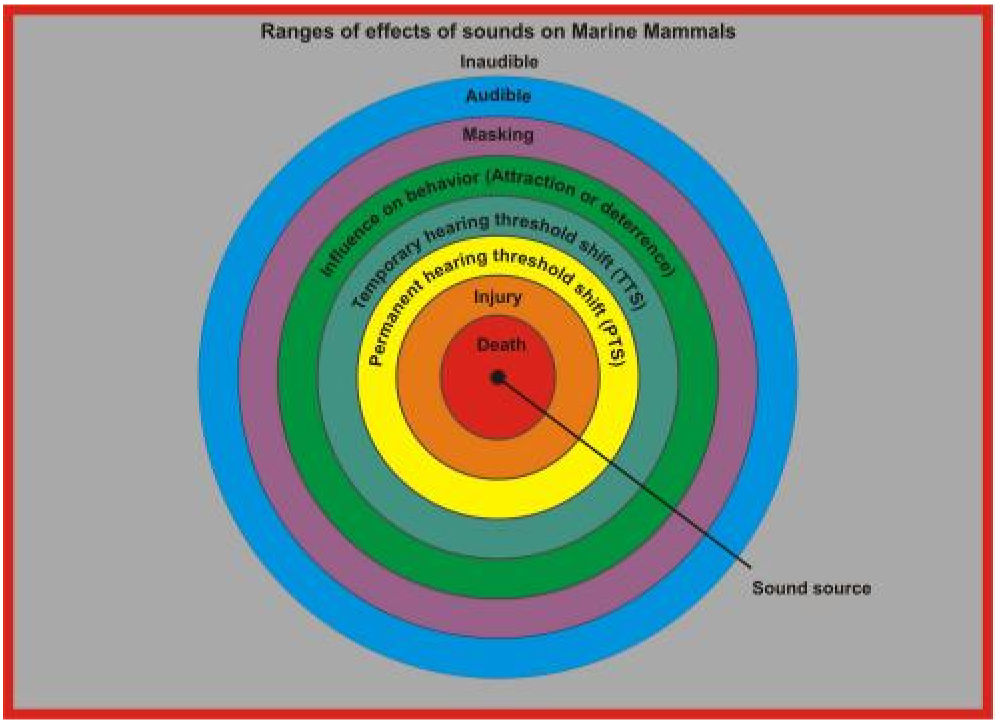

Fig. 3.1: *Schematic view of the effects of sound in relation to the received sound level (i.e. distance between an animal and a sound source).*

In Prins et al. (2008), three exposure criteria metrics were used to describe monopile-driving sounds. This method was proposed by Kastelein et al. (2008), and differs from Southall et al. (2007):

- The broadband sound level, Lp: the sound pressure level, summed over the analysis bandwidth range (PTS[2] onset level).
- The sound exposure level, SEL: the broadband sound level normalised to a 1-s period (TTS onset level).
- The equivalent continuous sound level, Leq: the steady dB-level, which would produce the same sound energy over a stated period of time as a specified time-varying sound. This parameter is only relevant for multiple strokes.

In 2008, the following was used as the standard for dose-response levels for marine mammals:

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[2] *The TTS threshold shift for hearing is a shift in the threshold for hearing sound, which, at low sound pressure values, is essentially a protection mechanism to the sensory organs of the ear. It creates a temporary increase in the lowest sound level to become audible in the organism. A comparison is the human loss of sensitivity for high frequencies due to listening too often to too loud music. Usually, TTS itself creates no damage and the functioning of the ear recovers fully. When prolonged or repeated, TTS can be become a permanent threshold shift (PTS).*

***Table 3.1:** Dose-response levels for porpoises and seals used in the 2008 Framework.*

	PTS-onset level	*TTS-onset level*	*TTS-onset level*	*Avoidance level*
Harbour porpoises	224 [1)]	160 [1)]	135 [2)]	97/108 [2)]
Harbour seals	215 [1)]	170 [1)]	145 [2)]	105 [2)] 120 [3)]
Metric	Lp	SEL	Leq	Leq
Units	dB re 1 μPa^2	dBw re 1 μPa^2.s	dBw re 1 μPa^2	dBw re 1 μPa^2
Weighting	No	Yes	Yes	Yes

dBw = weighted level
[1)] These levels are only valid for LF monopile driving transients, for which the spectrum peaks at around 200 Hz.
[2)] These levels refer to continuous noise
[3)] This level refers to multiple transients

Note that when mentioning dBw, meaning a frequency-weighted dB level, this weighting is species specific. Weighting needs to be done using the audiogram of the species for which the weighting is being carried out.

When using the piling noise from Q7 (now called *PAWP: Prinses Amalia Wind Park*) as a source, Kastelein et al. (2008) calculated the following distances for TTS and avoidance:

***Table 3.2:** Calculated radii for occurrence of piling effects on harbour porpoises and harbour seals – Temporary Thresholds Shift (TTS) and avoidance (based on the piling noise from OWF Q7, North Sea, 25 m water depth).*

	TTS after 1 stroke	TTS for driving 1 monopile	Avoidance for driving 1 monopile
Harbour porpoise	**40 m**	**500 m**	**12 km**
Harbour seal	**300 m**	**4 km**	**80 km**

These data were used in the nineteen AAs for the “second-round” OWFs at the DCS in 2008 and 2009.

3.4.2 Results shortlist experiments

The shortlist experiments were set up for better assessing the dose-response relationships: they were designed to improve TTS (temporary threshold shift) level understanding. The TTS studies on the porpoises and the seals consisted of four experimental studies:

1. The level/duration combinations of a noise band which cause TTS, the degree of TTS, and the rate of hearing recovery in porpoises.
2. The level/duration combinations of a noise band which cause TTS, the degree of TTS, and the rate of hearing recovery in harbour seals.
3. The level/duration combinations of playbacks of impulsive pile driving sounds, which cause TTS, the degree of TTS, and the rate of hearing recovery in porpoises.
4. The level/duration combinations of playbacks of impulsive pile driving sounds, which cause TTS, the degree of TTS, and the rate of hearing recovery in harbour seals.

It was originally planned to test four different noise bands capturing the frequencies where most of the energy of the piling noise is known to be present: octave band white noise bands centred at 0.5, 1, 2 and 4 kHz. The first set of experiments was carried out with the last frequency, the octave band white noise band centred at 4 kHz.

The results of the experiments showed the following:

1. Considering the sensitivity to noise bands, TTS in harbour porpoises (of over 2.5 minutes of octave-band white noise bands centred around 4 kHz) occurred at a SEL of ca. 152 and 162 dB re 1 μPa^2.s, depending on the Sound Pressure Level of the fatiguing sounds. This confirms the earlier assumption that harbour porpoises are particularly sensitive to high levels of sound exposure.
2. It appeared that SEL alone was not a proper metric for predicting the TTS onset level, since the TTS of the porpoises was more dependent on the duration of the exposure sound (and therefore a cumulative SEL) than on its level. These results are in line with earlier findings of Mooney (2009) measuring TTS of dolphins
3. Comparable to the studies in harbour porpoises, the studies on the narrow-band sensitivity of harbour seals showed the onset for TTS to occur at SEL of ca. 170 and 178 dB re 1 μPa^2.s, depending on the Sound Pressure Level of the fatiguing continuous noise sounds.
4. In the initial tests exposure to the broadband spectrum of (transient) piling sound (for which a recording was used from the OWEZ piling), no TTS of any significance could be induced. Probably, the SEL in these initial test was too low to induce TTS in porpoises. At these lower levels. avoidance behaviour was noted. As with the porpoise, the seals showed no TTS when confronted with the transient piling sound. One of the two seals showed clear avoidance behaviour. The exposure to higher levels of piling sound was not possible in the first phase of the project.

The translation of the results by Kastelein et al. (2011) into practical guidelines is not straightforward, since their experiments were not designed to assess behavioural effects of underwater noise on marine mammals. Therefore, below is the most relevant text from their report (important text has been underlined):

"*The results from the two noise band studies (sections 2.1. and 2.2. of this report) show that, despite their high frequency specialization, harbor porpoises are more vulnerable to TTS than harbor seals (for the noise bands tested).*"

"*Thus, it can be concluded that, though individual variation exists, the avoidance threshold level for seals is much higher than for porpoises, and that harbor porpoises are probably deterred further away from a pile driving site than harbor seals.*"

"*The results from the present project show that the predictions described in Prins et al. (2008) of distances from pile driving sites at which TTS starts to occur in harbor seals and harbor porpoises need to be adjusted (the predictions were based on the best available knowledge at the time). The TTS onset SPLs of harbor seals for frequencies around 4 kHz are higher than for the harbor porpoise (counter to what was expected from their audiograms). Also, the avoidance threshold SPL (from which avoidance distances can be calculated) was predicted to be lower for harbor seals (resulting in avoidance at greater distances) than for harbor porpoises. Although the present project cannot provide specific distances, because the studies were designed to measure TTS and not behavioural responses, it appears that the avoidance threshold SPLs are higher for harbor seals than for harbor porpoises (i.e. that harbour porpoises are deterred further away from pile driving sites than harbor seals). However, controlled behavioural response studies should be performed to quantify and characterize behavioural reactions as a function of sound exposure type.*

These experiments thus suggest two important (but tentative) findings:

1. The onset threshold sound pressure levels for the noise bands around 4 kHz and for the piling tested are higher for seals than for porpoises for both TTS and avoidance. This is counter to what was expected earlier from the audiograms and suggests that porpoises may be more sensitive to TTS due to piling than seals.
2. The fact that the piling noise triggered a behavioural reaction (avoidance) in both porpoises and seals while no TTS could be measured at the maximum SPL technically attainable (i.e. 120 min of 173 strikes/min at a peak SPL of 164 dB re 1 μPa; cumulative SEL: 183 dB re 1 $\mu Pa^2.s$) suggests that the cumulative SEL level for TTS due to piling is higher than the abovementioned 164/183 dB re 1 $\mu Pa^2.s$. The cumulative SEL for behavioural reactions lies presumably at least at this level or lower.

This information is quite relevant for translating the experimental results into practical guidelines for assessing piling noise effects on marine mammals. However, as the authors state, the experiments were designed to measure TTS, and not avoidance. Also, although noise bands did cause TTS in seals and porpoises, the (broadband) piling noise could not elicit TTS in these species. Further experiments need to be carried out to obtain more insight into avoidance behaviour in relation to piling noise.

3.4.3 Updated framework methodology

An improvement on the causality chain of underwater noise and the effects on marine organisms is described in the table below. The description of effects as a function of he distance to the underwater noise source as depicted in figure 3.1 above is no longer valid. Especially the assumption that behavioural changes occur at a larger distance than TTS seems no longer tenable (see e.g. the results from Kastelein et al. 2011). In stead, the table below is presented, which has also been used in the latest background document for further development of the MSFD descriptor 11 (Under water energy) developed by the Technical Subgroup on underwater noise and other forms of energy (Van der Graaf et al. 2012).

Table 3.3 Potential negative effects of sound on marine life

Impact	Type of effect
Physiological, non auditory	Damage to body tissue: e.g. massive internal haemorrhages with secondary lesions, ossicular fractures or dislocation, leakage of cerebro-spinal liquid into the middle ear, rupture of lung tissue
	Induction of gas embolism (Gas Embolic Syndrome, Decompression Sickness/DCS, 'the bends', Caisson syndrome)
	Induction of fat embolism
	Disruption of gas-filled organs like the swim bladder in fishes, with consequent damage to surrounding tissues
Auditory- (Sound Induced Hearing Loss)	Gross damage to the auditory system – e.g. resulting in: rupture of the oval or round window or rupture of the eardrum
	Vestibular trauma – e.g. resulting in: vertigo, dysfunction of coordination, and equilibrium
	Damage to the hair cells in fishes
	Permanent hearing threshold shift (PTS) – a permanent elevation of the level at which a sound can be detected
	Temporary hearing threshold shift (TTS) – a temporary elevation of the

	level at which a sound can be detected
Perceptual	Masking of communication with conspecifics
	Masking of other biologically important sounds
Behavioural	Stranding and beaching
	Interruption of normal behaviour such as feeding, breeding, and nursing
	Behaviour modified (less effective/efficient)
	Adaptive shifting of vocalisation intensity and/or frequency
	Displacement from area (short or long term)

As explained above, the results from the experimental work of Kastelein et al. (2011) are such that the assumptions on effect distances for TTS and avoidance for seals and porpoises seem to be incorrect. Basically, it means that the assumptions of which characteristics of sound cause what kind of effect need to be reconsidered. Although the audiogram gives us importance information on the sensitivity of mammal species to sound, it does not however translate easily into sensitivity levels to TTS. The latter is more related to the so-called dynamical range of the hearing system of mammal species.

Translating narrowband sensitivity (such as used in the studies by Kastelein et al. 2011) to broadband sensitivity (such as transient piling noise) and assessing the probability that TTS may occur at a specific distance from the sound source makes use of various, untested assumptions. One of the steps done is the weighting of the sound to express the sensitivity of the animal to the TTS level corresponding to the sound to which it is exposed. Currently, the audiogram is used as a proxy for deriving the organism's sensitivity for the TTS onset level. However, when more information will become available of the actual sensitivity of the animal for TTS, creating "equal loudness contours" (as the upper level of the dynamic range of the species hearing) would give a more accurate estimate for the species' sensitivity for TTS.

The audiograms of harbour seal and porpoise suggested seals to be more sensitive to TTS from 4 kHz narrow band. However, the findings of Kastelein et al. (2011) have shown the opposite: porpoises are more sensitive to TTS from the experimental narrow band sound than seals are. This is exemplary of the limited applicability of using the audiogram for deriving a species sensitivity for TTS.

There are many more issues to be dealt with, for which various solutions are possible (and suggested in literature). This needs to be studied further in detail before any progress can be made on defining threshold limits for practical use. The research program suggested by Kastelein in the Shortlist program has not yet been fully executed. This research program foresees in filling in important knowledge gaps in our understanding of the causality between underwater (piling) noise and the occurrence of TTS and possibly behavioural changes in marine mammals.

<u>Short-term solution</u>
A short-term solution is to (re)use the avoidance distances mentioned in the 2008 Framework, but with minor modifications, based on recent field observations of seals and porpoises during piling of offshore wind farms in the North Sea and the Baltic Sea.

Avoidance data on harbour porpoise due to piling show variable reactions, with a maximum observed avoidance reaction at ca. 20 km (Lucke 2010). Other studies show comparable or smaller avoidance distances during piling (e.g. Carstensen et al. 2006). It has to be stressed that effects can be expressed differently. In a recent publication by Brandt et al. (2011), reactions of porpoises to piling of Horns Rev II were measured with T-PODs. They measured a decrease in porpoise acoustic activity. A 100% decrease lasted for 24 to 72 hours at a distance of 2.6 km. This period gradually decreased with increasing distance and a no-effect level was detectable to at a mean distance of 17.8 km. As a maximum effect (worst-case) scenario, an avoidance distance of 20 kilometres seems appropriate. This is an increase in avoidance distance compared to what was used in the earlier AAs for the second-round OWFs (12 km).

Larger problems exist for assessing the underwater noise avoidance distances for seals. The data on the behaviour of tagged common seals before, during and after piling (Brasseur et al. 2012) did not show any convincing differences in seal behaviour between these periods. In the study referred to above by Lucke (2010), where the avoidance distance for porpoises due to was piling was estimated at ca. 20 kms, seals could be observed within this range during piling, so at an even smaller distance than 20 kms. It was not mentioned whether the seals were swimming with their heads below or above water. Seals may avoid underwater noise by emerging their ears from the water, during which they will not be able (or willing) to forage under water.
Kastelein et al. (2011) suggest that, based on their experimental findings, seals may not be more sensitive to underwater sound from piling than porpoises. In combination with the observations from Lucke (2010), there seems a basis for adapting the avoidance distance for harbour seals from 80 kms to 20 kms. Moreover, the empirical basis on which the 80 kms as avoidance distance for harbour seals were calculated in 2008 (Kastelein et al 2008) was quite small. Nevertheless, so is the empirical basis for assuming the 20 kms as a better alternative avoidance distance. So, this leads to a Catch 22 situation. Based on a worst-case approach, the current advice would be to maintain the avoidance distance of 80 kms, and to stress that additional field observations and experimental work is needed to elucidate this controversy.

It should be emphasised that the developments within the field of noise propagation modelling, and the assessment of noise effects on marine organisms is in full swing. Although currently there is a lack of a broadly accepted cause-and-effect model supported by recent experimental and field data, it is our expectation that this will change soon. Specifically regarding the description of piling noise, its propagation and its effects on marine organisms there is a large chance that in the near future, more knowledge will be available giving way to an improvement on what is suggested in this report as an update for the 2008 Framework. Studies that will supply an improvement on the understanding of the cause-and-effect chain, and are able to come to better assessments of piling noise effects on marine organisms need to be taken into account in future AAs. The assessments of noise effects on marine mammals in AAs will need to provide a transparent, internally consistent cause-and-effect model, which is supported by the most recent data from experimental and field studies, follow the latest insights in acoustic metrics and models and be broadly supported by the scientific society on underwater noise acoustics and ecological effects.

It further needs to be noted that as more and more OWFs are build in the (southern) North Sea, the chances of an accumulation of effects will grow. Some level of accumulation has been considered in the second-round OWF EIAs and AAs, but this was not based on actually existing OWFs; nor were any foreign OWFs considered. It is not possible to set up some concrete guideline on how to sum up the effects of OWFs on marine mammals. Much depends on the location, size and configuration of the OWFs. However, when planning an OWF, and writing the EIA and AA, the level of cumulative effects to be considered needs to be discussed with the proper authorities (“bevoegd gezag”).

4 Operational effects of OWFs

4.1 Effects on marine mammals

4.1.1 2008 Framework

In paragraph 7.2.4 of the 2008 Framework, a description is given of the possible effects of operational underwater noise from OWFs on marine mammals. Regarding harbour porpoises, effects on behaviour were found in experimental studies where porpoises were exposed to simulated operating noise from OWFs (Koschinsky et al. 2003). One study was able to show avoidance effects by porpoises in the field. Seals appeared not influenced by air-borne noise and visual effects produced by operating OWFs. When confronted with underwater noise from OWFs in an experimental set up, also seals changed their behaviour (Koschinsky et al. 2003).

4.1.2 Results shortlist experiment and other studies

Within the Shortlist monitoring and research framework, no experiments or studies were carried out specifically to better assess the effects of OWF operation on the behaviour of marine mammals.
After construction of OWF *OWEZ*, a study was performed to check for the use of the windfarm by porpoises. Contrary to what was expected, the data from this study point to an increased use of the area during operation of the windfarm compared to before the windfarm was constructed and compared to the areas around it while the farm was in use (Scheidat et al. 2011). However, in a recent review of the study, this conclusion could not be validated, due to the differences in sensitivity of the T-PODs used in the study of Scheidat et al. (Blacquière et al. 2012).

Tougaard & Carstensen (2011) showed no change in porpoise acoustic activity after construction of a small OWF north of Sprogø, Denmark.

Seals were known to react to piling during the construction of OWFs, and in many cases returned during the operation phase; only recently a study provided evidence of seals showing a structural change in distribution after construction (Skeate et al. 2012). In Lindeboom et al. (2011), it is mentioned that tagged and non-tagged seals were observed within the operating *OWEZ* wind farm, but according to what has been reported by Tougaard et al. (2006) for seals in the operating OWF *Horns Rev*, they did not exclude effects on their behaviour. Tougaard et al. (2006) clearly state that neither the tagged seals nor the visually observed seals showed any effect of the operating wind farm *Horns Rev*.

4.1.3 Updated framework methodology

Based on theoretical considerations, an effect of operating OWFs on local distribution (very close to OWFs) of seals and porpoises cannot be excluded. In the field, no negative effect of operating OWFs on seal and porpoise distribution has been observed. For harbour porpoises, any negative effects may be compensated for, such as has been suggested for *OWEZ* (Scheidat et al 2011). Seals also have been observed inside *OWEZ,* although at Scroby Sands, the harbour seals' foraging habitat seems to be partly taken over by competing grey seals (Skeate et al. 2012).

Overall, there seems to be no reason to change the conclusions from the 2008 Framework that operating OWFs have a negligible effect on the harbour seal and harbour porpoise distribution around OWFs.

It has to be noted here that what was stated in the chapter 3 about the sensitivity of harbour seals and porpoises to underwater noise from OWF piling, equally applies on the sensitivity of these species to underwater noise from operational OWFs. However, since the expectations of effect of operating wind farms, and field observations strongly suggest the effect of operational OWFs on seals and porpoises to be (near) negligible or even positive (see e.g. Lindeboom et al. 2011, Scheidat et al. 2011), there is no urgency to adapt the threshold distances from the 2008 Framework.

It needs to be emphasised that as more and more OWFs are build in the (southern) North Sea, the chances of an accumulation of effects will grow. Some level of cumulation has been considered in the second-round OWF EIAs and AAs, but this was not based on actually existing OWFs; nor were any foreign OWFs considered. It is not possible to set up some concrete guideline on how to sum up the effects of OWFs on marine mammals. Much depends on the location, size and configuration of the OWFs. However, when planning an OWF, and writing the EIA and AA, the level of cumulative effects to be considered needs to be discussed with the proper authorities (“bevoegd gezag”).

4.2 Effects on birds

4.2.1 2008 Framework

In Prins et al. (2008), the effects of the operational phase of OWFs on birds were described in three categories:

1. Possibility of collision of birds flying around for various reasons
2. Loss of foraging, resting or moulting habitat for seagoing birds
3. Avoidance of OWFs by seagoing and migrating birds causing larger energy expenditure (barrier effect)

Regarding the collision possibilities, Prins et al. (2008) presented three ways (route 1, 2 and 3) of calculating the possible number of collision victims: empirically (route 1 and 2) or by modelling (route 3). These three “models” were described in detail in Troost (2008). Route 1 made use of a simple correlation of actual collision victims (on land) with the rotor surface of wind farms. Route 2 calculated the risk of collisions by way of the so-called Bureau Waardenburg model, consisting of the collision risk from land-based wind farms and scaling them up taking into account bird fluxes, bird flying height, rotor surface etc. Route 3 made use of the ‘Band-model’. This model calculated the risk of collision of the rotor blades with an object flying through a rotor disk with a certain speed. This model was used in conjunction with the configuration of a wind farm (as developed in route 2) and replaced the empirical land-based collision risk from route 2.

In the 2008 Framework, the loss of habitat was determined by the number of bird days lost per year (i.e. densities with no wind farm present times the number of days the species is present during 1 year times the area lost due to the wind farm development). In the second-round AAs, habitat loss for local seabirds due to offshore wind farms was calculated in three steps. First, on-site local seabird densities need to be known, year-round. Second, the vulnerability of each species with regard to wind farms needs to be known. Seabirds’ densities have long been assessed in at-sea seabird surveys, both from aircraft and from ship. In the first series of Appropriate Assessments for Round 2 offshore wind farms in Dutch waters, both sources of information have been used. Species-specific levels of wind farm vulnerability were used, based on a range of relevant parameters regarding population size, at-sea behaviour and presumed speed of recovery after population decrease. The combination of seabird numbers present and their different vulnerabilities yielded estimates of general wind farm vulnerability, across the Dutch Continental Shelf. A third source of information comes from actual assessments of avoidance of wind farms by different species of seabirds. At the time of writing the second-round AAs, this information was very limited and only preliminary data from the Horns Rev - 1 OWF could be used.

The barrier effect can be expressed as the extra distance travelled by birds experiencing obstruction by the wind farm. The parameter is expressed as ‘bird distances’: a combination of (extra) distance flown, number of birds involved and the frequency of obstruction.

During the AAs for the second-round OWFs, these three effects were calculated. From the results, it became clear that the barrier effect caused negligible effects on migrating seabirds. Also the collision victims from migrating birds were negligible. These were calculated for a cumulative scenario of a fictitious OWF of 1000 MW at a location where the migration densities were highest. This does not mean there will never be a future effect due to the barrier effect or the collisions from migrations. It did focus the attention to the largest possible effects: mortality due to collisions from breeding shorebirds.

4.2.2 Results shortlist experiment and other studies

Compared to the Framework provided by Prins et al. (2008) major improvements were made on the data on breeding birds (see chapter 3 on the results from the tagging and the observations on lesser black-backed gulls), species specific collision risk modelling and cumulative effect modelling. This research was partly conducted in the context of the Shortlist program, and partly related to the work following from a legal dispute on the interpretation of collision victim modelling effects on population dynamics. Also, the results from the surveys and observations on the behaviour of species around OWF *OWEZ* and *PAWP* were used for an update on the assessment of loss of habitat. These results are discussed below.

Predicting numbers of collision victims: improvements in the Band model

Recently, the Band model has been improved by developing more guidance for the use of the Band model in offshore situations (Band 2011). This revised ‘extended Band model’ uses the same theoretical model to predict the collision probability for a single bird crossing the rotor-swept area, and in addition, provides a standard framework for estimating the total number of transits for the planned wind farm on the basis of bird densities (primarily from ship-based surveys), the number of turbines, levels of flight and turbine activity and the level of avoidance behaviour shown. Unlike the original Band model, this extended model constitutes a complete method for attaining collision estimates from data on bird abundance and in that sense could be regarded as a fourth route for estimating the rate of collisions with planned wind farms. It includes a spreadsheet and guidance documents to facilitate its use: www.bto.org/science/wetland-and-marine/soss/projects. This model has been developed for use with data on densities, such as those obtained with ship-based and aerial surveys, and uses a real bird density, along with the proportion of birds flying at rotor height, to estimate the number of transits per time. The use of bird densities, rather than flux, represents a key difference with the approach in routes 1, 2 and 3. Although the conversion from flux to density is possible for use in this model (Annex 2 in Band 2011), this again includes a number of assumptions, which increases the uncertainty of the results.

The applicability of the extended Band model (Band 2011) depends on both the availability of the underlying turbine and bird parameters that are needed in the sub-model and the availability of empirical bird density data within the study area. As with route-3, the estimates from the extended Band model are largely dependent on the avoidance rate applied. Determining the species-specific levels of avoidance exhibited by birds at sea remains an important factor for producing reliable collision rate estimates. The radar flux research of the OWEZ MEP has produced better estimates of species specific micro-avoidance and macro-avoidance rates which were applied in the cumulative effect model (Krijgsveld et al. 2011, Poot et al. 2011). These research results predict a very small till negligible effect by operational wind farms on both breeding and migrating birds.

The extended Band model (route 3 in the second-round OWF AAs, e.g. Arends et al 2008), developed under the framework of the Crown Estate Strategic Ornithological Support Services group, follows the same theoretical model to predict the collision probability for a single bird crossing the rotor-swept area as described in Band (2000) and Band et al. (2007), and promoted in guidance published by Scottish Natural Heritage. This revised model has been extended to create a standard framework for estimating the total number of transits for the planned wind farm on the basis of bird densities (primarily from ship-based or aerial surveys), the number of turbines, levels of flight and turbine activity and the level of avoidance behaviour shown. This model has been specifically developed for use in offshore situations. The approach is described in various steps in Band (2011):

1. Estimates for the numbers of flying birds in the area should be obtained, which assumes the absence of turbines and thus of avoiding action, displacement or attraction.
2. The level of flight activity can be used to estimate the potential number of transits through the rotor-swept area. The probability of collision for a single rotor transit can then be calculated.
3. These results can then be scaled up to yield the potential collision rate for the bird species in question, allowing for the proportion of time that turbines are not operational. Figures at this stage are still based on the assumption of current bird abundance and that no avoiding action is taken.
4. Finally a correction can be made for the proportion of birds likely to avoid the wind farm or its turbines (as an avoidance rate), either because they have been displaced from the site or because they take evasive action.

Habitat loss for local seabirds
The data needed for the loss of habitat have been updated in recent research. These data come partly from the studies carried out in the Shortlist program and partly from the studies performed in the monitoring program around the existing OWFs *OWEZ* and *PAWP*:

1. Density maps of seabirds in Dutch waters and at proposed wind farm sites
In the second-round AAs, seabird densities have been assessed from a combination of aerial surveys and from ship-based surveys. The aerial survey data stem from the so-called MWTL program. A recent evaluation of this program (Boon et al. 2010) showed several differences between the MWTL methodology and the Shortlist methodology: MWTL survey lines are fewer with larger distances between, and flight altitude is higher, which may lead to a lower possibility of determination for certain seabird species. However, the effect of these differences on the general picture of seabird density and distribution is not for every species notable. For instance, for the red-throated diver the differences between density and distribution may be attributed more to timing of the survey than to differences in methodology (Mervyn Roos, pers. comm.). The database of ship-based seabird survey has become very uneven in coverage in time and space, as work has concentrated on the two wind farms since

2004, leaving most of the remaining parts of the Dutch Continental Shelf un-surveyed for many years. Either databases, as stand-alone or in combination, thus have significant shortcomings if a new round of offshore wind farms needs to be evaluated. The Shortlist surveys (aerial and ship based, see Poot et al. 2011, van Bemmelen et al. 2011) have generated more recent, and high-resolution data, but for one year only and for only part of the Dutch Continental Shelf (DCS). New, high-quality seabird surveys across the entire DCS are thus urgently needed for proper future AAs. Note that both the Shortlist surveys and the surveys around the existing OWFs *OWEZ* and *PAWP* were more or less nearshore. Truly offshore parts of the DCS, such as the area in the northern part of the DCS (being a habitat for these birds), have not been surveyed in sufficient detail during the last decade.

2. Species-specific vulnerability parameters
Due to differences in life-history traits and at-sea behaviour, species differ in vulnerability for the presence of wind farms. Garthe et al. (2004) developed an index combining a set of parameters reflecting theoretical vulnerability. Leopold et al. (2010) slightly amended these indices and used them in a study of the DCS. Most of this work has been done before the first wind farms had become fully operational and before studies of actual bird behaviour in these parks were fully analysed. In addition, part of the parameters used, such as population sizes, have changed since. These are not further described here, and need to be re-assessed in future Appropriate Assessments.

3. Species-specific avoidance of wind farms
Some seabird species avoid the sea in and around offshore wind farms. The surface area of the wind farm, the size and density of turbines in the area and the species of seabird might determine the extent of avoidance. In the second-round AAs, parameters were based on the work around the OWFs at Horns Rev and Nysted in Denmark.

Recent research in and around the existing Dutch OWFs *OWEZ* and *PAWP* (Leopold et al. 2011) have indicated that avoidance is probably substantially less than in the studies at Horns Rev. For many species, birds are present within the wind farm, although in lower densities than outside the windfarm. The distance many bird species keep from the outer perimeter is zero or very small. In addition, the level of avoidance appears to be correlated to the turbine type installed. More studies are underway and in the near future, more avoidance data will be available, allowing for a meta-analysis of disturbance for a range of species and wind farms. In future AAs, additional calculations are needed to adjust the parameters used in the second-round AAs.

Note that some additional species of seabirds that were not considered in the second-round AAs might be relevant to future AAs (particularly if the northern part of the DCS needs to be considered) such as Fulmar and Puffin. Such species are protected within N2000 areas in the UK and effects on these areas should be considered (as has been done in the 2nd round AAs). These species do not occur in sufficient densities around *OWEZ* and *PAWP*, or the wind farms in Denmark, to estimate avoidance rates reliably.

4.2.3 Updated framework methodology

As described above, calculations for assessing possible collision victims and loss of habitat in future AAs need to be updated as follows:

1. **Collision victims**:
   Future AAs need to include the updated (SNH-)Band model as the standard methodology for calculating possible collision victims for migrating birds and foraging birds from breeding birds onshore. The other routes 1 and 2 did not show any methodological progress, and in comparison with the progress with route 3 (the Band model), have become less relevant for calculating possible collision victims. So far, no field measurements of collision victims in Dutch CP wind farms are available due to methodological limitations.
2. **Habitat loss**:
   In all three steps needed for calculating the loss of habitat for seagoing birds, progress has been made. Density maps have been updated with newer data and with a higher resolution. Species-specific vulnerability and population sizes can be re-assessed, as is the case for species-specific avoidance rates of OWFs. Models to assess cumulative effects on population dynamics of breeding birds are available.
3. **Barrier effect**:
   The barrier effect of OWFs has not been considered to give possible significant negative effects on birds. It is likely that this neither will be the case for each future OWF separately. However, when the Netherlands and its surrounding countries Belgium, United Kingdom and Germany continue with their plans for future OWFs in the southern North Sea, the cumulative effects of such OWFs can be considerably large. In the near future, the necessity of these calculations needs to be reconsidered.

The calculations described above are region specific and need to be carried out per planned OWF; there are no standard data to be used except maybe for species-specific avoidance rates of OWFs. E.g. the effect of turbine configuration on macro- and micro- avoidance rates is still unknown. Additional analyses are needed to calculate the rates. Other parameters need to be assessed per AA, because they are specific for the location where an OWF is planned, and dependent on the size and configuration of the turbines in a wind farm.

It needs to be emphasised that as more and more OWFs are build in the (southern) North Sea, the chances of an accumulation of effects will grow. Some level of accumulation of effects has been considered in the second-round OWF EIAs and AAs, but this was not based on actually existing OWFs; nor were any foreign OWFs considered. It is not possible to set up some concrete guideline on how to sum up the effects of OWFs on birds. Much depends on the location, size and configuration of the OWFs. However, when planning an OWF, and writing the EIA and AA, the level of cumulative effects to be considered needs to be discussed with the proper authorities ("bevoegd gezag").

5 Overview of updated framework methodologies

This chapter contains an overview of the advice given for the update of the 2008 Framework by Prins et al. (2008) for the guidance of the Appropriate Assessments of the environmental effects of OWFs.

5.1 Current status on knowledge of distribution of fish larvae, marine mammals and birds

5.1.1 Fish larvae

Improvements have been established on the spatio-temporal variability of fish eggs and larvae, the speed of development of the eggs to larvae and the various stages of the larvae. Such data could be used to improve the description of the behaviour of the larvae and to calibrate the larval transport models, which have been used in the 2008 Framework and the AAs. Thus, the quality of the results of the modelling exercises will be improved when the models are rerun with the new data on mortality (for sole, see paragraph 3.3.2) and when the larval distribution of the model is calibrated with the data on larval distribution from the surveys. Additional surveys will further improve this model.

5.1.2 Marine mammals

Harbour seal and grey seal

As a result of the shortlist surveys and the OWEZ results, an experimental relative density map of seals has been produced in Brasseur et al. (2012). This map can not yet be used for estimates of the assessment of effects of piling (e.g. when estimating proportions of the total amount of seals affected by piling); currently there are no reliable data on seal densities in Dutch coastal waters.

Harbour porpoise

Aerial surveys specifically for porpoises have delivered new data that should be used in future assessments of impacts of construction activities such as piling on harbour porpoises. Two pieces of information are relevant here: location-specific porpoise densities and the occasional presence of porpoise calves west of the Holland coast.

A realistic approach should consider the average to higher range of the extrapolated averages for a specific location and its surroundings as found in Geelhoed et al. (2011), taking into account the values as they have been observed in other studies (see table 8 in Geelhoed et al. 2011) and considering the interannual and seasonal variation as described in the porpoise conservation plan (Camphuysen & Siemensma, 2011).

Mother-calf pairs have been observed in July in areas that are relevant for the construction of OWFs. Although this is a first observation of calves in Dutch coastal waters, the relevance of our waters for porpoise calves should be taken into consideration in future impact assessment in combination with the assumed higher sensitivity of calves (or mother-calf pairs) for disturbance (due to the need for foraging). However, as for adults, no concrete figures can be given about the effect of disturbance by underwater noise on the fitness of the animals.

5.1.3 Birds

Regarding the breeding birds, the approach in the second round AAs was based on assumptions and observations on distribution, flux, flying height et cetera. For the studied species, the lesser black-backed gulls, the distribution over the area adjacent to their colonies seems to be very different from what was previously thought. Birds from the southern colony (in Volkerak) did not forage at sea, except for two flights in a whole breeding season (Gyimesi et al. 2011). The new spatial distribution of foraging birds from the Texel colony (Camphuysen 2011) seems sufficiently consistent to be used in the new calculations of possible bird collisions (other parts of the methodology are discussed in chapter 5 of this report). Other birds that were considered in the AA were the northern gannet and the storm petrel. Since these species do not breed on Dutch territory, no studies could be undertaken to tag them also with GPS transmitters. For these species, the assumptions used in the second-round AAs still seem appropriate.

5.2 Effects of underwater noise from OWF piling

5.2.1 Underwater noise: methodological standardisation

When it comes to using the concepts, terminology, metrics and standards in describing underwater noise from piling and operation of OWFs in future AAs, the advice given in De Jong et al. (2011) and Ainslie et al. (2011) should be followed (see their chapter 4). This would allow for comparison among different future studies and assessments

5.2.2 Underwater noise: source level and propagation modelling

Although the experiments at Kinderdijk were successful, the model development is not yet finished. The report on the model development contains no concrete results yet of any pragmatic value for the update of the 2008 Framework. Current research is aiming for the further development of this model. Moreover, it needs to be validated under realistic circumstances, i.e. the predictions from the model need to be verified by independent field measurements during piling operations.

5.2.3 Effects on fish larvae and trophic effects on birds and mammals

For sole, the worst-case assumption would be 100% larval mortality up to a distance of 400 m and 14% mortality at a distance of 400-1000 m from a ‘typical’ North Sea piling site. This adapted worst-case approach would lead to a reduction of ±50% of the effects estimated in the 2008 Framework for common sole. For other species than sole it is advised to maintain the 1000 m / 100% mortality criterion until new results may indicate other ranges

Additionally, to improve the modelling exercise of mortality and transport of larvae, it might be worthwhile to further develop the model (e.g. include natural mortality and include more species) and to validate/calibrate the model using the larval distribution data from the larval survey carried out in the Shortlist program (see above).

Regarding the trophic transfer of the reduced availability of prey fish for birds and mammals in Natura 2000 areas, there has been no methodological improvement compared to what has been done in the second-round AAs. For new AAs, it is advised to follow the procedure described in these AAs (e.g. Arends et al. 2008).

5.2.4 Effects on marine mammals

As a short-term solution it is advised is to (re)use the avoidance distances mentioned in the 2008 Framework, with a minor modifications for harbour porpoises, based on recent field observations of seals and porpoises during piling of offshore wind farms in the North Sea and the Baltic Sea.

Based on field studies, an avoidance distance of 20 kilometres seems appropriate. This is an increase in avoidance distance compared to what was used in the earlier AAs for the second-round OWFs (12 kms).

Although results or harbour seals are equivocal, the advice based on a worst-case approach is to maintain the avoidance distance of 80 kms, and to stress that additional field observations and experimental work is needed to elucidate this controversy.

5.3 Operational effects of OWFs

5.3.1 Marine mammals

Since the theoretical considerations of operating wind farms and field observations strongly suggest the effect of operational OWFs on seals and porpoises to be (near) negligible or even positive (see e.g. Lindeboom et al. 2011, Scheidat et al. 2011), there is no urgency to adapt the threshold distances from the 2008 Framework.

5.3.2 Effects on birds

The calculations for assessing possible collision victims and loss of habitat in future AAs need to be updated as follows:

1. *Collision victims*
   Future AAs need to include the updated (SNH-)Band model as the standard methodology for calculating possible collision victims for migrating birds and foraging birds from breeding birds onshore. This inclusion of ‘Route 3’ is in addition to the already standard use of ‘Route 2’, which should be kept as it was. Of course, within an AA, this is only relevant for species relating to Natura 2000 areas. Models to assess cumulative effects on population dynamics of breeding birds are available.
2. *Habitat loss:*
   Species-specific vulnerability and population sizes can be re-assessed, as is the case for species-specific avoidance rates of OWFs. Assessing habitat loss / disturbance is of course relevant only where these effects reach (directly or indirectly) to species of Natura 2000 sites. The AA has to specify this.
3. *Barrier effect:*
   The barrier effect of OWFs has so far not been considered to give possible significant negative effects on birds. It is likely that this will also not be the case for each future OWF separately. However, when the Netherlands and its surrounding countries Belgium, United Kingdom and Germany continue with their plans for future OWFs in the southern North Sea, the cumulative effects of such OWFs can be considerably large. In the near future, the necessity of these calculations needs to be reconsidered.

List of abbreviations

AA	Appropriate Assessment
DCS	Dutch Continental Shelf
EIA	Environmental Impact Assessment
Leq	Equivalent sound level (dB re 1 μPa^2)
Lp	Broadband sound pressure level (dB re 1 μPa^2)
MSP	Mean Squared Pressure Level
OWEZ	Offshore Windfarm Egmond aan Zee
OWF	Offshore Wind Farm
PAWP	Prinses Amalia Wind Park (*Princess Amalia Wind Farm)*
SEL	Sound Exposure Level (dB re 1 μPa^2.s)
SPL	Sound Pressure Level (dB re 1 μPa^2)
PTS	Permanent Threshold Shift
TTS	Temporary Threshold shift
T0	Period before wind farm construction
Tc	Period of wind farm construction
Top	Period of wind farm in operation
Tdec	Period of deconstruction of a wind farm

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