ENVIRONMENTAL IMPACTS OF OFFSHORE WIND FARMS IN THE BELGIAN PART OF THE NORTH SEA

ENVIRONMENTAL IMPACT MONITORING RELOADED

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EDITED BY:

Steven Degraer (steven.degraer@naturalsciences.be)
Robin Brabant (robin.brabant@naturalsciences.be)
Bob Rumes (bob.rumes@naturalsciences.be)
Laurence Vigin (laurence.vigin@naturalsciences.be)

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*S. Degraer[•*1], B. Rumes[•1], M. Alsebai[2], D. Botteldooren[3], R. Brabant[1], W. Courtens[4], E. Debusschere[2,5], J. Derweduwen[5], I. De Mesel[1], A. Erkman[6], J. Haelters[7], K. Hostens[5], F. Kerckhof[1], R.-M. Lafontaine[8], Y. Laurent[8], T. Moens[2], A. Norro[1], K. Persoon[5], J. Ranson[5], J. Reubens[2,9], A. Smith[10], E.W.M. Stienen[4], A. M. Ribeiro da Costa[5], E. Torreele[5], J. Vanaverbeke[1], S. Vandendriessche[5], N. Vanermen[4], M. Van de walle[4], H. Verstraete[4], L. Vigin[1], J. Vidao[10], M. Vincx[2], J. Wittoeck[5]*

[•]shared first authorship

[*]Corresponding author: Steven.Degraer@naturalsciences.be

[1] Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management (MARECO), Gulledelle 100, 1200 Brussels; 3de en 23ste Linieregimentsplein, 8400 Ostend; Vautierstraat 29, 1000 Brussels, Belgium.

[2] Ghent University, Biology Department, Marine Biology Research Group, Krijgslaan 281, Campus Sterre - S8, 9000 Ghent, Belgium.

[3] Ghent University, Department of Information Technology, Research Group Acoustics, Sint-Pietersnieuwstraat 41, 9000 Ghent, Belgium.

[4] Research Institute for Nature and Forest, Kliniekstraat 25, 1070 Brussels (Anderlecht), Belgium.

[5] Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Ostend, Belgium.

[6] Rijkswaterstaat, Poelendaelesingel 18, 4335 JA Middelberg, the Netherlands.

[7] Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Management Unit of the North Sea Mathematical Models (MUMM), 3de en 23ste Linieregimentsplein, 8400 Ostend, Belgium.

[8] Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Conservation Biology, Vautierstraat 29, 1000 Brussels, Belgium.

[9] Flanders Marine Institute (VLIZ), Wandelaarkaai 7, 8400 Ostend, Belgium.

[10] DeTect, Inc. 1022 West 23rd St., Suite 620, Panama City, FL 32405, USA.

INTRODUCTION

Offshore renewable energy development in the Belgian part of the North Sea has matured since our previous report in 2013. At present, nine Belgian projects representing a capacity of 2.2 GW were granted both a domain concession and an environmental permit. Three projects are operational, one is under construction, and the last five will need to be constructed in the near future, if Belgium is to meet its 2020 targets for renewable energy. These latter include the Mermaid project, which will generate a certain amount of energy from waves as well as wind. By 2018-2019, the number of wind farms constructed will have doubled with the realization of the Nobelwind, Rentel and Norther projects. The latter will entail the installation of the largest individual turbines (8.4 MW – reaching 187 m above mean sea level) in our waters. The near future may also see the first co-use of wind farm zones with aquaculture projects being developed in the areas of the C-Power and Belwind wind farms. In the meanwhile the electricity grid is undergoing necessary reinforcements, both onshore with the Stevin project and offshore with the proposed shared connection or 'plug-at-sea'. In the adjacent Dutch wind farm zone, the Borssele project entails the installation of 1.4 GW of wind energy and the resultant transboundary wind energy zone requires both consistent management measures and a comprehensive environmental monitoring program that adequately assesses cumulative environmental impacts.

To allow for a proper evaluation and auditing of the environmental impacts of offshore wind farms, the environmental permit includes a monitoring program to ensure (1) the ability to mitigate or even halt the activities in case of severe damage to the marine ecosystem and (2) an understanding of the environmental impact of offshore wind farms to support policy, management and design of future offshore wind farms. The former is tackled mainly by the basic monitoring program, the latter by the so called targeted monitoring program. In 2014 the existing basic environmental monitoring program was evaluated and a decision was made to focus on integrating work on several ecosystem components and streamlining research efforts (see below, reloading basic environmental impact monitoring).

The monitoring program targets physical (i.e. hydro-geomorphology and underwater noise), biological (i.e. hard substrate epifouling and fish communities, soft substrate macrobenthos, epibenthos and fish, seabirds and marine mammals), as well as socio-economical (i.e. seascape perception and offshore renewables appreciation) aspects of the marine environment although not all components are yearly studied or extensively reported on. The Operational Directorate Natural Environment (OD Nature) of the Royal Belgian Institute of Natural Sciences coordinates the monitoring and specifically covers hydro-geomorphology, underwater noise, hard substrate epifauna, radar detection of seabirds, marine mammals and socio-economic aspects. In 2014 and 2015, OD Nature further collaborated with different institutes to complete the necessary expertise in the following domains: seabirds (Research Institute for Nature and Forest, INBO), soft substrate epibenthos and fish (Institute for Agricultural and Fisheries Research, ILVO-Fisheries), and soft substrate macrobenthos (Marine Biology Research

Group, Ghent University). For details on the specific research strategies followed and methodologies used, one is referred to the individual chapters.

RELOADING BASIC ENVIRONMENTAL IMPACT MONITORING

The knowledge and expertise in relation to sampling technicalities and designs for offshore wind farm (OWF) monitoring gained from the Phase I basic monitoring (2005, 2008-2016; Degraer *et al.,* 2013) was revisited and discussed during a workshop with all scientists involved in the program, external experts and invitees from the OWF industry. The workshop focused on (1) How best to deal with variability (natural, anthropogenically induced) and spatio-temporal gradients?; (2) How to continue and optimise the basic monitoring program?; (3) How to plan the most appropriate sampling design for the basic monitoring program? An adapted monitoring program for the benthic and the pelagic realm was formulated, which excludes as far as possible sources of noise in the data by means of an adaptation of the sampling design. Management-relevant sources of variability in the data (i.e. benthic realm: e.g. distance to the coast, sedimentology, foundation type; pelagic realm: e.g. distance to the coast, seasonality) are used as explicit drivers for restructuring the monitoring program.

RESULTS OF THE ENVIRONMENTAL MONITORING PROGRAM

OPERATIONAL UNDERWATER SOUND EMISSION

Previous reports (e.g. Norro *et al.* 2013) analysed the underwater impulsive sound produced during construction activities. In the current report, the continuous underwater sound emitted by steel jacket and monopile foundation wind turbines is quantified, characterized and compared for low wind speeds (0-12 m/s). A maximum increase of SPL of about 20 dB re 1 µPa is observed at frequencies below 3 kHz. The addition of underwater sound increases with wind speed with a rate dependent on the type of foundation. For a mean wind speed of 10 m/s, a steel monopile will emit some 10 dB re 1 µPa more than a jacket foundation. Work is ongoing to expand this study to higher wind speeds and to quantify and qualify the cumulative effect of adjacent wind farms. Possible impacts on marine life like fish, marine mammals or invertebrates remain unclear mainly due to the lack of knowledge in disturbance or behavioral response levels for the species found at these sites.

EXPANSION OF SMALL-SCALE CHANGES IN MACROBENTHIC COMMUNITY INSIDE AN OFFSHORE WIND FARM?

Changes in hydrodynamics, presence of epifaunal coverage along the turbine and fisheries exclusion are expected to be the main causes influencing the macrobenthic community inside a wind farm. In this report we investigate whether previously observed changes in sediment characteristics and macrobenthic community (Coates *et al*., 2014) can also be observed at larger distances from the turbines. Stations in the close vicinity of the turbines (50 m distance, close samples) and further away (350-500 m distance, far samples) were sampled with a Van Veen grab in autumn 2015. No significant differences in abiotic factors are observed between the two distances. All samples are characterized by coarse sediments, with a low mud and total organic matter contents. Macrobenthic densities on the other hand differ significantly between the two distances with both higher densities and number of species for the far samples compared to the close samples. The latter are dominated by the amphipod *Urothoe brevicornis* and the mysid shrimp *Gastrosaccus spinifer*, while the amphipod *Bathyporeia elegans* and the polychaete *Spiophanes bombyx* are more abundant in far samples. Although this might be related to the turbine type, it remains unclear what underlying ecological processes are responsible for the difference in community structure between both distances as the current results are not consistent with results from previous studies. The current sampling design will be continued for the coming years. A targeted monitoring study will be required to elucidate changes in sedimentology and organic enrichment in the close vicinity of different turbine types.

EFFECT OF BELGIAN WIND FARMS ON THE EPIBENTHOS AND FISH OF THE SOFT SEDIMENTS

Many studies have demonstrated the reef effects on epibenthos and fish in the immediate vicinity of the turbine foundations (e.g. Reubens *et al.*, 2013, Bergström *et al.*, 2014), but the influence on demersal fish in the wider wind farm area is less clear (van Hal *et al.*, 2012; Bergström *et al.*, 2013). In Belgian wind farms, Vandendriessche et al. (2015) indicated several wider wind farm effects, including an increase in epibenthos biomass and densities and a possible ‘refugium effect’. By including the period 2013-2014, earlier observed positive short-term effects seem to have disappeared, and should be seen as a short-term reaction of opportunistic species directly after construction. Also, the earlier reported signals of a ‘refugium effect’ are no longer observed. For sandeel (*Ammodytes tobianus*), episodic increases and short-term positive effects on juveniles are observed, but no clear long-term sandeel trends are visible. Long-lived species are not yet encountered but may get a chance to establish and recover when the ongoing expansion of the wind farm area extends to one large continuous no-trawling area.

To investigate the effect of wind farms on the feeding behaviour of demersal fish, stomach content analyses were performed for lesser weever (*Echiichthys vipera*) and dab (*Limanda limanda*) in and around the C-Power wind farm. For both species there are no significant differences in stomach fullness inside or outside the wind farm. However, since the presence of the wind mill foundations, both fish species consume more prey species that are directly associated with hard substrates, both inside and in the direct vicinity of the wind farm. This demonstrates the expanding reef effect into the surrounding soft sediments.

THE EFFECTS OF HIGH INTENSITY IMPULSIVE SOUND ON YOUNG EUROPEAN SEA BASS (*DICENTRARCHUS LABRAX*), WITH SPECIAL ATTENTION TO PILE DRIVING

Pile driving generates strong impulsive noise that can affect the health and wellbeing of marine life. The impact of pile driving on young European sea bass (*Dicentrarchus labrax*), more specifically, the acute and delayed mortality, acute and chronic physiological stress responses and the impact of lower intensity impulsive sound on the fish behaviour were assessed through field and laboratory experiments (Debusschere, 2016). A field experiment at 45 m from the pile driving activity revealed no acute or delayed mortality but the fish showed strong acute secondary stress responses, a 50% decrease in oxygen consumption rate, in addition to behavioural responses as could be observed in laboratory experiments. Juvenile fish reduced their swimming activity and ceased all attacks on conspecifics at the onset of the impulsive sound exposure, but showed behavioural recovery within 25 minutes. The results also showed that the initial response change under repeated exposure. More research on multiple species and at population level are required as well as long-term data, especially on behavioural responses, in order to determine the ecological relevance of pile driving effects on young fish.

SEABIRD MONITORING AT OFFSHORE WIND FARMS IN THE BELGIAN PART OF THE NORTH SEA

Improvements to the modelling strategy of the long-term seabird monitoring program show significant avoidance by northern gannet (*Morus bassanus*) and common guillemot (*Uria aalge*) and attraction by great black-backed gull (*Larus marinus*) at the first two Belgian wind farms. Lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*) and Sandwich tern (*Thalasseus sandvicensis*) appear to be attracted to only one wind farm. While the avoidance of common guillemot and northern gannet seems readily interpretable from a disturbance perspective, it is still difficult to pinpoint the observed increases in seabird numbers, even more so because these are not always consistent between study sites. Gaining more insight in the diurnal and tidal-dependent variation in numbers and behaviour of birds occurring inside the offshore wind farms seems indispensable for understanding the observed patterns and learning whether birds come to the wind farms merely for roosting and the related stepping stone function, or whether offshore wind farms also offer increased food availability. This will need to be investigated through targeted research using bird radar data, GPS tracking data of tagged gulls, fixed cameras and/or visual observations from a fixed location inside the wind farm.

SEASONAL AND INTERANNUAL PATTERNS IN THE PRESENCE OF HARBOUR PORPOISES (*PHOCOENA PHOCOENA*) IN BELGIAN WATERS FROM 2010 TO 2015 AS DERIVED FROM PASSIVE ACOUSTIC MONITORING

Passive acoustic monitoring data of harbour porpoise from the period 2010 and 2015 reveal a significant seasonal trend in detections with peaks in late winter - early spring and late summer, consistent with both results of aerial surveys and strandings data. The experiences gained are used to design a strategy to monitor the effects of offshore wind farm construction and operation on harbour porpoises in Belgian waters.

ANTICIPATING FUTURE DEVELOPMENTS

UNDERWATER NOISE REGULATIONS FOR PILING NOISE IN BELGIUM AND THE NETHERLANDS

From 2017 onwards, new regulations with regard to impulsive underwater noise will make it necessary to use noise mitigation measures during piling in the Belgian wind farm zone and the adjacent Dutch wind energy zone of Borssele. However, these regulations are quite different and at times even contradictory and developers could benefit from an alignment of regulatory practices on a regional basis. Measurements of piling noise from constructed wind farms are used to extrapolate the anticipated noise levels of the next two wind farms to be constructed, and these are evaluated in relation to the regulations on underwater sound. Wind farm developers are already developing strategies for cost-effective piling noise reduction but uncertainty remains with regards to both the level of underwater noise produced during piling as well as with the effectiveness of the noise mitigation measures being applied. Our results indicate that a combination of noise mitigation measures will need to be used to comply with regulations.

RECREATIONAL FISHERMEN AND WIND FARMS

The closure of offshore wind farms for commercial fisheries combined with the installation of artificial hard substrates has favorably affected demersal and benthopelagic fish in the wind farm zones and could thus, in theory, provide opportunities for recreational fishermen. However, in Belgium, recreational fishermen are not allowed in the wind farm area and have to keep a minimum distance of 500 m from the turbines. As a result, less than 2% of Belgian recreational fishermen reported to go fishing in the larger wind farm area, even when 30% to 40% of the respondents either expected more fish, bigger fish or other fish species inside the wind farm. Data were derived from the annual fisheries Data Collection Framework survey for recreational fishermen. 40% of the respondents would consider fishing inside wind farms if it were allowed.

This is a clear indication that the enforcement of wind farm closure for fisheries and shipping is vital when aiming at the creation and/or restoration of nursing grounds in the area. However, the large distance to the wind farms will probably continue to limit fishing pressure, even if wind farms would (partly) be opened for recreational fisheries.

WIND FARMS AND THEIR INFLUENCE ON THE OCCURRENCE OF ICHTYOPLANKTON AND SQUID LARVAE

The expected large scale increase in wind farms is expected to influence both fish and cephalopod egg deposition by modifying the sea floor and providing additional egg deposition opportunities respectively. This is expected to manifest as higher densities of early life stages at the hard substrates (eggs) and in the water column (larvae) at the wind farms. This was investigated at the Thornton bank wind farm by repeatedly sampling three impact stations and three reference stations with a Bongo net from 2010 to 2013. The results do not show significant effects of the wind farm on fish eggs, fish larvae and squid larvae. However, the data provide good baseline information about ichthyoplankton and squid larvae at offshore stations that can be used in future monitoring.

DO WIND FARMS FAVOUR INTRODUCED HARD SUBSTRATA SPECIES?

Offshore wind farms, like other artificial structures in the marine environment, are hypothesised to favour introduced species and as such pose a threat to the native fauna. Previous reports described the colonization of this new habitat (Kerckhof *et al.,* 2010) and the emerging prominence of introduced species in the intertidal zone (Kerckhof *et al.,* 2011). In this report, we investigate introduced species on Belgian offshore wind farms with particular interest in (1) the position of introduced species on offshore wind farms in relation to other hard substrata in the Belgian part of the North Sea (BPNS), (2) the distribution of introduced species in the subtidal versus intertidal zone and (3) the potential of offshore wind farms for future flourishment of the introduced species. Overall eleven introduced and two cryptogenic species are observed on the wind turbines, seven of which are intertidal species and four are subtidal species. All but one introduced species observed on the offshore wind farms in Belgian waters (i.e. *Fenestrulina delicia*), is already known from the BPNS. In the subtidal zone, the offshore wind farms will only marginally contribute to the further spread of introduced species given the vast amount of both natural and artificial hard substrata already available in the North Sea, which already contain established populations of the same introduced species. However, for the intertidal zone, the wind farms may have the potential to substantially increase the risk of the further spreading of introduced species, given that offshore intertidal habitat still is relatively rare. It is however expected that offshore wind farms may significantly contribute only to the spread of clear water, intertidal introduced species, as such nuancing the introduction and invasion risk posed by offshore wind farms.

BIRD RADAR STUDY IN THE BELGIAN PART OF THE NORTH SEA: DEVELOPMENTS TO IMPROVE BIRD DETECTION

Dedicated bird radars are used in ornithological studies as they provide continuous data on a large scale for many years. However, the recorded radar data have a low taxonomic resolution and contain a lot of clutter i.e. records of objects other than birds (e.g. sea surface, ships, rain). A filter has been developed based on the differences in target characteristics as recorded by the radar, which allows removing as much clutter as possible from the vertical radar data. The filter tests showed very high scores for the criteria accuracy, sensitivity and specificity. However, a relatively high number of false positives remains in the model results. This will be improved in the future by including variables in the decision tree analysis which are linked to the bird track level, instead of only using the variables recorded by the radar which describe the single point records. This will result in a more accurate bird flux and therefore an improved outcome of bird collision models.

BATS IN THE BELGIAN PART OF THE NORTH SEA AND POSSIBLE IMPACTS OF OFFSHORE WIND FARMS

To evaluate and quantify the risk of offshore wind farms in the southern North Sea to bat populations we need first to determine the spatio-temporal distribution of bats in Belgian waters. During two full bat migration periods an automated acoustic recorder was installed on the Belgian research vessel ‘Belgica’ to record bats while the vessel is at sea at night. Over a hundred call sequences belonging to four different species were registered although calls were limited to only a few nights (Brabant *et al.,* 2016). In 2015 and 2016, an expanded network of nine Batcorders was collecting data in the Dutch and Belgian part of the North Sea and along the coastline. This detector network will increase our knowledge about the impact of offshore wind farms on bats as it will increase the number of detections of bats at sea and will allow direct comparison between data collected at the different locations, without seasonal or meteorological bias.

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CHAPTER 1

CHAPTER 1

RELOADING BASIC ENVIRONMENTAL MONITORING OF OFFSHORE WIND FARMS IN BELGIUM: PHASE II

S. Degraer[*]*, R. Brabant & I. De Mesel*

Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels; 3de en 23ste Linieregimentsplein, 8400 Oostende, Belgium.

[*]*Corresponding author: Steven.Degraer@naturalsciences.be*

SUMMARY

Lots of knowledge and expertise in relation to sampling technicalities and designs for offshore wind farm (OWF) monitoring were gained from the Phase I basic monitoring (2005, 2008-2016). Based on this knowledge, the sampling design for the basic monitoring, focusing on the detection of the long-term effects of OWFs, was revisited and discussed during a workshop with all scientists involved in the programme and invitees from the OWF industry. The workshop focused on (1) How to best deal with variability (natural, anthropogenically induced, spatio-temporal gradients)? (2) How to continue and optimise the basic monitoring programme? (3) How to plan the most appropriate sampling design for the basic monitoring programme? These issues were discussed in two subgroups covering the benthic and pelagic realm *sensu lato*; this to allow for a maximal

accommodation of the ecosystem component sampling programmes within each of the two realms. For each realm, distinction was made between variability that is of no interest in an offshore wind farm advisory setting (i.e. unexplained variation) that can either be excluded or that cannot be excluded, and variability in which we are interested and hence has to be an integral part of the monitoring design. All sources of variability were explored and categorized into one of these three types of variability. Possible sources of unexplained variation were excluded to the maximum by means of an adaptation of the sampling design. If this was not possible, these sources of variation were integrated in the monitoring programme and included as co-variables in the analysis. Management-relevant sources of variability in the data (i.e. benthic realm: e.g. distance to the coast, sedimentology, foundation type; pelagic realm: e.g. distance to the coast, seasonality) were used as explicit drivers for restructuring the monitoring programmes. An overview of the adapted monitoring programme for the benthic and the pelagic realm is presented.

1.1. INTRODUCTION

The first monitoring activities in the framework of the impact assessment of offshore wind farms in the Belgian part of the North Sea (BPNS) started in 2005. The objective was to gather reference data and to identify appropriate reference areas. The impact monitoring itself started in 2008, when the first six wind turbiness were constructed in Belgian waters. At first, the main focus was to come up with an appropriate methodology and monitoring design, to get at full speed from 2009 onwards. From then onwards, a distinction was made between basic and targeted monitoring. The basic monitoring is aimed at assessing the extent of the long-term impacts on the different aspects of the marine ecosystem and is therefore focusing on the *a posteriori*, resultant impact quantification. Targeted monitoring on the other hand deals with the understanding of the processes behind the impacts of a selected set of hypothesized cause-effect relationships highly relevant to the environmental impact assessment and is an important input for scientifically sound advice with regards to future projects. Only the basic monitoring programme is considered in this chapter.

The ministry responsible for the North Sea agreed to continue an integrated monitoring of the impact of offshore wind farms until at least 2023. Before the start of the second phase of the monitoring (2015 – 2023), the Operational Directorate Natural Environment (OD Nature) of the Royal Belgian Institute of Natural Sciences (RBINS) legally responsible for the execution of the monitoring programme, organised a workshop to evaluate how to optimise the basic monitoring programme. Over 30 participants from different research institutes, universities and the industry involved discussed for two days (28 – 29 October 2014) what has been achieved so far, what issues came up, how these could possibly be solved and hence, how to best continue the monitoring programme from 2016 onwards.

The workshop focused on (1) How to best deal with variability (natural, anthropogenically induced, spatio-temporal gradients)? (2) How to continue and optimise the basic monitoring programme? (3) How to plan the most appropriate sampling design for the basic monitoring programme? These issues were discussed in two subgroups covering the benthic and pelagic realm *sensu lato*. The benthic subgroup tackled the questions with regards to the ecosystem components sedimentology, macrobenthos and demersal fish. The pelagic subgroup covered (bentho-)pelagic fish, marine mammals, plankton, underwater sound as well as (sea)birds and bats.

The final conclusions allowed adjusting the Belgian basic monitoring programme where needed and set out the guidelines for the next phase of the monitoring. This chapter therefore aims at (1) providing an overview of basic monitoring programmes and their results until 2014; (2) scoping for a higher level of integration between the programmes; and (3) designing an enhanced basic monitoring programme for execution from 2015 onwards.

1.2. OVERVIEW OF THE MONITORED ECOSYSTEM COMPONENTS: 2005-2013

SEDIMENTOLOGY

The research of RBINS, OD Nature SUMO (Suspended Matter and Seabed Monitoring and Modelling) research team was aimed at quantifying the changes in turbidity and in the processes structuring the seabed during and after the construction of wind farms (turbine foundations and cable routes). Long-term measurements in combination with modelling techniques allowed predicting short- and long-term effects. Focus was also put on the dredging and sediment dumping activities related to the construction of the wind farms. Significant losses of sediment were observed, especially during the construction of the gravity based foundations.

Recent satellite images of turbidity wakes related to the wind turbines will contribute to quantifying the origin, dynamics and effects of these wakes. It is hypothesized that these wakes consist of recently accumulated biogenic deposits. This material will possibly be dispersed to a wider area due to these wakes.

SUMO is currently specializing in wake modelling and aims at using this knowledge in the impact monitoring of the wind farms. Because sediment wakes are produced by various anthropogenic activities, it is necessary to study the cumulative effects and to assess how the increase of fine sediments is buffered in the seabed, and how this is influencing the integrity of the bottom of the sea.

MACROBENTHOS OF THE SOFT SUBSTRATES

The research of the Marine Biology Research Group (Ghent University) focused on community structure, density, diversity and biomass of the macrobenthos of the soft substrates. Based on these data, the Benthos Ecosystem Quality Index (BEQI) was calculated, which is used by Belgium as an indicator within the Water Framework Directive and the Marine Strategy Framework Directive. The results showed that the macrobenthos (community composition, BEQI) is influenced by the disturbance due to the construction of a wind farm. This effect was however temporary. No large scale effects on the macrobenthic community could be observed during the operational phase of the wind farm. This might partially be explained by the fact that most samples were collected at the edge of the wind farms. Sampling locations inside the wind farms are therefore absolutely required in the next monitoring phase.

SOFT SUBSTRATE EPIBENTHOS AND ASSOCIATED FISH

The basic monitoring focused on wind farm effects and fringe effects of the redistribution of fisheries activities. This study executed by the Research Institute for Fisheries and Agriculture (ILVO), included several variables (density, biomass, diversity and species composition) of three ecosystem components (epibenthos, demersal fish and benthopelagic fish) in two seasons (spring and autumn), at two sandbanks (Thornton and Bligh Bank) and two sandbank habitats (sandbank tops and gullies). The density and length-frequency distribution of a few selected species were monitored in detail.

The data showed significant BACI-effects and significant effects within a specific year, both on the Thorntonbank and on the Bligh Bank. The number of ophiuroids (serpent stars) on the Bligh Bank in 2009 for instance, was significantly lower in the impact area compared to the reference area. Density of sole *Solea solea* was much higher in 2012 at the edge of the wind farm on the Bligh Bank, compared to the reference area. Dab *Limanda limanda* specimens were significantly smaller in the impact area on the Thorntonbank in 2012, than in the reference area.

Taking into account that the wind farms are relatively new and that monitoring of the epibenthos and demersal fish has only been possible for three years, it is of great importance to continue the monitoring of this ecosystem component.

EPIFAUNA OF THE HARD SUBSTRATES

The basic monitoring of the epifauna on the hard substrates executed by the Marine Ecology and Management section (MARECO) of RBINS, focused on the intertidal and subtidal (-15 m) parts of the turbine foundation and the rocks of the scour protection. Visual surveys and qualitative samples were used to study the intertidal, while video sequences and photographs completed quantitative samples in the subtidal and the collection of rocks from the scour protection. Both in Belwind and in C-Power, we always tried sampling at the same turbine. This was done seasonally.

The number of non-indigenous species (NIS) found in the intertidal samples was proportionally high (50%). The subtidal fouling community stabilised rapidly, with a dominance of a limited number of species and seasonal dynamics. The proportion of NIS in the subtidal samples was rather low. Differences in the fouling community between the Thornton Bank and the Bligh

Bank might be caused by the location of the foundation along the onshore-offshore gradient and/or by the type of substrate (concrete versus steel wind turbine foundations). The rocks of the scour protection harbor a larger number of species and this community is still developing.

HARD SUBSTRATE ASSOCIATED FISH

Hard substrate fish monitoring was conducted by UGent's Marine Biology Research Group between 2009 and 2012 at a gravity-based foundation (GBF) in the C-Power wind farm and focused on the community structure of the fish associated with the hard substrates. A hard substrate (shipwreck) and a soft substrate (sandbank) were assigned as control areas. The samples were collected every two weeks or every month with a fishing rod and by divers (visual observation; only at the GBF).

The samples, which contained 24 species in total, were dominated by Atlantic cod *Gadus morhua* and pouting *Trisopterus luscus*. The density of both species was much higher around the GBF compared to the shipwreck and the sandbank. The abundance of both species however varies seasonally, with highest densities in autumn. Cod specimens were mainly individuals from year class 1 and 2, for pouting this was year class 0 and 1. Year class 0 cod specimens were encountered in spring (May – June) in both C-Power and Belwind in several years. These individuals were circa 5 cm and therefore became benthopelagic only very recently.

SEABIRD

The impact of offshore wind farms on the density and distribution of seabirds was studied by the Research Institute for Nature and Forest (INBO) by means of a BACI design. Ship-based seabird surveys were conducted along fixed monitoring tracks through impact and reference areas following an international standard methodology. Three years of 'post-impact' monitoring on the Bligh Bank and surrounding areas showed that Northern gannet *Morus bassanus*, guillemot *Uria aalge* and auk *Alca torda* avoid the wind farm and that the numbers respectively decreased with 85%, 71% and 64%. The number of lesser black-backed gull *Larus fuscus* and herring gull *Larus argentatus* increased with a factor 5.3 and 9.5, respectively. The 'post-impact' monitoring on the Thorntonbank is currently ongoing.

The ecological motives explaining the attraction of certain species are unclear at this point, but aside from an increased availability of roosting locations, an increased food availability is a most plausible explanation. It is important to mention that the attraction of seabirds in the wind farms results in a higher risk of collision with the structures.

Aside from the seabird surveys, there is also a continuous monitoring of birds to study the impact of wind farms, making use of a bird radar (executed by MARECO). The goals of this study are (1) to assess to what extent wind farms act as a barrier to local and migrating birds and (2) to quantify the temporal variability (e.g. seasonal, diurnal) in bird fluxes through the wind farm area.

Based on the results of the visual surveys and the radar measurements we estimated the number of birds colliding with the turbines, using a mathematical bird collision risk model (CRM). The number of casualties per turbine per year [lower and upper 95% confidence intervals] in the wind farm at the Bligh Bank for the six most dominant seabird species is estimated at 1.8 [0.4; 12.5]. During one night of intense passerine migration, the CRM estimated 28 collision victims in the wind farm at the Thorntonbank.

UNDERWATER SOUND

The underwater sound level was measured by MARECO before and during the construction of the wind farms. The background level at these locations is about 100 dB re 1µ Pa SPL. During the construction of monopile and jacket foundation, steel piles are hammered into the seabed. This is creating excessive underwater sound levels, varying between 189 to 196 dB re 1µ Pa (zero to peak level (Lz-p), normalized at 750 m distance). These sound levels exceed the background level at a distance up to 70 km from the piling location.

MARINE MAMMALS

The monitoring of marine mammals executed by MARECO, is limited to the harbour porpoise *Phocoena phocoena*, as this is the only common species in the BPNS and it is regarded as most sensitive to underwater sound.

Three methods were used: Passive Acoustic Monitoring (PAM), Line Transect (aerial) Surveys (LTS) and (tested in 2014) Strip Transect (aerial) Surveys (STS; digital). PAM results in a (corrected) measure of presence – absence of porpoises at a certain location. LTS and STS render density and distribution figures. By the end of 2015, 3605 days of PAM data were collected (2010 – 2014 at four locations). 22 aerial surveys covering the entire BPNS were conducted. This resulted in valuable spatio-temporal data on distribution, number and presence of harbour porpoises. There are clear indications of disturbance during piling activities.

1.3. TOWARDS A BASIC MONITORING PROGRAMME PHASE II

DEALING WITH IMPACT-INDUCED VERSUS SPATIO-TEMPORAL GRADIENT-INDUCED VARIABILITY

To determine the ecological impact of an activity (i.e. offshore wind farm), the impact of that activity on a certain response variable (e.g. the density of a species) or multivariate community structure is investigated. The impact might be the change through time or the different evolution compared to a (not impacted) control or reference area. Both are often combined in ecological studies in a so-called BACI (Before-After Control-Impact) design. This allows comparing trends in the response variable.

Natural fluctuations of the response variable are causing variability in the data which is not linked to the investigated impact (i.e. statistical noise). Understanding the natural variability of the response variable is essential to include the right covariates, aiming to explain part of the data variability. Including the right covariates results in a lower chance of mistakenly interpreting a change in the response variable as an impact, while actually it is caused by an effect of (one of) the covariate(s). It also narrows confidence intervals and thus increases the statistical power.

The different sources of variation influencing the different ecosystem components were identified during the workshop. For instance, seabird density is influenced by e.g. seasonality, time of day, meteorological circumstances, the onshore-offshore gradient, fisheries activities, etc. All these co-variables are to be accounted for when assessing the impact of offshore wind farms on the seabird density.

WHICH VARIABLES INFLUENCE THE MARINE ENVIRONMENT AND HOW CAN THEY INFLUENCE THE IMPACT ASSESSMENT OF OFFSHORE WIND FARMS

Three types of variability were distinguished:

1. Variability in which we are not interested and which can be excluded with an appropriate sampling design (i.e. unexplained variation that can be excluded);
2. Variability in which we are not interested and which cannot be excluded (i.e. unexplained variation that cannot be excluded);
3. Variability in which we are interested in function of rendering advice in the framework of future wind farms and which should be covered by the basic monitoring programme.

The different sources of variation identified during the workshop, were allocated to one of these three groups and color-coded (1=red; 2=orange; 3=green; annex I).

Benthic Realm

Sources of unexplained variation to be excluded

For the benthic ecosystem components monitoring programmes several possible sources of unexplained variation in the data and therefore preferably to be excluded from the analysis, were identified. Seasonal variability and diurnal variability should be excluded because these do not contribute to our knowledge relevant to management advice. The same holds true for the variation linked to ‘distance to a turbine’. These sources of variation can be excluded or at least reduced by adjusting the sampling design.

Sources of unexplained variation that cannot be excluded

An understanding of the effect of year-to-year variability, hydrodynamics, suspended particulate matter and other human activities do not contribute to our knowledge relevant to management advice but are difficult to exclude from the analysis and will therefore be adopted as co-variables in the monitoring programmes.

Variability relevant for advisory purposes

Other variables are to be included in the analysis, because understanding of this variability is of great importance with respect to rendering advice for future projects. For instance, the different types of foundations which are used at present (i.e. jacket, monopile and gravity-based foundations) should be incorporated in the sampling design. This is also the case for the configuration of turbines in the wind farm, as the orientation relative to the dominant tidal current is important for the resulting sediment transport and consequent ecological effects. The scale of the project has an influence on the hydrodynamics and sedimentology, and is an important variable in the way offshore wind farms act as a stepping stone for (non-indigenous) species living on e.g. the foundations and scour protection.

Sediment type and the nearshore-offshore gradient are also important variables to include, because the location of the wind farms are likely to trigger different impacts. For example, very different faunal communities are present along the nearshore-offshore gradient. It is essential to include this gradient in the sampling design to understand the impact of the OWFs on these different communities. Sediment type is an important variable determining the macrobenthic community structure.

The pelagic realm

variability to be excluded

The pelagic realm subgroup identified ‘diurnal variation’ and ‘distance to a turbine’ as variables causing variability in the data and which should be excluded. Diurnal variation is not of importance when assessing the impacts of OWFs for most ecosystem components and can easily be excluded by sampling only during daytime. For birds (night time migration) and bats however, diurnal variability is of course relevant and should therefore be included in the analysis. Distance to a turbine is considered less relevant in the basic monitoring, except for underwater noise impact assessment during piling activities.

Variability that cannot be excluded

Several variables linked to temporal variability (e.g. year-to-year variability, tidal variability) are included as co-variable in the analysis, because it is not possible (or very difficult) to exclude these. Other human activities are also considered as co-variables which cannot be excluded.

The wind speed affects the operational underwater sound, being louder at higher wind speeds because of the higher rotation speed of the turbines.

Variability relevant for advisory purposes

The following variables should be included in the analysis, because these do contribute to our management-relevant knowledge of OWF impacts: nearshore-offshore gradient, seasonality, time/effect interaction, wind farm configuration and scale, and wind speed. The nearshore-offshore gradient is of particular importance as there are different faunal communities living to the Southern part of the Belgian renewable energy zone (e.g. Norther concession area) and North of the Thorntonbank (e.g. Belwind concession area). This is the case for e.g. seabirds, fish, plankton, marine mammals, bats. The Belgian wind farm zone also crosses the boundary between the turbid coastal waters and the clearer offshore waters of the English Channel. Its impact on pelagic fish is unknown at present. Telemetry data of fish might provide insight here. The bird research should focus on both the nearshore community (e.g. sandwich tern, common tern, little gull) and the offshore community (e.g. auk, guillemot). This approach would also allow assessing the effect of the foundation type.

Seasonality is of great importance for both birds and pelagic fish, but for different reasons. The seabird community is very different in the different seasons: in May and June large numbers of terns reside in the area (mainly nearshore, birds directive Annex I species); in September and October there is intense migration of little gull (birds directive Annex I species); in November intense migration of northern gannet occurs (mainly offshore); in winter, large numbers of auks and guillemots reside in the area. This is why monthly seabird surveys are required year-round. Accounting for seasonality in the analysis is necessary to be able to give specific advice about the expected effects and possible mitigating measures, e.g. terns are sensitive to collisions but are mainly present nearshore. At present, we lack knowledge on the distribution of pelagic fish except for some anecdotic observations and assumptions (e.g. Atlantic horse mackerel is regularly seen in the wake of the turbines; hard substrates around the turbines are of importance for eggs and larvae of pelagic fish; sea bass is attracted by the turbine foundations; do wind farms have an effect on the distribution of herring and sprat?). To gain more knowledge, a year-round monitoring (catches with nets and/or sonar imagery) is required.

A time/effect interaction is of potential importance for birds and marine mammals. For instance, in Denmark habituation was observed in the response of red throated divers to wind farms. Just after the construction of the wind farm they avoided the park completely. After some years they came back to forage at the edge of the wind farm, possibly attracted by the higher food availability inside the wind farms. Similar habituation was also observed in the behavior of seals (recent telemetry study).

Wind farm configuration and scale are important variables to take in account in the impact studies on birds and fish. Large, connected wind farms might have a larger refugium effect for fish. For birds, this might create a barrier to migration if flight corridors are not foreseen. The configuration of turbines of a wind farm, more specific the number of turbines per unit surface area, is also influencing the impact on birds.

1.4. ADJUSTMENTS/IMPROVEMENTS OF THE SAMPLING DESIGN FOR THE BASIC MONITORING PHASE II

The relevant sources of variation for the benthic and pelagic ecosystem components are identified and we distinguished between variation we want to understand in function of rendering advice and variation we do not need to understand in such advisory context (i.e. sources of unexplained variation). The latter can partially be excluded by adjusting the sampling design. The part which cannot be avoided is adopted as co-variable. Taking account of all this, a sampling design including the number of samples and timing of sampling was developed.

ADJUSTMENTS/IMPROVEMENTS OF THE SAMPLING DESIGN FOR THE BASIC MONITORING PHASE II OF THE BENTHIC REALM

All possible combinations of substrate type and type of foundation, along the on-/offshore gradient are presented in tables 1 and 2, per ecosystem component (table 1: demersal fish, epibenthos of the soft substrate, macrobenthos and hyperbenthos; table 2: epibenthos of the hard substrate).

Table 1. Sampling options and choices for the benthic ecosystem components (except epibenthos of the hard substrate). GBF = gravity based foundation, JF = jacket foundation, MP = monopile.

Timing	Autumn													
On-offshore	Nearshore						Midshore				Offshore			
Sediment type	Fine?		Coarse		Cobble?		Coarse				Coarse			
Foundation type	unknown		Unknown		Unknown		GBF		JF		MP		JF	
Distance from foundation	Far	close	Far	close	Far	close	Far	close	Far	close	Far	close	Far	close
Demersal fish/ epibenthos soft sediments	▨	▨	▨	▨	▨	▨			•		•		o	
Macrobenthos	▨	▨	▨	▨	▨	▨		•	•	•	•	•	o	o
Hyperbenthos	▨	▨	▨	▨	▨	▨			•		•		o	

Table 2. Sampling options sand choices for the epibenthos of the hard substrate.

Timing	Autumn									
On-offshore	Nearshore		Midshore				Offshore			
Foundation type	Unknown		GBF		JF		MP		JF	
Depth (subtidal/in-tertidal)	Sub	Inter	Sub	Inter	Sub	Inter	Sub	Inter	Sub	Inter
Epibenthos hard sub	▨	▨	•	•	•	•	•	•	o	o

Legend	
▨	Not yet elaborated due to high uncertainty of design of wind farm
▒	Not relevant
•	Effect included in monitoring
o	Will be included if situation will be present in the future

Seasonal variability is excluded by sampling only in autumn instead of sampling twice a year for the benthic ecosystem components. To rule out diurnal variability, samples will be collected as much as possible during daytime.

Until 2014, the variation along the nearshore-offshore gradient was focused on two points only (i.e. the Thorntonbank and the Bligh Bank), but this will be expanded in the new sampling design to three points along the gradient. Practically, this implies that it is not necessary to monitor every ecosystem component in each individual wind farm. Most efforts will be done inside Belwind, C-Power and Norther, respectively representing the offshore, midshore and nearshore location.

The aspect distance from a turbine was also added in table 1, as this cannot be entirely excluded from the analysis. It will however be reduced by sampling at two fixed distances from the turbines (i.e. “far” or “close” from/to a turbine). This is also important in the development of the sampling design. This distance will be different for the different ecosystem components, taking the practical restrictions into account of what is technically feasible. It is, for example, technically impossible to measure the effects close to a turbine for epibenthos and demersal fish as it impossible to trawl close to the turbines. The distance aspect or sediment type is not applicable to hard substrate epifauna (i.e. the fouling on the foundations), but here a distinction between intertidal and subtidal is made.

The phase I results of the macrobenthic study showed that the construction phase has a clear impact on the macrobenthic community, but that the impact disappeared during the exploitation phase. This can be due to the fact that there is no impact on the macrobenthos during the exploitation or that the sampling design (few sampling locations with several replicates per location) was not appropriate to detect it. Targeted monitoring however indicated that the macrobenthic community is impacted in the proximity of turbines. Therefore the sampling design will be adjusted in the phase II. From now on a randomized design will be used, which means that more locations inside the wind farms will be sampled but only one sample per location will be collected. The total number of samples will be more or less equal to the phase I monitoring. To determine the effect of the turbines, samples will be collected 'far' (ca. 250m) and 'close' (ca. 50m) to the turbines. Macrobenthic samples will be collected from communities typical for coarse sediments and fine silt sediment (i.e. *Abra alba* and *Ophelia borealis* communities) and possibly also from communities associated with natural gravel beds (at the Norther concession, to be investigated). In practice, samples will therefore be collected at the concession areas of Norther, C-Power and Belwind. It is still to determine which sediment types are present in the Norther concession area, so all options are left open (coarse sand, fine sand and silt, gravel). Combined with type of foundation (GBF, JF and MP) this leads to nine possible combinations. Depending on the seabed survey and the chosen type of foundation, the appropriate options will be selected.

Beam trawl samples to collect the epibenthic fauna and demersal fish species used to be collected in spring and autumn. In the phase II monitoring design this will be reduced to once a year (in autumn), to rule out seasonality. It is not necessary to collect samples in every wind farm along the near-offshore gradient. Considering the knowledge and experience gained from the C-Power and Belwind monitoring, sampling in these wind farms will be preserved. As we know that the Norther concession area holds an entirely different (nearshore) faunal community, it will be necessary to also collect beam trawl samples in that area.

The hyperbenthos (i.e. small sized bottom-dependent animals that live just above the seabed) was not monitored in the past. This was identified as a gap in the data during the workshop. A feasibility study to determine whether or not it is useful to include this ecosystem component in the monitoring programme, will be conducted.

In short, the benthic basic monitoring of phase II will focus on autumn samples to be collected only in three of the (future) eight wind farms, i.e. Norther (nearshore), C-Power (midshore) and Belwind (offshore).

ADJUSTMENTS/IMPROVEMENTS OF THE SAMPLING DESIGN FOR THE BASIC MONITORING PHASE II OF THE PELAGIC REALM

The monitoring of the pelagic ecosystem components will also focus on two to three (depending on the ecosystem component) wind farms along the on-/offshore gradient.

For those ecosystem components it was decided that monitoring will continue until stabilization of the effects occurs. It will continue thereafter for two more years to confirm the stabilization and will then be stopped, if there were at least five years of post-construction monitoring. After a break of five years, the yearly monitoring is restarted for a minimum of three years. Consequently, the seabird surveys in the Belwind wind farm were stopped at the end of April 2015 since we monitored five years post-construction and the effects stabilized. The seabird surveys in Belwind will restart in 2021. The methodology of the monthly seabird surveys as applied in the first phase of the monitoring will however be continued, but the focus will move to the Thorntonbank (C-Power) and the area to the South of the Thorntonbank (Norther concession area). The surveys on the Lodewijckbank (Northwind) are stopped because of the presence of an intermediate community between the nearshore (Norther and C-Power) and the offshore (Belwind) locations. The radar research on the Thorntonbank will be continued year round.

Harbour porpoises are monitored year-round with passive acoustic monitoring devices (C-Pods). Aerial surveys of the entire Belgian part of the North Sea are conducted four times a year. In the future, seals will be tagged with Vemco telemetry tags and GPS/GSM tags; this provided availability of funding.

The (bentho-)pelagic fish community is an ecosystem component which has not yet been investigated within the basic monitoring programme. Whether pelagic fish are attracted to the underwater structures of OWFs therefore remains an open question. It is also expected that the exclusion of fisheries inside the OWFs will have a large effect on the (bentho-)pelagic ecosystem. A preliminary study using a fish-finder sonar (and possibly other techniques) to monitor (bentho-) pelagic fish will be initiated.

Acoustic telemetry tags in cod individuals proved that cod is attracted to the OWFs. The OWFs are of importance especially for younger individuals (one and two years old), showing a high site fidelity. This telemetry study will be continued to study the importance of OWFs also for older individuals.

Bat recorders are installed on the research vessel Belgica, the Belwind platform and a turbine in the C-Power wind farm to study the distribution and density of bats at sea and inside the wind farms. Possibly more detectors will be installed in the future.

Plankton is not being monitored because an impact is unlikely. This might however be different for fish larvae, but this will be the subject of a targeted monitoring action.

Underwater noise measurements are continued inside the operational wind farms and the relationship between wind speed and underwater noise will further be investigated. Measurements during the construction of new wind farms will be conducted.

The sampling location along the nearshore offshore gradient and the timing for the (bentho-)pelagic ecosystem components are summarized in table 3.

Table 3. Sampling location and timing for the (bentho-)pelagic ecosystem components.

ecosystem component	on-offshore gradient			timing
	nearshore	midshore	offshore	
seabirds	•	•		monthly
seabirds radar		•		continuous
marine mammals – C-Pods	•	•	•	continuous
marine mammals – aerial survey	•	•	•	4 times/year
bats		•	•	continuous
(bentho-)pelagic fish – sonar study	•	•	•	monthly
(bentho-)pelagic fish - telemetry	•	•	•	continuous

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[1]Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Oostende, Belgium

[2]Ghent University, Ghent University, Biology Department, Marine Biology Research Group, Krijgslaan 281, Campus Sterre - S8, 9000 Ghent, Belgium

[3]NORTHER NV, John Cordierlaan 9, 8400 Oostende, Belgium

[4]Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Gulledelle 100, 1200 Brussels; 3de en 23ste Linieregimentsplein, 8400 Oostende; Vautierstraat 29, 1000 Brussels, Belgium.

[5]Otary Offshore Energy, Slijkensesteenweg 2, 8400 Oostende

[6]Imares, Landsdiep 4, 1797SZ DEN HOORN TEXEL, The Netherlands

[7]Research Institute for Nature and Forest , Kliniekstraat 25, 1070 Brussel (Anderlecht)

[8]Parkwind, Sint-Maartenstraat 5, 3000 Leuven, Belgium

2

CHAPTER

CHAPTER 2

OFFSHORE RENEWABLE ENERGY DEVELOPMENT IN THE BELGIAN PART OF THE NORTH SEA - 2016

R. Brabant[•,*]*, B. Rumes*[•] *& S. Degraer*

Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels, Belgium.

[•] *shared first authorship*

[*] *Corresponding author: Robin.Brabant@naturalsciences.be*

ABSTRACT

Offshore wind farms are expected to contribute significantly to the Belgian 2020 targets for renewable energy. Today, 182 turbines are operational in the Belgian part of the North Sea. In the next few years, an additional 234-342 turbines may be installed. With 238 km² reserved for offshore wind farms in Belgium and 344 km² in the adjacent Dutch Borssele, cumulative ecological impacts may however be expected. These impacts both positive and negative, triggered an environmental monitoring programme focusing on various aspects of the marine ecosystem components, but also on the human appreciation of offshore wind farms. This report provides an overview of the offshore renewable energy development in the Belgian part of the North Sea.

2.1. OFFSHORE RENEWABLE ENERGY IN BELGIUM

The European Directive 2001/77/EC on the promotion of electricity produced from renewable energy sources in the internal electricity market, imposes a target figure for the contribution of the production of electricity from renewable energy sources upon each Member State. For Belgium, this target figure is 13% of the total energy consumption, which must be achieved by 2020. Offshore wind farms in the Belgian part of the North Sea (BPNS) are expected to make an important contribution to achieve that goal.

With the Royal Decree of 17 May 2004, a 264 km² area within the BPNS is reserved for the production of electricity from water, currents or wind. It is located between two major shipping routes: the north and south traffic separation schemes. In 2011, the zone was adjusted on its Northern and Southern side in order to ensure safe shipping traffic in the vicinity of the wind farms. After this adjustment the total surface of the area amounted to 238 km².

Prior to installing a renewable energy project, a developer must obtain (1) a domain concession and (2) an environmental permit. Without an environmental permit, a project developer is not allowed to build and exploit a wind farm, even if a domain concession was granted.

In order to stimulate the development of wave energy in Belgium, the Mermaid project obtained its domain concession license only on condition that a certain amount of energy would be generated from waves as well as from wind.

When a project developer applies for an environmental permit an administrative procedure, mandatory by law, starts. This procedure has several steps, including a public consultation during which the public and other stakeholders can express any comments or objections based on the environmental impact study (EIS) that is set up by the project developer. Later on during the permit procedure, the Management Unit of the North Sea Mathematical Models (MUMM), a Scientific Service of the Operational Directorate Natural Environment (OD Nature) of the Royal Belgian Institute of Natural Sciences, gives advice on the acceptability of expected environmental impacts of the future project to the Minister responsible for the marine environment. MUMM's advice includes an environmental impact assessment, based on the EIS. The Minister then grants or denies the environmental permit in a duly motivated decree.

The environmental permit includes a number of terms and conditions intended to minimise and/or mitigate the impact of the project on the marine ecosystem. Furthermore, as required by law, the permit imposes a monitoring programme to assess the effects of the project on the marine environment.

At present, nine projects were granted a domain concession and an environmental permit (from South to North: Norther, C-Power, Rentel, Northwind, Seastar, Nobelwind, Belwind, Northwester II & Mermaid) (Table 1). When all Belgian wind farms are built, there will be just under 500 wind turbines in the Belgian part of the North Sea. The entire area with its nine parks will have a capacity of 2200 MW and cover up to 10 % of the total electricity needs of Belgium or nearly 50 % of the electricity needs of all Belgian households.

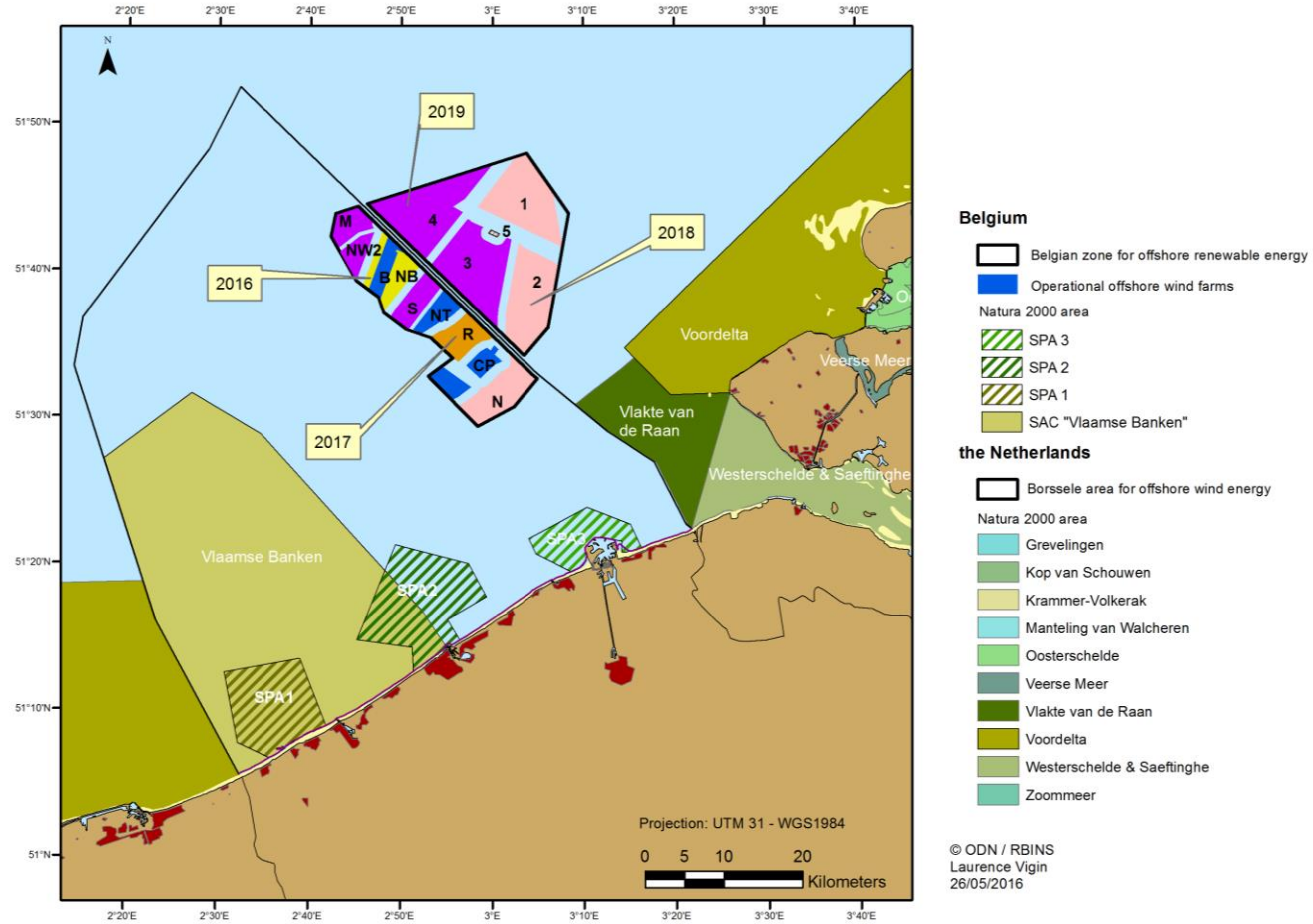

Figure 1. Map of the Belgian zone for offshore renewable energy, the Dutch Borssele offshore wind area and Natura 2000 areas in the vicinity. Already constructed wind farms are indicated in blue (CP: C-Power, NT: Northwind and B: Belwind), wind farms under construction in 2016 in yellow (NB: Nobelwind), 2017 in orange (R: Rentel), 2018 pink (N: Norther, 1 and 2: Borssele 1 and 2) and 2019 in purple (S: Seastar, NW2: Northwester2, M: Mermaid, 3 and 4: Borssele 3 and 4)

Table 1. Overview of wind farms in the Belgian part of the North Sea (situation on March 18th, 2016)
*: number of turbines and/or total capacity still to be decided; **: including 5 MW of wave energy.

Project		Number of turbines	Capacity (MW)	Total capacity (MW)	Concession obtained	Environmental permit obtained	Status
C-Power	phase 1	6	5	325	YES	YES	Phase 1 operational since 2009
	phase 2 & 3	48	6.15		YES	YES	Phase 2 and 3 operational since 2013
Belwind	phase 1	55	3	171	YES	YES	Phase 1 operational since 2011
	Alstom Demo project	1	6		YES	YES	Demo turbine operational 2013
Nobelwind		50	3.3	165	YES	YES	Construction ongoing (2016-2017)
Northwind		72	3	216	YES	YES	operational since 2014
Norther		47-100*	3-10	258 – 470*	YES	YES	Constructions foreseen to start in 2018
Rentel		47 – 78*	4-10	289 – 468*	YES	YES	Construction foreseen to start in 2017
Seastar		41	4-10	246*	YES	YES	Construction foreseen to start in 2019
Mermaid		27-41	4-10	232-266 + 5**	YES	YES	Construction foreseen to start in 2019
Northwester 2		22 - 32	3-10	217-224	YES	YES	Construction foreseen to start in 2019

2.2. MARINE SPATIAL PLAN AND AQUACULTURE

On 20 March 2014 Belgium approved a new marine spatial plan for the Belgian Part of the North Sea by Royal Decree. The new plan lays out principles, goals, objectives, a long-term vision and spatial policy choices for the management of the Belgian territorial sea and the Exclusive Economic Zone (EEZ). Management actions, indicators and targets addressing marine protected areas and the management of human uses including commercial fishing, offshore aquaculture, offshore renewable energy, shipping, dredging, sand and gravel extraction, pipelines and cables, military activities, tourism and recreation, and scientific research are included. The current marine spatial plan is valid for a period of six years and thus in 2020 a new plan will be formulated. This will allow the government to take into account new developments in the field of marine renewable energy.

In the current marine spatial plan two zones are dedicated to sustainable aquaculture. These are both situated within the operational Belwind and C-Power windfarms. In December 2015, the Aquavalue project formulated a roadmap for integrated aquaculture for Flanders and defined on a technical and economical level four possible pilots for integrated aquaculture in Belgium. These included two pilots in the wind farms: one involves bivalve and sea weed aquaculture, and the other the herding of conditioned sea bass.

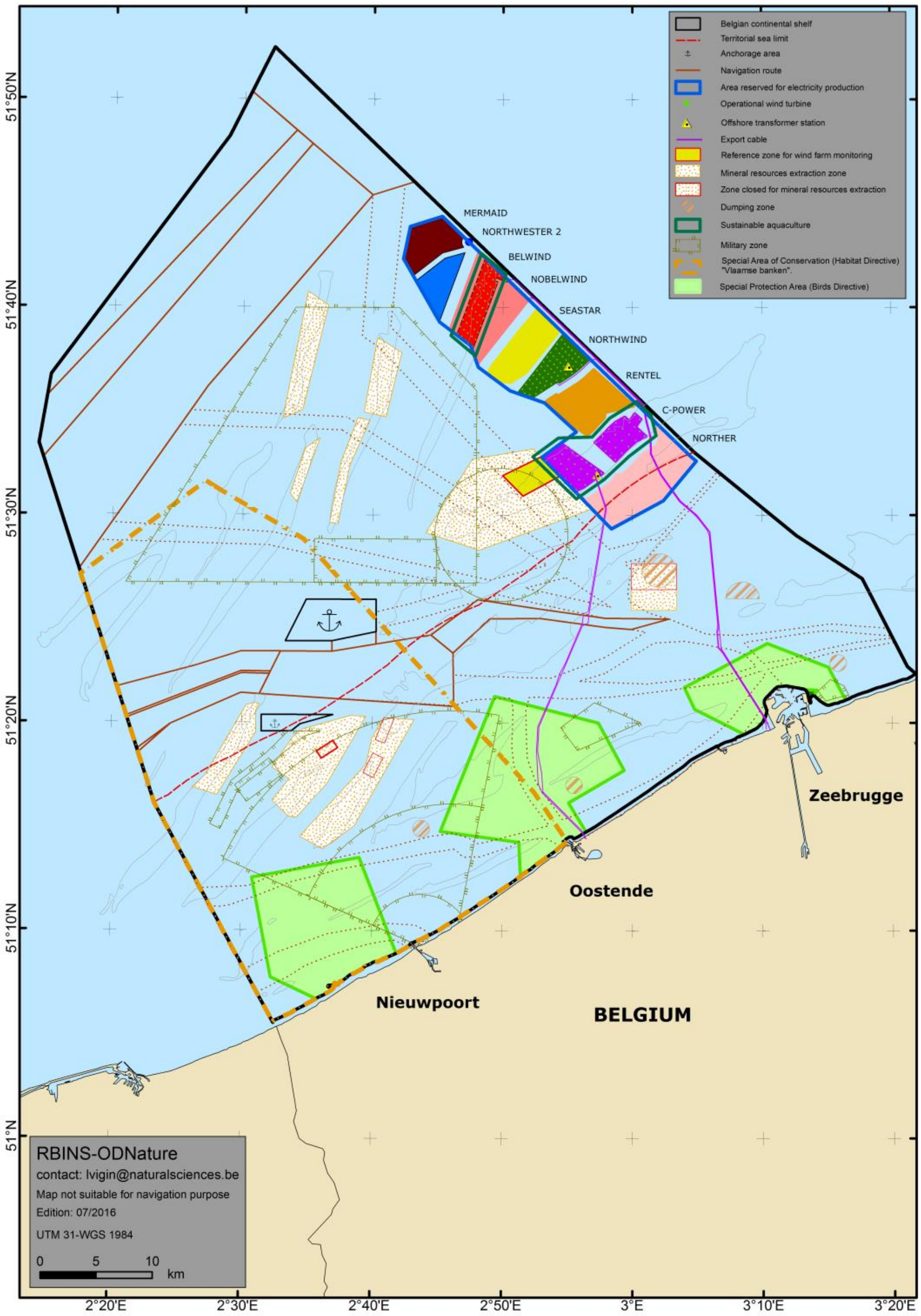

Figure 2. Marine spatial plan of the Belgian Part of the North Sea.

2.3. GRID REINFORCEMENT AND A ‘PLUG AT SEA’

The first three offshore wind farms were connected to the electricity grid by a limited strengthening of the existing high-voltage grid. For the next six projects to be built a comprehensive network upgrade is necessary. To meet this necessity, Elia launched the Stevin project which includes a new power station near the port of Zeebrugge and a high voltage network from Zeebrugge to Zomergem. It is expected to be finished in 2018.

The three operational wind farms each ensure the export of their electricity to the onshore grid. Several proposals have been formulated to develop a shared connection, a so-called ‘plug-at-sea’ which would allow the remaining projects to share an export connection and would allow for integration in an as yet to be developed international offshore grid. The first project of this nature, the Belgian Offshore Grid, included a meshed grid with two offshore high voltage stations, one of which was to be located on an artificial island and six export- and/or interconnection cables (Figure 3). Currently a more reduced setup – a Modular Offshore Grid (MOG) - consisting of a single Offshore switch Yard (OSY) located near the Rentel concession is being considered, which would connect four of the remaining wind farms to the grid.

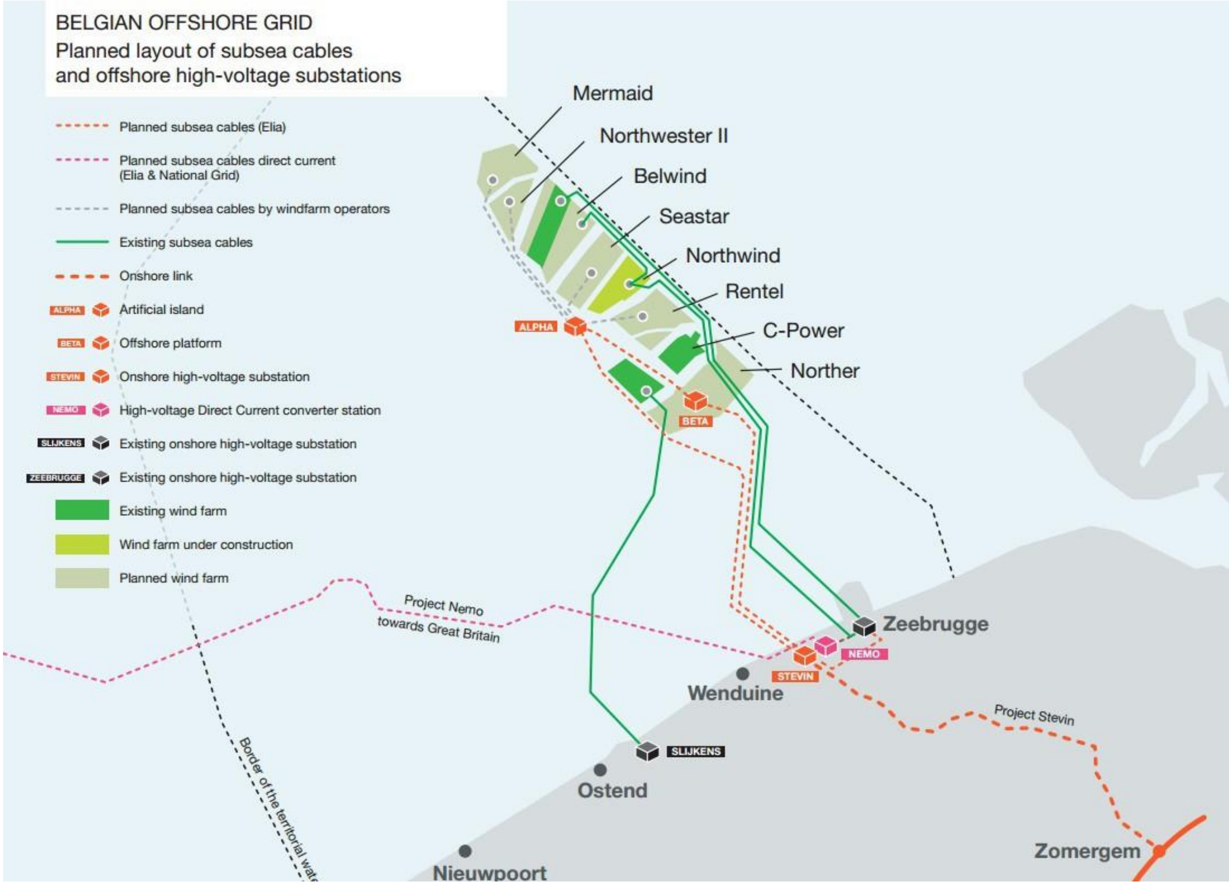

Figure 3. Initial design for the Belgian Offshore Grid (BOG).(Source: www.G-tec.eu).

CHAPTER
3

CHAPTER 3

QUANTIFICATION AND CHARACTERISATION OF BELGIAN OFFSHORE WIND FARM OPERATIONAL SOUND EMISSION AT LOW WIND SPEEDS

A.Norro[*] *& S. Degraer*

Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels, Belgium.

[*] *Corresponding author: Alain.Norro@naturalsciences.be*

ABSTRACT

Offshore renewable energy installations contribute to the continuous underwater sound that has been identified as an environmental concern under the EU Marine Strategy Framework Directive. This study quantified, characterised and compared the continuous underwater sound emitted by steel jacket foundation and monopile Wind turbines during operation at low wind speed (0-12 m/s). The operational sound emitted by a monopile founded and a jacket founded wind farm in the BPNS showed a maximum increase of SPL of about 20 dB re 1 µPa. Spectral analysis showed that this increase occurs at frequencies below 3 kHz. Steel monopile foundations even when equipped

with a less powerful generator, emitted significantly more underwater sound than jacket foundations. The addition of underwater sound is increasing with wind speed with a rate dependent of the type of foundation, with monopiles showing a stronger increase with wind speed than jacket foundations. Possible impacts on marine life like fish, marine mammals or invertebrates remain unclear mainly due to the lack of knowledge in disturbance or behavioural response levels for the species that could be found on these sites. Future challenges are to expand the study to higher wind speeds (study ongoing) and to quantify and qualify the additional sound pressure of a larger wind farm or a series of adjacent smaller wind farms (i.e. cumulative effects).

3.1. INTRODUCTION

According to the Marine Strategy Framework Directive (MSFD) EU Member States have to determine, achieve and control good environmental status for their marine waters by 2020 (EU Directive 2008/56/EC). As part of the MSFD, EU Member States are requested to ensure the “introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment”. This target specifically refers to anthropogenic activities undertaken at sea that indeed may generate underwater sound that could be harmful to marine life (Dekelin et al., 2014). Besides loud, low and mid frequency impulsive sounds (as produced by e.g. pile driving; Norro et al., 2013a), concern is also raised about continuous low frequency sound (Commission Decision 2010/477/EU). Offshore renewable energy installations are one of the human activities contributing to this continuous sound (Dekelin et al, 2014).

The implementation of wind farms at sea generates underwater sound. Four different phases are distinguished during the life of an offshore wind farm: 1. before implantation phase or initial situation; 2. construction phase; 3. operational phase during electricity production; and 4.dismantlement or decommissioning phase (Nedwell et al., 2004). The sound generated differs relative to these four phases. For the Belgian part of the North Sea (BPNS), several studies already exist documenting sound emission during some of these phases. The initial situation at the Thorntonbank was documented by Henriet et al. (2006), while Haelters et al. (2009) studied the T_{-1} condition at the Bligh Bank site. The sound produced during the construction phase was documented by Haelters et al. (2009) for the six gravity-based foundation (GBF) Wind turbines at the Thorntonbank and by Norro et al. (2010) for construction by piling as applied at the Bligh Bank and Thorntonbank (C-Power phases II and III). The sound produced during the operational and dismantlement phases remains yet to be quantified.

During operation of a wind farm, vibration is produced by the rotation of the wind turbines through all related parts, such as the gearbox and other moving parts. This vibration is transmitted to the water by the support structure or foundation like a steel monopile, jacket or GBF, as such producing underwater sound. Clearly, the underwater sound produced by an operating Wind turbine is much lower than the sound emitted during their construction; this particularly when pile driving is used (COWRIE, 2010). However, the construction sound lasts for a limited period of time (typically few weeks, e.g. C-Power phase II), while the operational sound is produced throughout the full operational phase of the wind farm that is expected to be about or more than 20 years. Measurements of operational sound in various offshore wind farms showed a higher than the background sound intensity (Boesen and Kjaer, 2005; Andersson et al., 2011). A 6 MW monopile-based wind turbine for example is audible up to at least 20 km distance (Marmo et al., 2013). In a more focused report, Betke (2006) documented the emitted sound of a 2 MW turbine using a spectral analysis. The highest sound pressure levels are observed near frequencies of 150 Hz and 300 Hz with a sound pressure level of 118 dB and 105 dB re 1 μPa, respectively. No increase of sound pressure level above background level was observed for frequencies above 800 Hz. Comparison with data measured in Sweden (Utgrunden wind farm cited by Betke, 2006) showed a similar pattern. Uffe (2002) further demonstrated that concrete foundations and steel pile foundations show different spectral features and that the sound emitted by both types of foundation is stronger than the ambient sound only for the frequencies below 1kHz (steel pile being noisier). Nedwell et al. (2007) however nuanced the increased sound level concluding that the increase in level of sound is not greater than what may be expected from the natural variation in the background sound level that may occur as a marine mammal moves or during bad weather conditions. Still, a probable negative impact risk labelled moderate to high for marine mammals and moderate for fish and benthos is expected (Bergström et al., 2014).

The objective of this paper is to further contribute to the knowledge on operational wind farm sound emission, and to quantify and characterise the underwater sound emitted by steel jacket foundation wind turbines (C-Power phase II and III wind farm, Thorntonbank) and monopile wind turbines (Belwind phase 1 wind farm, Bligh Bank) during the operational phase.

3.2. MATERIAL AND METHODS

MEASUREMENTS METHODOLOGY

Based on Norro et al. (2013), measurements were performed from a drifting rigid hull inflatable boat (RHIB) inside the wind farm and hence in the vicinity of the Wind turbines at eleven occasions (Table 1). All equipment like engine or echosounder was turned off in order to avoid any interaction with the hydrophone. The geographic position and time was recorded with a handheld GPS GARMIN GPSMap60 at a rate of one position every 5 s. At the start and the end of each measurement a reference signal was recorded. The clock of the recorder was synchronised beforehand with the GPS-time (UTC).

Table 1. Location, date and recording time of the operational underwater sound measurements used in this study.

Location	Date	Foundation type	Info on records
Belwind	11/7/2011	steel monopile	1*20 min
Belwind	3/4/2012	steel monopile	2*20 min
C-Power	2/4/2012	jacket	2*20 min
C-Power	29/4/2013	jacket	1*20min
Belwind	30/4/2013	steel monopile	2*20 min
Belwind	5/5/2014	steel monopile	2 * 20 minutes
C-Power	6/5/2014	jacket	2 * 20 minutes
Northwind/C-Power	31/7/14	steel monopile /jacket	3 of various length
Belwind	26/5/15	steel monopile	1 * 10 min usable
Northwind	26/5/15	steel monopile	3*20 min
Northwind	30/6/15	steel monopile	3*20 min

ACOUSTIC MEASUREMENT EQUIPMENT

At every occasion, at least one Brüel & Kjær hydrophone (type 8104) was deployed at a depth of 10 m. A Brüel & Kjær amplifier (Nexus type 2692-0S4) was connected between the hydrophone and the recorder in order to allow for an amplification of the signal. A reference signal was used together with the output sensitivity of the Nexus to calibrate the recorded signal. The signal was recorded using an audio MARANTZ Solid State Recorder (type PMD671). It was operated with the highest possible sampling rate of 44.100 Hz. The signal was recorded in WAVE format (.wav) on Compact Flash cards of 2 GB (Sandisk Ultra II). Batteries powered all equipment.

WEATHER CONDITIONS DURING FIELD WORK

Weather conditions encountered during fieldwork featured wind of Bft 1-4 and a sea state ranging from 1 to 2-3.

Onsite real time weather data were not available at the time of data analysis. We used the real time wind data measured at the Westhinder that is located some 25 NM away both sites, instead (real time measurements from Meetnet Vlaamse Banken- afdeling KUST). These data are three hourly averaged data of wind speed at 10 m height and wind direction.

ANALYSIS OF THE RECORDINGS

The reference tones accompanying every record and used for calibration were excluded from the analysis and the complete remaining part of the record was used for further analysis. In case of clear interference or when the hydrophone was removed from the water to avoid collision with a foundation, short parts of the record were excluded. In some occasions a record was rejected mainly because of strong interference in the signal.

Sound pressure level (SPL) and zero to peak level (L_{z-p}) were calculated, plotted against wind speed (discriminating between monopile and jacket foundations) and analysed using a linear regression model written in Matlab or R. Both, linear models obtained for wind effect on sound pressure levels generated by steel monopiles and jackets were further examined. An ANCOVA analysis to test for statistical difference of both models was performed in R.

A spectral analysis of the signal in the form of the third octave band spectrum of the underwater SPL was performed. For every selected record, the spectra were computed using MATLAB routines built according to the norm IEC1260.

3.3. RESULTS

The regression analyses for the jacket foundations revealed two statistically significant regression models (SPL slope: p = 0,0026; L_{z-p} slope: p = 0,002) (Figure 1), i.e.

SPL=1,1 * wind speed + 122,5

L_{z-p} =0,96 * wind speed + 144,3

For steel monopiles, a significant regression model could be found only for SPL (slope: p = 0,01), i.e.

SPL=1,9 * wind speed + 120,3

The ANCOVA test showed that the interaction between type of foundation and SPL was highly significant (p = 0,0037).

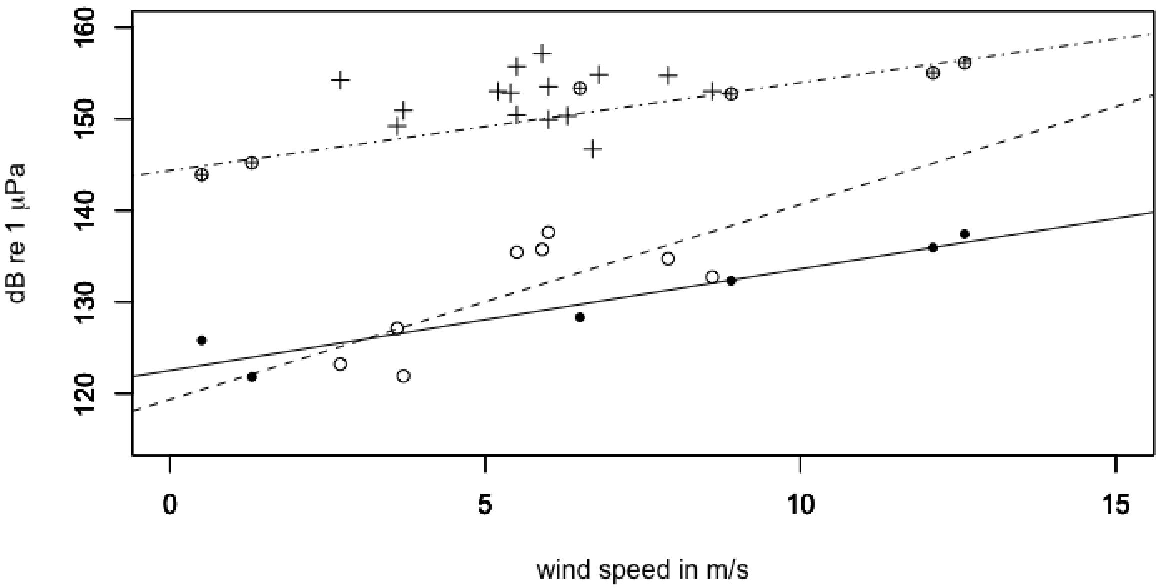

Figure 1. Operational sound pressure levels (SPL, lower part) and zero to peak level (L_{z-p}, upper part) versus wind speed. Linear regression models presented show only those having a significantly different slope. ○, monopile SPL; ●, jacket SPL; +, monopile L_{z-p}; crossed circle, jacket L_{z-p}. Plain line, linear model jacket SPL; dashed line, linear model monopile SPL. Dot dashed line for linear model jacket L_{z-p}. Linear model monopile L_{z-p} not presented because statistically not significant.

For jacket foundations, most of the energy was produced between 60 and 600 Hz (Figure 2). Above 600 Hz a decay was observed. For steel monopiles, it appears that the ranges of emitted frequencies extended to 3 kHz before a decay was observed for some spectra (Figure 3). A peak was observed at 5 kHz, but only for one record. The spectral analysis of the signal in the form of the third octave band spectrum of SPL did not allow isolating specific peaks that could discriminate between the type of foundation.

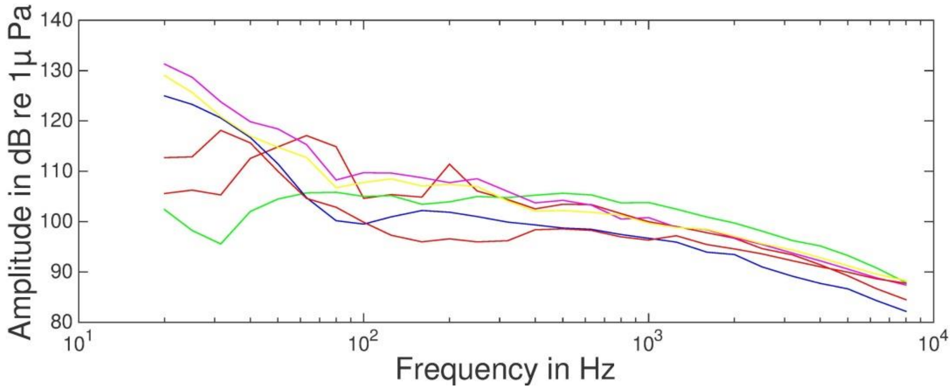

Figure 2. Spectral analysis (1/3 octave band spectra) of the jacket foundation recordings (C-Power wind farm, Thorntonbank).

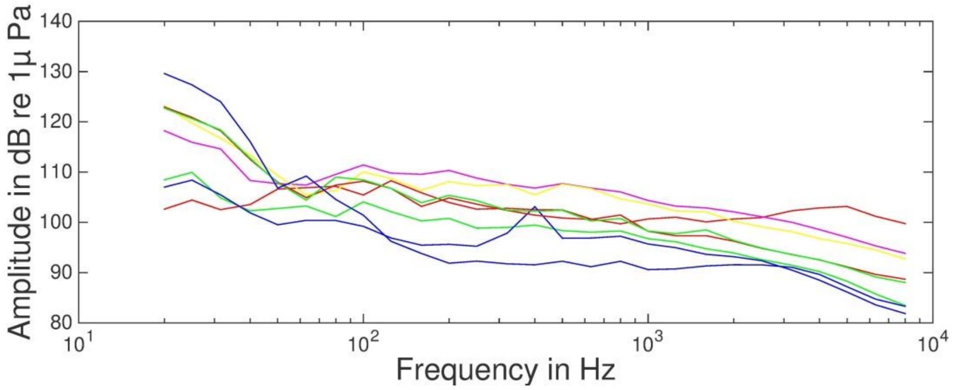

Figure 3. Spectral analysis (1/3 octave band spectra) of the monopile foundation recordings (Belwind wind farm, Bligh Bank).

3.4. DISCUSSION

Our study demonstrated SPL and L_{z-p} to be correlated with wind speed at low wind speed conditions (not demonstrated for steel monopile foundations L_{z-p}). The emitted underwater sound further increases more intensely with wind speed for steel monopile foundations than for jacket style foundations, confirming that the observed increase in underwater sound is not solely due to weather conditions but intrinsic to the presence of the wind farms. Both study sites indeed are very close to each other (10 NM) and present similar wind, bathymetric and sedimentary conditions. The hypothesis proposed by Norro et al. (2013b) that steel monopile foundations emit higher SPL than jacket foundation hence could be validated. For a mean wind speed of 10 m/s, we can now predict that a steel monopile will emit some 10 dB re 1µPa more than a jacket foundation.

Our findings also allow assessing the sound addition above the background levels in the wind farms. For the jacket foundations installed at the Thorntonbank, the background SPL correspond to 122 dB re 1µPa (Henriet et al. 2006), from which we can take that the jacket foundations increase SPL by 11 dB re 1µPa at a wind speed of 10 m/s. For the steel monopiles at the Bligh Bank, a 19 dB re 1 µ Pa increase of SPL above the 120 dB re 1µ Pa background level (Haelters et al. 2009) can be found at a wind speed of 10 m/s.

Wind by itself participates to ambient sound (Kerman et al., 1983; Dalh et al., 2007). Elevation of underwater sound solely due to the wind speed effect can be evaluated. Here, we used a model developed for shallow water by Murugan et al. (2011). An increase of underwater sound at a wind of 10 m/s is about 4 dB re 1µ Pa. It typically appears at a 1 kHz frequency.

COMPLIANCE WITH THE EU MSFD DESCRIPTOR FOR LOW FREQUENCY SOUND.

Sound emitted by an operating wind farm has to comply with the indicator 11.2 ‘continuous low frequency noise’ .This indicator proposes to identify trends in the ambient noise level within the 1/3 octave bands 63 and 125 Hz (centre frequency) (re 1µPa RMS; average noise level in these octave bands over a year) measured by observation stations and/or with the use of models if appropriate (Van der Graaf et al, 2012).

The trend referred to here however, is to be evaluated based on a yearly mean underwater sound, which – in absence of continuous measurements at different locations – remains to be assessed using validated models.

We can approximate from Norro et al. (2013a) that few kilometres are needed to reduce levels of about 140 dB re 1µPa to 120 dB re 1µ Pa. The sound produced by an operating wind farm could hence be detected at such distance, which accords with Andersson (2011).

POSSIBLE IMPACT ON THE MARINE LIFE

Up front, it should be remembered that during the operational phase of a wind farm relatively low additional underwater sound seem to be generated; this certainly compared to the construction phase using pile driving (190 dB re µPa at 750 m for piling steel monopile foundation) (e.g. Norro et al., 2013a). Nevertheless, it should be emphasised that these underwater sound emissions will be continuously present throughout the complete operational phase of the wind farm that currently is set at a minimum of 20 years.

The impact on marine life if any, will be related to the level and the frequency spectrum of the emitted underwater sound. Marine life with a hearing capacity matching frequencies from 60 Hz to 3 kHz may be impacted. This corresponds to some fish and marine mammals while effects on invertebrates remain mostly unknown (Sole et al.2013). The levels concerned here are low and impact if any will most probably be mainly masking or behavioural. Marine biologists still are at the early stage of such impact evaluation and virtually no validated thresholds are published today.

The small increase in sound in the immediate vicinity of Wind turbines in operation is very unlikely to cause a behavioural response for marine species (Bergström et al., 2014), as was demonstrated for European sea bass *Dicentrarchus labrax*, Atlantic cod *Gadus morhua*, common dab *Limanda limanda*, Atlantic herring *Clupea harengus*, Atlantic salmon *Salmo salar*, bottlenose dolphin *Tursiops truncatus*, harbour porpoise *Phocoena phocoena* and common seal *Phoca vitulina* (Nedwell et al., 2007). Also Betke (2006) expects the sound emitted by the Horn Rev during operation no longer to be heard by harbour porpoises from 100 m distance from the turbine, but yet highlighted caution is needed due to the limited knowledge available on the topic. Clearly, while bottlenose dolphins and harbour porpoises would be aware of various components of the wind farm operational sound up to a 200 m distance, the measured levels were considered insufficient to cause any hearing damage (Ward et al., 2006). Sigray and Andersson (2011) studying particle motion around operational Wind turbines, concluded that behavioural reactions of fish are possible in the very close vicinity of the Wind turbine (1-5 m). Whether the 20 dB re 1 µPa increase as it was observed for steel monopiles, may create such behavioural response hence yet remains an open question.

PERSPECTIVE

While we now start having a proper view on sound emitted by operational wind farms, these data are solely derived from measurements in single wind farms. The question raising today is what the additional sound pressure of a larger wind farm or a series of adjacent smaller wind farms would produce. In the BPNS for example, the zone reserved for energy production is a compact zone of approximately 20 NM long and 4 NM wide that may accommodate no less than eight wind farms. Such a question could be solved by the use of an acoustic model validated for the zone of interest and combined with the collection of field data to compare with the model results.

It further remains to be investigated whether the linear models of sound to wind speed as developed in this study, can also be applied to higher wind speeds. Actions for such analysis are currently ongoing.

ACKNOWLEDGEMENTS

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4

CHAPTER

CHAPTER 4

EVALUATING UNDERWATER NOISE REGULATIONS FOR PILING NOISE IN BELGIUM AND THE NETHERLANDS

B. Rumes[*1], A. Erkman*[2] & J. Haelters*[1]

[1]Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels; 3de en 23ste Linieregimentsplein, 8400 Oostende Belgium.

[2]Rijkswaterstaat, Poelendaelesingel 18, 4335 JA Middelberg, the Netherlands.

[*]Corresponding author: Bob.Rumes@naturalsciences.be

ABSTRACT

There is concern about possible effects on the marine ecosystem of high levels of underwater noise generated during pile driving for the construction of offshore wind farms. As a result, various national governments in Europe have identified limits of underwater sound levels, as such imposing in many cases the use of noise mitigation measures. In this paper we compare the regulations with regard to impulsive underwater noise in the Belgian wind farm zone with those in the Dutch wind energy zone of Borssele. These (planned and existing) wind farms are situated at opposite sides of the maritime border between both countries. These regulations are quite different and at times even contradictory and developers could benefit from an alignment of regulatory practices on a regional basis. Measurements of piling noise from constructed wind farms are used to extrapolate the anticipated noise levels of the next two wind farms to be constructed, and these are evaluated in relation to the new regulations on underwater sound. Wind farm developers are already developing strategies for cost-effective piling noise reduction but uncertainty remains with regards to both the level of underwater noise produced during piling as well as with the effectiveness of the noise mitigation measures being applied. Our results indicate that a combination of noise mitigation measures may need to be used to comply with the new regulations.

4.1 INTRODUCTION

At the end of 2015, 11.6 GW of offshore wind capacity was operational in the Southern North Sea and a further 20.3 GW was consented and scheduled to be constructed in the next decade (EWEA, 2016). During 2015, more capacity was installed than ever before and work was carried out on 22 offshore wind farms in Europe (EWEA, 2016). Understanding the environmental impact of offshore wind farms is necessary to support policy and management of this publicly subsidized industry. Environmental impact monitoring of offshore wind farms has been ongoing since 2000 (Danish Energy Agency, 2013), and the effect of piling noise on marine mammals, and in particular the harbour porpoise (*Phocoena phocoena*), is recognized as one of the major environmental drivers for underwater noise regulations as it concerns a species sensitive to sound (Lepper *et al.,* 2008), legally protected nationally and internationally (Dolman et al., 2016) and the effect of piling noise has been demonstrated to extend over a large distance (Brandt et al., 2011; 2012, Haelters et al., 2015). Potential effects of piling noise on marine mammals range from auditory masking, behavioural disturbance, physiological stress, hearing loss (temporary or permanent) up to physical injury or death (Lucke et al., 2009).

In the European Marine Strategy Framework Directive (MSFD; 2008/56/EC) member states should aim to achieve or maintain good environmental status (GES) by 2020 at the latest. For the introduction of energy, including underwater noise, GES requires anthropogenic underwater noise to be at levels that do not adversely affect the marine environment. To implement the MSFD for anthropogenic impulsive sounds, Belgium adopted an interim criterion of a maximum zero to peak noise level ($L_{z\text{-}p}$) of 185 dB re 1 µPa at 750 m from the source (Anonymous, 2012a). In the Netherlands, it has been argued that, lacking certain information on the impact of impulsive sounds on the marine ecosystem, no general criterion could be defined in 2012. Additional studies were since conducted to address these knowledge gaps (Anonymous, 2012b). The Netherlands however agreed that, mitigating measures should be taken at a case by case basis for activities such as piling and seismic investigations, to prevent negative impacts on the marine fauna (Anonymous, 2012b). This difference in approach in neighbouring countries is not surprising, as so far all European member states which have defined GES for underwater noise have used different approaches (Dekeling, 2015).

In practice, underwater noise regulations for individual projects in both Belgium and the Netherlands are to a large extent stipulated in the *environmental permit* (Belgium) and in the *Kavelbesluit* (The Netherlands) (Table 1).

In this chapter, measurements of piling noise (zero to peak sound pressure level Lz-p and unweighted sound exposure level SEL) from constructed wind farms are used to extrapolate the anticipated piling noise levels of the next two Belgian wind farms to be built (Figure 1) and these are evaluated in relation to the Belgian and Dutch regulations in order to determine what level of noise mitigation will be needed.

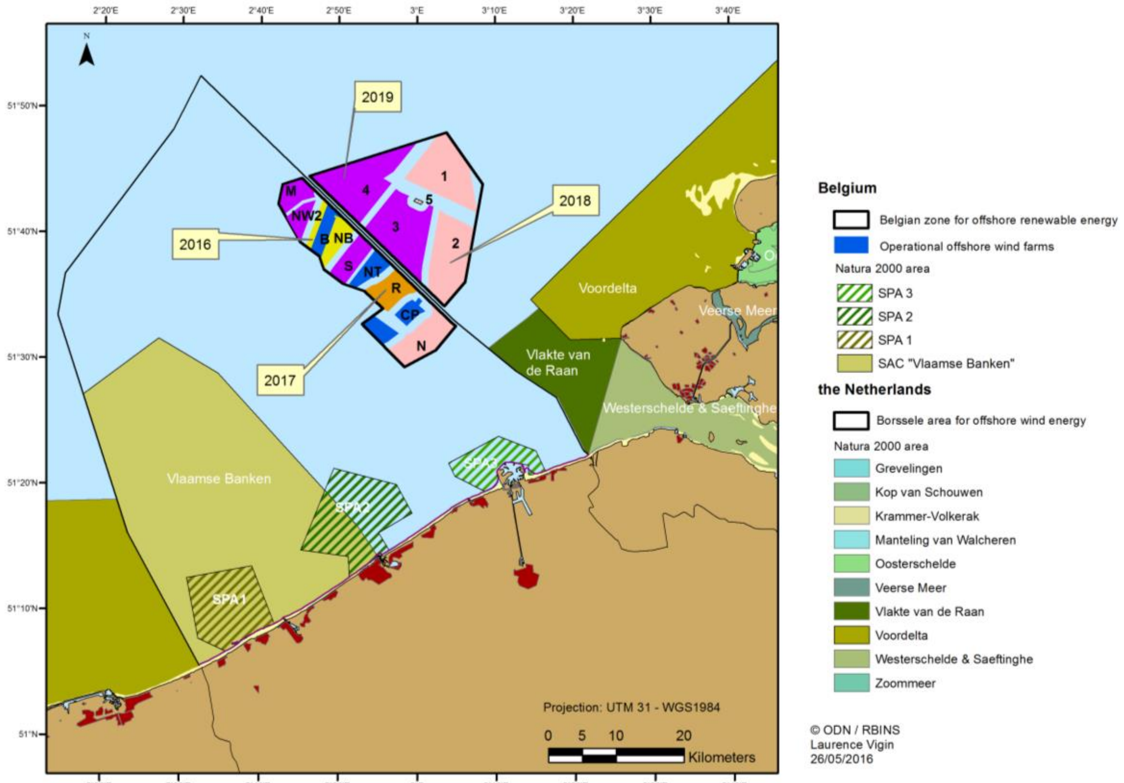

Figure 1. Map of the Belgian zone for offshore renewable energy, the Dutch Borssele offshore wind area and Natura 2000 areas in the vicinity. Already constructed wind farms are indicated in blue (CP: C-Power, NT: Northwind and B: Belwind); wind farms under construction in 2016 in yellow (NB: Nobelwind); in 2017 in orange (R: Rentel); in 2018 in pink (N: Norther, 1 and 2: Borssele 1 and 2); and in 2019 in purple (S: Seastar, NW2: Northwester2, M: Mermaid, 3 and 4: Borssele 3 and 4).

Table 1. Overview of the underwater noise regulations for wind farm construction in Belgium and the Netherlands (Borssele) (data Rumes et al., 2011; 2012; Ministerie van Economische Zaken, 2015)

	Belgian wind farm zone	Borssele
	Measures to limit or monitor the introduction of impulsive sound	
Noise restriction	Lz-p @ 750m: 185 dB re µPa	SEL @ 750m: 160-172 dB re µPa²s 1)
Noise mitigation	Yes, if limit is exceeded	Yes, if limit is exceeded
Noise monitoring	Ad hoc inspections, by government	Continuous, by permit holder
	Measures to limit the impact of piling on marine mammals	
Seasonal piling restriction	No piling from January 1st to April 30th	No piling from January 1st till and including May 31st 2)
Acoustic deterrent device	Yes, starts 30 min prior to piling	Yes, starts 30 min before piling
Piling starts with soft start	Yes	Yes
Marine mammal inspection prior to piling	Yes, by permit holder	No

1) As function of the number of turbines that is to be installed and the period of the year.

2) Only for projects with more than 76 wind turbines per single wind farm of ~350 MW

4.2 DATA SOURCES AND ANALYTICAL METHODS

Underwater noise levels were recorded at distances ranging from 250 m to 14 km from the pile driving location during the installation of steel monopiles (5.0 m diameter) at both the Bligh Bank (Belwind) and the Lodewijk Bank (Northwind), and of pin piles (1.8 m diameter) at the Thorntonbank (C-Power). Measurements of piling noise were performed using a Brüel & Kjær hydrophone (type 8104) which was deployed at a depth of 10 m, suspended from a drifting Rigid Hull Inflatable Boat (RHIB) (Norro et al., 2012). To avoid interaction with the hydrophone, the engine, radar and echosounder were turned off. For more details: see Haelters et al. (2009). Zero to peak sound pressure level (Lz-p SPL), unweighted SEL, cumulative SEL and 1/3 octave spectra were computed in order to quantify the underwater noise emitted during piling. These data were combined with SEL and SPL data aggregated by Bellmann (2014) to derive two functions which express SPL and SEL in relation to pile diameter in SPSS (IBM Corporation). Pile diameter was chosen since it is known well beforehand and both Parvin et al. (2006) and Betke & Matuschek (2010) previously found a proportionate increase in SPL with increasing pile diameter.

These functions were then used to extrapolate the anticipated underwater noise levels for the next two wind farms to be built in the Belgian wind farm zone: Rentel and Norther (Figure 1, Table 2).

Table 2. Characteristics of the planned Rentel and Norther wind farms (data Rentel and Norther, may be subject to change).

	Rentel	Norther
Anticipated period of piling	May to September 2017	May to September 2018
Foundation type	Monopile	Monopile (+ 1 Jacket for OTS)
Number of foundations	43	45
Pile diameter	7.2 – 7.8 m	6.5 – 8.0 m
Pile wall thickness	60 – 105 mm	60 – 90 mm
Noise restriction in permit	Lz-p @ 750m: 185 dB	Lz-p @ 750m: 185 dB

4.3 RESULTS

From a wide range of underwater noise measurements during pile driving work without noise mitigation systems (23 and 29 *in situ* measurements of SEL and SPL respectively with pile diameters between 0.7 and 6.0 m) two logarithmic trend curves were derived which express SPL and SEL as a function of pile diameter:

$$\text{SPL Lz-p @ 750 m} = 181.8 + 10.536 \cdot \ln(\text{pile diameter in m}) \qquad (R^2 = 0.73)$$

$$\text{SEL @ 750 m} = 158.7 + 11.124 \cdot \ln(\text{pile diameter in m}) \qquad (R^2 = 0.78)$$

This is at best a rough approximation since other factors such as local geology, thickness of the pile wall, and hydraulic hammer energy also influence the noise levels generated during piling (Betke & Matuschek, 2010; Fricke & Rolfes, 2015).

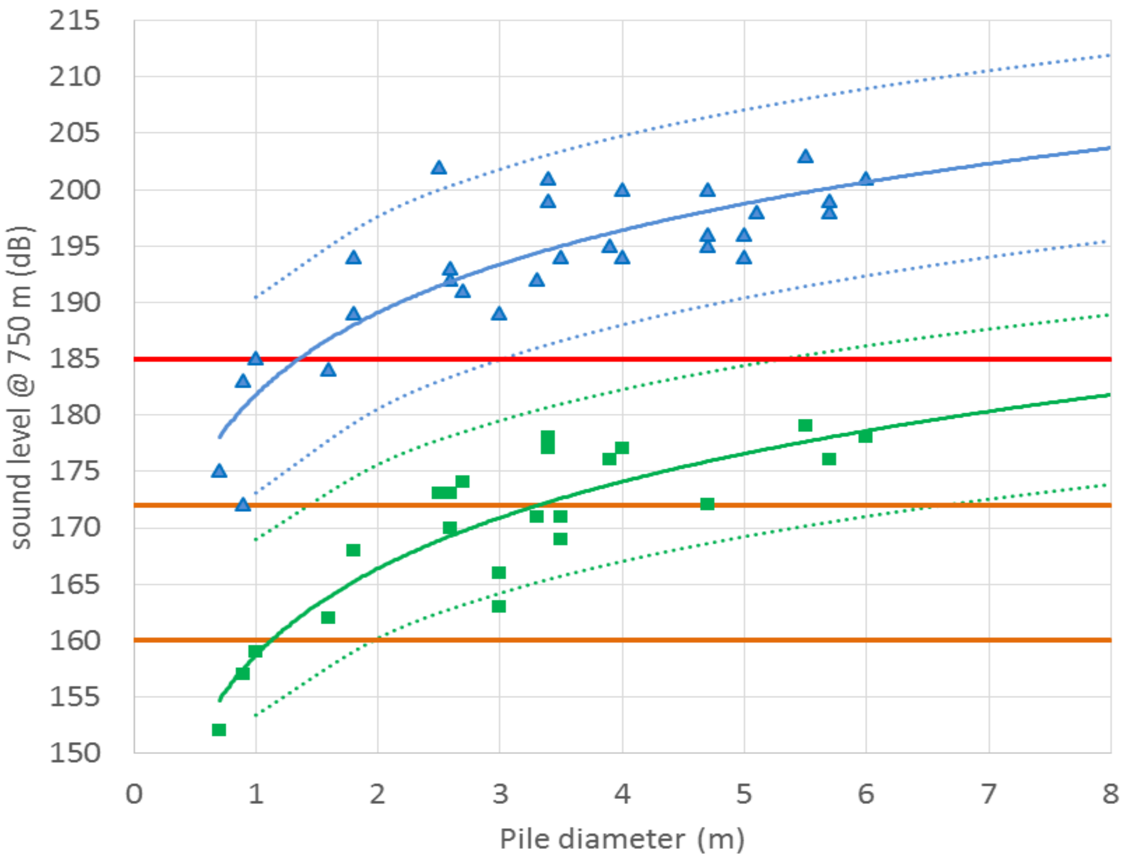

Figure 2. Zero to Peak Sound Pressure Levels (Lz-p @ 750m) (blue) and Sound Exposure Levels (SEL @ 750 m) (green) measured during pile driving as a function of pile diameter in relation to the Belgian Lz-p threshold (red) and the variable Dutch SEL threshold (orange – upper and lower end of range). 95% confidence intervals indicated by dashed lines (SEL and SPL data from Bellmann, 2014 and overview listed in Rumes et al., 2015).

If we apply these equations to the pile diameters foreseen in the as yet to be constructed wind farms we end up with a range of noise levels that exceeds both the Belgian and Dutch legislation by up to 19 dB (Table 3).

Table 3. Anticipated noise levels (Lz-p and SEL @ 750m) for the Rentel and Norther offshore wind farms and their relations to underwater noise thresholds for wind farm construction in Belgium and the Netherlands (Borssele).

	Rentel	Norther
Pile diameter	7.2 – 7.8 m	6.5 – 8.0 m
Anticipated noise level Lz-p @ 750m	203 dB	202 – 204 dB
Anticipated noise level SEL @ 750m	181 - 182 dB	180 - 182 dB
Minimal noise reduction to comply with Belgian limits	18 dB (Lz-p)	17 – 19 dB (Lz-p)
Minimal noise reduction to comply with Dutch limits*	18 – 19 dB (May) (SEL) 12 – 13 dB (June-August)	17 – 19 dB (May) (SEL) 11 – 13 dB (June–August)

* Not required by the Belgian environmental license

4.4 DISCUSSION

UNCERTAINTY REGARDING THE NOISE LEVEL EXTRAPOLATIONS

The logarithmic trend curves that were derived which express SPL and SEL as a function of pile diameter give at best a rough approximation since other factors such as local geology, thickness of the pile wall, and hydraulic hammer energy also influence the noise levels generated during piling (Betke & Matuschek, 2010; Fricke & Rolfes, 2015).

It should be noted that monopiles with diameters exceeding 7.0 m have yet to be installed, that noise mitigation systems have been used for the piling of all piles exceeding 6.0 m, and that the relation between pile diameter and noise levels thus remains uncertain for these XL (extra large) monopiles. As such, our estimates should be interpreted with considerable caution. However, it is clear that noise mitigation measures will need to be used to comply with conditions of the environmental license.

POSSIBLE NOISE MITIGATION MEASURES AND THEIR IMPACT

A wide range of noise mitigation systems has been developed and tested in offshore wind farms since Germany and Denmark both adopted piling noise level restrictions in 2012. These can be roughly categorized as bubble curtain systems, shell-in-shell systems, and others of which the Hydro Sound Damper and AdBm acoustic resonator are best documented.

A bubble curtain is formed around a pile by freely rising bubbles created by compressed air injected into the water through a ring of perforated pipes encircling the pile. Various types of bubble curtains exist (Little Bubble Curtains, Big Bubble Curtains, Double Big Bubble Curtains) and they are currently the most widely used techniques of noise mitigation. In Little Bubble Curtains (LBC) perforated pipes surround the pile in a close fit. LBC are less suitable in areas with strong currents as sound leakages may occur when bubbles drift away. A big bubble curtain (BBC) is a ring of perforated pipes positioned on the sea floor around the foundation to be piled. Compressors located on the construction vessel or on a platform feed air into the pipe. The air passes into the water column by regularly arranged holes. Freely rising bubbles form a large curtain around the entire structure, even during running tides, thus shielding the environment from the noise source (Koschinski & Lüdemann, 2013). Double Big Bubble Curtains (DBBC) add a second of ring of perforated pipes around a BBC. Noise reductions of 5 – 14 dB SEL, 10 – 15 dB SEL and 14 – 18 dB SEL have been found for LBC, BBC, DBBC respectively (Bellman *et al.,* 2015). Both the Rentel and Norther intend to deploy a Big Bubble Curtain (BBC) during piling to mitigate the impacts of excessive underwater noise (Figure 3). It is quite clear that a single noise mitigation measure, big bubble curtain, will in all likelihood not in itself suffice to comply with the national noise regulations. Koschinski & Lüdemann (2013) state that “a BBC is the best-tested and the most thoroughly proven noise mitigation technique for foundations of OWFs, but caution that certainty in noise reduction level cannot be guaranteed.”

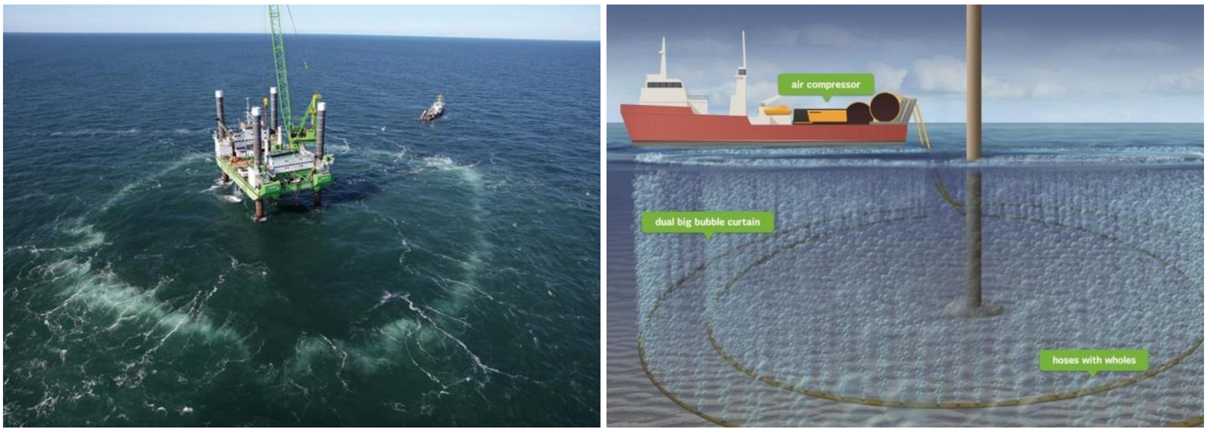

Figure 3. Left: Big Bubble Curtain in operation at Borkum West II. Note the presence of the BBC installation vessel (upper right corner) which also powers the compressors (Trianel GmbH). Right: Schematic of dubble big bubble curtain (DanTysk.com).

Shell-in-shell systems require encasing of the pile by an additional structure and thus reflect a part of the noise back inside. Various systems have been developed using additional layers containing air (foam, composites or bubbles freely rising inside) and the space between the pile and the casing can be water filled (with or without air bubbles) or dewatered. By combining several principles of noise reduction (shielding/reflection, absorption, scattering by air bubbles), shell-in-shell systems have a high theoretical noise reduction potential that is assumed to significantly exceed that of a BBC. They come however with a higher cost to developers as the heavy weight of most isolation casings requires a special design of the jack-up-rig, and as the time required to install the casing significantly increases construction time.

Hydro Sound Damper systems use fishing nets with air filled elastic balloons and special polyethylene foam elements with high dissipative effects to reduce continuous and impact noise (Elmer & Savery, 2014). Although this system is promising, with acoustic reductions of 9 dB (SEL) on average, and up to 15 dB Lz-p (Bruns *et al.,* 2014), it was not selected by the developers. It lengthens the construction time per pile because it needs to be fixed to the piles and doubts remain as to its application in an area with strong tidal currents. The AdBm Noise Abatement System consists of arrays of tuneable air-filled acoustic resonators which are deployed in a collapsible framework (Lee *et al.*, 2014). Initial tests show acoustic reductions of up to 37 dB Lz-p for these air-filled acoustic resonators (AdBm, 2014) but a full scale field deployment has yet to take place.

Other measures which can be taken to reduce the noise levels generated during piling are directly related to the technical aspects of the piling operation. These include, but are not limited to, prolonging the pulse duration (Neuber & Uhl, 2012), reducing blow energy used (Bellman et al., 2015), and using an over-dimensioned pile driver at only 2/3 of its maximum power (Nehls et al., 2007). Although studies suggest that these measures, separately, all result in a fairly limited reduction of noise levels (Bellman et al., 2015), they have the advantage that they do not greatly impact construction timing and can be used in combination with other noise mitigation measures in order to comply with the legal noise limit. Rentel has indicated that an over-dimensioned pile driver and a reduction in blow energy will be used in addition to the BBC.

In The Netherlands the so-called BLUE Piling Technology is being developed. It uses the combustion of a gas mixture under a water column located in a reservoir on top of the pile to create a pressure increase which accelerates the water upwards and causes a downward force pushing the pile into the soil. The water column then falls back again, delivering a second blow. The exhaust gases are released and the cycle is repeated. This technology would deliver much lower noise levels than a conventional hydraulic hammer (www.fistuca.com).

4.5 FURTHER STEPS

A great deal of uncertainty still exists on both the anticipated underwater noise levels for piling of the XL monopiles as well as on the level of noise reduction that can be achieved by the measures currently being proposed by the Belgian wind farm developers. It is likely that a combination of noise mitigation measures will be needed to comply with national regulations. An in-depth underwater noise monitoring programme will be needed to determine the effectively produced noise levels.

In addition to underwater piling noise restrictions, both the Belgian and Dutch government have formulated a number of measures to prevent and limit the impact of piling noise on marine mammals. These include seasonally variable noise limits or restrictions, the use of acoustic deterrent devices prior to piling, and the use of a soft start procedure. All these measures are intended to minimise the number of marine mammals exposed to piling noise. Currently, these regulations are not streamlined and at times even contradictory for the Belgian wind energy area and the Dutch Borssele zone. For example, the seasonal piling restriction in the Borssele zone lasts up to the end of May rather than April for Belgian wind farms, but can be avoided if the 350 MW wind farm consists of more than 76 foundations. Developers and the marine fauna would benefit from the alignment of regulatory practices on a regional basis. As the Belgian and Borrsele wind farms are all located relatively close to each other, (partly) concurrent piling periods at multiple parks with similar noise restrictions will benefit the marine environment (as opposed to either consecutive piling periods or wildly dissimilar noise restrictions, which are in conflict with the noise restrictions in the neighbouring country).

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CHAPTER

5

CHAPTER 5

“A WIND OF CHANGE” IN RECREATIONAL FISHERIES?

RECREATIONAL FISHERMEN AND WIND FARMS: CURRENT USE AND PERCEPTION

S. Vandendriessche[*1], *K. Persoon*[1], *E. Torreele*[1], *J. Reubens*[2,3] *& K. Hostens*[1]

[1]*Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Oostende, Belgium.*
[2]*Flanders Marine Institute (VLIZ), InnovOcean site (entrance: warehouse 48), Wandelaarkaai 7, 8400 Ostend, Belgium.*
[3] *Ghent University, Ghent University, Biology Department, Marine Biology Research Group, Krijgslaan 281, Campus Sterre - S8, 9000 Ghent, Belgium.*

[*]*Corresponding author: Sofie.Vandendriessche@ilvo.vlaanderen.be*

ABSTRACT

Offshore wind farms create opportunities for recreational fishermen in Belgium, since the presence of hard substrates and the closure for trawling create a favorable habitat for fish. After the construction in 2008, a concentration of anglers was observed in the vicinity of the first wind farm during monitoring. In the following years, however, the interest of anglers for the wind farms seemed to disappear. To elucidate the evolution in the relation between recreational angling intensity and wind farms, this study aimed to assess how Belgian recreational fishermen perceive wind farms, how often they visit them and why, and which fish species they (expect to) catch. Data were derived from the annual DCF survey for recreational fishermen. Less than 2% of the sea anglers reported to go fishing in the larger wind farm area, even when 30 to 40 percent of the respondents either expected more fish, bigger fish or other fish species. The main reasons to stay away from wind farms is because entering the wind farms themselves is not allowed, because the distance to the wind farms is relatively large, because charter vessels do not offer fish trips to wind farms, and because wind farms are protection zones and nursery areas for fish. 40% of the respondents would consider fishing inside wind farms if it were allowed, mainly because they expect more or other fish. This is a clear indication that the enforcement of wind farm closure for fisheries and shipping is vital when aiming at the creation and/or restoration of nursing grounds in the area. However, the large distance to the wind farms will probably continue to limit fishing pressure, even if wind farms would (partly) be opened for recreational fisheries.

5.1. INTRODUCTION

Wind farms function as artificial reefs, harboring high biomasses and acting as aggregation and/or production sites for different fish species (Pickering & Whitmarch, 1997; Reubens *et al.*, 2013a; Reubens *et al.*, 2013b). This presents opportunities for both commercial and recreational fisheries. At the same time, this reef effect could negatively affect fish stocks, as aggregated fish are easier to catch (Rose & Kulka, 1999), thereby worsening overfishing on stocks already under pressure. Hence, the evolution of fisheries effects should be closely monitored in relation to wind farm development.

In Belgium, offshore wind farms are closed for vessels not involved in wind farm maintenance or scientific research, or associated with the government (KB Veiligheidsafstanden 11/04/2012) . This means that fishing is prohibited based on safety concerns. A side effect of this measure is that the Belgian wind farms act like a small scale MPA (Marine Protected Area). Due to the absence of commercial trawlers and the relatively high abundances of gadoid species such as cod *Gadus morhua* and pouting *Trisopterus luscus,* the immediate vicinity of wind turbine foundations is an ideal location for recreational angling.

During the early years of construction and operation, ship-based monitoring surveys indicated that recreational fishermen aggregated in the vicinity of the first wind turbines (Fig 1, map 2008-2009). Although the intensity of surveys further increased in 2010-2011, observations of anglers decreased and the link with wind farms seemed mostly gone. This might be due to a number of reasons: the wind farms are quite far for angling day trips, less fish is caught than expected, anglers have to respect a safety distance of 500m and, as such, cannot fish as close to the hard substrates and structure as with wrecks, etc. (Vandendriessche *et al.*, 2013).

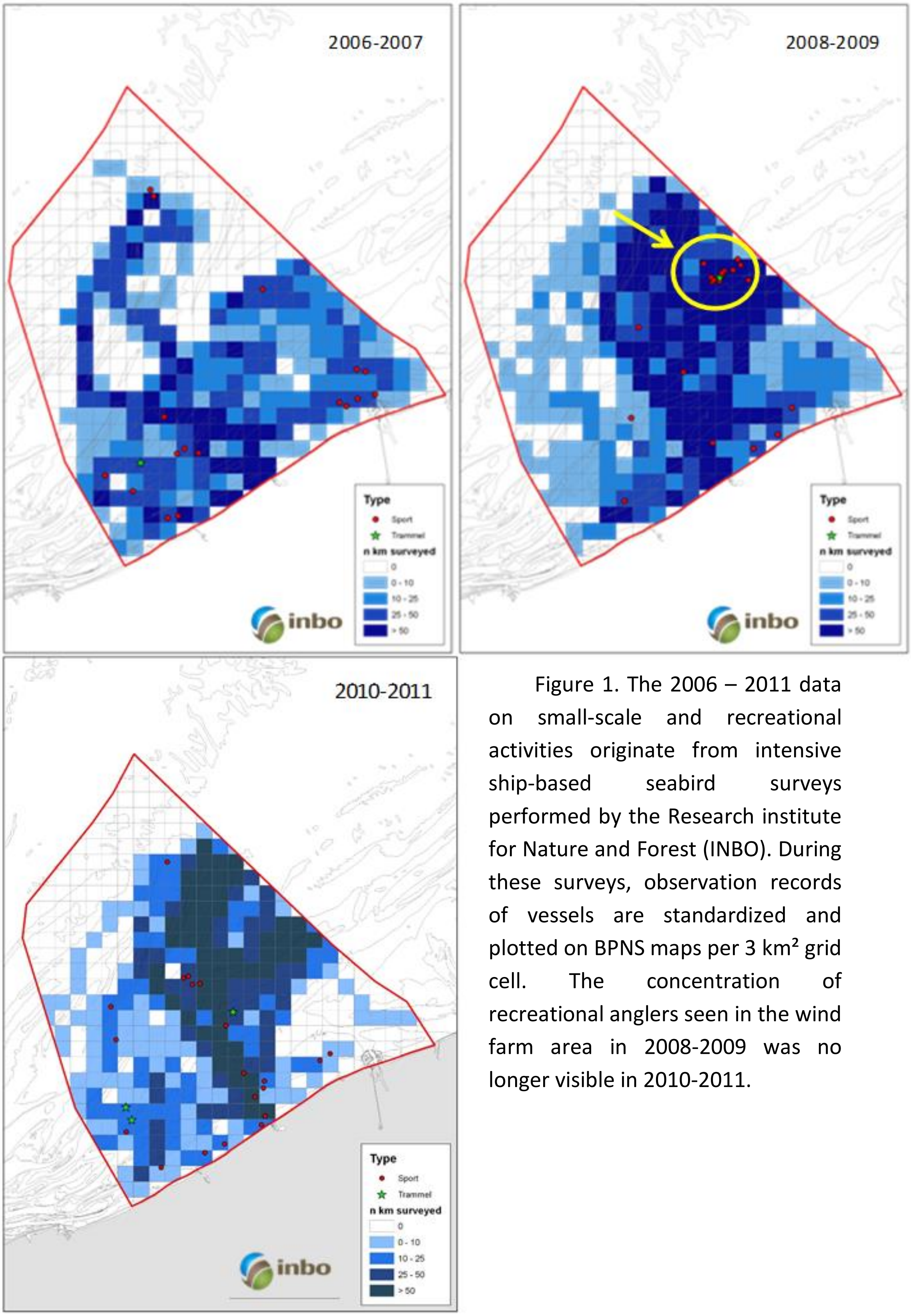

Figure 1. The 2006 – 2011 data on small-scale and recreational activities originate from intensive ship-based seabird surveys performed by the Research institute for Nature and Forest (INBO). During these surveys, observation records of vessels are standardized and plotted on BPNS maps per 3 km² grid cell. The concentration of recreational anglers seen in the wind farm area in 2008-2009 was no longer visible in 2010-2011.

To elucidate the current relation between recreational angling intensity and wind farms, this study aimed to assess how Belgian recreational fishermen perceive wind farms, how often they visit them and why, and which fish species they (expect to) catch.

5.2. MATERIAL AND METHODS

To gather information on recreational fishing at wind farms, specific questions were included in the 2015 questionnaire for recreational fishermen at sea. This questionnaire is an annual obligation within the Data Collection Framework (DCF, EC 199/2008). Under this framework, European Member States (MS) collect, manage and make available a wide range of fisheries data needed for scientific advice. The DCF distinguishes between commercial and recreational fisheries, with recreational fisheries being defined as 'non-commercial fishing activities exploiting living aquatic resources for recreation or sport.' The questionnaire survey was carried out in Belgium for the first time in 2013-2014 (Zenner *et al*, in prep).

The design of the initial survey was adapted in 2015 based on recommendations from respondents, scientists and an international working group on recreational fisheries (WGRFS). Additional questions were included to address the effect of wind farms on the distribution of recreational fisheries. A full version of the survey is available in the Annex. Both paper versions and an online survey were distributed among recreational fishermen and were promoted by means of flyers distributed from charter vessels, social media and through recreational fisheries organizations (e.g. VZW Sportvisserij Vlaanderen and VZW Zeevissport). The questionnaire was designed and distributed in Dutch. All responses were stored, structured and analyzed using MS Access. Only the questions and responses that are relevant with regard to wind farms are discussed in this report[1].

5.3. RESULTS

GENERAL (RESULTS OF Q1)

A total of 224 completed questionnaires was received and analyzed. The majority of the respondents were men (98.7%). The age of the respondents ranged between 15 and 79, but almost half of the respondents (47%) were older than 55. Most respondents originated from the provinces West-Vlaanderen (91), Oost-Vlaanderen (59), Antwerpen (42) and Vlaams-Brabant (20). Fishing is mainly done by angling from the shore or at sea, or by recreational shrimp fishing.

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[1] A full report on the results of the questionnaire survey is available on request, contact co-author Els Torreele

ANGLERS AT SEA (RESULTS OF Q4.1)

Our observations show that in the vicinity of Belgian wind farms, recreational fishing is only done by angling. 119 respondents (53%) indicated to go angling at sea, mostly over soft sediments (sandbanks) or ship wrecks. The main target species for angling are cod (*Gadus morhua*), dab (*Limanda limanda*), sea bass (*Dicentrarchus labrax*), sole (*Solea solea*), whiting (*Merlangius merlangus*), mackerel (*Scomber scombrus*), flounder (*Platichthys flesus*) and plaice (*Pleuronectes platessa*). Only 2 of the sea anglers (1.7%) indicated to go fishing in the larger wind farm area (at safety distance from the nearest turbine). Their reasons to fish in this area are (1) there is abundancy of fish and (2) there is little competition. One of these fishermen reported that he caught the following species in the larger wind farm area: sea bass, pouting (*Trisopterus luscus*), whiting, saithe (*Pollachius virens*), dab, sole, plaice, flounder and other species. One respondent indicated to fish at a UK wind farm for rays and sharks.

PERCEPTION ON WIND FARMS (RESULTS OF Q7)

Question 7 was not limited to sea anglers, and was answered at least partly by 116 respondents (52%). Most fishermen indicated not to go fishing in the vicinity of the wind farms, or to stay well away from the turbines. Based on the answers to the open question, following reasons can be listed:

- Entering the wind farms is not allowed for safety reasons (33 respondents)
- The Belgian wind farms are too far away from harbors (24 respondents)
- Charter vessel do not go there (11 respondents)
- Wind farms are protection zones and nursery areas for fish and should be respected as such (5 respondents)
- There is a lot of noise (1 respondent)

Answers to the question Q7.2, whether respondents perceived or expected changes in fish in the vicinity of wind farms, were often positive: 30 to 40 percent of the respondents either expected more fish, bigger fish or other fish species (Figure 2).

To the question whether respondents would consider fishing inside wind farms if it were allowed and why (not), 40% of the respondents answered positively, mainly because they expect more or other fish. A few respondents indicated that they would consider fishing in wind farms because of “the challenge of fishing there”, “the peace and quiet at the wind farms” or “ to check whether it is true what scientists say about the wind farms”. 32% indicated not to consider fishing at wind farms even if it were allowed, mainly to protect fish stocks. Other reasons are because the wind farms are too far away and fuel is expensive, and due to safety considerations.

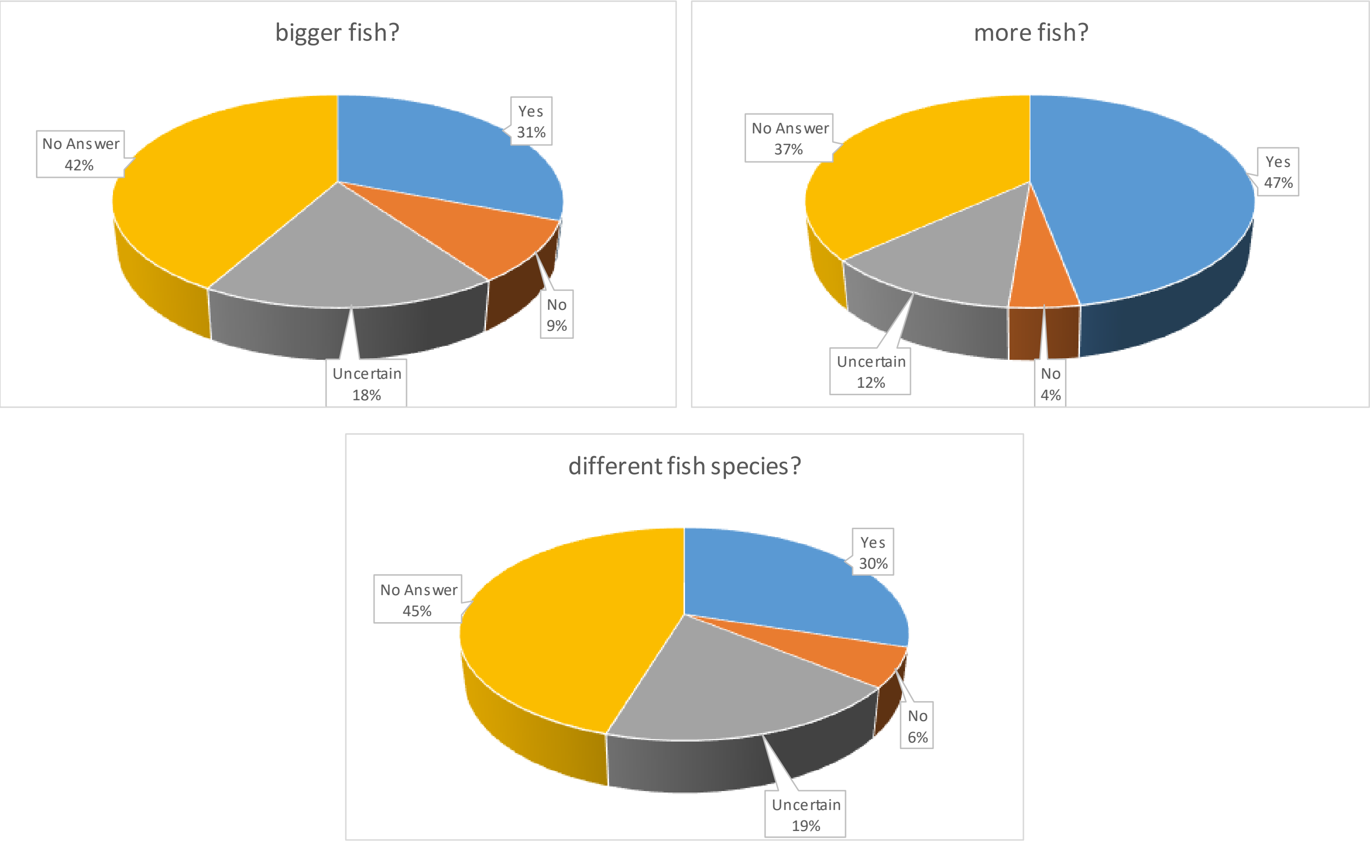

Figure 2. Pie charts representing answers of respondents to Q7.2 of the questionnaire: “Do you perceive or expect an effect of wind farms on fish? Do you expect (1) bigger fish, (2) more fish, (3) other fish species?

5.4. DISCUSSION

SAMPLE SIZE

Survey-based studies ideally result in a representative sample of the population under study. In the present study, the representativeness of the analyzed survey results is uncertain since

1. information about the size and configuration of Belgian recreational fisheries is largely lacking since recreational fishermen have no registration obligation (Van Winsen *et al*, 2016)
2. the survey was distributed in Dutch only, so a number of French-speaking recreational fishermen probably did not respond to the questionnaire
3. the survey was mainly promoted as an online questionnaire (a paper version was available on request)

Still, the response rate was considered to be quite high. The majority of the respondents was not sampled in the previous DCF-surveys: 79% of the respondents had never filled out a survey concerning their fishing activities before, indicating that the targeted public is increasingly aware of the annual survey. Still, taking into account that the factor for upscaling to the whole population is unknown, some careful conclusions can be made based on the survey results (Van Winsen *et al*, 2016).

CURRENT USE OF THE BELGIAN WIND FARM AREA BY RECREATIONAL FISHERMEN

Whether it is due to the safety measures around wind farms, the relatively large distance from the coast in relation to fuel costs, or another reason, recreational fisheries are currently almost non-existing in the larger vicinity of wind farms according to the survey results. Remarks of several respondents suggest that the reserve status of wind farms is respected by recreational fishermen. Since wind farms are closed for commercial trawling as well, the fishing pressure within the wind farms and their safety buffer can be considered extremely low compared to fishing grounds within the Belgian part of the North Sea (see also Vandendriessche *et al,* 2013). Fishing effort just outside the safety buffer is mainly limited to commercial fisheries, since only two recreational fishermen report to fish in the vicinity of wind farms. Unfortunately, a question on how close these fishermen go to the turbines was not included. Such a question should be considered for future questionnaires, especially since biological research (Degraer *et al*, 2013) indicates that the reef effect takes place in the immediate vicinity of the foundations and that effects further away from the turbines are limited. Hence, anglers respecting the 500m safety buffer will probably not notice an effect and will not return.

Low fishing pressures are not universal for European wind farms. They can be expected in Germany and in the Netherlands[2], where wind farms are closed for fisheries as well. In Denmark, on the other hand, passive fisheries such as angling is allowed in some of the wind farms, and in the UK, fisheries are allowed, including trawling (Verhaeghe *et al*, 2011). Nevertheless, little information is available on the extent of the use of these wind farms by recreational fisheries, mainly because only larger vessels are usually included in monitoring surveys and effects analyses (e.g. Mackinson *et al*, 2006).

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[2] Starting in 2017, vessel transit and multiple usage (including recreational fisheries and professional fisheries that do not affect seafloor integrity) will be allowed in operational wind farms in the Netherlands. For details and restrictions see Ministerie van Infrastructuur en Milieu, 2015.

PERCEPTION ON WIND FARMS AND CONSIDERATIONS ABOUT FUTURE USE

The survey results indicated that there is an enthusiasm among recreational fishermen to fish within the wind farms (if it was allowed), mainly due to the expected positive effect of wind farms on fish populations (more fish, bigger fish, other species). Almost 40% of the respondents indicate that they would go fishing within a wind farm if it was allowed. This is a clear indication that the enforcement of wind farm closure for fisheries and shipping is vital when aiming at the creation and/or restoration of nursing grounds in the area. With regard to species like cod *Gadus morhua,* which is recovering but still has not achieved sustainable levels in the North Sea, Reubens *et al.* (2013b) already suggested that the wind farms should remain closed to all types of fishing.

On the other hand, wind farms create opportunities for multiple stakeholder use. Based on a case study with an offshore wind power facility in the Adriatic, Fayram and de Risi (2007) stated that creating a limited entry for recreational fishery and excluding commercial fishing from the area surrounding offshore wind turbines, may aid in controlling total harvest and may benefit several important stakeholder groups: (1) recreational and commercial fishermen in terms of higher recreational catch rates and potentially higher overall yield, (2) fisheries managers in terms of more precise control of recreational fisheries harvest, and (3) owners of offshore wind power facilities in terms of reduced risk of damage to infrastructure due to fishing activity. Similarly, Verhaeghe *et al* (2011) suggested that angling for seabass could be compatible with wind farms. For this species, but also for species such as cod and pouting, which appear to be attracted to the Belgian wind farms (Reubens *et al*, 2013a) and have been reported to be caught by anglers in the vicinity of the wind farms, high yields could be expected for recreational anglers.

In the Netherlands, motivations for opening operational wind farms for vessel transit and multiple use (e.g. recreational shipping and aquaculture) are mainly a more efficient use of marine space and opportunities for biodiversity and a sustainable use of the North Sea (Ministerie van Infrastructuur en Milieu, 2015). In Belgium however, the questionnaire survey results indicate that the large distance between the wind farms and the Belgian harbors (e.g. ± 27km between the Thorntonbank wind farm and the harbor of Zeebrugge) is an important reason not to go fishing there. Consequently, the distance to the existing wind farms will probably continue to limit fishing pressure, even if wind farms would (partly) be opened for recreational fisheries. If new wind farms would be constructed closer to shore, however, the relation between recreational fishing intensity and wind farms could change substantially.

ACKNOWLEDGEMENTS

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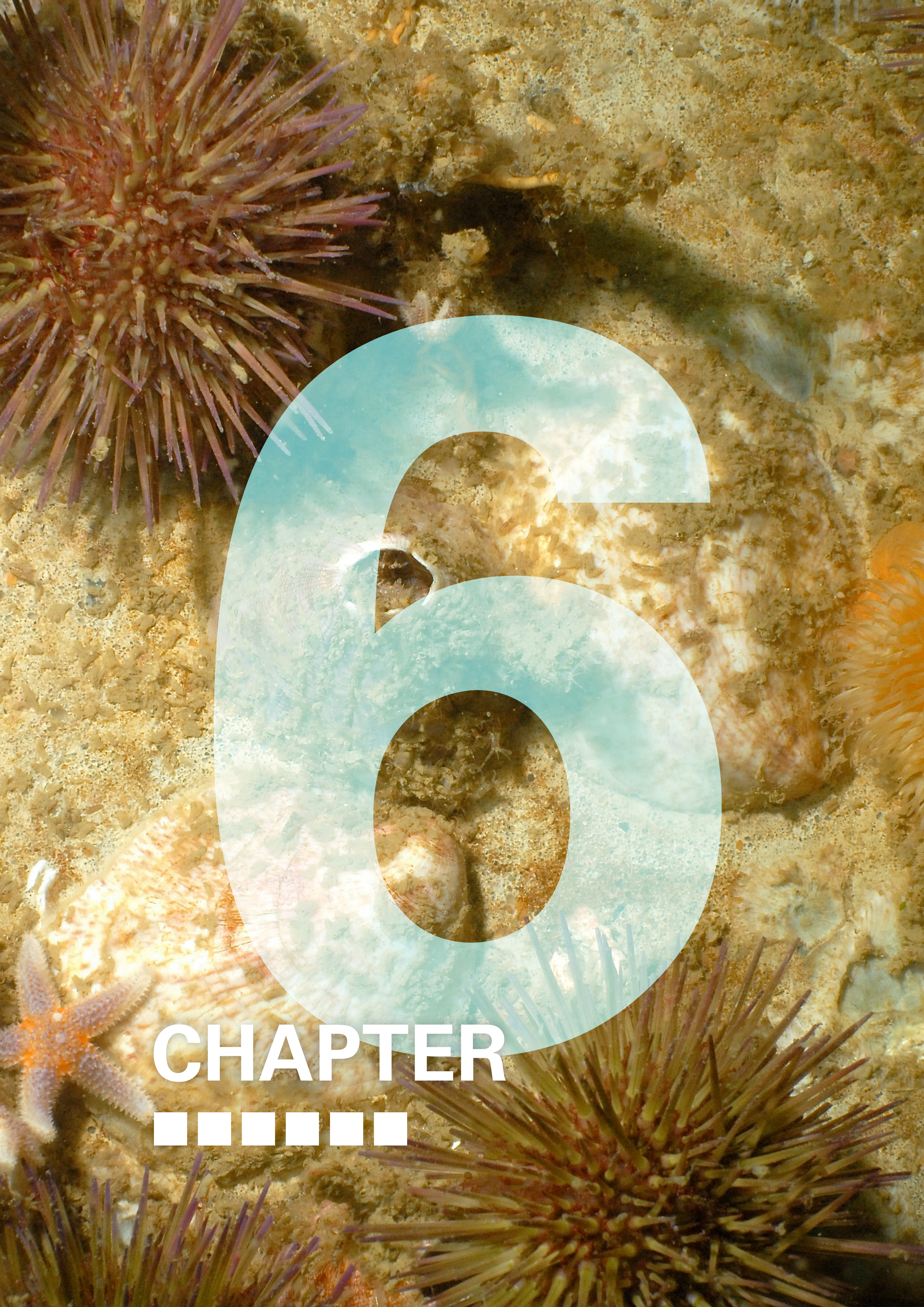

6

CHAPTER

CHAPTER 6

DO WIND FARMS FAVOUR INTRODUCED HARD SUBSTRATA SPECIES?

F. Kerckhof[*], I. De Mesel & S. Degraer

Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels; 3de en 23ste Linieregimentsplein, 8400 Oostende, Belgium.

[*]*Corresponding author: Francis.Kerckhof@naturalsciences.be*

ABSTRACT

Offshore wind farms, like other artificial structures in the marine environment, are hypothesised to favour introduced species and as such pose a threat to the native fauna. However, this has so far never been investigated for offshore wind farms. In this study, we investigated introduced species on Belgian offshore wind farms with particular interest in (1) the position of introduced species on offshore wind farms in relation to other hard substrata in the Belgian part of the North Sea (BPNS), (2) the distribution of introduced species in the subtidal versus intertidal zone and (3) the potential of offshore wind farms for future flourishment of the introduced species. Therefore we compared different hard substratum communities, both natural and man-made, on the relative importance of introduced species in the subtidal and intertidal communities.

Overall we detected eleven introduced and two cryptogenic species on the wind turbines, seven of which are intertidal species (i.e. *Balanus (Amphibalanus) improvisus, Crassostrea gigas, Elminius (Austrominius) modestus, Hemigrapsus sanguineus, Jassa marmorata, Megabalanus coccopoma* and *Telmatogeton japonicus*) and four are subtidal species (i.e. *Corophium (Monocorophium) sextonae, Crepidula fornicata, Diplosoma listerianum* and *Fenestrulina delicia*). We found that, all but one introduced species observed on the offshore wind farms in Belgian waters (i.e. *F. delicia*), were already known from the BPNS. Clear colonisation patterns occurred in both wind farms and this can be considered a confirmation that the observed patterns are consistent and may hence be expected similar in other wind farms in the southern North Sea. In the subtidal zone, the offshore wind farms will only marginally contribute to the further spread of introduced species given the vast amount of both natural and artificial hard substrata already available in the North Sea, which already contain established populations of the same introduced species. However, for the intertidal zone, the wind farms may have the potential to substantially increase the risk of the further spreading of introduced species, given that offshore intertidal habitat still is relatively rare. Wind farms will indeed drastically increase the available habitat to intertidal introduced species. It is however expected that offshore wind farms may significantly contribute only to the spread of clear water, intertidal introduced species, as such nuancing the introduction and invasion risk posed by offshore wind farms.

6.1. INTRODUCTION

In the last decades, the predominantly sandy coastline along the southern North Sea underwent drastic changes under the influence of human activities. The number of coastal defence works increased all along the coastline (www.kustveiligheid.be), ports are expanding and other large infrastructural works are taking place (www.maasvlakte2.com; www.vlaamsebaaien.com). Even offshore waters are undergoing a major industrial development, especially with the increase of offshore wind farms (www.ewea.org; Rodrigues et al., 2015). The proliferation of all these man-made structures resulted in an overall hardening of the coast and its offshore waters.

The hardening is still ongoing. Wind farms extend further offshore and will in the future occupy large areas of the shallow waters of the North Sea (www.ewea.org). Some of the hard substrata such as wind farms create completely new habitats in the marine ecosystem. It is often postulated that wind farms, like other artificial structures in the marine environment, would favour introduced species and as such pose a threat to the native fauna (Glasby et al., 2007; Bulleri & Chapman, 2010, Kerckhof et al., 2011). For instance, wind turbine foundations create an intertidal zone, formerly non-existent in offshore North Sea waters (Kerckhof et al., 2010).

The increased availability of man-made hard substrata, together with the increased activities of vectors such as shipping, not only allows a much faster and more intense transport of certain species all over the globe but the migrants now find more suitable habitat to settle and to prosper in regions beyond their original distribution. This is explicitly the case in coastal habitats, including estuaries and harbours (Carlton, 1996b; Wolff 1999; Wolff, 2005; Galil et al., 2009).

In this study we quantified the importance of the hard substrata created by wind farms to introduced species with the specific aim of exploring the risk of wind farms to contribute to the further spread of introduced and potentially invasive species throughout the North Sea. To this extent, we focused on introduced species on Belgian offshore wind farms and investigated (1) the relative dominance of introduced species in the subtidal versus intertidal zone of offshore wind farms, (2) the colonisation pattern of introduced species on offshore wind farms and (3) the position of introduced species on offshore wind farms in relation to other hard substrata in the Belgian part of the North Sea (BPNS).

6.2. MATERIAL & METHODS

INTRODUCED SPECIES: WHAT'S IN A NAME

In this study, we defined introduced species as non-indigenous species that are introduced in a certain region – in this case the North Sea – by historical human intentional or unintentional activities (e.g. Carlton, 1996a) across natural dispersal barriers. This means that they originate from areas around the globe that are non-adjacent to the North Sea. These areas include the Mediterranean, the Black and Caspian Seas (Wolff, 2005). Thus, range-expanding species, i.e. species that are spreading into the North Sea from adjacent regions where they occur indigenously, were excluded from this study.

Additionally, we took into account cryptogenic species. Cryptogenic species are species of which the status – indigenous or non-indigenous – cannot be scientifically proved (Carlton, 1996a). The cryptogenic species included in this study meet most of the attributes proposed by Chapman and Carlton (1991), e.g. having a wide-spread occurrence in harbours and other coastal habitats, association with human mechanism(s) of dispersal. Introduced and cryptogenic species are further collectively called introduced species.

In this study, we only considered macrofaunal (retained by a 1 mm mesh-sized sieve) introduced species in the BPNS encompassing coastal harbours. We included euryhaline (>30 psu) and polyhaline (18-30 psu) species, and excluded mesohaline and oligohaline species living in brackish waters below 18 psu.

DATA AVAILABILITY

Belgian offshore wind farms

We extracted a species list for the subtidal and intertidal community on the wind turbines – foundations and scour protection – from the database with all available data from the C-Power and Belwind wind farms. Hard substrata macrofauna was collected from the C-Power and Belwind wind farms, located in the Belgian offshore renewable energy zone (see Brabant et al., 2011). The C-Power wind farm (six concrete gravity-based foundations (GBF), 49 jacket foundations) is located on the Thornton Bank some 30 km offshore. The Belwind wind farm (during the study period: 56 steel monopiles and 1 jacket foundation) is situated on the Bligh Bank at about 50 km off the coast. Both banks belong to the Zeeland Banks system (Cattrijsse & Vincx, 2001). The samples were collected late 2008-2015 from a selected set of wind turbines: D5 and D4 at the C-Power site and BBB8, BBC2 and BBC8 at the Belwind site. The samples included scrape samples on the turbine foundations and stones gathered from the scour protection (Kerckhof et al., 2011).

Out of the species pool of all species identified, we eliminated those species that were usually not associated to hard substrata such as infaunal or pelagic macrofauna that accidently occurred in the samples. This yielded a list with genuine hard substratum species. We further also classified the species according to their observed prevalent occurrence in the sub- or intertidal zone. In this study, intertidal species are those species living in the eulitoral and splash zone. Species were considered intertidal if they solely or predominantly inhabit the eulitoral and or splash zone, while species mainly having a sublitoral distribution and only occurring occasionally in the infralitoral fringe (i.e. lower mussel zone) were considered true subtidal species (e.g. Hayward and Ryland, 1990; Hiscock et al., 2005; http://www.marlin.ac.uk/).

The SACFOR scale (Connor & Hiscock 1996) was used to score the relative abundance of the offshore wind farm introduced species.

Other Belgian hard substrata

We compiled a list of introduced species associated with hard substrata in Belgian waters based on an inventory of all introduced species in Belgian waters that was assembled using various available sources e.g. Kerckhof et al. (2007) and the various Belgian reports submitted to the ICES Working Group on Introductions and Transfers of Marine Organisms (WGITMO) (ICES, 2001 – 2016).

Subsequently, we allocated the introduced species to the different habitats within which these occur. To that extent, we screened different datasets and publications dealing with the fauna on Belgian artificial hard substrata such as ship wrecks (Zintzen, 2007; Zintzen, 2010), coastal defence structures (Daro, 1969; Engledow et al., 2001; Volckaert et al., 2003; Volckaert et al., 2004; Kerckhof et al., unpubl. data EMBOS), harbour works (Derweduwen et al., 2014) and offshore buoys (Kerckhof & Cattrijsse, 2001; Kerckhof F., unpubl. data) for the presence of introduced species. The list of species detected in the subtidal samples consisted of 148 species of which 144 were considered as true subtidal species. Intertidally, we identified 30 species of which 16 were classified as true intertidal species.

We further classified the introduced species as established, non-established or extinct. We considered a species established when the species has been regularly observed (i.e. not restricted to a single observation) with several individuals, suggesting a viable and self-sustaining population for that species in the BPNS and its adjacent waters. If the introduced species did not meet these criteria it was considered non-established. We considered an introduced species as extinct if, after a prolonged period of presence, the species has not been detected anymore during the last five years even after dedicated inspection of its habitat.

6.3. RESULTS

INTRODUCED VERSUS NON-INTRODUCED SPECIES IN OFFSHORE WIND FARMS

In the intertidal zone of the offshore wind farms, six species were introduced: *Crassostrea gigas*, *Elminius (Austrominius) modestus*, *Megabalanus coccopoma*, *Jassa marmorata*, *Hemigrapsus sanguineus* and *Telmatogeton japonicus*, and one is cryptogenic: *Balanus (Amphibalanus) improvisus*. In the subtidal, three introduced species (i.e. *Crepidula fornicata*, *Corophium (Monocorophium) sextonae* and *Fenestrulina delicia*) and one cryptogenic species (i.e. *Diplosoma listerianum*), were detected.

The introduced species share is relatively high in the intertidal zone (i.e. 23 %) (Figure 1A), while in the subtidal the share is very low (i.e. 2.7 %). If the true intertidal species allocation is considered (Figure 1B), the introduced species share in the intertidal is even higher (i.e. 43 %), while with 2.8 % the share remains very low in the subtidal community.

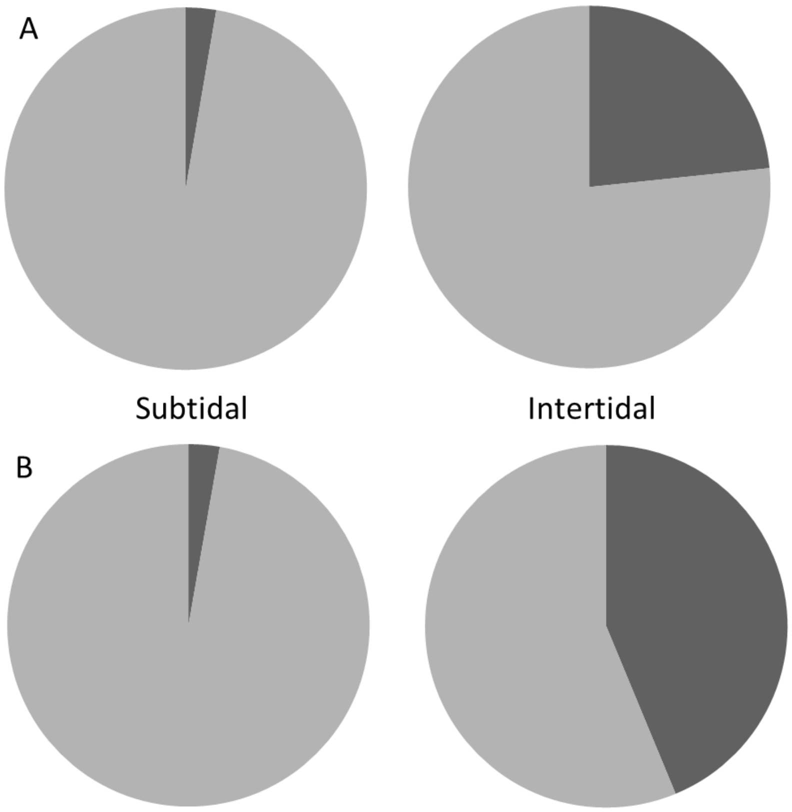

Figure 1. Intertidal and subtidal introduced versus non-introduced species richness on Belgian offshore wind farms. A, subtidal and intertidal species allocation as detected in the samples. B, true subtidal and true intertidal species allocation. Dark grey, introduced species; light grey, non-introduced species.

INTRODUCED SPECIES COLONISATION PATTERN ON OFFSHORE WIND FARMS

Subtidally, two introduced species were present from the start in both wind farms, i.e. *C. fornicata* and *C. sextonae*, but only the abundant *C. fornicata* persisted after year one (Tables 1 and 2). Two other species came in only after three years, i.e. *D. listerianum* (abundant in both wind farms) and *F. delicia* (rare and only on the C-Power scour protection).

Table 1. Colonisation pattern of intertidal (IT) and subtidal (ST) introduced species (*, cryptogenic) at the C-Power gravity-based foundation (CP) and the Belwind monopile (BW) wind farms. Semi-quantitative abundances using SACFOR scale: S, superabundant; A, abundant; C, common; F, frequent; O, occasional; R, rare.

			Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8
*Balanus improvisus**	IT	CP			O					no data
		BW	O				no data		no data	no data
Crassostrea gigas	IT	CP			O	O	O	F	F	no data
		BW					no data	F	no data	no data
Elminius modestus	IT	CP	A	A	A	A	A	A	A	no data
		BW	C	C	C	C	no data	C	no data	no data
Hemigrapsus sanguineus	IT	CP			F	F		F		no data
		BW					no data	O	no data	no data
*Jassa marmorata**	IT	CP	C	C	C	C	C	C	C	no data
		BW	C		O		no data	C	no data	no data
Megabalanus coccopoma	IT	CP	C							no data
		BW	F				no data		no data	no data
Telmatogeton japonicus	IT	CP	S	S	S	S	S	S	S	no data
		BW	S	S	S	S	no data	S	no data	no data
Corophium sextonae	ST	CP	R							
		BW	F						no data	no data
Crepidula fornicata	ST	CP	A	A	A	A	A	A	A	A
		BW	F	F	F	F	A	A	no data	no data
*Diplosoma listerianum**	ST	CP			R	O	O	O	F	F
		BW			A		S	F	no data	no data
Fenestrulina delicia	ST	CP			R	R	R			
		BW							no data	no data

Elminius modestus, *T. japonicus* and *J. marmorata* were early colonizers of the intertidal zone, all three persisting commonly to superabundantly throughout the study period (Tables 1 and 2). Another early colonizer in both wind farms, i.e. *M. coccopoma*, disappeared after one year in both wind farms. Later on in the succession, the C-Power intertidal zone became home to *C. gigas* and *H. sanguineus* from the third year onwards, while these species were not detected in Belwind until after six years. No clear succession pattern can be deducted for *B. improvisus* that was only found in low numbers on two occasions.

Table 2. Colonisation time and persistence of the introduced species at the C-Power and the Belwind wind farms.

	Early / late coloniser	**Persisting / non-persisting**
Balanus improvisus	Data deficient	
Corophium sextonae	Early	Non-persisting
Crassostrea gigas	Late	Persisting
Crepidula fornicata	Early	Persisting
Diplosoma listerianum	Late	Persisting
Elminius modestus	Early	Persisting
Fenestrulina delicia	Data deficient	
Hemigrapsus sanguineus	Late	Persisting
Jassa marmorata	Early	Persisting
Megabalanus coccopoma	Early	Non-persisting
Telmatogeton japonicus	Early	Persisting

INTRODUCED SPECIES ON OFFSHORE WIND FARMS IN RELATION TO OTHER HARD SUBSTRATA

The list of introduced species associated with hard substrata in the BPNS consisted of 32 species of which five (four barnacle species occurring only on navigational buoys and one bryozoan species found only in a marina) are considered non-established (Table 3). One species *Megabalanus coccopoma* is classified as extinct because it has not been reported since 2010. 27 other introduced species were observed regularly and in fair number and hence are currently considered established on hard substrata in the BPNS.

The largest number, 24 species, is found in harbours, of which 23 are established, followed by navigational buoys on which 15 introduced species were found, of which eleven are considered established, followed by coastal defence works with eleven introduced species of which nine are considered established. On the contrary, the lowest number was found on the wrecks and the reef balls. On the wrecks only three introduced species were found including the only two that were also detected on the reef balls. The number of introduced species was very low on the natural hard substrata, none were reported before 1910 while only one, *C. fornicata,* was detected in recent studies. The latter species is almost omnipresent, lacking so far only from intertidal offshore wind farm.

Table 3. List of introduced species associated with different hard substrata in the BPNS

species	**higher taxon**	**coastal defence**	**harbours**	**buoys**	**wrecks**	**wind inter**	**wind sub**	**gravel (Gilson, 1900)**	**gravel (Houziaux, 2005 & Hinders, 2013)**	**reefbals**
Aplidium glabrum	Ascidiacea		x							
Balanus amphitrite	Cirripedia	x	x	x						
Balanus improvisus	Cirripedia	x	x	x		x				
Balanus reticulatus	Cirripedia			x						
Balanus trigonus	Cirripedia			x						
Balanus variegatus	Cirripedia			x						
Bugula neritina	Bryozoa		x							
Bugula stolonifera	Bryozoa		x							
Bugula simplex	Bryozoa		x							
Boccardia proboscidea	Polychaeta	x								
Boccardiella hamata	Polychaeta		x							
Botrylloides violaceus	Ascidiacea		x							
Caprella mutica	Amphipoda		x	x						
Corophium sextonae	Amphipoda				x		x			
Crassostrea gigas	Bivalvia	x	x	x		x				
Crepidula fornicata	Gastropoda	x	x	x	x		x		x	x
Diplosoma listerianum	Ascidiacea		x		x		x			x
Elminius modestus	Cirripedia	x	x	x		x				
Fenestrulina delicia	Bryozoa						x			
Ficopomatus enigmaticus	Polychaeta		x							
Haliplanella lineata	Actinaria		x							
Haliclona xena	Porifera		x							
Hemigrapsus sanguineus	Decapoda	x	x	x		x				
Hemigrapsus takanoi	Decapoda		x							
Jassa marmorata	Amphipoda	x	x	x		x				
Megabalanus coccopoma	Cirripedia	x		x		x				
Megabalanus tintinnabulum	Cirripedia			x						
Molgula manhattensis	Ascidiacea		x							
Petricola pholadiformis juv.	Bivalvia	x	x	x						
Styela clava	Ascidiacea		x							
Telmatogeton japonicus	Diptera	x	x	x		x				
Tricellaria inopinata	Bryozoa		x							

The introduced species assemblages on the different types of hard substrata are different. Crustaceans and molluscs are dominant on all artificial hard substrata, while tunicates and bryozoans remain largely restricted to harbour environments (Table 4). So far no introduced porifera, annelids nor cnidarians were detected on the offshore wind turbines in the BPNS.

Table 4. Number of introduced species, grouped into higher taxa, on different types of hard substrata in Belgian waters. Number of established introduced species in parentheses.

Higher taxa	Total	Wind farms	Harbours	Coastal defence	Buoys
Chordata,Tunicata	5 (5)	1 (1)	5 (5)	0 (0)	0 (0)
Arthropoda, Crustacea	13 (8)	6 (5)	7 (7)	4 (4)	11 (6)
Cirripedia	8 (3)	3 (2)	3 (3)	8 (3)	8 (3)
Amphipoda	3 (3)	2 (2)	2 (2)	2 (2)	2 (2)
Decapoda	2 (2)	1 (1)	2 (2)	1 (1)	1 (1)
Arthropoda, Hexapoda	1 (1)	1 (1)	1 (1)	1 (1)	1 (1)
Bryozoa	5 (4)	1 (1)	4 (3)	0 (0)	0 (0)
Molusca	3 (3)	2 (2)	3 (3)	3 (3)	3 (3)
Bivalvia	2 (2)	1 (1)	2 (2)	2 (2)	2 (2)
Gastropoda	1 (1)	1 (1)	1 (1)	1 (1)	1 (1)
Porifera	1 (1)	0 (0)	1 (1)	0 (0)	0 (0)
Annelida	3 (3)	0 (0)	2 (2)	1 (1)	0 (0)
Cnidaria	1 (1)	0 (0)	1 (1)	0 (0)	0 (0)

6.4. DISCUSSION

PATTERNS OF INTRODUCED SPECIES IN BELGIAN OFFSHORE WIND FARMS

Introduced species occur in the subtidal and intertidal zones of Belgian offshore wind farms. Their presence is particularly noticeable in the intertidal zone with a percentage of no less than 23 %, or 43 % if only true intertidal species are considered (Figure 1A, 1B). The intertidal zone, as occurring now in the wind farms, is a new habitat in the offshore environment. It hence is no surprise that here species that were formerly not present offshore including introduced species thrive. The fact that introduced species however tend to prevail in the intertidal zone compared to the subtidal zone may be explained by species-poor and environmentally harsh environments such as the intertidal but also brackish water environments being particularly sensitive to introductions (Wolff, 1999; Wolff, 2005; Ruiz et al., 1997). The subtidal offshore wind farm hard substrata represent a more benign environment, where introduced (and other non-indigenous) species may have less opportunities for establishing.

Clear colonisation patterns occur in both wind farms. The predominantly similar pattern as observed in both wind farms can be considered a confirmation that the observed patterns are consistent and may hence be expected similar in other wind farms in the southern North Sea. The time of colonisation after wind farm construction as quantified in this study however needs nuancing. Not observing a species for example does not necessarily mean that the species is not present as it may simply have been missed during sampling. Many species indeed occur patchily hampering a reliable observation of their absence. This is particularly the case when the species is relatively rare. Some presumed absences in the database may hence be interpreted as such (e.g. *H. sanguineus* at C-Power in year 5 or *J. marmorata* at Belwind in year 4). The exact timing of arrival is further complicated by gaps in the data series. For example *C. gigas* was first observed at Belwind in year 6. However, no sampling occurred in the 5th year at Belwind and judging from the size and the growth rings of the *C. gigas*, these individuals are likely to have settled already during the previous year. Taking account of these facts justifies the smoothening of the data base interpretation as done is this study.

Notwithstanding the need to smoothen the data, both late true intertidal colonisers (e.g. *C. gigas* and *H. sanguineus*) consistently arrived later at Belwind compared to C-Power. This may be explained by the longer distance the Belwind wind farm is away from the coastline, where the major source populations for these species occur. It hence is not surprising that the likelihood of their pelagic larvae to reach the offshore constructions is lower at Belwind.

EXPLORING THE RISK OF WIND FARMS TO CONTRIBUTE TO THE FURTHER SPREAD OF INTRODUCED AND POTENTIALLY INVASIVE SPECIES THROUGHOUT THE NORTH SEA.

All introduced species on offshore wind farms except one, de bryozoan *F. delicia* (De Blauwe et al. 2014), were already known from Belgian waters. For the subtidal zone, the offshore wind farms will only “marginally” contribute to the further spread of introduced species given the vast amount of both natural and artificial hard substrata already available in the North Sea. These already host established populations of the same introduced species.

The largest number of introduced species is found in harbours and on navigation buoys followed by coastal defence works where the number equals that of the wind turbines. On all these man-made structures both the intertidal and subtidal habitats are present. Indeed, the number of introduced species proved to be particularly high in the intertidal zone on the wind turbines. The higher number in coastal waters accords with observations elsewhere (Wolff, 1999; Wolff, 2005; Ruiz et al., 1997) and illustrates that these habitats are prone to new introductions. The lowest number is found on the wrecks and on the reef balls, lacking an intertidal zone and also on the subtidal natural gravel beds.

For the intertidal zone, the wind farms have the potential to substantially increase the risk of further species introductions and introduced species spread given that – besides offshore wind farms – offshore intertidal habitat still is relatively rare. Wind farms will hence drastically increase the available habitat to obligate intertidal introduced hard substrata species for which offshore habitat did not exist in the southern North Sea until recently. Offshore wind farms hence make outer coast environments more susceptible to invasion by those species that have invaded to date (McQuaid and Arenas, 2009).

In case intertidal introduced species become invasive within the offshore wind farm, this should not necessarily be considered a problem, given the artificial origin of these communities. The problem of invasiveness hence only poses when potentially invasive introduced species reach natural rocky shore communities (formerly not exposed to these species) with the help of offshore wind farms.

However, as all except for one introduced species detected on offshore wind farms were already found on coastal artificial hard substrata they may already profit from an increased connectivity as a consequence of the coastal hardening. These species may as such spread throughout the North Sea and "invade" natural rocky shore communities, irrespective of offshore wind farms. This hypothesis may however only hold true for subtidal species and intertidal species thriving in turbid coastal waters, which consequently can have source populations on the coastal hard substrata, and not for species in need of intertidal, clear water habitat. In the latter case, offshore wind farms would pose a threat to the further spread of these species, potentially providing stepping stones onto the natural clear water, intertidal rocky shore communities along i.e. the eastern Scottish and Norwegian coasts.

The arrival of new introduced species on the offshore wind turbines can be expected. The wind farms are susceptible to the arrival and subsequent establishments of new introduced species. In the area, many vessels operate that often have been working or still alternately work in the offshore industry elsewhere around the globe. They could carry with them as fouling or in ballast water many exotic organisms. Climate change could enhance the introduction process, because warm water species could, after their initial introduction now find climatic conditions more suitable. Therefore, continued monitoring is needed as is also requested in the frame work of the EU Marine Strategy Framework Directive.

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7

CHAPTER

CHAPTER 7

EXPANSION OF SMALL-SCALE CHANGES IN MACROBENTHIC COMMUNITY INSIDE AN OFFSHORE WIND FARM?

J. Reubens[*], *M. Alsebai & T. Moens*

Ghent University, Biology Department, Marine Biology Research Group, Krijgslaan 281,
Campus Sterre - S8, 9000 Ghent, Belgium.

[*]*Corresponding author:* *Jan.Reubens@UGent.be*

ABSTRACT

The presence of offshore wind farms in the marine environment has some impacts on the macrobenthic community living in the natural sandy sediments. Changes in hydrodynamics, presence of epifaunal coverage along the turbine and fisheries exclusion are expected to be the main causes influencing the macrobenthos. In this study it was investigated whether changes in sediment characteristics and the macrobenthic community occurred inside a wind farm in the Belgian part of the North Sea. Both stations in the close vicinity of the turbines (50 m distance, close samples) and further away (350-500 m distance, far samples) were sampled with a Van Veen grab in autumn 2015.

No significant differences in abiotic factors were observed between the two distances. All samples were characterized by coarse sediments, with a low mud and total organic matter content. Macrobenthic densities on the other hand differed significantly between the two distances. Densities and number of species were higher for the far samples compared to the close samples. The latter were dominated by *Urothoe brevicornis* and *Gastrosaccus spinifer*, while *Bathyporeia elegans* and *Spiophanes bombyx* were more important in far samples. It is currently unclear what underlying ecological processes are responsible for the difference in community structure between both distances. Further, the current results are not consistent with results from previous studies, which might be related to the turbine type used. Therefore it is recommended to continue following the current sampling design for the coming years. In addition, it would be interesting to perform a targeted monitoring study to investigate potential changes in sedimentology and organic enrichment in the close vicinity of different turbine types.

7.1. INTRODUCTION

Since the early 2000s offshore wind farms are built all across the North Sea. As of June 2015 there were 3072 wind turbines present in European waters, in 82 wind farms across 11 countries (Ho and Mbistrova 2015). With the construction of these wind turbines, artificial hard substrates are introduced into the natural sandy environments (i.e. reef effect). This reef effect causes large impact on the marine environment at different scales (Petersen and Malm 2006). Biodiversity and ecosystem functioning are influenced and as a result these effects have environmental costs and benefits (Andersson et al. 2009, Langhamer 2012) including habitat alteration, changes in sediment characteristics, underwater noise and hydrodynamics. All these direct changes on the ecosystem influence community structure and trophic interactions in the marine environment, e.g. rapid colonization of hard substrates by an epifaunal community (De Mesel et al. 2013, De Mesel et al. 2015); changes in community composition of soft substrate macro- and epibenthos, demersal and benthic fish (Reubens et al. 2013, Reubens et al. 2014, Vandendriessche et al. 2015); changes in spatio-temporal distribution and migration routes of demersal fish, seabirds and marine mammals (Reubens et al. 2014, Haelters et al. 2015, Vanermen et al. 2015).

In this report we focus on the possible effects on the macrobenthic community in offshore wind farms. As stated by Kröncke (2011) and Kröncke et al. (2011) the main natural factors structuring macrobenthic species distribution and communities are temperature, the influence of different water masses, sediment type and food supply of the sediment. There is a natural temporal and spatial variability in presence of macrobenthic communities (Ysebaert and Herman 2002). Besides, anthropogenic stressors such as commercial fishing, dredging and eutrophication may play a role in structuring the macrofauna as well (Kröncke et al. 2011). Thus, one might expect that changes in sediment type, changes in food supply of the sediment and fisheries exclusion will have a major influence on the macrobenthic community present in offshore wind farms.

Macrobenthos is an important component of the marine environment to be monitored for potential reef effects. It provides us with direct information on how soft, sandy sediments and their inhabitants are changing (Coates 2014). The effects on macrobenthos can scale up to the food web, as many macrobenthic species are an important food source for demersal fish species (Vandendriessche et al. 2015). Changes in macrobenthic communities has the potential to alter food web energy flows (Dannheim et al. 2014).

For offshore wind farms a distinction can be made between construction and operation effects related to the macrobenthos (Coates 2014, Coates et al. 2015). During the construction, dredging activities have a direct effect on the macrobenthic assemblages by the removal of sediments. This leads to decreased abundance, diversity and biomass of the benthic organisms (Boyd et al. 2003, Coates et al. 2015). However, the effects on the macrobenthic community are rather small as they show a high recovery potential after disturbance and are restricted to the impacted sites (Coates et al. 2015). Effects related to the operational phase of the wind farms on the other hand, develop on a much slower pace, can be (long-)lasting and act over a larger spatial scale due to the lasting habitat alterations (Van den Eynde et al. 2013, Vanhellemont and Ruddick 2014, De Mesel et al. 2015, Coates et al. 2016).

Coates et al. (2014) revealed changes in sedimentology up to 50 m distance from wind turbines. Grain size significantly reduced and organic matter content increased close to the turbines. The changes in grain size were the result of changing hydrodynamics. In the wake of the turbines, there is a decreased current flow, which prevents the re-suspension of finer sands. The increase in organic matter results from the epifouling organisms. Epifauna present on foundations contribute to the organic matter input on the seabed by sedimentation of faeces and detritus, and filtering suspended particulate matter out of the water (Maar et al. 2009). In addition, the refinement of the sediment reduces the pore-water flow within the sediments (Janssen et al. 2005), which results in less organic matter being flushed (Coates 2014). The changes in these environmental characteristics triggered changes in the macrobenthic community. Density and diversity increase and a shift in species dominance was observed (Coates et al. 2014).

The small-scale enrichment and fining of the sediment around wind turbines is the result of the prevailing hydrodynamics and epifaunal coverage. However, it is hypothesized, that in the longer term an expansion of these changing environmental characteristics could be facilitated due to the prohibition of beam trawling inside the wind farms (Coates 2014).

Now, three years later, it is investigated whether: 1) the small scale changes observed by Coates et al. (2014) are still present and 2) changes in the environmental characteristics and macrobenthic community expanded to larger distance from the turbines.

7.2. MATERIAL & METHODS

STUDY AREA

Within the Belgian part of the North Sea (BPNS) an area of 238 km² is reserved for the production of renewable energy. This area is subdivided in several concession areas (Brabant et al. 2013). The current study was conducted in the concession area of the offshore wind farm 'C-Power', which is located on the Thorntonbank sandbank (fig. 1). The wind farm consists of 54 turbines. The first six (constructed in 2008) were built on gravity-based foundations. The other 48 turbines have a jacket foundation and were constructed between 2011 and 2013 (Brabant et al. 2013).

SAMPLE DESIGN, COLLECTION AND TREATMENT

Effect of distance from turbine

A systematic stratified sampling design was adopted (fig. 1). Samples were collected in autumn 2015 at two distances (close and far) from the wind turbines. The close samples (n = 16) were taken at a distance of approximately 50 m from the turbines on the South-West side. If sampling at South-West direction was not possible (to comply with a minimum distance of 50 m from infield electricity cables) samples were taken at the North-East site of the turbines. The far samples (n = 32) were gathered in the middle between the four surrounding wind turbines. Here, distances ranged between 350 and 500 m from the turbines (fig. 1). The close samples were gathered on October 23th and November 3th, 2015, while the far samples were collected on October 6th and 7th, 2015.

Initially, a two-way spatial (close vs far) and temporal (present vs 2011 and 2012 (Coates et al. 2014)) comparison of samples was planned. Too many differences in sampling strategy (Table 1) however, resulted in a one-way spatial comparison only.

Table 1. Overview of differences in sampling design between 2011-2012 and 2015.

	2011-2012	**2015**
Season	Spring	Autumn
Replication	3 replicates at one location	Samples as replicates
# samples	1	16 close 32 far
Gradient	Taken into account	Not taken into account

Samples were collected by means of a Van Veen grab (0.1m²), sieved alive onboard over a 1 mm sieve table and subsequently preserved in an 8 % formaldehyde-seawater solution. In the laboratory, samples were stained with Rose Bengal and rinsed over a 1 mm sieve. All macrobenthic animals were identified to species level, whenever possible. Individuals were counted and biomass (blotted wet weight, mg) was determined for every species per sample.

From the grab sample, a subsample was taken with a core (Ø 3.6cm) to obtain information on grain size distribution, total organic matter (TOM) content and mud content. Median grain size was determined on dried samples (dried at 60°c) using a laser diffraction method with a measuring range of 0.02 - 2000 µm (Malvern Mastersizer 2000G, hydro version 5.40). Sediment fractions larger than 2000 µm were quantified using a 2 mm sieve. TOM was determined per sample by weighing the difference between the dry weight (48 h at 60°C) and the ash weight (2h at 500°C).

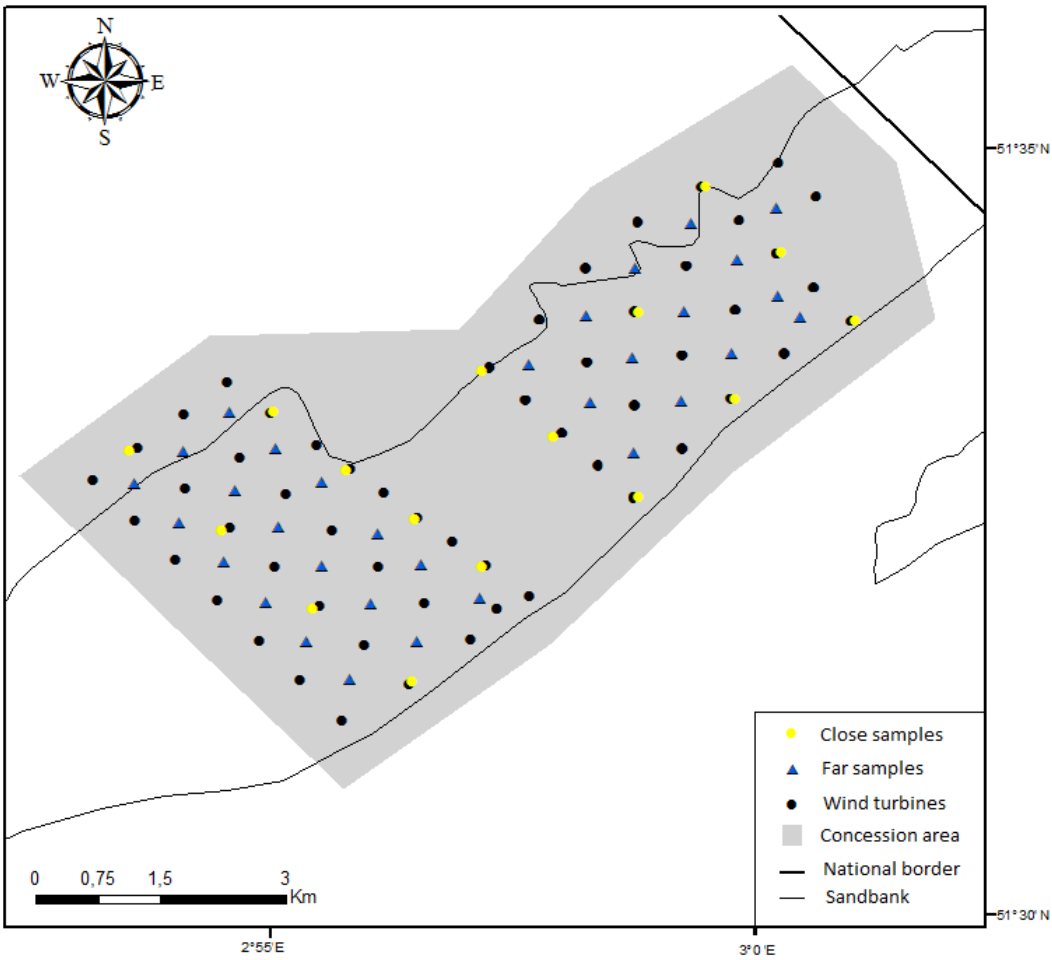

Figure 1. Overview of the C-Power concession area with indication of the close (yellow dots) and far (blue triangles) sampling locations.

Differences in median grain size over time

Although no direct comparison of biotic samples on temporal scale was possible, we investigated potential differences in median grain size over time at the C-Power concession area. Within the wind farm monitoring programme, samples on median grain from the Thorntonbank are available since 2005. However, due to construction works and safety issues, no samples could be collected within the concession area between 2011 and 2014. Data is available for 2008-2010.

DATA ANALYSIS

Effect of distance from turbine

Rare species were not removed from the dataset, as the presence of these species might be a first indication that something is changing in the macrobenthic community (not evaluated in this report). The abundance (ind m^{-2}), number of species (S) and Pielou's evenness were calculated. One-way Anovas were performed to detect any significant differences between the distances. Levene's test was used to control for homogeneity of variance, while the shapiro test was used for normality. If needed data were log-transformed.

Permutational Anova (Permanova) with a fixed one-factor (distance) design was used to investigate the effect of distance on the macrobenthic community composition. Permanova makes no explicit assumptions regarding the distribution of original variables (Anderson et al. 2008). It was decided to use Type III sums of squares as the design was unbalanced. Number of permutations was set to 9999 and unrestricted permutation of raw data was performed as there was only one factor. The multivariate analysis of abundance data was based on a Bray-Curtis resemblance matrix and performed on fourth root transformed data. The resemblance matric Euclidean distance was applied for the multivariate analysis of the environmental variables (Grain size, TOM and sediment fraction > 2mm) after normalization. Homogeneity of multivariate dispersions was tested using the PERMDISP routine, using distances among centroids. Principal Coordinates Analysis (PCO) was run to visualize the data. Furthermore, a distance-based linear model (DistLM) based on Adjusted R^2 and Stepwise criterion was carried out to investigate the relationship between the macrobenthic community and the environmental variables. Variables were tested for multi-collinearity (Anderson et al. 2008). Mud was excluded from the analysis, as data remained skewed (even after transformation). In addition a similarity percentage (SIMPER) routine was done to specify the role of individual species in separation between groups of samples and the closeness of samples within a group (Clarke and Gorley 2006).

All analyses were performed in the Plymouth Routines in Multivariate Ecological Research (PRIMER) programme (version 6.1.11) with the PERMANOVA add-on software (Clarke and Gorley 2006, Anderson et al. 2008) and in R (version 3.2.2) (Team 2015). A significance level of $p = 0.05$ was used in all tests. Results are expressed as mean ± standard deviation (SD).

Differences in median grain size over time

Since the assumptions of parametrical statistical approaches were not fulfilled, not even after log-transformation of the data, the non-parametric Kruskal-Wallis test was used to compare median grain size between years. Analyses were performed in R (version 3.2.2) (Team 2015). A significance level of $p = 0.05$ was used in all tests. Results are expressed as mean ± standard deviation (SD).

7.3. RESULTS

EFFECT OF DISTANCE FROM TURBINE

All samples consisted of coarse sediments (median grain size > 300 µm). At the close samples median grain size ranged from 301 to 515 µm, while at the far samples it ranged from 306 to 518 µm. The mud content was zero in most samples. Only two far samples had a mud content of 0.5 and 0.9 % respectively. TOM content remained low in all samples with a mean percentage of 0.59 ±0.16 at close and 0.76 ± 0.37 at the far distance. The sediment fraction over 2 mm ranged from 0.2 to 5.8 % at the close samples, while at the far samples it ranged from 0.1 to 9.2 % (Table 2 and Fig. 2). A multivariate analysis on the normalized abiotic data (Grain size, MUD, TOM and >2mm) revealed that there were no significant differences between the two distances (1-factor Permanova: p = 0.34; Permdisp: p= 0.28).

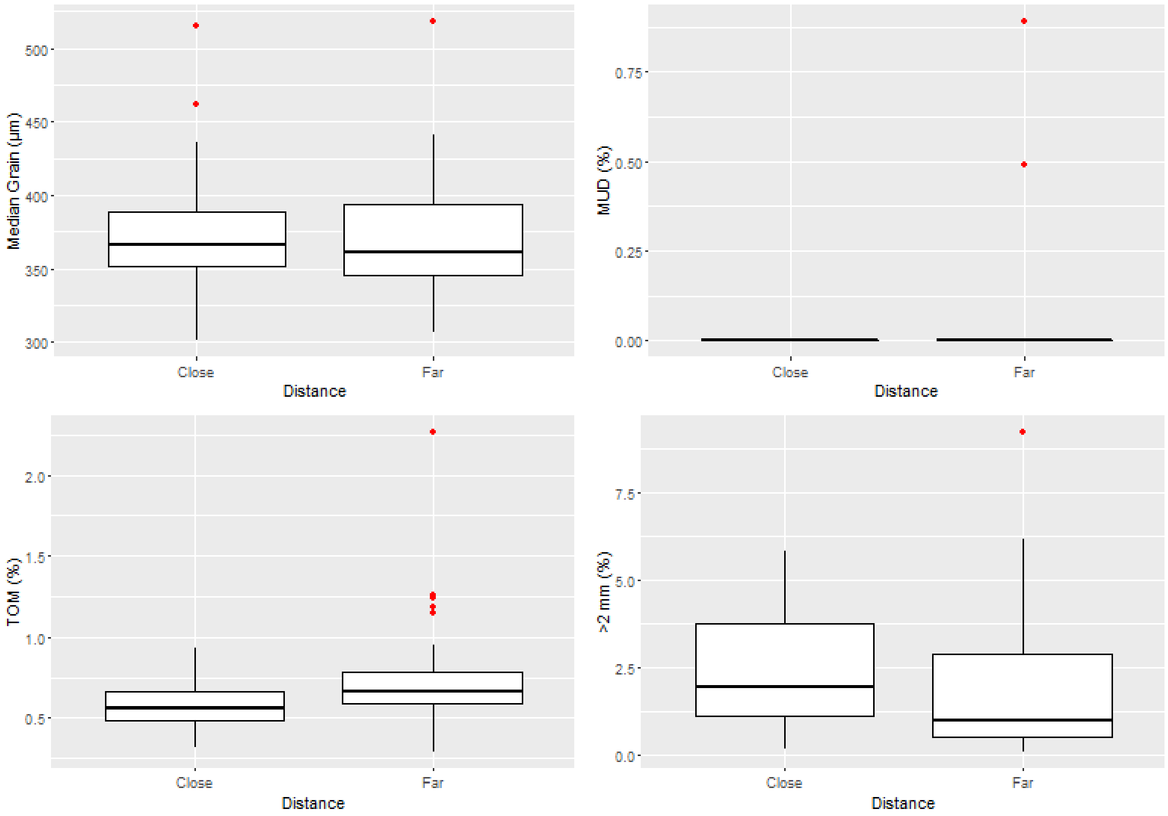

Figure 2. Box plots of the median grain size (Median_Grain), mud content (MUD, volume %), total organic matter (TOM) and sediment fraction above 2mm (>2mm, mass %) per sampling sites. Red dots represent the outliers.

Abundance and number of species present were low in all samples of both sites (Table 2). However, average abundance was higher at the far samples (492 ± 263 ind m^{-2}) than at the close samples (319 ± 195 ind m^{-2}). The same trend was observed for the number of species in the samples (far: 13 ± 4; close: 9 ± 4). A significant difference between the sites was found both for abundance (one-way Anova, p = 0.01) and number of species (one-way Anova, p = 0.0008). Mean eveness was slightly higher in the far samples (0.84 ± 0.08) compared to the close ones (0.81 ± 0.09) (Table 2, Fig. 3), but this yielded no significant differences (one-way Anova, P = 0.23). The multivariate analysis on the macrobenthic community structure revealed a significant effect of distance (Permanova, p=0.001), as visualized by the PCO analysis (Fig. 4). Permdisp was not significant (p= 0.945), thus the significant differences between the two sites are not the result of a dispersion effect.

Table 2. Overview of number of stations and calculated community descriptors (mean ± SD) of the two distances (close-far) sampled at the C-Power wind farm in 2015. * indicates whether significant differences were observed

	Close	**Far**
# Samples	16	32
Species abundance N (ind m-²) *	319.38 ± 195.01	492.81 ± 263.01
Number of species S *	8.56 ± 3.53	12.88 ± 4.10
Evenness	0.81 ± 0.09	0.84 ± 0.08
Median grain size (µm)	378.39 ± 53.39	373.14 ± 43.01
Mud content (%)	0	0.04 ± 0.18
Total organic matter (%)	0.59 ± 0.16	0.76 ± 0.37
Sediment fraction > 2mm (%)	2.50 ± 1.78	1.99 ± 2.22

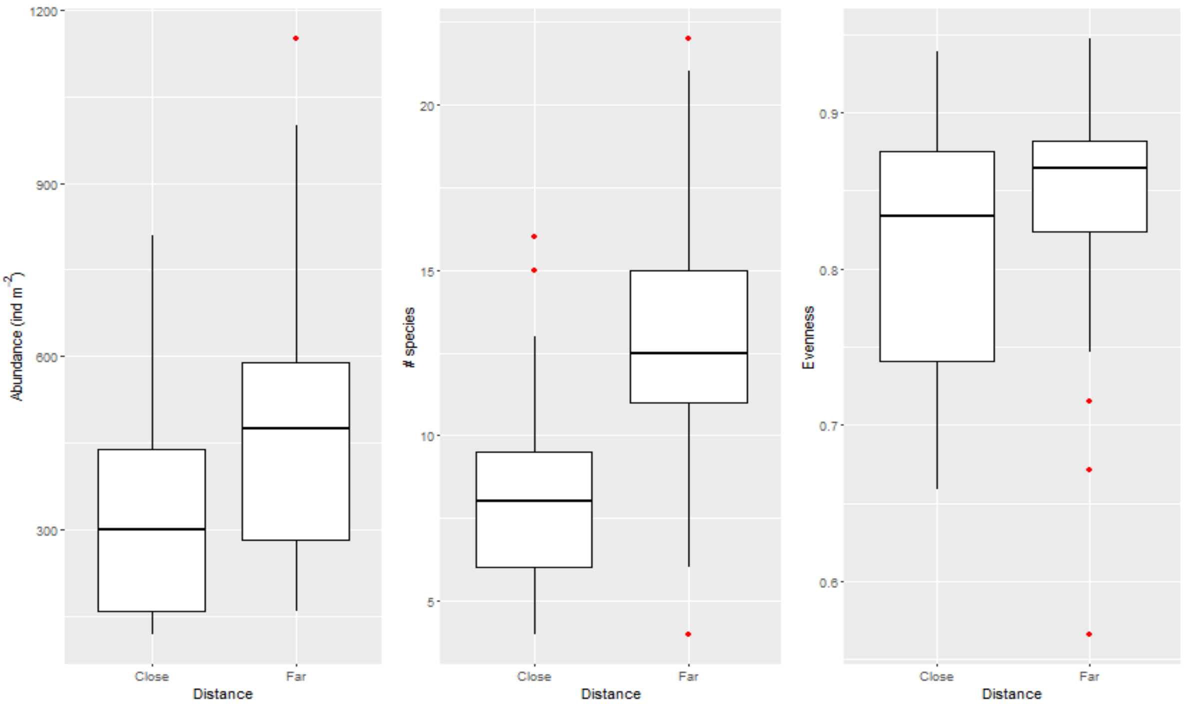

Figure 3. Box plots of the abundance, number of species and evenness per sampling site. Red dots represent the outliers.

A DistLM was carried out to investigate the relationship between the macrobenthic community and the environmental variables. The DistLM revealed that only grain size has a significant relationship with the multivariate data and explained 5.7 % of the variation in the community structure. All three environmental variables together explained only 10.5 % of the variation. Thus some other variables, which are key to explaining the community differences, are missing.

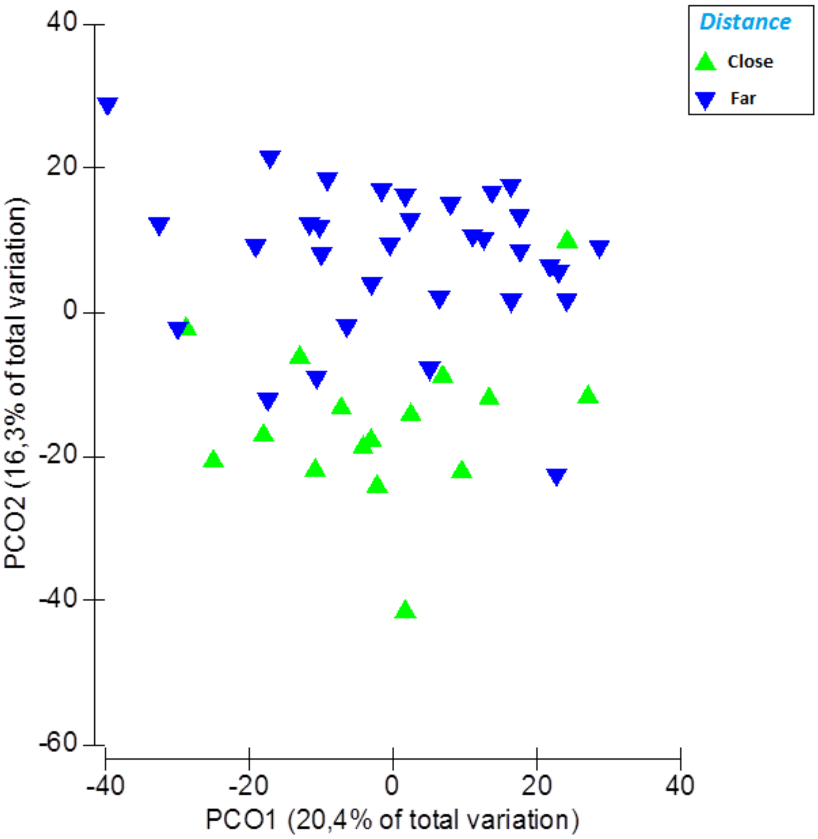

Figure 4. PCO (Principal Coordinates analysis) plot based on Bray-Curtis resemblance matrix of the fourth root transformed macrobenthic densities at two distances from the wind turbines.

In addition SIMPER analysis was run to specify the dominant species in the community of both groups of samples (Clarke and Gorley 2006). Average similarity between the close samples was 49%. Main contributing species to this similarity were: *Urothoe brevicornis* (28 %), *Nephtys spec. (36 %)* and *Gastrosaccus spinifer* (17%). Average similarity between the far samples was 51 % and this was made up of 35% from *Nephtys spec.*, 13 % from *Bathyporeia elegans*, 11 % from *Spiophanes bombyx*, 9 % from *U. brevicornis* and 8 % from *G. spinifer*. Average dissimilarity between the two groups was 55 %. *U. brevicornis* (7 %), *B. elegans* (6 %) and *S. bombyx* (6 %) were the three most contributing species to this dissimilarity. Many other species contributed to a lesser extent.

DIFFERENCES IN MEDIAN GRAIN SIZE OVER TIME

Data on median grain size was available for the years 2008, 2009, 2010 and 2015 (Table 3 and Fig. 5). This data relates to far samples only.

Mean medain grain size did not differ much between the years. 2015 has the highest medain grain size (373.14 ± 43 µm), while in 2010 it was lowest (347.91 ± 45 µm). The non-parametric Kruskal-Wallis chi-squared test revealed that there are no significant differences in medain grain size among the different years (p= 0.43).

Table 3. Overview of number of stations and Median grain size (mean ± SD) sampled over the years.

Year	# Samples	Median grain size (µm)
2008	26	360.23 ± 33.60
2009	30	371.02 ± 70.78
2010	4	347.91 ± 45.44
2015	32	373.14 ± 43.01

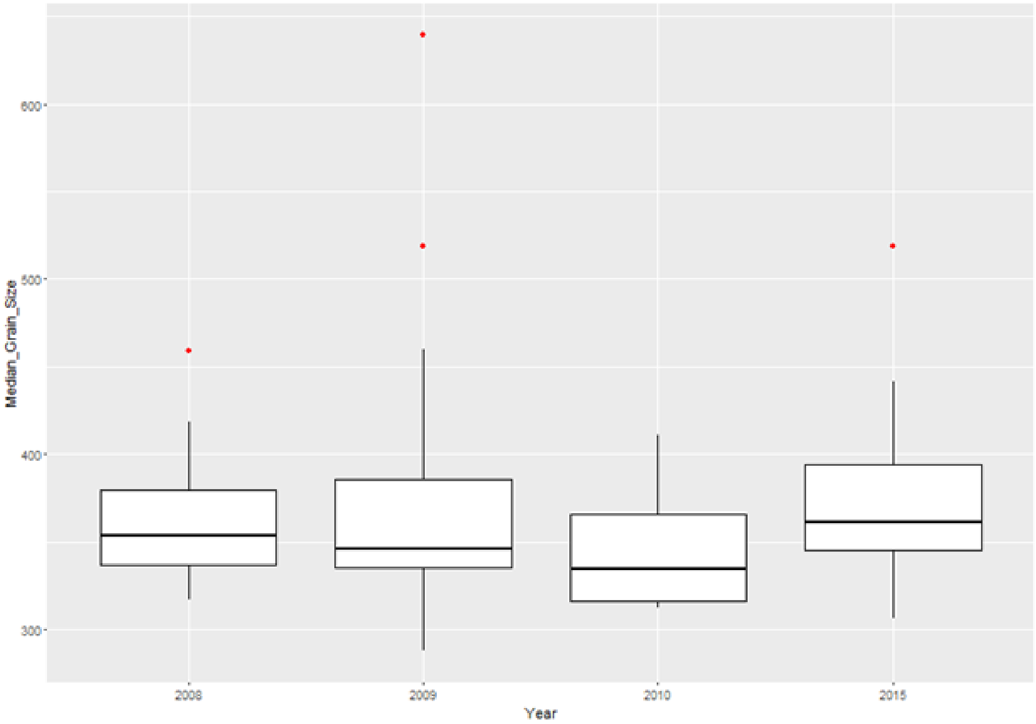

Figure 5. Boxplots of median grain size for the factor ‘Year’. Red dots represent the outliers.

7.4. DISCUSSION

Close to gravity-based turbines, small-scale enrichment and fining of the sediment occurs (Coates et al. 2014), which are the result of playing hydrodynamics and epifaunal coverage. These changes result in changes of the macrobenthic community and were observed up to 50 m distance from the turbines. In the current study however, no changes in sediment characteristics were observed close to the turbines. The currently measured values are in line with preconstruction values (Reubens et al. 2009, Coates and Vincx 2010). In addition, the comparison of median grain size over the years (2008-2015) did not yield any significant differences. The discrepancy between the current work and the one of Coates et al. (2014) might be due to the turbine type involved. The latter was performed around one gravity-based foundation. These foundation types have a large concrete base that largely effect local current flow. Decreased current flows in the wake of the turbine will prevent the re-suspension of finer sands and enriched TOM close to the turbines. In the current study we took close samples near 16 turbines. However, 13 out of the 16 turbines are jacket foundations, having an open structure allowing the main current flow to pass through the construction (Lancelot et al. 1987). In addition, the work of Coates (2014) was performed in late spring, shortly after the Phaeocystis bloom. When the bloom dies of there is an increase in deposition of organic material to the bottom (Lancelot et al. 1987). At locations with reduced currents (such as in the wake of gravity-based turbines) the organic material can accumulate. The possible influence of turbine type will be investigated in more detail in future work.

Another variable that cannot be ruled out to explain differences between the close and far samples is the time lag in sampling. The far samples were gathered in the beginning of October, while the close ones were collected the end of October/beginning of November. 6 and 7 October, surface seawater temperature was 16.15 °C on average, while on November 3th, temperature dropped to 13.8 °C. Temperature is known to structure macrobenthic communities (Kröncke 2011, Kröncke et al. 2011).

Changes in the environmental characteristics and the macrobenthos not only occur in close vicinity of offshore wind turbines, but might also occur at a larger distance due to the fisheries exclusion (Hiscock et al. 2002). Trawl fisheries cause physical disruption of the seabed through contact of the gear components with the sediment. As a result sediment resuspension into the water column occurs in the wake of the gear (Depestele et al. 2015). Mainly the smaller particle sizes are resuspended. These types of fisheries thus prevent smaller sediment fractions to settle down on the seabed. In addition, intensive trawling activities can significantly affect mortality, diversity and species composition of macrobenthic communities (Piet et al. 2000, Jennings et al. 2001). Due to the prohibition of trawling inside offshore wind farms, species sensitive to physical disturbance might get the chance to recover (e.g. bivalve species, tube building terebellids, echinoderms) (Rijnsdorp et al. 1998). Next to macrobenthic species, also epibenthic species and fish benefit from the fisheries closure as higher numbers and larger individuals can be observed in these zones (Vandendriessche et al. 2015).

In contrast to the findings on the abiotic factors, a clear difference in macrobenthic community was found between the close and the far samples. Currently it is unclear what causes these differences. SIMPER analyses revealed that *U. brevicornis* and *G. spinifer* thrive better closer to the turbines while *B. elegans* and *S. bombyx* were more abundant in far samples. All four species are known to be widely distributed along the BPNS. *Urothoe brevicornis* and *B. elegans* prefer medium to coarse-grained sediments with a low mud content, while *G. spinifer* and *S. bombyx* can cope with a wider range of sediment types (Degraer et al. 2006). Thus, the relative abundance of these species is no direct indication for specific habitat changes.

In addition, samples at the far distance yielded more species and higher densities on average than the close samples, once again contrasting the results of Coates et al. (2014). The lower abundances and number of species near the turbines might again be related to the turbine type. Personal observations, while performing scientific dives, at the jacket foundations revealed that this turbine type is heavily fouled by blue mussels (*Mytilus edulis*), which is in accordance to different other studies in the North Sea and Baltic Sea that investigated fouling assemblages at offshore structures (Zettler and Pollehne 2006, Joschko et al. 2008, Wilhelmsson and Malm 2008). The observed *M. edulis* densities have been confirmed by F. Kerckhof (pers. comm.) and it seems to be a stable community as high densities were observed in 2012, 2013 and 2014.

Commonly, beneath suspended mussel cultures, there is an increased sedimentation rate, TOM and total organic carbon (TOC) while oxygen levels reduce. These effects result in reduced infaunal diversity and abundance (Chamberlain et al. 2001), which is in line with the current findings. However, the BPNS is characterized by a well-mixed water column, thus reduced oxygen levels are not expected in these waters. In the long run, it might be that long lasting shell debris (originating from ceased individuals) may lead to coarser sediments. These shells can potentially serve as attachment sites for sessile reef forming organisms (Krone et al. 2013).

7.5. CONCLUSIONS AND RECOMMENDATIONS

It can be concluded that the installation of offshore wind turbines induces changes in the macrobenthos. Results from the current study revealed that differences in macrobenthic community were observed between the close and far samples. As no differences in sedimentology were present, it is unclear what underlying ecological processes are responsible for these community changes. It might be related to changing hydrodynamics, presence of an epifaunal community on the turbines, fisheries exclusion inside the wind farm or a combination of these factors.

The current results are not consistent with results from previous studies, which might be related to the turbine type used. This study was performed in a wind farm dominated by jacket foundations, while the previous study focused on effects near one gravity-based foundation. Jackets have an open structure, allowing the main current flow to pass through. Gravity-based foundations on the other hand obstruct currents and areas with a lower current flow are generated in the wake of the turbine. These differences in flow velocity influence colonization potential of epifaunal species and sediment and TOM resuspension. In addition, the fisheries exclusion inside the wind farm might give macrobenthic species that are sensitive to disturbance a chance to recover. Although no clear trend was observed yet, this reason cannot be ruled out.

As the current study revealed that some differences in the macrobenthic community are present between the close and far samples, but cannot be explained by specific ecological processes, it is recommended to continue to current sampling design and take samples close to the turbines. In addition it would be interesting to perform a targeted monitoring study on the sedimentology and enrichment potential in the close vicinity of the turbines. In addition to Coates et al. (2014) this should include different foundation types as the current results suggest that the turbine type might play an important role in the habitat structuring. We suggest using the sampling design of Coates et al. (2014) and sample at a gravity-based foundation, a monopile and a jacket foundation.

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8

CHAPTER

CHAPTER 8

EFFECTS OF BELGIAN WIND FARMS ON THE EPIBENTHOS AND FISH OF THE SOFT SEDIMENT

J. Derweduwen[*]*, S. Vandendriessche & K. Hostens*

Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Oostende, Belgium.

[*]*Corresponding author: Jozefien.Derweduwen@ilvo.vlaanderen.be*

ABSTRACT

This chapter focuses on the changes in epibenthos and demersal fish of the soft substrates in and around the C-Power and Belwind wind farm. The time series graphs from Vandendriessche *et al.* (2015) were extended and scanned for non-parallelisms between reference and wind farm trend lines. Also size distribution graphs were drawn and analysed. The analyses showed differences between wind farm and reference areas for the period 2013-2014.

The positive short-term effects from Vandendriessche *et al.* (2015) seemed to be a reaction of opportunistic species (i.e. common starfish (*Asterias rubens*), green sea urchin (*Psammechinus miliaris*), brittle star (*Ophiura ophiura*)) since the observed effects disappeared shortly after. The positive short-term effects on plaice (*Pleuronectes platessa*) and sole (*Solea solea*) could be explained by natural variations in the ecosystem. The earlier reported signals of a 'refugium effect'

are no longer observed. The decreasing trend for dab (*Limanda limanda*) continued, resulting in a net emigration from the wind farm. Sandeel (*Ammodytes tobianus*) displayed episodic increases and short-term positive effects on juveniles. No long-term sandeel trends were visible.

Long living species were not yet encountered but may get a chance to establish and recover with the expansion of the wind farm area to a large continuous no-trawling area.

8.1. INTRODUCTION

The construction of wind farms implies the introduction of artificial hard structures into the soft sediment. Many studies already demonstrated the reef effects on epibenthos and fish in the immediate vicinity of wind turbines (May, 2005; Peterson & Malm, 2006; Wilhelmsson *et al.*, 2006a; Wilson *et al.*, 2007; Wilhelmsson *et al.*, 2009; Andersson *et al.*, 2009; Andersson & Öhman, 2010; Reubens *et al.*, 2011; Simon *et al.*, 2011; Krone, 2012; Langhamer, 2012; Bergström *et al.*, 2013; Krone *et al.*, 2013; Reubens *et al.*, 2013, Bergström *et al.*, 2014). The surrounding natural soft sediment and its associated macrofauna also seems to be affected by the artificial hard structures and by the absence of fisheries (Barros *et al.*, 2001; Duineveld *et al.*, 2007; Simon *et al*, 2011; Coates *et al.*, 2014; Dannheim *et al.*, 2014; Gutow *et al.*, 2014; Bergman *et al.*, 2015; Coates *et al.*, 2015; Coates *et al.*, 2016), e.g. shifts in macrobenthic assemblages, higher densities of species sensitive to trawling activities, changes in species or community energy flow.

A Dutch study (van Hal *et al.*, 2012) found no significant wind farm effects or effects of fisheries exclusion on the abundance and community structure of demersal fish, including whiting. Similarly, Bergström *et al.* (2013) revealed no large-scale wind farm effects on benthic fish diversity and abundance. A German study (Gutow *et al.*, 2014) described notable changes in epibenthic biomass and abundances, resulting in differences between reference areas and the wind farm area, A Danish study of Stenberg *et al.* (2015) again noted an overall positive wind farm effect on fish abundance, mainly on a small spatial scale (close to the turbines). At the level of key fish species, e.g. whiting (*Merlangius merlangus*) however, wind farm effects seemed limited: there were similar length distributions and catch levels in the wind farm and the reference area. Similarly, populations of the sand-dwelling species dab (*Limanda limanda*) and sandeel (*Ammodytidae* spp.) were not altered by the wind farm.

In Belgium, Vandendriessche *et al.* (2015) indicated several wind farm effects, including an increase in epibenthos biomass and densities. The higher sole densities in the wind farm and changes in length-frequency distributions for dab (*L. limanda*) and plaice (*Pleuronectes platessa*) may signal a ‘refugium effect’. Positive short-term effects on sandeel densities were both described by van Deurs *et al.* (2012) and Vandendriessche *et al.* (2015).

Edge effects due to changes in fisheries intensity or ‘spillover’ from the wind farm could not be demonstrated in Belgian wind farms (Vandendriessche *et al.,* 2015). However, such effects will probably emerge as soon as the wind farm area is becoming a single entity and the effects of fisheries exclusion will further develop. For this reason, edge effects are not within the scope of this study and will be further investigated once the construction of this large wind farm is completed.

The present study focuses on wind farm effects (combined effect of the wind farm presence and fisheries exclusion) on those epibenthic and demersal fish species that showed remarkable changes in density, biomass and/or size distribution in Vandendriessche *et al.*, 2015.

The research question for this study is:

"Are the previously observed wind farm effects still present and expanding?", including the subquestions

- Are there significantly different densities of epibenthic and fish species in the wind farm compared to the reference area for the years 2013-2014?
- Are there shifts in size distribution of certain species in the wind farm compared to the reference area?

8.2. MATERIAL AND METHODS

In 2013 and 2014, beam trawl samples were taken within the wind farms, i.e. between the turbine rows, just outside the concessions and at reference stations away from the concessions, both in spring and autumn. A number of stations could not be sampled due to bad weather conditions and logistic problems (Figure 1, Table 1 and Figure 2). Up till now, no samples could be taken on the Lodewijckbank (Northwind) due to the fact that no straight line of 1 Nm can be fished because of the orientation of the infield cables. Epibenthos and demersal fish are organisms living on or in the vicinity of the sea bottom and which can efficiently be sampled with this shrimp trawl. They were sampled with an 8-meter shrimp trawl (22 mm mesh in the cod end) equipped with a bolder-chain in the ground rope. The net was towed over 1 nautical Mile, approximately covering 15 minutes at an average speed of 3.5 to 4 knots in the direction of the current. Data on time, start and stop coordinates, trajectory and sampling depth were noted to enable a correct conversion towards sampled surface units. The fish tracks are more or less positioned following depth contours that run parallel to the coastline, thereby minimizing the depth variation within a single track, except for track 2 and track 3 in the C-power concession area due to the positioning of the electricity cables.

All samples gathered in 2013 and 2014 have been processed (on board and in the lab). All data are entered in the ILVO database (developed and maintained in close cooperation with VLIZ), and were delivered to the Belgian Marine Data Centre for archiving.

Due to serious logistic problems with the R.V. Belgica, no samples could be taken in 2015.

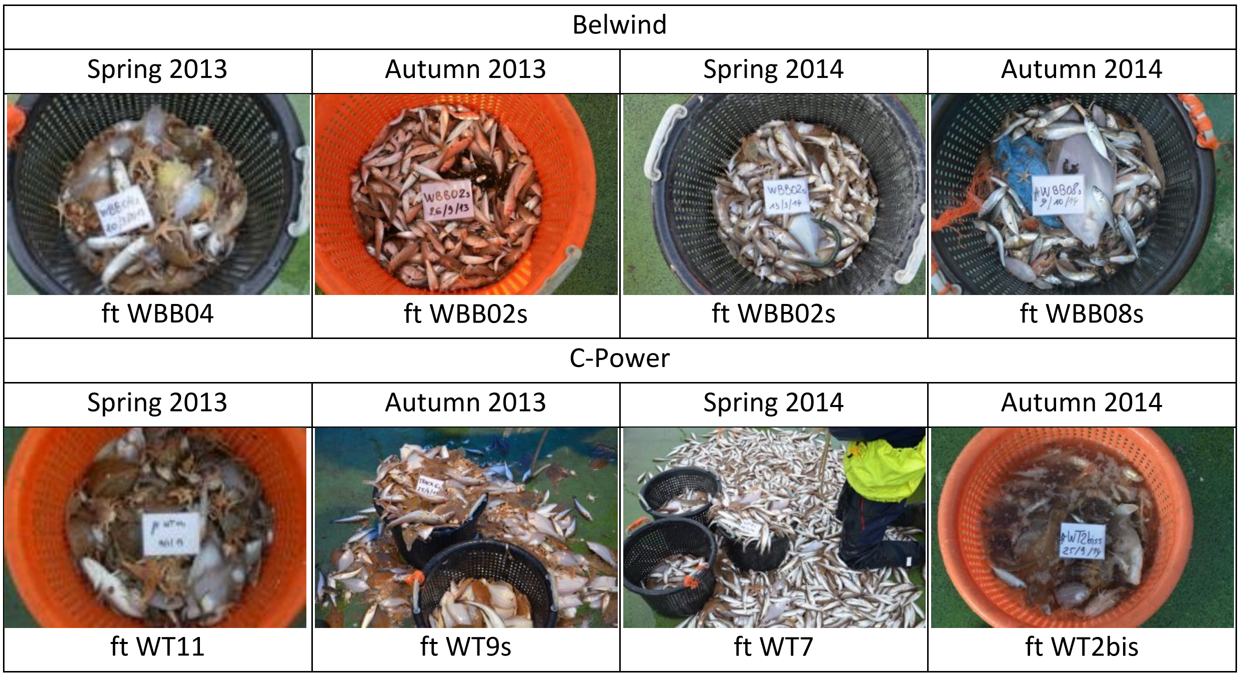

Figure 1. Some impressions of the beam trawl catches in and around the C-power and Belwind wind farms.

Table 1. Stations per sandbank system, with indication of sampling activities in spring and autumn.

sandbank system	station	imp/ref/fri	top/gully	spring 2014	autumn 2014
Gootebank	WG2	ref	top	X	X
	330	ref	gully	X	X
Thorntonbank (C-Power)	WT1(bis)	ref	gully	X	X
	WT2(bis)	ref	top	X	X
	WT3	ref	gully	X	X
	WT7	fringe	gully	X	X
	WT9	fringe	gully	X	X
	WT10	fringe	gully	X	X
	WT11	fringe	gully	X	X
	track 2	impact	top	X	
	track 3	impact	top	X	
	track 5	impact	top	X	X
	track 6	impact	top	X	X
Lodewijckbank	BZN01	impact	top		
Bligh Bank (Belwind)	WBB01	ref	gully	X	X
	WBB02	ref	top	X	
	WBB03	ref	gully	X	X
	WBB04	fringe	gully	X	X
	WBB05	impact	gully	X	X
	WBB06a	impact	top	X	
	WBB06b	impact	top	X	
	WBB07	impact	gully	X	X
	WBB08	fringe	gully	X	X
Oosthinder	WOH01	ref	gully	X	
	WOH02	ref	top	X	
	WOH03	ref	gully	X	

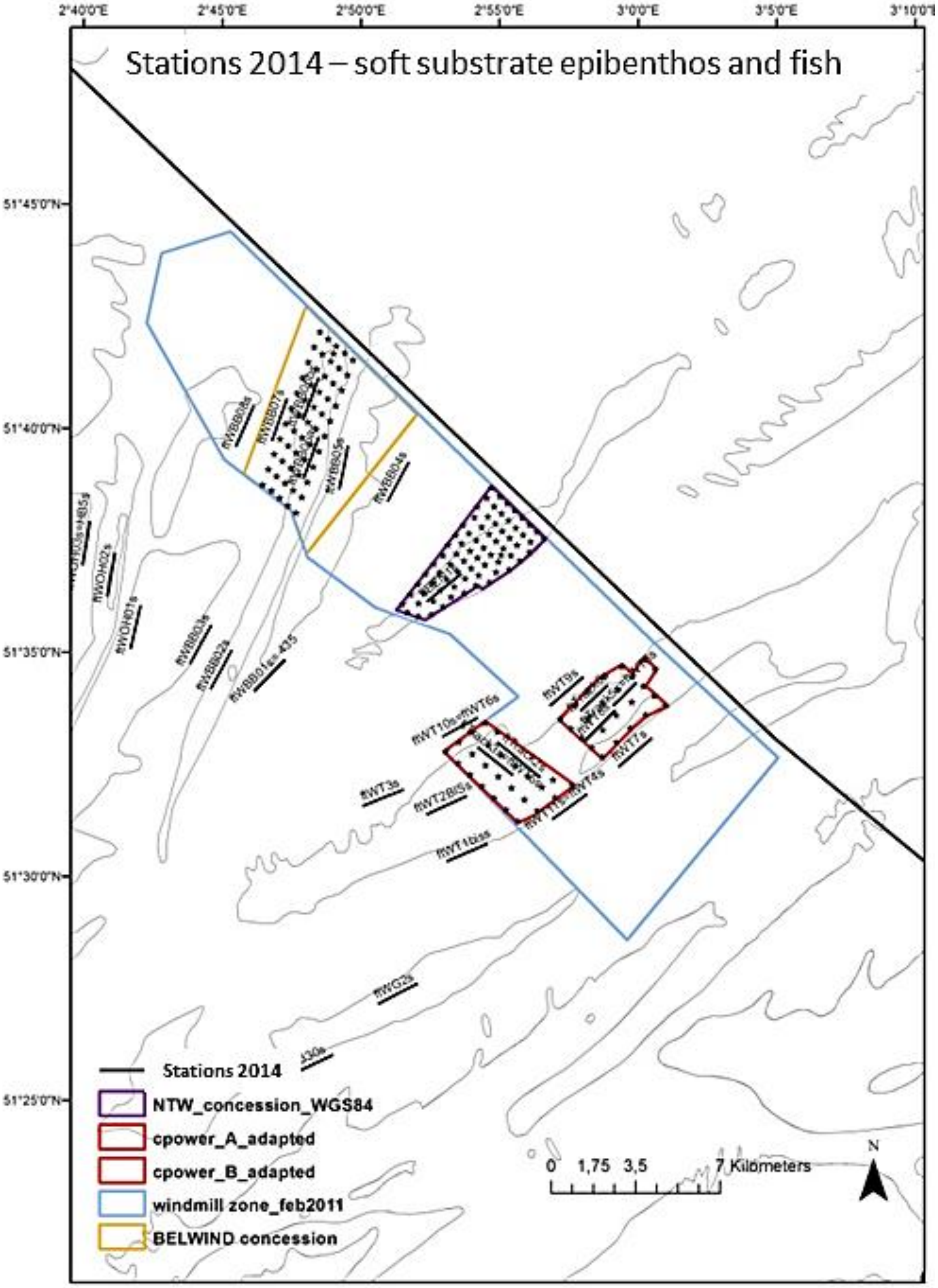

Figure 2. Map showing the 2014 sampling stations at the wind farm concession areas of C-power, Belwind (and Northwind).

The time series graphs from Vandendriessche *et al.* (2015) were expanded in this study.

If clear non-parallelisms occurred in the density time series graphs for the period 2013-2014, statistical analyses were performed with "area" (Control/Impact) as factor. If significant results were found, statistical analyses were executed for 2013 and 2014 separately. The Plymouth routines in multivariate ecological research (PRIMER)e-package + PERMANOVA add-on, version 6.1.6 (Anderson *et al.*, 2007) were used.

8.3. RESULTS

GENERAL

An exploratory overview of the average densities of the five most abundant species in the wind farm (impact) and the surrounding reference (ref) area for 2013 and 2014, is given in Figure 3.

Lesser weever (*Echiichthys vipera*) was a dominant species in the Belwind area (Bligh Bank) (both in autumn and spring) (Figure 3 left) and a subdominant species in the C-Power area (Thorntonbank) in autumn (Figure 3. right up). Density differences between impact and reference area emerged but the pattern was not unambiguous. The hermit crab *Pagurus bernhardus* was also important in the Belwind area (Bligh Bank) and in both seasons but in lower densities and with hardly no differences between impact and reference area.

The soft-bottom community of the C-Power area (Thorntonbank) was dominated by the common starfish (*Asterias rubens*) in autumn (Figure 3. right up) and by the brown shrimp (*Crangon crangon*) and the common starfish in spring (Figure 3. right below). The common starfish densities showed higher values in the wind farm, but also high standard errors. Detailed graphs and analysis on this species are described in paragraph 3.2.

Figure 4 (left) indicates wind farm effects on the epibenthos biomass at the sand bank tops in the Belwind area (Bligh Bank) with increased values at the wind farm top stations, both in autumn (up) and spring (below). From 2011 onwards however, these biomass values decreased and evolved towards (spring) or even below (autumn) the reference top values.

A similar trend is visible in the C-Power area (Thorntonbank) in autumn (Figure 4. right up), the epibenthos biomass was higher at the wind farm top stations (purple line) and declined from 2012 onwards to similar values as the reference top stations (light blue line).

Epibenthos density graphs showed similar patterns (figures not shown).

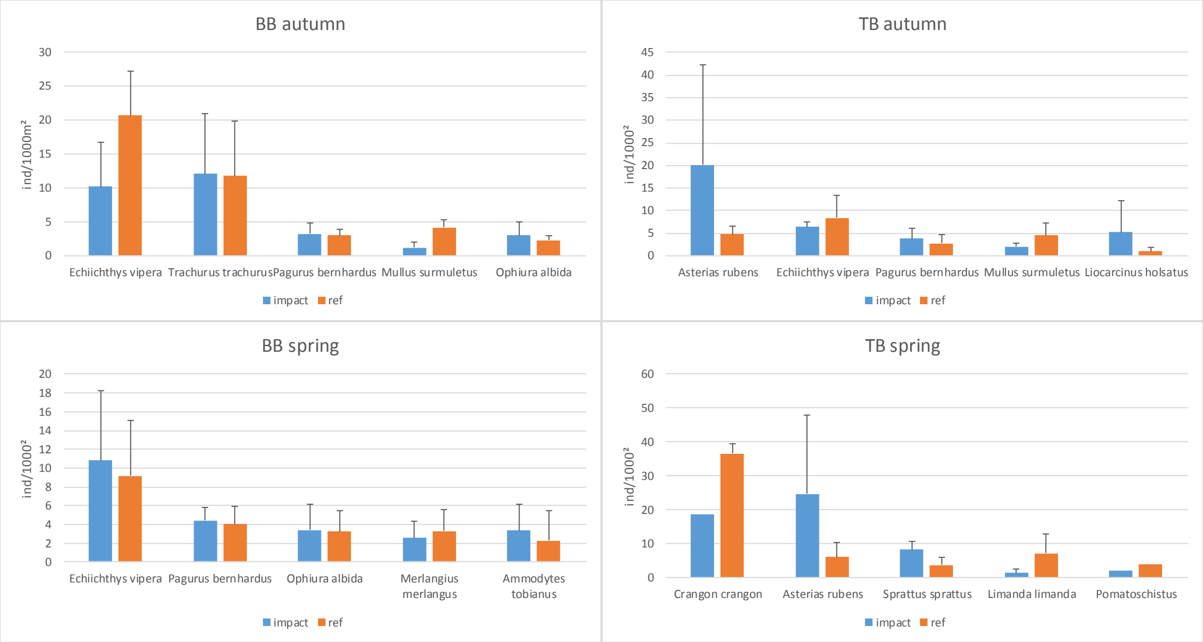

Figure 3. Average densities (ind/1000m²±SE) of the five most abundant species for 2013 and 2014 together, for the Belwind (Bligh Bank) (left) and C-Power area (Thorntonbank) (right) and in autumn (up) and spring (below).

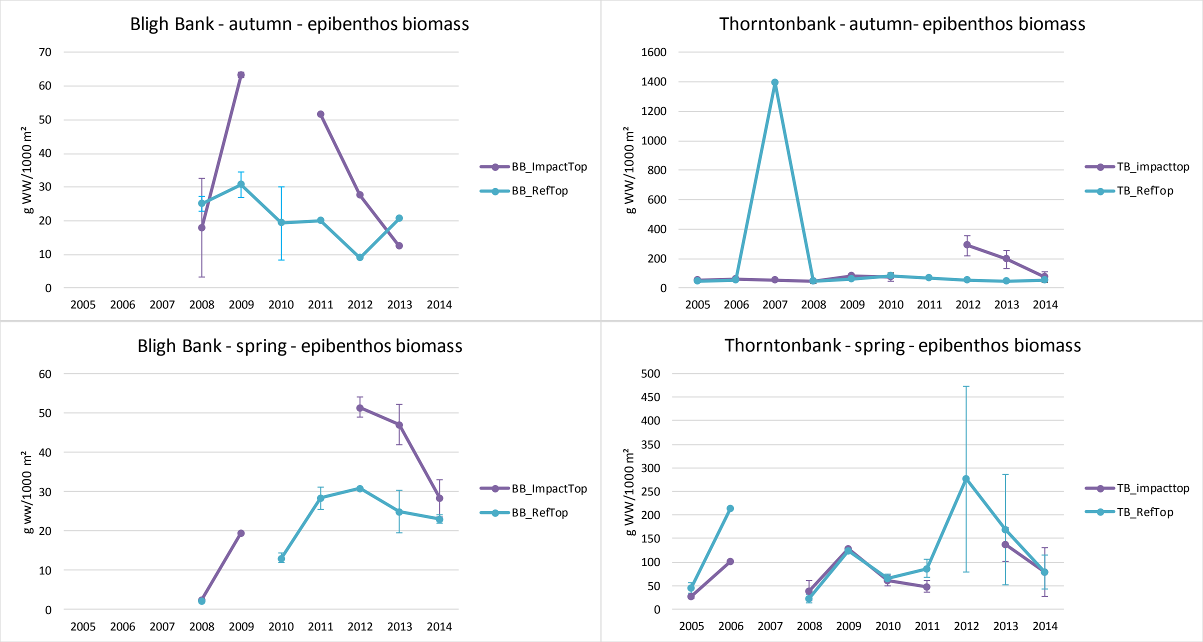

Figure 4. Epibenthos biomass (average gWW/1000 m^2±SE) for the Belwind (Bligh Bank) (left) and C-Power area (Thorntonbank) (right), in autumn (up) and spring (below) between 2005 and 2014.

DENSITY AT SPECIES LEVEL

Common starfish (*Asterias rubens*) & green sea urchin (*Psammechinus miliaris*)

The high densities of **common starfish** (*A. rubens*) in the Belwind area (Bligh Bank) in 2011 suggested a significant wind farm effect (in spring) and significant effects within years (in autumn) (Vandendriessche *et al.*, 2015). From 2011 onwards however, an overall decrease in common starfish densities occurred (Figure 5), in both wind farms (C-Power not shown), seasons and sandbank systems. Both reference and impact densities were very low in 2014, with no significant differences between reference and impact values. This might indicate that the previously observed wind farm effect was a temporary phenomenon.

A similar pattern appears for **green sea urchin** (*P. miliaris*) in the Belwind area (Bligh Bank) (Figure 6): high densities in the wind farm area and a declining trend from 2011 onwards.

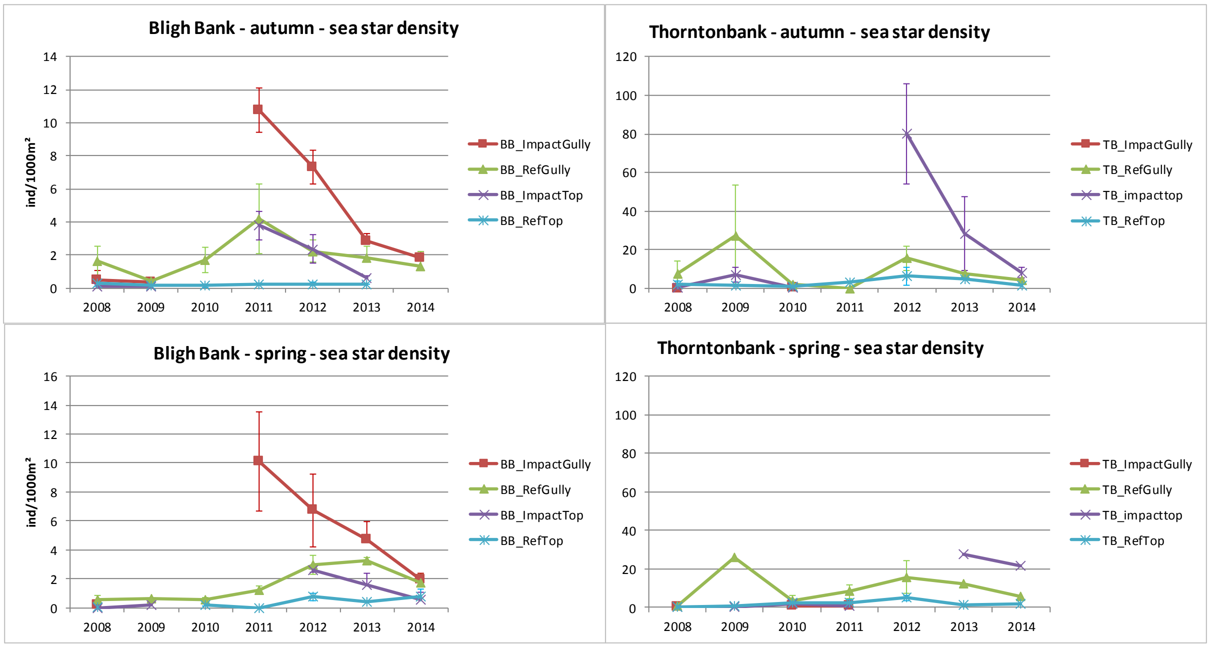

Figure 5. Average common starfish (*A. rubens*) density (ind/1000 m^2±SE) for the Belwind (Bligh Bank) (left) and C-Power area (Thorntonbank) (right), in autumn (up) and spring (below) between 2008 and 2014.

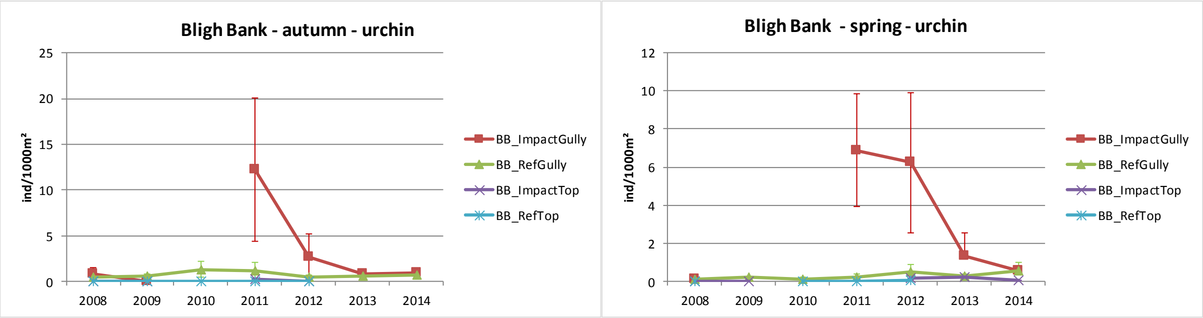

Figure 6. Average green sea urchin (*P. miliaris*) density (ind/1000 m^2±SE) for the Belwind area (Bligh Bank) in autumn (left) and spring (right) between 2008 and 2014.

Brittle star (*Ophiura ophiura*)

A wind farm effect on **brittle star** densities of the gullies was observed in 2009 (Vandendriessche *et al.*, 2015): densities in the Belwind wind farm (Bligh Bank) dropped dramatically. After the wind farm construction, the population recovered and followed the same trend as the reference population.

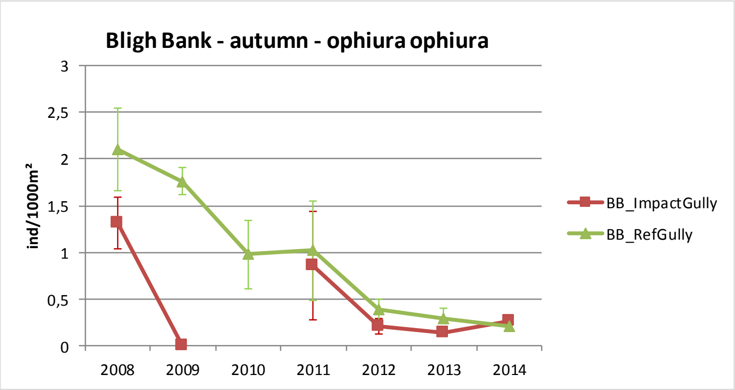

Figure 7. Average brittle star (*O. ophiura*) density (ind/1000 m^2±SE) for the Belwind area (Bligh Bank) in autumn between 2008 and 2014.

Dab (*Limanda limanda*)

For autumn **dab** densities in the Belwind area (Bligh Bank) (Figure 8. left), there is a downward trend from 2008 onwards. Non-parallelisms between the autumn wind farm and reference densities occurred, both at the tops (between 2011-2013) and in the gullies (between 2013-2014). This may indicate a wind farm effect on the density of dab. For the period 2013-2014, the autumn density differences between wind farm and reference gully stations turned out to be significant (p=0,03) and more specifically, the density difference in 2014 (p=0,01). In spring 2012 (Figure 8. right), the high density of dab in the wind farm in 2012 and the subsequently steep decline between 2012 and 2013 is striking. No differences between impact and reference densities occurred in spring for the period 2013-2014.

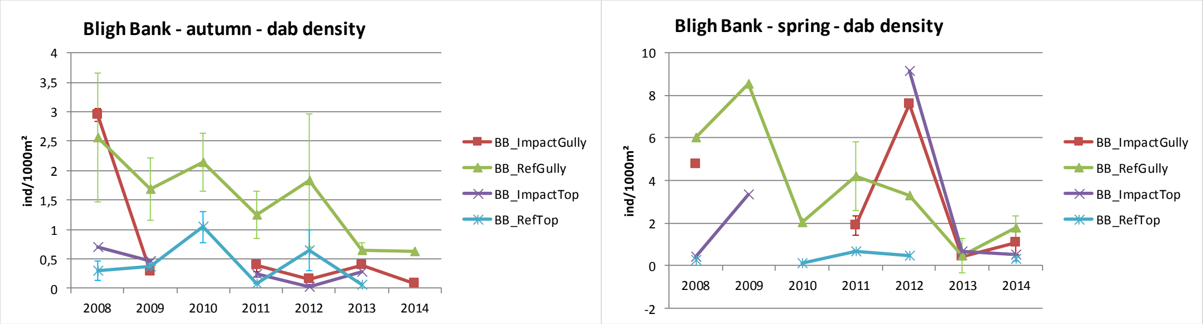

Figure 8. Average dab (*L. limanda*) density (ind/1000 m^2±SE) for the Belwind area (Bligh Bank) in autumn (left) and spring (right) between 2008 and 2014.

Plaice (*Pleuronectes platessa*)

A general increase in **plaice** density was established over the years. From 2011-2012 onwards however, densities generally decreased again towards similar or even lower values than those before the construction of the wind farm (Figure 9). In 2013-2014, spring plaice densities were higher, but not significantly, at the wind farm tops compared to the reference tops.

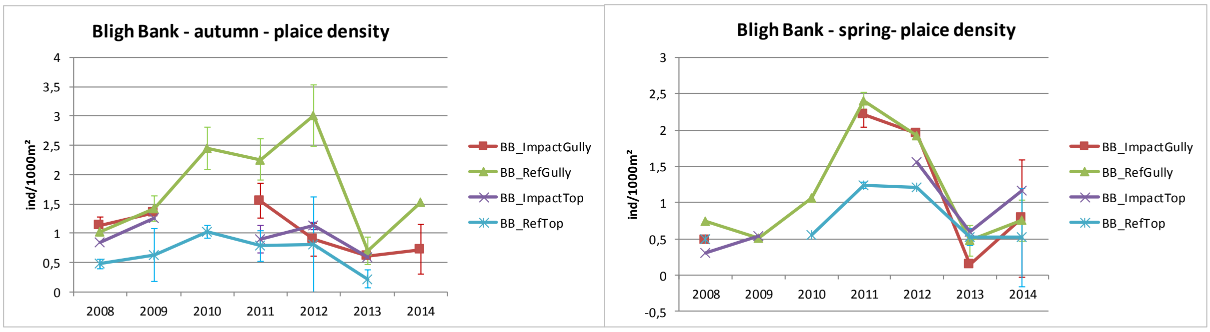

Figure 9. Average plaice (*P. platessa*) density (ind/1000 m^2±SE) for the Belwind area (Bligh Bank) in autumn (left) and spring (right) between 2008 and 2014.

Sole (*Solea solea*)

Higher sole densities were observed in the Belwind area (Bligh Bank) wind farm area in spring 2011 and 2012, both in the gullies and at the sandbank tops (Figure 10). From 2013 onwards however, this difference between impact and reference stations disappeared and sole densities reached approximately the same values. The previously observed wind farm effect on the density of sole (cfr. Vandendriessche *et al.*, 2015) seems to have been a temporary phenomenon.

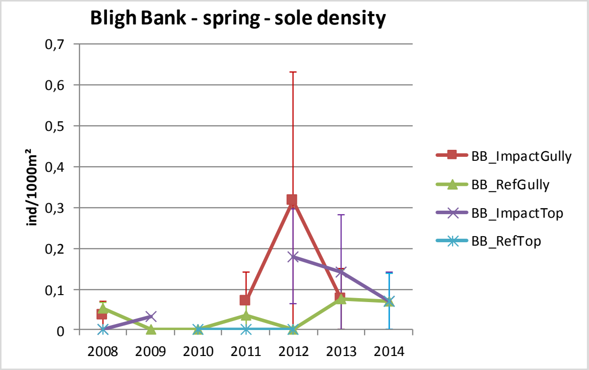

Figure 10. Average sole (*S. solea*) density (ind/1000 m^2±SE) for the Belwind area (Bligh Bank) in spring between 2008 and 2014.

Sandeel (*Ammodytes tobianus*)

Over the years, sandeel densities showed episodic increases at both wind farms, in both seasons, at both impact and reference stations (Figure 11). Non-parallelisms in the time series included higher autumn densities at the reference stations than at the impact top stations in the C-Power area (Thorntonbank) in 2014 (Figure 11. right up). The opposite pattern was observed for the differences in spring densities in the C-Power area (Figure 11. right below). Also in the Belwind area (Bligh Bank), a non-parallelism occurred (Figure 11. left below): lower reference densities in 2012, higher reference values in 2013 and again lower reference densities in 2014, compared to the gradually decreasing impact densities. These non-parallelisms may signal a wind farm effect on the sandeel densities, but statistical analyses showed no significant differences between reference and impact sandeel densities for 2013 and 2014.

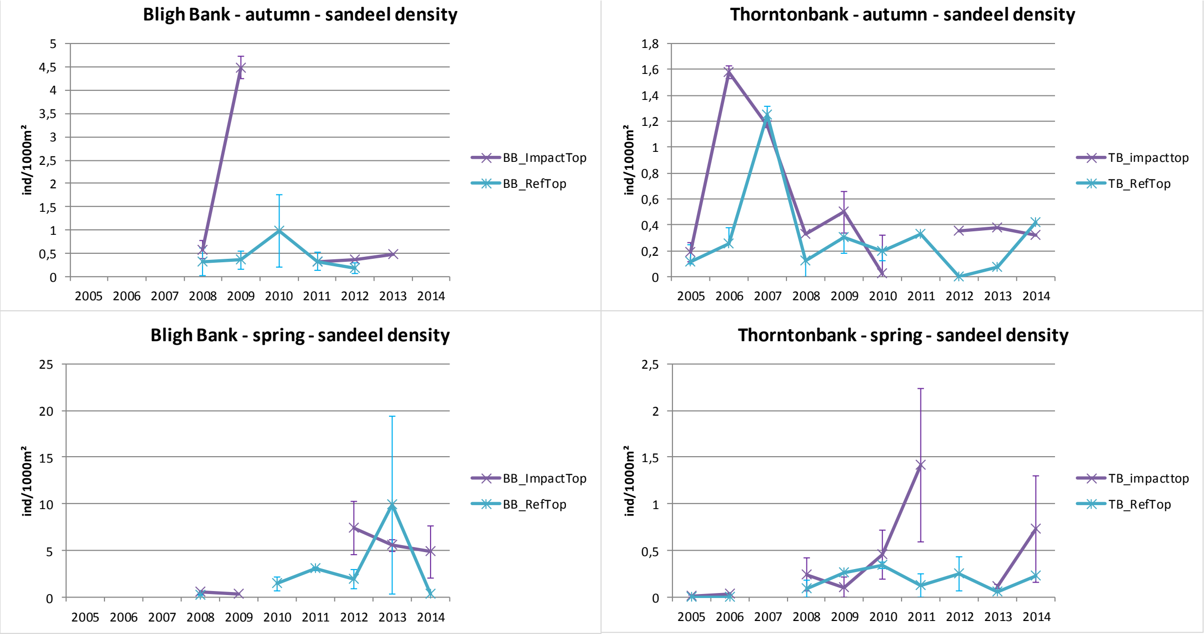

Figure 11. Average sandeel (*A.tobianus*) density (ind/1000 m^2±SE) for the Belwind (Bligh Bank) (left) and C-Power area (Thorntonbank) (right), in autumn (up) and spring (below) between 2005 and 2014.

Time series graphs were also made for whiting (*Merlangius merlangus*), swimming crab (*Liocarcinus holsatus*) and brown shrimp (*Crangon crangon*). However, no significant changes could be denoted.

SIZE DISTRIBUTION

Dab (*Limanda limanda*)

Figure 12 shows the size distribution of dab between 2008 and 2014. From 2008 to 2013, two size classes could be distinguished, both in reference and impact areas. However, densities decreased dramatically over the years, first in the impact area but also in the reference area. The smallest size class completely disappeared in 2014 which automatically leads to a higher average length of 22 cm, in both areas.

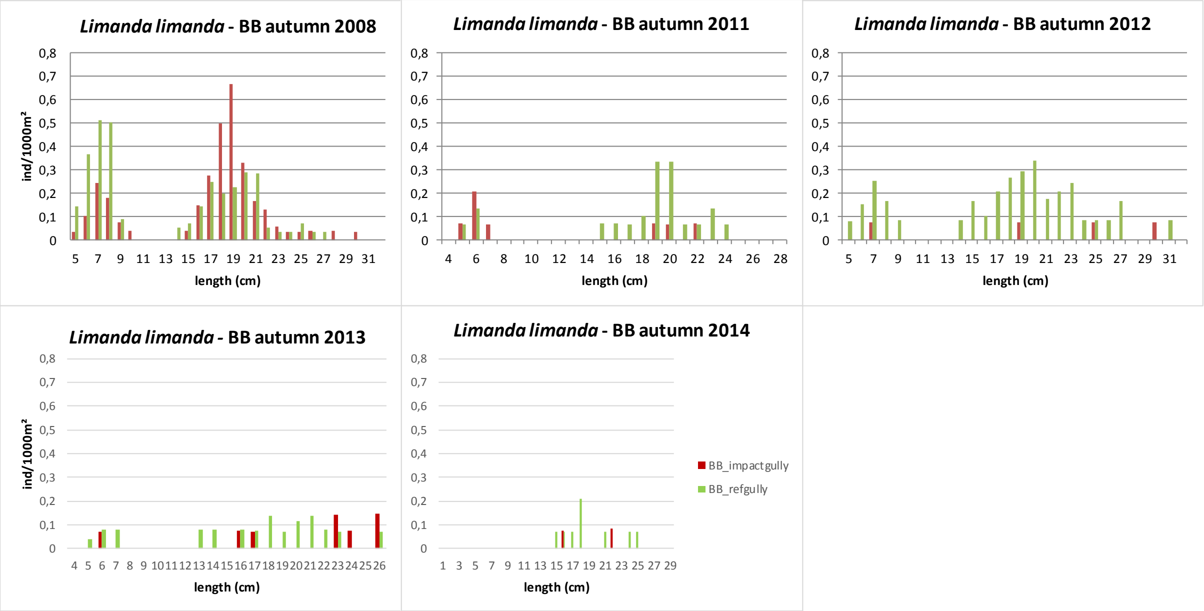

Figure 12. Length-frequency distributions of dab (*L. limanda*) at the Belwind (Bligh Bank) gully stations in autumn 2008, 2011, 2012, 2013 and 2014.

Sandeel (*Ammodytes tobianus*)

Time series graphs of the size distribution of **sandeel** from Vandendriessche *et al*. (2015) could not be complemented for the impact and reference stations of autumn 2013 and 2014 in the Belwind area (Bligh Bank) due to missing data.

Spring data on size distributions are represented in Figure 13. In the Belwind wind farm area (Bligh Bank) (Figure 13. up), no changes in sandeel size distribution occurred. In the C-Power area (Thorntonbank) (Figure 13. below) however, there was a shift towards smaller individuals in the wind farm area.

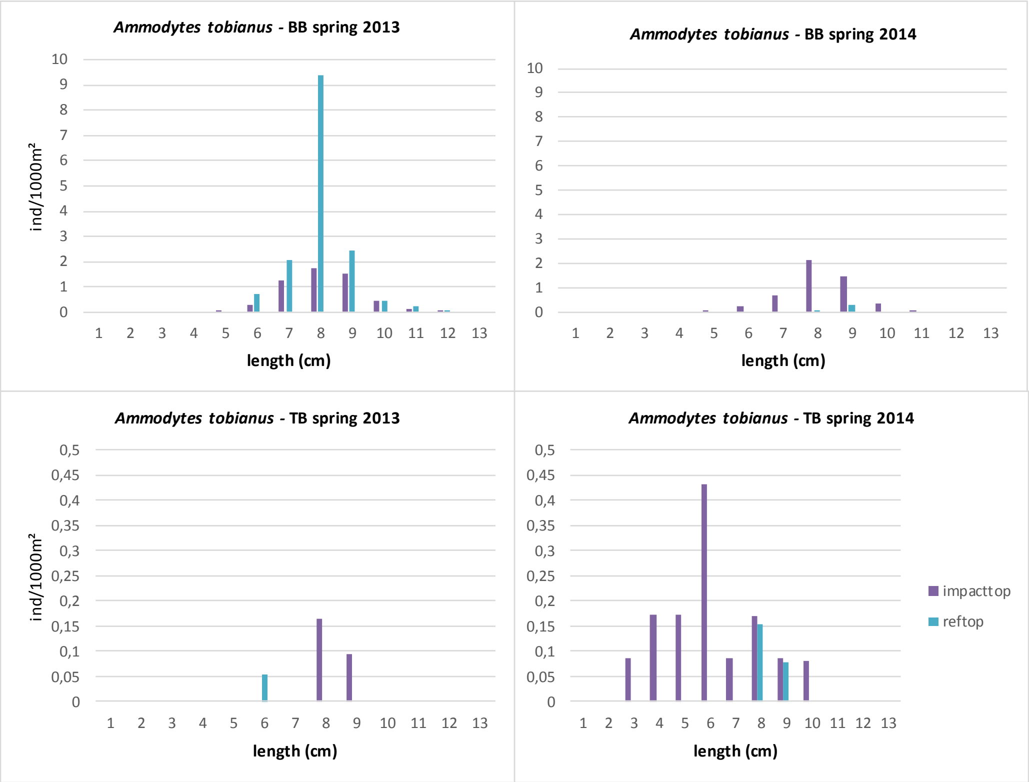

Figure 13. Length-frequency distributions of sandeel (*A. tobianus*) for the Belwind (Bligh Bank) (up) and C-Power area (Thorntonbank) (below) in spring 2013 and 2014.

8.4. DISCUSSION

GENERAL

The observed wind farm effects on soft bottom epibenthos, demersal and benthopelagic fish described in Vandendriessche et al. (2015) were further investigated in this study. This was done by extending existing time series graphs and size distribution graphs and by scanning for non-parallelisms between reference and wind farm trend lines. These analyses showed differences between wind farm and reference areas for the period 2013-2014.

EPIBENTHOS BIOMASS

The epibenthos biomass values showed remarkable post-construction increases in Vandendriessche et al (2015) (data 2008-2012). However, the extended time series show that these increases in both the C-Power (Thorntonbank) and Belwind (Bligh Bank) wind farms only lasted for a couple of years. In 2013-2014, biomass values at the wind farms decreased to comparable or lower values compared to the reference areas. This might indicate that the observed wind farm effect was a temporary phenomenon. Similarly, Gutow et al., (2014) found a significant wind farm effect on the epifauna biomass, which disappeared again the year after. In Gutow’s study however, the reference and wind farm values further diverged. It is not yet clear whether this was a transient development.

COMMON STARFISH (*Asterias rubens)*

The previously observed wind farm effect on the common starfish (A. rubens) densities (i.e. higher densities in the wind farm) was mainly due to a recruitment of small individuals (Coates et al., 2014; Vandendriessche et al, 2015). From 2012 onwards, the wind farm starfish densities strongly decreased to values similar to the ones recorded at the reference stations. This may be due to a lower recruitment following unfavourable environmental conditions (Coates et al., 2014) or the decreased food availability.

The positive wind farm effect on common starfish seems to have been a temporary phenomenon.

This phenomenon of large numbers of juvenile starfish alternated by a low number of large individuals is observed on the foundations as well (Kerckhof et al., 2012).

GREEN SEA URCHIN (*Psammechinus miliaris*)

Similar to the common starfish, the high wind farm densities of green sea urchin (P. miliaris) in the Belwind wind farm (Bligh Bank) drastically declined to similar values as in the reference area. This may be due to a:

- Infringements: the species is sensitive to physical damage by trawling (Lokkeborg, 2005). However, the data from RBINS-OD Nature do not show an increased number of violations to the trawling prohibition for the period 2013-2014 and most infringements seems limited to the safety zone.
- Dislodgment: De Mesel et al. (2015) observed large feeding fronts of the sea urchin on the turbines, which may be an indication of urchin concentrations on the turbines.

BRITTLE STAR *(Ophiura ophiura)*

A sudden decline in brittle star (O. ophiura) densities was caused by the construction of the Belwind (Bligh Bank) wind farm. From 2011 onwards, wind farm densities recovered and both reference and wind farm densities displayed comparable densities with a naturally varying pattern. In this case, the wind farm effect seemed to be a temporary construction effect.

DAB (*Limanda limanda)*

A general decreasing trend in autumn dab (L. limanda) densities occurred from 2008 onwards, both in the Belwind wind farm (Bligh Bank) and the reference area. Dab seemed to move away from the wind farm and its reference area until only a few adult individuals remained. However, the significantly lower autumn densities in the impact area and the non-parallelism between impact and reference densities may suggest a wind farm effect, i.e. a higher net emigration from the wind farm. In spring 2012, there was a temporarily higher attraction/production of dab in the wind farm. Similarly, Leonhard et al. (2011) also observed short-term changes in dab densities after deployment of the Danish wind farm Horns Rev 1. These changes mostly reflected the general trend of this fish population in the North Sea (Leonhard et al., 2011). Long-term wind farm effects on dab were not encountered, both in this study and in Stenberg et al., 2015.

OTHER FLATFISH

For sole (Solea solea) and plaice (Pleuronectes platessa), there seemed to be a temporarily higher attraction/production in the Belgian wind farm area. Lindeboom et al. (2011) denoted a significant increase of sole inside the Dutch OWEZ wind farm. However, telemetry experiments indicated that the majority of sole movements took place at spatial scales larger than the wind farm area and that no large scale avoidance nor attraction occurred (Winter et al., 2010).

In general, a short residence time of adult flatfish in the wind farm was already hypothesized by Winter et al. (2010), Lindeboom et al. (2011) and Vandendriessche et al, (2015). Altered flatfish densities and size distributions (e.g. no large individuals of plaice or turbot (Psetta maxima), in contrast to Vandendriessche et al., 2015) indicate that the previously reported ‘refugium effect’ was rather limited. In 2013-2014, the Belgian wind farms were still rather small and discontinuous. From 2015 onwards, the wind farm area is becoming a larger and continuous area. With the expansion of the wind farm area to a continuous area in the future, the area may act as a no-trawling zone. Short-term positive effects are expected to occur with the construction of every new wind farm. We may also expect long-term positive effects since the wind farms constitute a sanctuary area for trawling-sensitive organisms. For example, the likely increase of dense Lanice conchilega reefs in the wind farm area could create an ecological important large-scale ‘refugium’ for higher trophic levels (Coates et al., 2016). Juvenile fish will have a higher chance to survive and even older, bigger fish will improve survival rates (Langhamer, 2012).

However, environmental parameters should also be considered here. Temperature may cause inter-annual variability in catchability: high temperatures may reduce the gear efficiency because of higher escape rates induced by increased activity in dab and plaice (Bolle et al, 2001). So, the fact that the temperature at the Belwind wind farm (Bligh Bank) was approximately one degree higher in 2013 and 2014 (17,0-17,3 C°) than in 2012 (16,0-16,1 °C) may partly explain the decreased densities of dab and plaice for 2013 and 2014.

SANDEEL (*Ammodytes tobianus)*

In this study and in Vandendriessche et al. (2015), episodic increases of sandeel (A. tobianus) occurred with slightly positive effects on juvenile sandeels. Leonhard et al. (2011) and Van Deurs et al. (2012) also observed a positive short-term wind farm effect on the densities of sandeel which were mainly related to changes in sediment composition and predator abundance.

The fining of the sediment in the immediate vicinity of the turbines (Coates et al., 2014) and the sandeel's preference for sand habitats (Van Deurs et al., 2012), suggest that sandeels are moving away from the turbines. However, this hypothesis should be further investigated to be confirmed. The patchy sandeel distribution, shifts in predator abundance, changes in pelagic activity and changes in recruitment due to changes in zooplankton availability during the larval stage may also offer an explanation for the observed changes (Arnott & Ruxton, 2002; Frederikson et al., 2006; Van Deurs et al., 2012). The significant attraction of herring gull (Larus argentatus), a piscivorous bird species, to the Belwind wind farm (Bligh Bank) (Vanermen et al., 2015) may be linked to the decreasing sandeel densities from spring 2012 to 2014.

Still, no significant long-term effects on this species could be detected, in this study nor in the studies of Van Deurs et al. (2012) and Stenberg et al. (2015).

Since sandeel plays an inevitable key role in the North Sea ecosystem (Leonhard et al., 2011) and has been nominated as a candidate indicator species of the health of the North Sea Ecosystem (Rogers et al., 2010), it is important to further monitor this species with a more suitable sampling strategy for quantitative estimations of sandeel densities.

LONG LIVING SPECIES

Due to the prohibition of beam trawling in the wind farms, vulnerable species (e.g. Lanice conchilega and Echinocyamus pusillus) are getting the opportunity to recover in the Belwind wind energy concession zone (Coates et al., 2016). Long living species vulnerable to trawling (e.g. Ostrea edulis and Sertularia cupressina at Horns Rev (Anonymous, 2006) have not yet been encountered in the Belgian wind farms. This may be attributed to the occurrence of infringements in the past (Vandendriessche et al. 2011) and -to a lesser extent- in the recent years, combined with the fact that long living species grow extremely slowly and thus have a highly limited and prolonged recovery capacity (Clark et al., 2016). Once these long living species re-establish and recover, overall habitat complexity and biodiversity will increase and far-reaching positive effects may be expected.

FUTURE MONITORING

The patterns observed so far should be considered as short-term effects. They most probably reflect the initial stages of the ecological change and succession. Some impacts may not have been detected yet because they are still not developed to the extent needed to become detectable. Long-term monitoring remains an important tool to detect changes in the epibenthos and fish community. To know whether these changes are caused by the presence of the turbines or by fisheries exclusion, specific experiments and targeted monitoring (such as diet study Derweduwen et al., 2016) are needed to gain important knowledge on cause-effect relationships (Callaway et al., 2007;

Lindeboom et al., 2011; Lindeboom et al., 2015).

8.5. CONCLUSION

The positive wind farm effects on the epibenthos biomass, common starfish, green sea urchin and the negative wind farm effects on the brittle star reported in Vandendriessche et al. (2015) seemed to be short-term reactions of opportunistic species. The disturbance effects have faded in 2013-2014 and the ecosystem is again subordinate to natural fluctuations.

Similarly, the earlier reported signals of a 'refugium effect' are no longer observed for sole and plaice. The overall decreasing trend in densities is a result of natural variations (e.g. higher temperature in 2013-2014).

The negative trend in dab densities further declined with a significant higher net emigration from the Belwind wind farm in autumn.

Episodic increases in sandeel densities were encountered with short-term positive effects on juvenile sandeels. However, no significant long-term effects could be detected. Therefore, a more suitable sampling strategy for quantitative estimations of sandeel densities is necessary.

Long living species (e.g. Ostrea edulis and Sertularia cupressina) were not yet encountered in the Belgian wind farms. The expansion of the wind farm area to a large continuous, no-trawling area in the future and more time to recover may favour those species. Once these long living species re-establish and recover, overall habitat complexity and biodiversity will increase and far-reaching positive effects may be expected.

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CHAPTER 9

CHAPTER 9

WIND FARMS AND THEIR INFLUENCE ON THE OCCURRENCE OF ICHTHYOPLANKTON AND SQUID LARVAE

S. Vandendriessche, A. M. Ribeiro da Costa & K. Hostens*

Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Oostende, Belgium.

Corresponding author: Sofie.Vandendriessche@ilvo.vlaanderen.be

ABSTRACT

Changes in the seafloor structure induced by the introduction of wind farms are expected to influence fish populations depositing their eggs on the seafloor, since they are known to require a specific substrate for spawning. Hard substrate creation can also influence egg deposition opportunities for invertebrates, such as the commercially important cuttlefish and different species of squids. So, wind farm construction is expected to have positive effects on fish and invertebrate species that require hard substrates for spawning, and this is expected to manifest as higher densities of early life stages at the hard substrates (eggs) and in the water column (larvae) at the wind farms. This was investigated at the Thorntonbank wind farm by repeatedly sampling three impact stations and three reference stations with a Bongo net from 2010 to 2013. The results do not show significant effects of the wind farm on fish eggs, fish larvae and squid larvae. Nevertheless, the data provide good baseline information about ichthyoplankton and squid larvae at offshore stations that can be used in future monitoring. When planning future monitoring activities at this site, adaptations to the design should be incorporated and alternative methods for quantifying squid larvae should be considered.

9.1. INTRODUCTION

With the establishment of the 2020 targets by the EU, wind farms have been constructed throughout the North Sea, with more than 500 turbines being foreseen just in the Belgian part. Changes in the seafloor structure induced by the introduction of wind farm turbines are expected to influence fish populations depositing their eggs on the seafloor, since they are known to require a specific substrate for spawning. Hard substrate creation can also influence egg deposition opportunities for invertebrates, such as the commercially important cuttlefish and different species of squids. So, wind farm construction is expected to have positive effects on fish and invertebrate species that require hard substrates for spawning, and this is expected to manifest as higher densities of early life stages at the hard substrates (eggs) and in the water column (larvae) at the wind farms.

Herring is one of the species which is expected to benefit from hard substrate addition since it requires rocks, vegetation or gravel for spawning. In the Belgian part of the North Sea (BPNS) the known spawning grounds of *Clupea harengus* have disappeared, but the introduction of wind turbines in the area may trigger their recovery and even establish new ones (Di Marcantonio *et al.,* 2006). In addition to fish species, squids (Cephalopoda) are also expected to increase in density with the introduction of wind farms since they require hard substrate for spawning and they usually deposit their eggs on rocky bottoms (e.g. loliginid species, Hastie *et al.,* 2009). Some of the species previously observed in the North Sea include *Sepia officinalis*, *Sepiola atlantica, Loligo vulgaris* and *Allotheutis subulata.* Several monitoring reports concerning the effects of wind farms on marine biota have included observations regarding (adult) squids. A short-term decrease in squid density (Degraer, 2014; Vandendriessche *et al.,* 2013b) and a long-term increase, although limited, in species richness have been reported (Lindeboom *et al.,* 2011 in Rumes *et al.,* 2013). Nonetheless, there is still a need to increase the information available regarding the effects of wind farm construction on this group. Their increasing importance for fisheries and as food for other organisms make them important targets for research.

The expected effects of hard substrate creation at wind farms on spawning activities of fish and squids have triggered a study to (a) gather baseline information on the composition and variability concerning ichthyoplankton and squid larvae at offshore stations and (b) determine whether wind farms influence the density of ichthyoplankton and squid larvae.

9.2. METHODS

A total of six stations were sampled on the Thorntonbank, three within the C-Power concession area (impact) and another three in the reference area (control). The positions of the stations were approximate and were slightly adapted based on weather conditions and safety guidelines given by the wind farm companies. At the impact stations, we tried to obtain a minimal distance to the nearest turbine, thereby respecting the safety buffer of 500m .

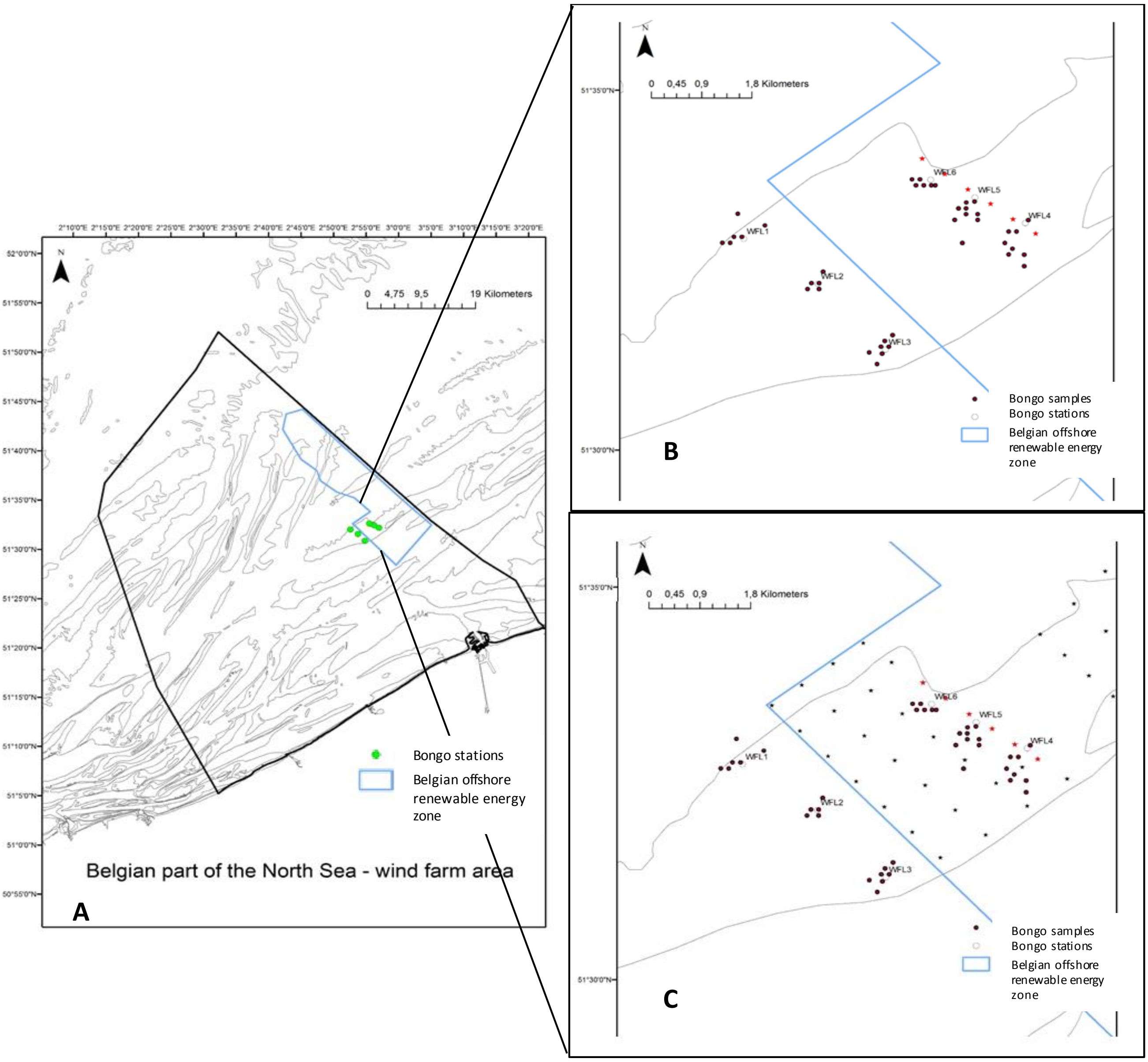

Figure 1. A) Overview of the Belgian part of the North Sea (BPNS) with the delimitation of the Belgian offshore renewable energy zone (blue) and the six sampling stations (green) at the Thorntonbank; B) detailed location of sampling stations, both in the reference (WFL1, WFL2, WFL3) and impact (WFL4, WFL, WFL 6) areas in the beginning of the study period; C) detailed location of the same sampling stations in the end of the study period, with the newly installed foundations(small black dots).

Figure 2. Bongo nets

Sampling was carried out from 2010 until 2013, on board of the research vessels Zeeleeuw and Belgica. Samples were taken opportunistically: monthly when possible, but at least twice a year (in spring and autumn). At each station, a Bongo net (Figure 2) was deployed, fitted with 500 µm mesh nets, a fly weight and a flow meter (Smith *et al.,* 1968). At first, the nets were towed in an oblique continuous haul from the bottom to the surface, but this occasionally resulted in a very small filtered water volume. Consequently, we adapted the sampling method to 10 min undulating tows from March 2011 onwards. At each station, a CTD was used to obtain vertical profiles of temperature and salinity and to assess the level of water mixing. Turbidity was measured with a Secchi disk and chlorophyll *a* (Chl a) data were obtained from fixed fluorimeters on board of the research vessels Zeeleeuw and Belgica. Missing data were obtained from Modis/Meris satellite data [3] and from Van Ginderdeuren (2013).

All samples were rinsed on board and transferred into a 4% formaldehyde solution, since this allows for the best fixation of the melanophores (essential in identification), and body morphology (Munk and Nielsen, 2005). In the laboratory, samples were transferred to an ethanol solution for lab analyses. Using a Leica® stereomicroscope, all ichthyoplankton and squids (Cephalopoda) were identified to the lowest possible taxonomical level. Identification was done with the aid of Munk *et al.* (2005), Russell (1976) and Sweeney *et al* (1992). All larval individuals were identified and counted. When larvae were very abundant (more than 100 individuals of the same family), a subsample of 100 individuals was identified per larval stage and extrapolated. Fish eggs present in the samples were counted. The pseudoreplicates of each haul were kept separate during processing, but the results were averaged before analysis.

The obtained dataset was highly asymmetric, since not all seasons were analyzed in each year and the number of sampling events was not uniform, giving rise to an unbalanced design. Statistical analyses were performed using the package PRIMER 6 with PERMANOVA (Clarke 1993, Clarke and Gorley 2006). In first instance, the impact of wind farms was tested using a crossed design including control/impact and sampling event. Temporal patterns were then further explored using season and year as factors. In all analyses, a type III sum of squares (SS) was used to correct for an unbalanced design. In case of significant interaction effects, pairwise tests were done to detect the specific differences between groups. When the number of unique permutations in the output file turned out to be lower than 100, the Monte Carlo test option was selected *a priori*.

Community analysis was done on fish larvae data only; data of fish eggs and squid larvae were subjected to univariate analyses only. The larval density data were fourth-root transformed and a zero-adjusted Bray-Curtis similarity matrix (Bray and Curtis, 1957) was constructed.

In terms of biodiversity analysis, the DIVERSE function was used on raw density data. Species richness was obtained by calculating the number of species (S) and the expected number of species ES(50), while the Shannon-Wiener index (H', $\log_e$) was used to calculate species diversity. Although the latter is known to be sensitive to sample size, it was maintained to allow further comparisons with previous studies. Spatial and temporal differences for species diversity were also verified using PERMANOVA, based on Euclidean distance resemblances.

The environmental variables (water temperature, Chl *a*, salinity and secchi depth) were correlated with the patterns in species composition and abundance. The first step was to normalize the abiotic data and to check for collinearity using a draftsman plot. A Euclidean distance resemblance matrix was drafted and distance-based linear models (DistLM; step-wise model with BIC criteria) were used to see which predictor variables (or combination of) best explained the data patterns.

9.3. RESULTS

GENERAL RESULTS

The 66 samples analyzed in this study were spread over different years and seasons. The data were consequently unbalanced, with information from 8 different months (see table 1). Within each sampling event, samples were collected in both the control (reference) and the impacted areas (wind farm).

Table 1. Description of the included sampling events. During each event, three stations were sampled per location (reference and control).

Year	Month	Season	Cruise
2010	March	Winter	Zeeleeuw 10-090
2010	April	Spring	Zeeleeuw 10-210
2010	June	Spring	Zeeleeuw 10-310
2010	July	Summer	Zeeleeuw - 10-410
2010	September	Summer	Belgica 2010/25b
2010	December	Autumn	Zeeleeuw 10-750
2011	January	Winter	Zeeleeuw - 11/050
2012	January	Winter	Belgica 2012/01
2012	February	Winter	Belgica 2012/6a
2013	March	Winter	Belgica 2013/08A
2013	September	Summer	Belgica 2013/26A

Within the Bongo net samples, 38 species of fish and squids were encountered (29 identified to species level). Two groups of squids were found, more precisely *Sepiola atlantica* (Figure 6) and the species complex Loliginidae (Figure 7), which is composed of the species *Loligo vulgaris* and *Allotheutis subulata*. These species are indistinguishable at larval stage. In terms of fish larvae, the most abundant species were Ammodytidae sp. (42%), *Clupea harengus* (36%) and Clupeidae sp. (8%). Table 2 gives an overview of the encountered species list, with average and maximum densities, and monthly occurrence. Some specimens could not be identified to species level mostly due to their small size and/or degree of degradation (e.g. disappearing melanophores).

Figure 3. Image of the squid *Sepiola atlantica*

Figure 4. Image of Loliginidae sp.

Table 2. List of all taxa found in this study with scientific and common names, their average density (ind.m^{-3}), maximum density (ind.m^{-3}) and seasonal occurrence.

Fish taxa/species	**CommonName**	**Average density**	**Maximum density**	**Seasonal occurence**
Pisces eggs		22,04	192,59	Dec-April, Jun, Jul, Sep
Pisces sp.		0,38	12,03	Feb, April, Jun, Jul, Sep
Ammodytidae				
Ammodytidae sp.	sandeel	0,52	7,69	Feb-April, Jun
Ammodytes marinus	Raitt's sandeel	<0.1	2,33	Jan-March
Hyperoplus lanceolatus	greater sandeel	<0.1	0,20	March, April
Ammodytes tobianus	lesser sandeel	<0.1	1,52	Jan, March, Dec
Hyperoplus immaculatus	Corbin's sand eel	<0.1	0,20	March
Bothidae				
Arnoglossus laterna	scaldfish	<0.1	<0.1	March, Sep
Callionymidae				
Callionymus reticulatus	reticulated dragonet	<0.1	0,98	Jun, Jul
Callionymus lyra	common dragonet	<0.1	0,34	Jul
Callionymus sp.	dragonet	<0.1	0,32	Jan, Jul
Carangidae				
Trachurus trachurus	Atlantic horse mackerel	0,11	2,25	Jun, Jul, Sep, Dec
Clupeidae				
Clupea harengus	Herring	0,60	5,40	Dec-April, Jul
Clupeidae sp.	shads	0,59	17,33	Jan-April, Jun, Jul
Sprattus sprattus	Sprat	0,30	5,38	Feb, Jun, Jul, Dec
Sardina pilchardus	European pilchard	<0.1	0,80	Jul
Cottidae				
Myoxocephalus scorpius	short-horn sculpin	<0.1	0,20	March
Gadidae				
Gadus morhua	cod	<0.1	<0.1	March
Merlangius merlangus	whiting	<0.1	0,20	March
Gadidae sp.	codfishes	<0.1	1,28	Feb, March, Jun, Jul
Pollachius pollachius	pollack	<0.1	0,35	Jul
Trisopterus luscus	Bib	<0.1	0,10	Feb, Dec
Pollachius virens	saithe	<0.1	<0.1	Feb
Gobiidae				
Pomatoschistus microps	common goby	<0.1	<0.1	Sep
Pomatoschistus minutus	sand goby	<0.1	<0.1	Sep
Gobiidae sp.	gobies	<0.1	0,59	Jun, Jul, Sep, Dec
Gobius niger	black goby	<0.1	0,20	Jul, Sep
Pomatoschistus pictus	painted goby	<0.1	0,32	March, Jul, Sep
Liparidae				
Liparis liparis	sea snail	<0.1	<0.1	Jan, March
Pleuronectidae				
Pleuronectes platessa	plaice	<0.1	0,24	Jan-March
Pleuronectidae sp.	Righteyed Flounders	0,21	5,31	Jan, March, Jun
Limanda limanda	dab	<0.1	<0.1	Feb, March
Soleidae				
Solea solea	common sole	<0.1	0,79	Jan, April, Jun, Jul
Syngnathidae				
Syngnathus rostellatus	Nilsson's pipefish	<0.1	0,05	Sep
Trachinidae				
Echiichthys vipera	lesser weever	<0.1	0,32	Jul
Triglidae				
Chelidonichthys lucerna	tub gurnard	<0.1	0,29	Jul

Squid taxa/species	**CommonName**	**Average density**	**Maximum density**	**Seasonal occurence**
Loliginidae				
Loliginidae sp.	pencil squids	<0.1	<0.1	Sep
Sepiolidae				
Sepiola atlantica	Atlantic bobtail	<0.1	<0.1	Jan

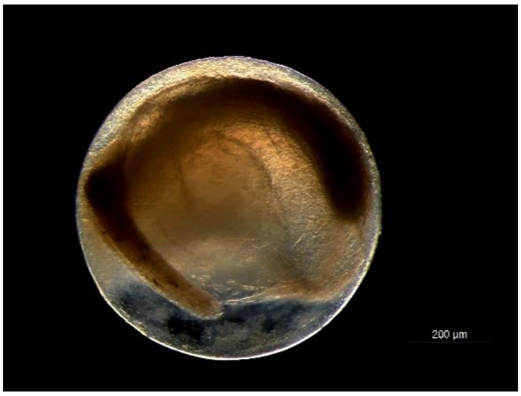

Figure 5. Image of a fish egg.

Overall, the trends observed for fish eggs and fish larvae were similar, with peaks around March (2010, 2013) and June (2010). The maximum egg density was recorded in March 2013 with 73.73 ind.m^{-3} and fish larvae had their major peak in June, reaching 0.37 ind.m^{-3}. Squid larvae, on the other hand, were only present on two sampling occasions: January 2012 (density so small it is not visible in Figure 9) and in September 2013, where they reached their maximum density of 0.02 ind.m^{-3}. In these two occasions the species were different, in January the species observed was *Sepiola atlantica* while in September it was Loliginidae sp.

WIND FARM IMPACT

PERMANOVA tests in a *CI x event* design were done for total densities of fish eggs, fish larvae and squid larvae, for fish larvae composition and for densities of the most abundant fish species and groups (Clupeidae sp., Pleuronectidae sp., *Clupea harengus*, *Sprattus sprattus*, Ammodytidae sp., Gobiidae sp.) and for fish larvae diversity (number of species S, expected number of species ES, Shannon-Wiener diversity index). Significant interaction effects were seen for *C. harengus* densities, for fish larvae composition and for the expected number of species (see table 3). Pairwise tests, however, indicated that these were more the result of event-related factors rather than wind farms related impact.

Table 3. PERMANOVA results of overall tests including treatment (control versus impact) and sampling event. Significant results are indicated as bold.

	Source	df	SS	MS	Pseudo-F	P(perm)	unique perms
composition fish larvae	CI	1	586,94	586,94	0,74762	0,5672	9938
	ev	10	79387	7938,7	15,693	0,0001	9872
	CIxev	10	7850,8	785,08	1,5519	**0,0072**	9863
	Res	44	22258	505,87			
	Total	65	1,10E+05				
density *Clupea harengus*	CI	1	1,17E-02	1,17E-02	3,88E-02	0,8549	9835
	ev	10	12,528	1,2528	45,264	0,0001	9942
	CIxev	10	3,0176	0,30176	10,903	**0,0001**	9943
	Res	44	1,2178	2,77E-02			
	Total	65	16,775				
Density Ammodytidae sp.	CI	1	8,66E-02	8,66E-02	2,2004	0,1719	8006
	ev	10	15,345	1,5345	61,853	0,0001	9953
	CIxev	10	0,39344	3,93E-02	1,5859	0,1364	9920
	Res	44	1,0916	2,48E-02			
	Total	65	16,917				
Density Clupeidae sp.	CI	1	7,10E-03	7,10E-03	6,49E-02	0,8025	9812
	ev	10	11,026	1,1026	14,851	0,0001	9946
	CIxev	10	1,0946	0,10946	1,4744	0,1722	9937
	Res	44	3,2667	7,42E-02			
	Total	65	15,395				
density Gobiidae sp.	CI	1	3,78E-02	3,78E-02	0,87298	0,397	946
	ev	10	1,9468	0,19468	5,2808	0,002	999
	CIxev	10	0,43314	4,33E-02	1,1749	0,341	999
	Res	44	1,6221	3,69E-02			
	Total	65	4,0399				
density *Sprattus sprattus*	CI	1	2,92E-02	2,92E-02	0,33289	0,5473	2587
	ev	10	7,9397	0,79397	16,612	0,0001	9946
	CIxev	10	0,8768	8,77E-02	1,8345	0,0638	9927
	Res	44	2,1029	4,78E-02			
	Total	65	10,949				
density Pleuronectidae sp.	CI	1	3,55E-02	3,55E-02	3,7022	0,0816	2498
	ev	10	5,3453	0,53453	9,1838	0,0001	9932
	CIxev	10	9,60E-02	9,60E-03	0,16494	0,998	9917
	Res	44	2,561	5,82E-02			
	Total	65	8,0379				
total density fish larvae	ci	1	0,15066	0,15066	3,0968	0,0868	9845
	ev	10	17,779	1,7779	36,544	0,0001	9936
	cixev	10	0,89055	8,91E-02	1,8305	0,0834	9924
	Res	44	2,1406	4,87E-02			
	Total	65	20,961				
fish larvae S	CI	1	1,52E-02	1,52E-02	4,66E-03	0,9637	3719
	ev	10	299,27	29,927	15,553	0,0001	9935
	CIxev	10	32,485	3,2485	1,6882	0,121	9948
	Res	44	84,667	1,9242			
	Total	65	416,44				
fish larvae ES(50)	CI	1	1,2273	1,2273	1,4261	0,2635	961
	ev	10	104,12	10,412	42,95	0,0001	9921
	CIxev	10	8,6061	0,86061	3,55	**0,0014**	9957
	Res	44	10,667	0,24242			
	Total	65	124,62				
fish larvae Shannon-Wiener	CI	1	3,13E-02	3,13E-02	0,17072	0,685	9807
	ev	10	14,994	1,4994	13,316	0,0001	9942
	CIxev	10	1,8352	0,18352	1,6297	0,1287	9947
	Res	44	4,9547	0,11261			
	Total	65	21,816				
total density fish eggs	ci	1	0,33141	0,33141	3,9081	0,0522	9823
	ev	10	77,041	7,7041	90,848	0,0001	9934
	cixev	10	0,9596	9,60E-02	1,1316	0,3721	9937
	Res	44	3,7313	8,48E-02			
	Total	65	82,063				
total density squid larvae	ci	1	6,67E-05	6,67E-05	1,56E-02	0,9071	9774
	ev	10	0,56116	5,61E-02	13,13	0,0002	9616
	cixev	10	3,64E-02	3,64E-03	0,85108	0,6767	9923
	Res	44	0,18805	4,27E-03			
	Total	65	0,78564				

TEMPORAL VARIATION

Since "event" appeared to have more important structuring effect on fish eggs, fish larvae and squids than wind farm presence, we further explored the temporal variation within the samples (Figure 6). In terms of fish larval density, we observed a significant seasonal and inter-annual variability, and an interaction between the two (PERMANOVA SxY: pseudo F of 4.72, p<0.001). The species contributing most to the dissimilarity between seasons were Ammodytidae sp., Clupeidae sp., *Ammodytes marinus, Clupea harengus* and Pisces sp.. In terms of interannual variability, the main contributors were Ammodytidae sp., Clupeidae sp., *Ammodytes marinus, Clupea harengus* and Gobiidae sp.

For fish egg density, a significant inter-annual and inter-seasonal difference was also detected, but without interaction between the terms (PERMANOVA pseudo F of 3, p=0.0869). Seasonal variability was statistically significant between all seasons with the exception of winter versus spring, while inter-annual variability was significant among all years except between 2010 and 2013.

Squid larval densities were significantly different between seasons and years and there was a significant interaction between the two. However, the extremely low encounter rate of these larvae hampered further statistical exploration.

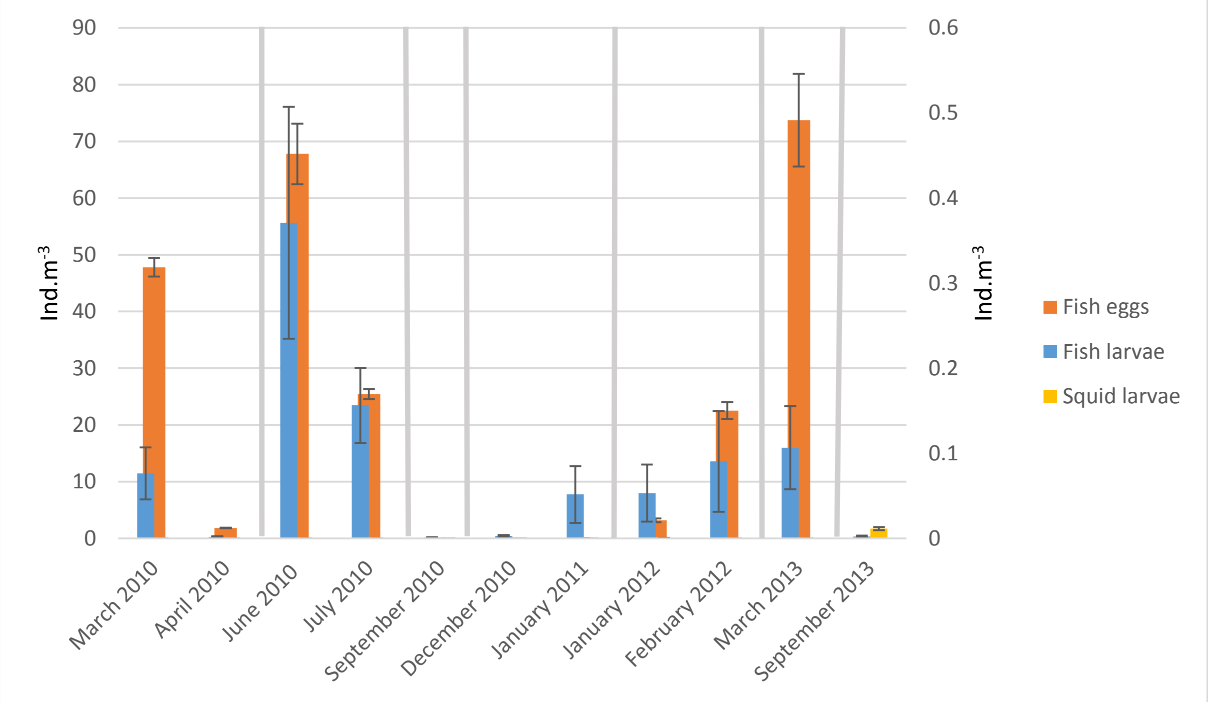

Figure 6. Density (ind.m^{-3}) of fish eggs (left axis) and fish and squid larvae (right axis) per sampling event. Grey vertical lines represent the periods (months) where data is missing.

Species richness *S* ranged between 0.3 to 7 species per sampling event, while the expected number of species in 50 individuals (ES_{50}) varied between 0 and 4. Shannon Wiener diversity (H') ranged between 0 and 1.6 (Figure 7). Species richness S showed both inter-seasonal and inter-annual significant differences. The seasonal differences were observed between winter-autumn and spring-autumn, while the annual ones were between 2011-2012, 2011-2013 and 2012-2013. ES_{50} on the other hand only exhibited a significant variability between seasons and not years. This asymmetry was between winter-autumn and summer-autumn and it was similar to the one observed in S. The Shannon-Wiener index showed interannual variability, with the differences lying between 2010-2011, 2011-2013 and 2012-2013.

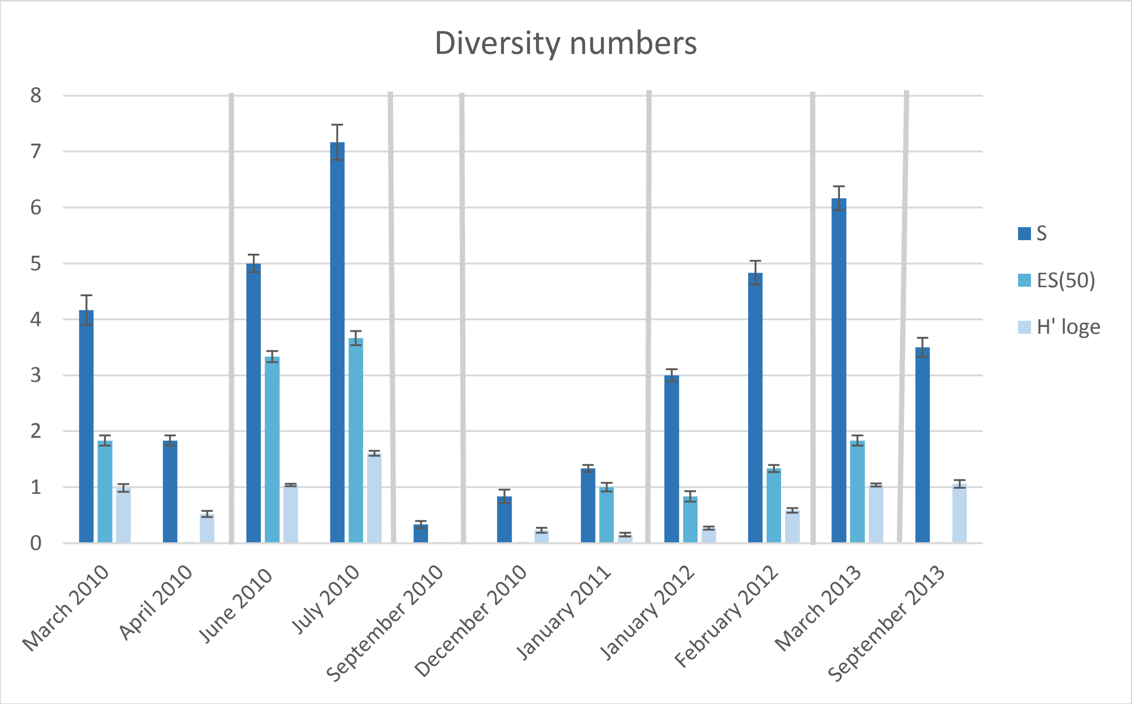

Figure 7. Column chart of diversity numbers for all sampling events considered in this study. Diversity represented as the number of species (S), the expected number of species in 50 individuals (ES(50)) and the Shannon-Wiener diversity index (H'). Grey vertical lines represent the periods (months) where data is missing.

Figure 8. Larvae of *Sprattus sprattus*, *Clupea harengus*, *Ammodytes marinus* and *Trachurus trachurus*.

The analysis of community structure of fish larvae also showed a clear temporal structure: in January the community composition was mainly *Clupea harengus* (87.82%) and Clupeidae sp. (10.29%). February was dominated by Ammodytidae sp. (87.35%) and *Clupea harengus* (9.62%). In March, *Ammodytes marinus* (23.85%) Ammodytidae sp. (40.19%) *Clupea harengus* (27.82%) had the highest relative abundance. In April the group with higher abundance was "other", which refers to amongst others *Arnoglossus laterna, Callionymus lyra, Pollachius pollachius,* and *Syngnathus rostellatus*. Besides this group, *Clupea harengus* represented 14.29% of the total community and Clupeidae sp. 21.43%. In June the group with highest abundance was also "other", but in terms of the species selected, Clupeidae sp. had a relative abundance of 38.45%, Pleuronectidae sp. 14.79% and *Sprattus sprattus* 12.01%. In July it was mainly *Sprattus sprattus* (39.33%) and *Trachurus trachurus* (20.08%), with a smaller percentage of *Clupea harengus* (9.62%). September was dominated by Gobiidae sp. (48.87%) and December by *Clupea harengus* (20%), *Sprattus sprattus* (20%) and Gobiidae sp. (20%). Since the individuals identified as Clupeidae sp. were the ones where a proper identification to species level was not possible, the interpretation of figure 9 suggests that they were most likely *C. harengus* in the first half of the year and *S. sprattus* in the second half. However, it's important to keep in mind the discontinuity of this study when interpreting the results.

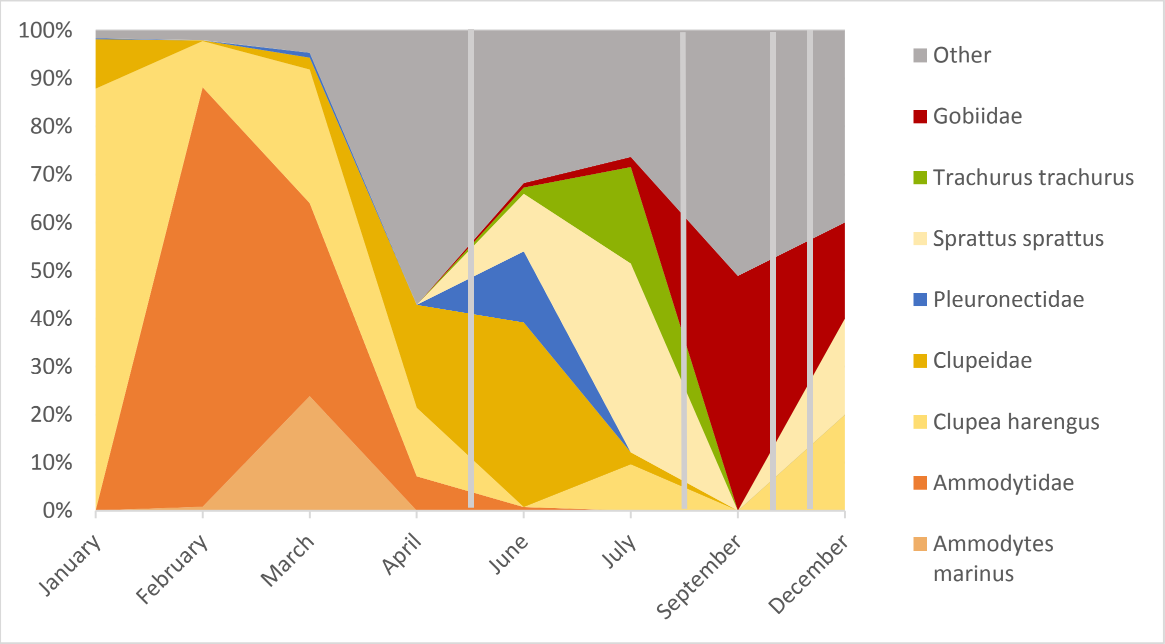

Figure 9. Relative seasonal abundance (%) of the most abundant fish larvae taxa. Grey vertical lines represent the periods (months) where data is missing.

ENVIRONMENTAL VARIABLES AND THEIR INFLUENCE ON FISH LARVAE

The profiles of all environmental variables show that the water column was generally well-mixed throughout the year. Water temperature showed peaks around July and September. Values fluctuated between 4 ºC in February 2012 (winter) and 17 ºC in July 2010 (summer). Salinity was practically constant throughout the year and ranged between 31 PSU and 34 PSU, with lowest values in March 2013 and highest values in January 2012 (Figure 10). Chl *a* and secchi depth (a measure of turbidity) showed largely opposite profiles. Chl *a* varied between 0.14 µg/L (June 2010) and 6.01 µg/L (April 2010) and Secchi depth had as lowest value 2 m (March 2013) and highest 8 m (June 2010).

To investigate the relationship between environmental variables and observed densities, a DistLM model was used. The output showed that the model that best fitted the data was the one including water temperature, salinity and secchi depth. Between the three variables, water temperature most explained the variation in the data (19%, p=0.0001), while Secchi depth was responsible for 12% (p=0.0001) and salinity for less than 0.1% (p=0.0001).

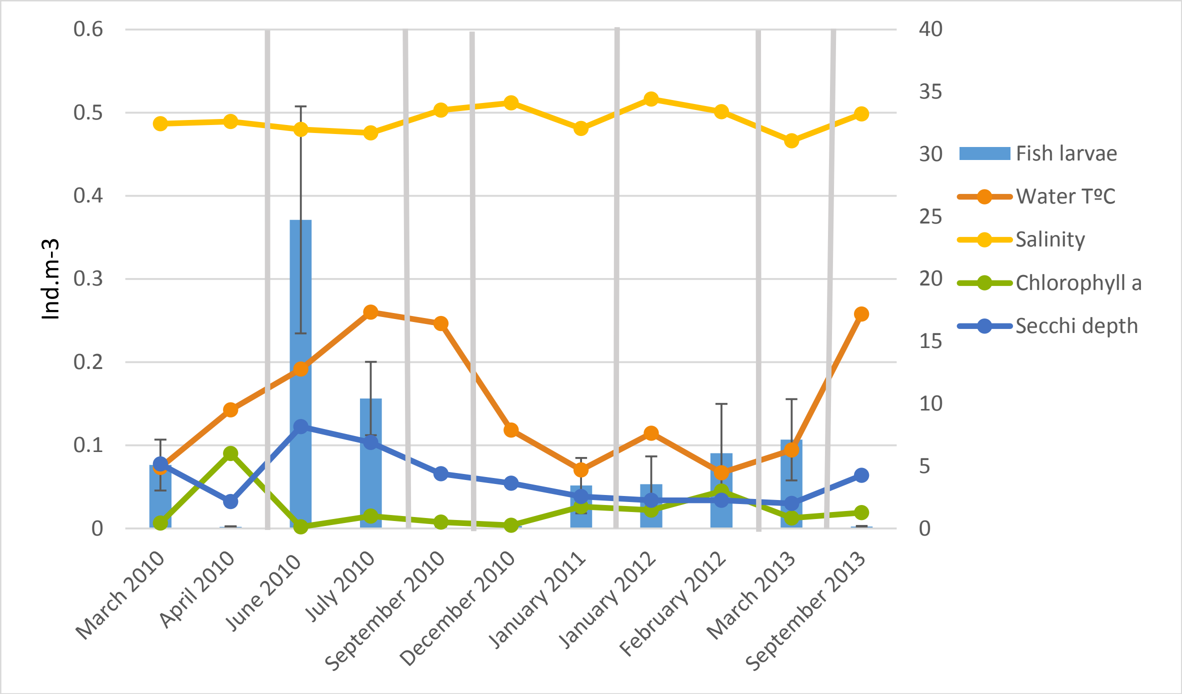

Figure 10. Fish larval densities (ind.m-3) per sampling event, averaged over all stations (with SE) in the left axis. On the right axis: Water temperature (ºC), salinity (PSU), chlorophyll a (µg/L) and secchi depth (m). Grey vertical lines represent the periods (months) where data is missing.

A distance-based redundancy analysis (dbRDA) plot was combined with a Multi-Dimensional Scaling (MDS) plot of fish larvae composition in order to visualize, in context, the results from the DistLM analysis. This ordination plot illustrates the relationship between the predictor (environmental) variables that best explain the fish larvae density data variation. The vectors within the circle represent the effect of the explanatory variables included in the model, with the length of the vector corresponding to how much a variable explains the data. The longer the vector the best a certain variable explains it (Anderson *et al.,* 2008). Accordingly, the variables which best explained the fish larvae density data were water temperature, Secchi depth and salinity (Figure 11).

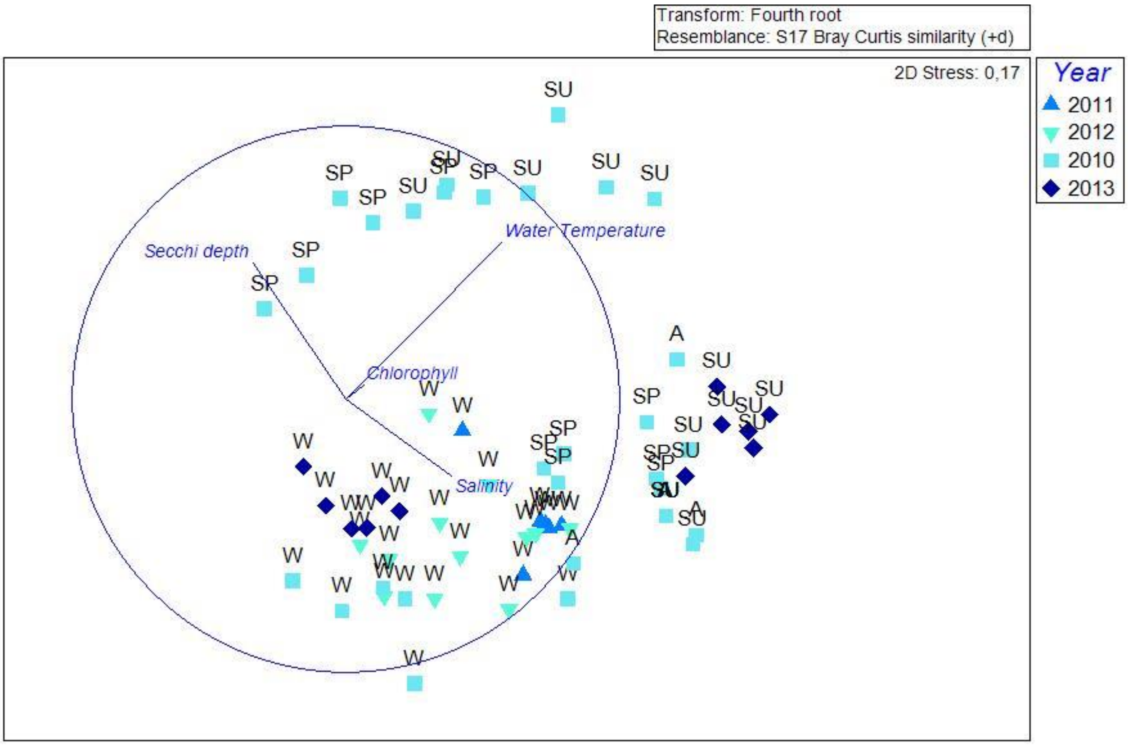

Figure 11. Two-dimensional non-metric MDS plot (stress value=0.17) of all samples, with indication of different shades of blue for year (2010-2013) and different symbols for spring (April-June), summer (July-September), autumn (October-December) and winter (January-March). Adjusted to the MDS plot is the distLM output graph (dbRDA plot), with the environmental variables that best explain the outcome. The bigger the line the more likely that variable is to explain the pattern observed in the MDS plot.

9.4. DISCUSSION

GENERAL

This study focused on the offshore zooplankton community of the Thorntonbank, and a total of 38 fish and 2 squid species were found within the sampling area. Similar results were obtained in previous studies conducted in the BPNS and the larger North Sea (e.g. Dewicke et al., 2002; Taylor et al., 2007; Van Ginderdeuren et al., 2013), with the exception of the presence of larvae of *Gobius niger*, which were not recorder yet in Belgian waters. Adults, however, are found abundantly near wind turbine foundations (Andersson and Öhmann, 2010).

WIND FARM IMPACT

Up to now, research has been mainly focusing on the impacts of wind farms on the adult stages of fish (e.g. Derweduwen *et al.,* 2010; Lindeboom *et al.,* 2011; Reubens *et al.,* 2010, 2011; Vandendriessche *et al.,* 2011-2013a). The present study, on the other hand, specifically focused on the impact of wind farms on the early life stages of fish and squids in order to detect the specific effects of the introduction of turbines (hard substrate) on such an important part of their life cycle. Potential effects include an increase in eggs and larvae of species which require hard substrates to spawn (e.g. herring, gobies), and a decrease in the species which need sandy bottoms.

Despite the expected effects of the wind farm on fish and squid larvae, PERMANOVA tests did not detect significant differences between the two sampling locations: control (reference area) and impact (concession area), in terms of density, species composition and diversity. The absence of statistical evidence of impact, however, should be seen in the light of the following facts:

- The continuous construction of turbines after the beginning of this study (see Figure 1), leading to an expansion of the concession area which almost reached the reference area by the end of the study period. This caused the conditions between the two sampling locations to become increasingly similar with time, reducing the probability of finding significant differences among them. The reference stations could have been affected by the disturbance caused during the construction phase, which influenced the usual dynamics of the area. However, effects take longer to establish with increasing distance from the impact so some distinction between the sampling stations (control and impact) would probably still exist but the disturbance decreased the probability of that difference being significant. Therefore, in future studies it is imperative to move the reference area further away from the C-Power concession in order increase the possibility of detecting the impacts of the introduction of such artificial structures while still maintaining similar environmental conditions (to avoid confounding factors).
- The large distance between turbine rows which delays the detection of changes within the sandy substrate, and the safety measures (i.e. distance from turbines) which prevented the detection of effects in the areas where they most occur (i.e. closer to the turbines, Coates *et al.,* 2012).
- The wind farm is relatively recent and it has been shown that stable biotic communities take some time to be fully established following the introduction of an artificial structure (new habitat) (i.e. 3 to 5 years) delays the detection of impacts (Degraer *et al.,* 2012; Jensen, 2002; Gray, 2006; Petersen and Malm, 2006). Therefore, effects of spawning substrate are still small but may increase over the following years and with the increase of the number of turbines.

In terms of squid larvae, an increase in density within the wind farms in comparison with the reference area was expected, due to their preference for hard substrate as spawning ground. The results, however, showed that the densities were rather low and that only few individuals were caught with the Bongo net. Hence, other sampling methods should be considered in order to capture more individuals and allow a proper comparison between different locations. One of these methods could include visual census (performed by divers) of spawning adults and egg clusters. Diver observations are already being carried out within the framework of the environmental monitoring of the wind farms in the BPNS (Kerckhof *et al.,* 2012), and an additional focus on squids and their egg clusters could be considered. Egg clusters have been seen on video footage (Francis Kerckhof, pers. comm.) and on loggers recovered from the wind farm area (Jan Reubens, pers. comm.) but these were not investigated in detail. Moreover, literature has been mainly focusing on adults and on the species with high commercial value (e.g. *Loligo forbesi*) and there is a general lack of information regarding the community composition, prey, spawning periods, geographic range and temporal distribution of these animals and their young. There is a need to expand the focus of research and encompass a wide range of species. A fundamental baseline study on the distribution of larval and juvenile squids in the southern North Sea is desirable. An impact study of wind farms and other anthropogenic activities can only be successful when detailed information on the ecology of these species is available.

Moreover, other than detecting the anthropogenic impact of wind farms on the density and diversity of ichthyoplankton and squids, it is important to confirm if the wind farms are, in fact, spawning grounds. Usually this can be detected by mapping the distribution of species and observing which areas have higher aggregation of individuals, the presence of both eggs and larvae, and the presence of mature individuals with running eggs or sperm (Ellis *et al.,* 2012). Therefore, ichthyoplankton studies, such as this one, provide valuable information for tracing the changes derived from the introduction of a new type of substrate (creation of a new spawning ground), but should be complemented with more information (e.g. spawning stock biomass).

OTHER SOURCES OF VARIATION

Fish larvae

Significant wind farm effects were not detected, but significant differences among seasons and years were very clear, reflecting a temporal variability in density, composition and diversity of fish larvae. The temporal variability was best reflected in the succession of species throughout the year with *Clupeidae* present year-round, alternating between *C.harengus* and *S. sprattus. Ammodytidae* were mostly confined to February and March, while *Pleuronectidae* appeared mainly in June, followed by *T. trachurus* and *Gobiidae* in the last months of the year. Other than this seasonal succession, also interannual differences were detected. These types of variability are related to the time of spawning and to the time at which the main food sources of fish larvae are most available, so that the hatching matches the period of higher food availability. Since fish larvae are small in size, their swimming capacities are not yet well developed (i.e. slow swimming speed), which makes finding prey more difficult. Therefore, the higher the abundance in prey, the higher the encounter rates will be and feeding will succeed instead of fail (Bailey *et al.,* 2004). This synchronization is vital for the recruitment success of high trophic level species in the temperate southern North Sea (Hjort 1914, Cushing 1990, Eilertsen and Wyatt 2000, Kirby *et al.* 2007).

Fish larvae are very susceptible to not only biological, but also physical changes in the environment (Taylor *et al.,* 2002, Voigt *et al.,* 2003, Hays *et al.,* 2005). These changes affect their physiological metabolism and reproduction, so individuals tend to search for appropriate conditions (e.g. temperature, salinity) which may differ to some degree between species (Rose, 2005; van der Kooij *et al.,* 2008; Munk *et al.,* 2009; Hillgruber *et al.,* 1997, Erftemeijer *et al.,* 2009). These environmental conditions also have the capacity to affect fish larvae indirectly, by affecting the timing of both phyto- and zooplankton blooms (food sources). For instance, temperature and salinity have a major effect on the stratification or mixing of the water column, which consequently influence the nutrient supply to the layers occupied by phytoplankton. Moreover, warmer temperatures are known to affect the timing of both blooms (Genner *et al.,* 2010; Smayda *et al.,* 2004; Beare *et al.,* 2002).

Surface water temperature during the study period ranged between 4ºC in February 2012 (winter) and 17ºC in July 2010 (summer). Similar results were obtained by Van Ginderdeuren *et al.* (2013) with a minimum of 2ºC also in February and a maximum of 21ºC in August, and O'Brien *et al.* (2011) with the minimum temperature being recorded in February and the maximum in August. Salinity ranged from 31 to 34 PSU, remaining constant throughout the year, which matched the observations made by Van Ginderdeuren *et al.* (2013) where the range was from 30-35 PSU. These results are identical to the "typical values for salinity and temperature of water masses in the North Sea" (OSPAR report, adapted from NSTF, 1993). Secchi depth, as a measure of turbidity, showed an opposite pattern in relation to Chl *a*. This is logical since an increase in phytoplankton density increases the amount of particles in the water, thereby decreasing visibility (increasing turbidity). An increase in turbidity corresponds to a lower secchi depth. However, it is not accurate to assume an exclusive correlation between Chl *a* and secchi depth, since visibility/turbidity is also affected by other factors (e.g. sediment resuspension).

In this study, the model which best fitted the density of fish larvae was the one with all variables except Chl *a*. This was not expected, since fish larvae are dependent on phytoplankton, and Chl *a* is an indirect measure of phytoplankton abundance. Many studies, performed on merohyperbenthos (i.e. individuals which spend only part of their early-life history on the water layer closer to the seabed, including fish larvae), have come to the conclusion that the two variables which explained the data best were Chl *a* and temperature (Dewicke *et al.,* 2002; Russell 1976; Lindley 1998). In the present study, large chl *a* differences were not expected, since control and impact stations were quite close to each other. Nevertheless, chl *a* values were obtained from in situ fluorimeter measurements. In the case of missing or aberrant measurements, values were obtained from satellite data. The spatial and temporal resolution of these data were not always optimal, which could have obscured small-scale spatial variation. Hence, chl *a* was not retained in the statistical model. However, the remaining variables within the model were expected to have an influence on the ichthyoplankton data. For instance, Harris and Cyrus (1996) reported temperature and turbidity (secchi depth) as the most relevant explanatory variables in terms of fish larvae density.

Fish eggs

A significant inter-seasonal and inter-annual variability was also detected in terms of fish egg density. They were observed throughout the study period and showed a similar pattern as fish larvae, with peaks occurring in March (2010, 2013) and June (2010). The year-round occurrence of fish eggs was expected since no distinction was made between eggs from different species, and spawning periods are spread throughout the year with occasions where the spawning period of more than one species overlap. This may explain the higher densities in those months. These observations supported the ones made by Dewicke *et al*. (2002), who found the highest numbers of fish eggs around the same time of the year in the Flemish Banks, and by Beyst *et al*. (2001) where fish eggs were observed around March and April. Like fish larvae, the peaks in egg density were most likely related with the synchronization between the time of spawning and of higher food availability for the newly-hatched larvae. The analysis of fish eggs is important since it provides valuable insight in terms of spawning of important fish species, sometimes more precise than the data obtained from adult individuals (Fox *et al.,* 2005). This is of extreme importance for fisheries biology and management in order to, for instance, outline important habitats and determine the cause of shifts in population abundance (Munk *et al.,* 2009). Therefore, in future studies, the identification to species level should be conducted not only for larvae, but for eggs as well.

9.5. CONCLUSION

Based on the data of 2010-2013, no clear evidence could be provided for positive impacts of wind farms on early life stages of fish and squid. The study, however, provides a good baseline for future monitoring and allows to fine-tune research methodology. Recommendations for future studies include an optimal geographical spacing of the sampling stations, and an extension of data collection through visual census. Additionally, data on early life stages should be complemented with data on spawning stock biomass.

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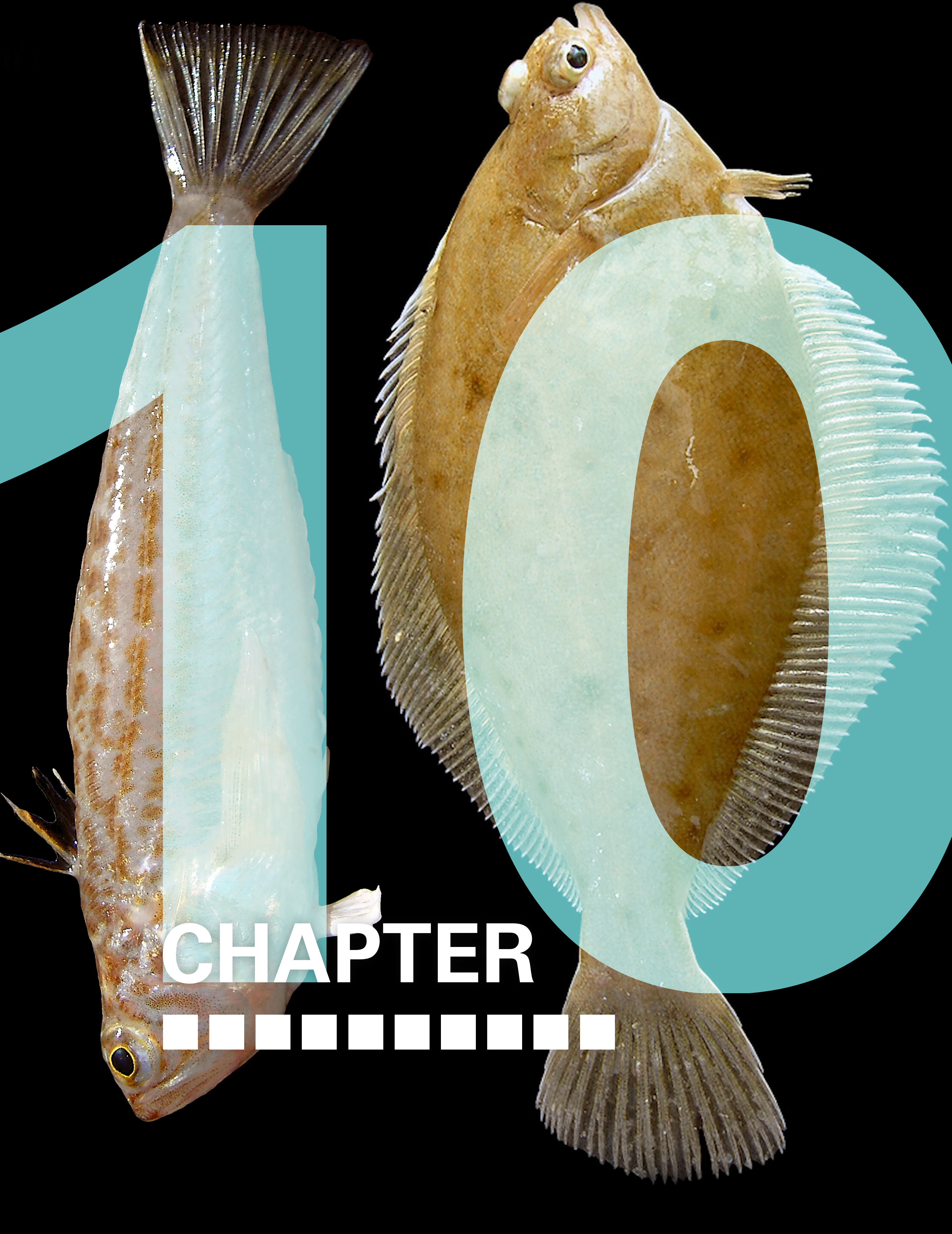

10

CHAPTER

CHAPTER 10

FEEDING BEHAVIOUR OF LESSER WEEVER (*ECHIICHTHYS VIPERA*) AND DAB (*LIMANDA LIMANDA*) IN THE C-POWER WIND FARM

J. Derweduwen, J. Ranson, J. Wittoeck & K. Hostens*

Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Oostende, Belgium.

**Corresponding author: Jozefien.Derweduwen@ilvo.vlaanderen.be*

ABSTRACT

This chapter focuses on the feeding behaviour of lesser weever (*Echiichthys vipera*) and dab (*Limanda limanda*) in and around the C-Power wind farm. To find out if the presence of wind farms is influencing the feeding behaviour of both demersal fish, stomach content analyses were performed for both demersal fish species originating from the wind farm and from a nearby control area. Results on stomach fullness, frequency of occurrence and numerical percentage of prey taxa, prey diversity and prey species composition are discussed.

The fullness index and prey diversity of lesser weever was not affected by the presence of the wind farm. However, the diet composition did change: lesser weever consumed significantly more of the species

Jassa herdmani - which is typically associated with hard substrates and highly available in the wind farm- in both the control and to a greater extent in the impact area.

The fullness index of dab also displayed no significant differences. The prey diversity and the diet composition of dab however, were influenced by the presence of the wind farm. The ampipods *Nototropis swammerdamei* and *J. herdmani* and the tube-building polychaete *Lanice conchilega* were responsible for those differences. The latter species is a well-known ecosystem-engineer with the potential to enhance habitat complexity and heterogeneity. Its presence might have led to a significant higher prey diversity in the wind farm and hence in the diet of dab.

The differences in feeding behaviour between wind farm and control area can in part be related to the presence of the wind farm, its associated fauna and the expanding reef effect.

10.1. INTRODUCTION

With the construction of wind farms, artificial hard substrates are introduced into a natural sandy environment, and act as artificial reefs (Petersen & Malm, 2006; Langhamer, 2012). These hard substrates are in general rapidly colonized by an epifaunal community (Petersen & Malm, 2006; Kerckhof *et al,* 2010; De Mesel *et al.,* 2013; De Mesel *et al,* 2015) which may provide food for fish that aggregate around these structures (May *et* al, 2005; Reubens *et al,* 2011; Reubens *et al,* 2013; Wilhelmsson *et al*, 2006). Also demersal fish from the surrounding soft substrates profit from the presence of the wind turbines.

In 2012, a feeding behaviour study on dab *Limanda limanda* was performed which showed that dab had fuller stomachs in the C-Power wind farm compared to the control area (Derweduwen *et al.*, 2012). Furthermore, the hard substratum species *Phtisica marina* solely occurred in the stomachs of fish originating from the wind farm area. Other diet studies on pouting (Reubens *et al,* 2011; Reubens *et al*, 2013) indicated that the amphipod *Jassa herdmani* and the crab *Pisidia longicornis* - both (sub)dominantly present on the foundation of the wind farms (Kerckhof *et al,* 2010; De Mesel *et al*, 2013; De Mesel *et al.,* 2015) - were important prey species. However, these prey species were not found in the diet study of Derweduwen *et al.* (2012). This is probably due to the fact that the hard substratum epifaunal community was not yet fully developed and stable two years after construction (Jensen, 2002; Gray, 2006; Petersen and Malm, 2006) and hence the effects on the soft sediment were also still limited (Bergström *et al,* 2012; Bergström *et al.*, 2013, Wilhelmsson *et al.,* 2006; Vandendriessche *et al*, 2013; Vandendriessche *et al.*, 2015). In this study, which is conducted five years after construction, we may expect different effects of the windmills and its hard substrate community on the demersal fish of the soft sediment.

Both the artificial reef effect associated with the physical presence of the turbine foundations and scour protection (Reubens *et al*, 2011; Reubens *et al*, 2013; Derweduwen *et al*, 2012) and the exclusion of fisheries activities from wind farms and their safety buffers may change the food availability and subsequent diet of demersal fish within the wind farm (Berkenhagen *et al.*, 2010; Kaiser & Ramsay, 1997; van Hal *et al.*, 2012).

The research questions we want to answer in this study are the following.

- Do demersal fish have fuller stomachs inside versus outside the wind farm and 5 years after versus before the construction of the wind farm? In other words, do they feed more inside the wind farm, after it was constructed?
- Do fish have a different diet composition inside versus outside the wind farm and before versus 5 years after the construction of the wind farm?
- Do demersal fishes feed on hard substratum species associated with the wind farm constructions?

10.2. MATERIAL AND METHODS

SAMPLING

In autumn 2009, 2010, 2012 and 2013 samples for stomach analyses were collected at several impact locations within the C-Power wind farm, located on the Thornton bank in the Belgian part of the North Sea and at an adjoining reference location using an 8m shrimp trawl (see chapter 8 'Effects of Belgian wind farms on the epibenthos and fish of the soft sediment').

Per station, a number of specimens of lesser weever (*Echiichthys vipera*), dab (*Limanda Limanda*) and whiting (*Merlangius merlangus*) were collected and injected with formaldehyde (35 %) for preservation. The length of lesser weever varied between 70 and 155 mm and the length of dab varied between 123 and 270 mm. All individuals of both species were subdivided into three length categories: small (S), medium (M) and large (L) (see Table 1). The specimens were stored in formaldehyde (8 %) until analysis.

Table 1. Length categories (in mm) for the two studied fish species, dab (L. limanda) and lesser weever (E. vipera).

	Dab	**Lesser weever**
Small	<151	<101
Medium	151-170	101-130
Large	>170	>130

LABORATORY TREATMENT

The intact stomachs were removed by cutting above the oesophagus and below the large intestine. An incision was made along the longitudinal axis and the contents were emptied on a sieve (0.125 mm), rinsed and put into a Petri dish with a few drops of deionised water. All prey items encountered in the stomachs, were counted and identified using a binocular microscope. Prevailing protocols for accreditation were followed

(BELAC – ISO 16665), using the current determination keys and the correct names based on WoRMS (Vandepitte *et al.* 2010) If possible, prey items were identified to species level. Some prey items were classified into a higher taxonomic level (e.g. order) due to fragmentation or partial digestion.

Both fish and stomach contents were placed into separate vials for further investigation and subsequent drying. After identification, the stomach contents were placed in pre-weighed porcelain or aluminium foil cups, dried at 60°C for 48 hours, weighed, incinerated in ceramic cups at 500°C for 2 hours and cooled to room temperature in a desiccator for 2 hours before weighing again in order to obtain ash weights and to calculate ash free dry weights (AFDW) of the stomach contents. An overview of all the analysed fishes is given in annex 2.

DATA ANALYSIS

Analyses were done for lesser weever and dab but not for whiting since there were not enough control samples for this species. For the analysis of the stomach content data, several indices were used. The fullness index (FI) was used, where S_i is the ash-free dry weight (AFDW) of the stomach content in milligram (mg) and W_i is the ash-free dry weight (AFDW) of the fish (mg).

$$FI = \frac{S_i}{W_i} x100$$

For a number of fishes, only the Wet Weight (WW) was determined. WW of the fish was then converted to AFDW with the common formula AFDW ≈ 20% of WW (Edgar and Shaw, 1995; Van Ginderdeuren, 2013).

Also the percentage of empty stomachs was calculated for each fish species and station.

The frequency of occurrence and numerical percentage of prey items were calculated to characterise the stomach contents (Hyslop, 1980). The frequency of occurrence (*FO%*) calculates the percentage of the total number of stomachs in which a specific prey species occurs where FO_i is the number of stomachs in which the species '*I*' occurs, and FO_t is the total number of full stomachs.

$$FO\% = \frac{FO_i}{FO_t} x100$$

The diet composition was expressed as a numerical percentage (*N%*):

$$N\% = \frac{number\ of\ individuals\ of\ prey\ type\ i}{total\ number\ of\ ingested\ prey\ items} x100$$

The prey species richness in fish stomachs was estimated by S, the number of species in a stomach. The Shannon-Wiener Index H'(log*e*) was used to calculate prey species diversity.

Statistical analyses were performed using the Plymouth routines in multivariate ecological research (PRIMER)e-package + PERMANOVA add-on, version 6.1.6 (Andersen *et al.*, 2007). Prior to multivariate analysis the prey abundance data were standardised (De Crespin de Billy *et al.* 2000) and a similarity matrix was constructed using the Bray-Curtis index of similarity. For the community analysis, the multivariate techniques SIMPER (similarity percentages procedure) and PCA (Principal Component Analysis) were used to investigate the feeding strategy of lesser weever and dab and to highlight the important prey items in their diet.

The statistical analyses are based on the "Before After Control Impact" (BACI)-design (Smith *et al.,* 1993). The analysed factors are "time", "area" and "length category". The factor "time" has two levels: Before *versus* After the construction of the wind farm, also noted as B and A. The "After"-period implies the presence of the 3-dimensional wind farm (from spring 2011 onwards).

The factor "area" also has two levels: Control *versus* Impact, also noted as C and I. An effect of 'time" solely gives an indication of natural temporal variation, both in control and impact areas. An effect of "area" demonstrates natural spatial variation, both before and after the construction of the wind farm. An interaction between "time" and "area" indicates that there is a wind farm effect on the prey density, diversity or species composition. Pair-wise tests then could reveal where the differences are situated.

The factor "length category" has three levels: Small, Medium and Large. Since this factor has no significant effect on prey species diversity, prey species composition nor fullness index, all length categories were pooled for further analyses.

10.3. RESULTS

LESSER WEEVER (*ECHIICHTHYS VIPERA*)

Fullness Index (FI) and % empty stomachs

In general, the percentage of empty stomachs was relatively low, especially after the construction. There were more empty stomachs in the impact area than in the control area, both before (23% *vs.* 15%) and after the construction (16% *vs.* 6%) (Table2).

The fullness index FI ranged between 1.1 (±0.3) (AC) and 1.7 (±0.8) (BC) (Figure 1 left). Although the effect of 'length category' on the fullness index was not significant, the representation per length category (Small, Medium and Large) shows a slightly more detailed picture (Figure 1 right). In general, lesser weever had fuller stomachs in the impact area (I). (Figure 1).

Also the factors 'time' (B/A) and 'area' (C/I) seemed to have no effect on the fullness index of lesser weever.

Table 2. Percentage of empty stomachs of lesser weever, Before (B) and After (A) the construction of the wind farm, in Control (C) and Impact (I) areas.

		% empty stomachs
B	C	15
	I	23,21
A	C	6,38
	I	15,66

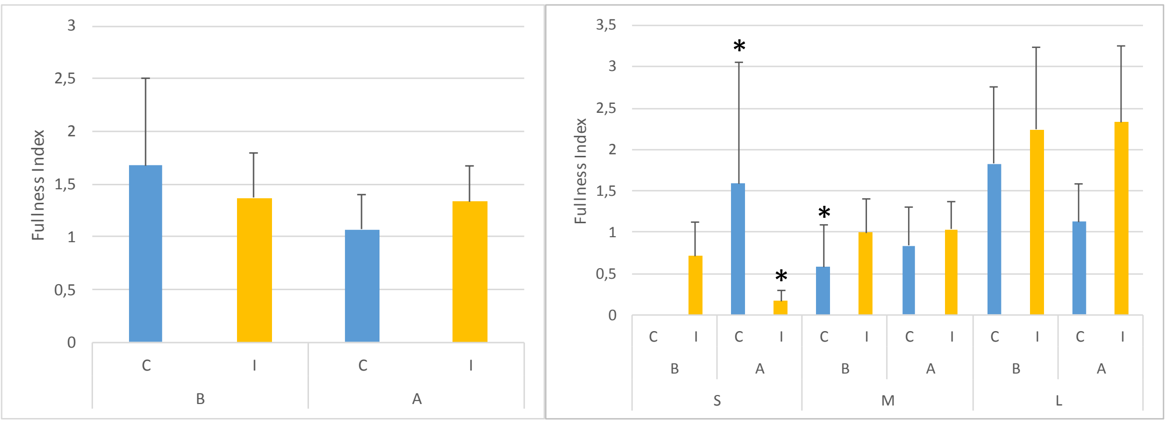

Figure 1. Fullness Index (±SE) for lesser weever (E. vipera), Before (B) and After (A) the construction of the wind farm, in Control (C) and Impact (I) areas (left) and also for small (S), medium (M) and large (L) individuals (right). * encompasses less than five samples for that combination of factors.

Diversity of the diet

The number of species nor the Shannon-Wiener Index H'(loge) was affected by the factors 'time' and 'area' nor by the interaction between those factors. So, no significant differences could be demonstrated between control and impact area, between before and after the construction of the wind farm, nor between any combination of those factors. This implies that the wind farm did not affect the diversity of the diet of lesser weever (Figure 3).

Notable however, is the species *Pisidia longicornis*, a hard-substratum Decapoda which only occurred in the wind farm area, after it was constructed (Figure 2).

Figure 2. *Pisidia longicornis*.

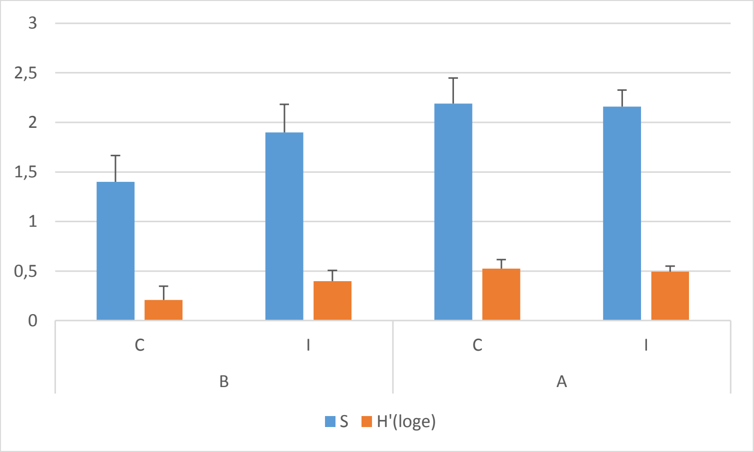

Figure 3. Average number of prey species/stomach S (±SE) and Shannon-Wiener Index H' (±SE) for lesser weever (E. vipera) before (B) and after (A) the construction of the wind farm, in control (C) and impact (I) areas.

Numerical percentage (N%) and Frequency of occurrence (FO%)

Before (B) the construction of the wind farm Decapoda (mainly Brachyura sp.) and Mysida (mainly *G. spinifer* (Figure 5 left) and unidentified Mysidae sp) were numerically the most important prey taxa in the diet of lesser weever (*E. vipera*) (Figure 4). Those taxa were also the most frequently encountered ones (see frequency of occurrence (FO%) (Table 3). After the construction of the wind farm, Decapoda and Mysida were still important prey taxa. However, the Amphipoda became more important, especially in the wind farm (I) were they dominated the diet of lesser weever with a numerical percentage (N%) of 57 % and a frequency of occurrence (FO%) of 79 % (Figure 4 and Table 3). This could mainly be attributed to the dominance (average number of 3±0.47) of the hard substrate Amphipoda *Jassa herdmani* (Figure 5 right) after the construction of the wind farm, both inside (AI) and outside the wind farm area (AC) (Figure 6). Indeed, a significant difference (p=0.01) in numbers of *J. herdmani* could be detected for the factor 'time' but no difference in numbers of *J. herdmani* were found between control and impact area.

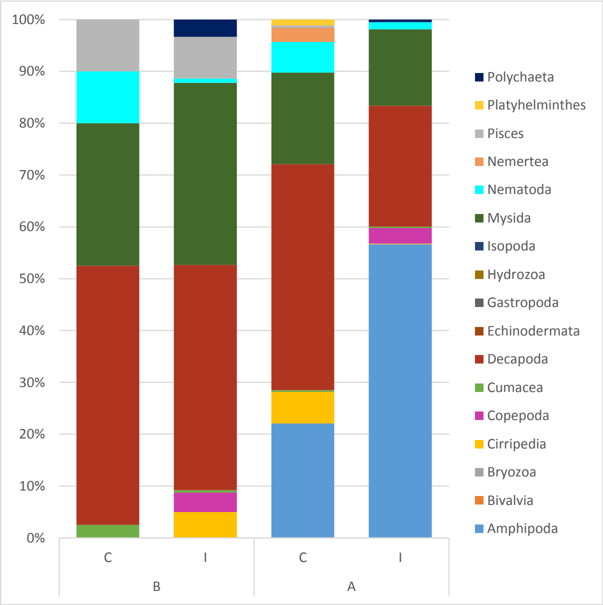

Figure 4. Representation of the diet composition of lesser weever (E. vipera) based on numerical percentages (N%) of prey items, before (B) and after (A) the construction of the wind farm, in control (C) and impact (I) areas.

Table 3. Frequency of occurrence (FO%) of the different prey taxa of lesser weever (E. vipera), Before (B) and After (A) the construction of the wind farm, in Control (C) and Impact (I) areas.

	B		A	
	C	I	C	I
Amphipoda	-	-	29,7	79
Bivalvia	-	-	-	-
Bryozoa	-	-	-	-
Cirripedia	-	6,7	8,1	6,5
Copepoda	-	6,7	-	1,6
Cumacea	10	3,3	-	3,2
Decapoda	60	56,7	67,6	43,6
Echinodermata	-	-	-	-
Gastropoda	-	-	-	-
Hydrozoa	-	-	-	-
Isopoda	-	-	-	-
Mysida	40	50	37,8	40,3
Nematoda	10	3,3	13,5	1,6
Nemertea	-	-	-	-
Pisces	10	23,3	2,7	-
Platyhelminthes	-	-	2,7	-
Polychaeta	-	3,3	-	1,6

Figure 5. *Gastrosaccus spinifer* (left) and the hard-substratum species *Jassa herdmani* (right) © Hans Hillewaert.

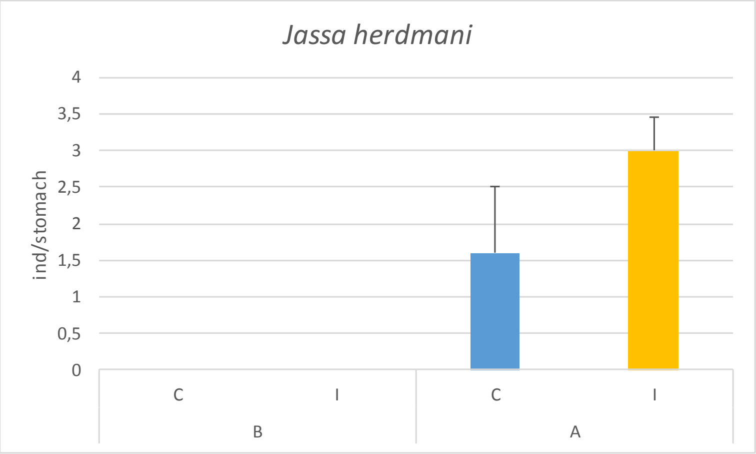

Figure 6. Average number (ind/stomach±SE) of J. herdmani before (B) and after (A) the construction of the wind farm, in control (C) and impact (I) areas.

Community analysis

Statistical analyses revealed several important differences in diet composition of lesser weever, both between control and impact areas (p=0.0004) and before and after construction (p=0.0001). Also, the overall BACI-effect turned out to be significant (p=0.003) which implies that the wind farm had a substantial effect on the diet composition of lesser weever (Table 4).

In the wind farm, the species community differed significantly before and after the construction (BI-AI; p=0.0001). Notable differences were also detected between control and impact areas after the construction (AC-AI; p=0.0001). Both phenomena could mainly be explained by two species: *J. herdmani* and *G. spinifer.* The former was clearly more abundant in the wind farm after the construction (AI) (see first paragraph). The latter however, showed a higher numerical abundance in the wind farm before (BI) than after (AI) the construction and had also a higher abundance after the construction in the control area (AC) than in the impact area (AI).

The PCA-plot illustrates above-mentioned and gives an indication of the most important prey species/taxa (Figure 7), which are also represented in Table 4.

Table 4. p-values for the different factors and their interaction effect on the diet composition of lesser weever (E. vipera) and the characteristic species/taxa for each group (AI= after impact; AC= after control; BI= before impact; BC = before control) based on SIMPER-analyses.

FACTOR	**Pair-wise tests**	**p**	**Group**	**Characteristic species/taxa**
Time (B/A)		0.0001	AI	*J. herdmani, G. spinifer*
Area (C/I)		0.0004	AC	*J. herdmani, G. spinifer, Processa modica,* Caridea sp.
BA x CI		0.003	BI	Mysidae sp., *G. spinifer,*
	B/A within I	0.0001	BC	*G. spinifer*
	B/A within C	0.09		
	C/I within A	0.0001		
	C/I within B	0.07		

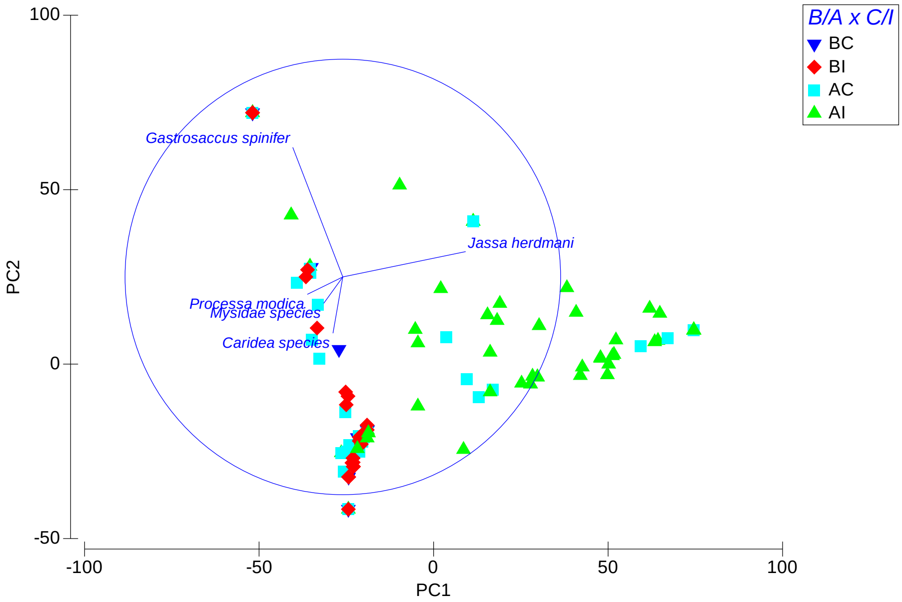

Figure 7. PCA (Principal Component Analysis) plot based on numerical prey abundances of lesser weever (E. vipera) with indication of the most important prey taxa. Axes 1 and 2 explain 24.4% and 13.2% of the total variation respectively.

DAB (*LIMANDA LIMANDA*)

Fullness Index (FI) and % empty stomachs

The percentage of empty stomachs varied between 11 and 18 % and was approximately equal before and after the construction and in control and impact areas (Table 5).

The fullness index varied between 0.03 (±0,01) AC) and 0.29 (±0.11) (BI) (Figure 8 left). The 'length category' again had no significant effect on the fullness index of dab but was visualised in Figure 8 (right) to get a more detailed image. The fullness index was generally higher in the wind farm area (I) than in the control area (C), both before and after the construction (Figure 8). However, this overall difference in fullness index between wind farm and control area was not significant.

The fullness index was lower ‘after’ construction than ‘before' (Figure 8 left). In the impact area however, this was only the case for the ‘large’ individuals (Figure 8 right). Still, the factor ‘time’ was not significant for the interpretation of the fullness index values.

Table 5. Percentage of empty stomachs of dab (L. limanda), Before (B) and After (A) the construction of the wind farm, in Control (C) and Impact (I) areas.

		% empty stomachs
B	C	17,65
	I	11,11
A	C	14,29
	I	15,15

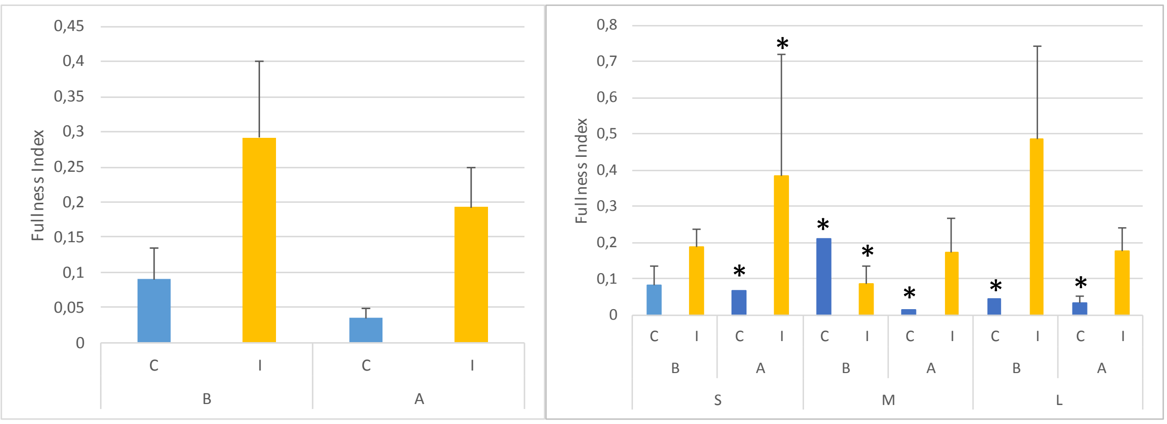

Figure 8. Fullness Index (±SE) for dab (L. limanda) Before (B) and After (A) the construction of the wind farm, in Control (C) and Impact (I) areas (left) and also for small (S), medium (M) and large (L) individuals (right). * encompasses less than five samples for that combination of factors.

Diversity of the diet

The average number of prey species was significantly (p=0.01) higher in the fish stomachs originating from the impact area (S=4±0.5) than in those originating from the control area (2±0.1). The same is true for the Shannon-Wiener diversity index H’ (0,9±0,1 *versus* 0,3±0,1, p= 0.001). This difference was most explicit after the construction of the wind farm (A) (Figure 9).

Species or taxa which only occurred in fish stomachs originating from the wind farm area (I) and were not present in fish stomachs originating from the control area (C) were *N. swammerdamei, L. conchilega, J. herdmani* Calanoida *sp.,* Copepoda *sp.,* Brachyura juvenile, Hydrozoa *sp., Liocarcinus pusillus,* Palaemonidae *sp.,* Eteoninae *sp.,* Gammaridea *sp., Abludomelita obtusata* and Echinodermata.

The factor ‘time’ nor the interaction between ‘area’ and ‘time’ had a significant effect on the prey diversity indices. Still, when comparing ‘Before’ and ‘After’ construction, it seems that the number of prey species has declined in the control area and increased in the wind farm area.

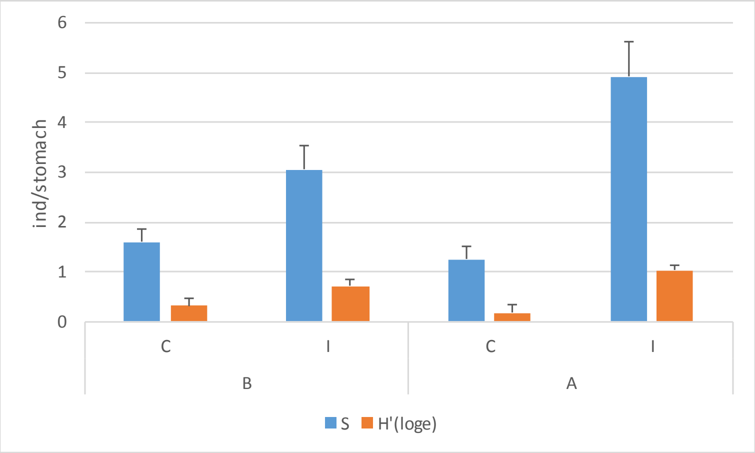

Figure 9. Average number of prey species/stomach S (±SE) and Shannon-Wiener Index H’ (loge) (±SE) for dab (L. limanda) before (B) and after (A) the construction of the wind farm, in control (C) and impact (I) areas.

Numerical percentage (N%) and Frequency of occurrence (FO%)

The diet composition of dab varied a lot, both for the factor ‘area’ (Control/Impact) as for the factor ‘time’ (Before/After). Nematoda were only present before the construction of the wind farm, particularly in the control area (BC) (N%=41.67 and FO%=44.44). Decapoda were numerically the most dominant taxon (N%=48.18) in the impact area before the construction (BI). Also the frequency of occurrence was the highest (FO%=65) of all taxa (Figure 10 and Table 6). Decapoda completely disappeared from the diet of dab in the control zone after construction. Cirripedia occurred solely after construction, especially in the control area (AC) where they were present in half of the stomachs (N%=37.5 and FO%=50) and were of equal importance as the Polychaeta, both numerically as concerning the frequency of occurrence.

The relatively high numerical percentage of Amphipoda in the impact area and the absence of that taxon in the control area is striking (Figure 10 and Table 6), particularly their dominance after the construction (N%=76.92 and FO%=51.68) is remarkable. This difference (between C and I) could particularly be attributed to the relatively high number of the Amphipoda *Nototropis swammerdamei* (5±1.68) in the wind farm (Figure 11). Moreover, there was a significant wind farm effect (BACI-effect) on the numerical abundance of *N. swammerdamei* (p=0.02).

Jassa herdmani (2±1.16) and *Lanice conchilega* (1±0.44) were also important species and occurred only in the wind farm and not in the control area (see previous paragraph and Figure 11 and Figure 12).

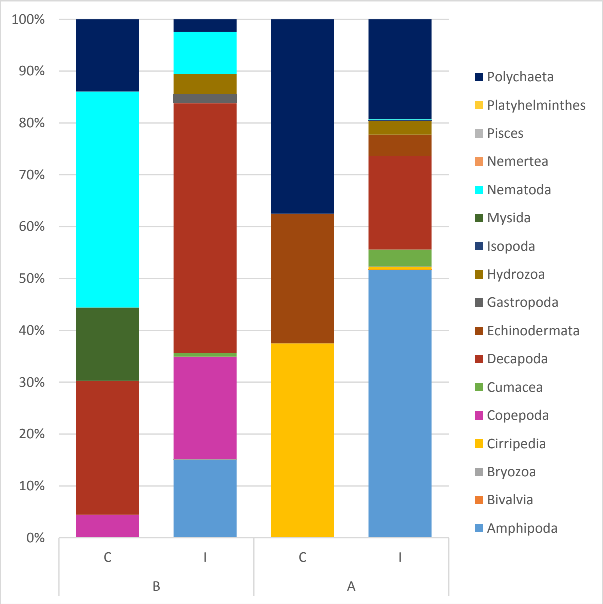

Figure 10. Representation of the diet composition of dab (L. limanda) based on numerical percentages (N%) of prey items, before (B) and after (A) the construction of the wind farm, in control (C) and impact (I) areas.

Table 6. Frequency of occurrence (FO%) of the different prey taxa of dab (L. limanda) Before (B) and After (A) the construction of the wind farm, in Control (C) and Impact (I) areas.

	B		A	
	C	I	C	I
Amphipoda	-	35	-	76,9
Bivalvia	-	-	-	3,9
Bryozoa	-	5	-	-
Cirripedia	-	-	50	7,7
Copepoda	11,1	40	-	3,9
Cumacea	-	5	-	11,5
Decapoda	33,3	65	-	50
Echinodermata	-	-	25	7,7
Gastropoda	-	15	-	-
Hydrozoa	-	20	-	15,4
Isopoda	-	-	-	7,7
Mysida	22,2	-	-	7,7
Nematoda	44,4	20	-	3,9
Nemertea	-	-	-	-
Pisces	-	-	-	-
Platyhelminthes	-	-	-	-
Polychaeta	33,3	10	50	61,5

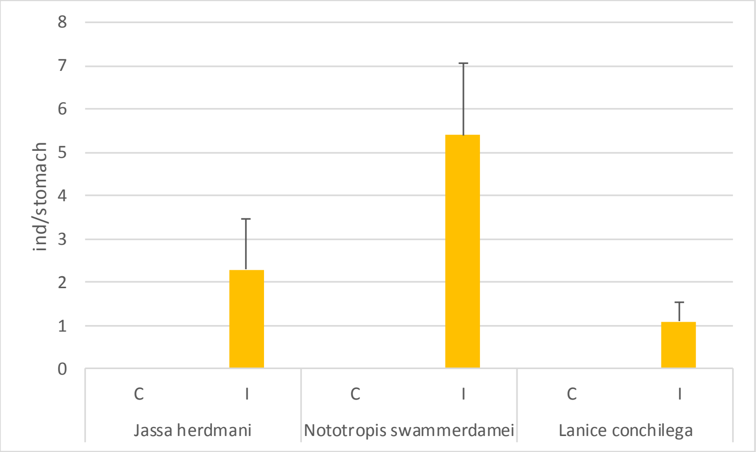

Figure 11. Average number (ind/stomach±SE) of J. herdmani,, N. swammerdamei and L. conchilega in the stomachs of dab (L. limanda) before (B) and after (A) the construction of the wind farm, in control (C) and impact (I) areas.

Figure 12. *Nototropis swammerdamei* (left) and *Lanice conchilega* (right) © Hans Hillewaert (left) and P. Legranche (right).

Community analysis

Statistical analyses of the prey species composition of dab indicated a significant interaction between 'time' (B/A) and 'area' (C/I) (p=0.0001), which implies that the wind farm does have an effect on the prey species composition.

Before the construction of the wind farm, the prey species composition differed significantly in control and impact areas (BC-BI) due to higher abundances of Nematoda and Brachyura in the control and impact area, respectively (p=0.005). After the construction however, the dominance of Cirripedia in the control area and of *N. swammerdamei* in the impact area were responsible for the significant difference between 'areas' (AC-AI) (p=0.0005). Looking into more detail to the impact area, differences in prey species composition before and after the construction (BI-AI) were particularly caused by the dominance of *N. swammerdamei* after the construction (A) and of Brachyura before the construction (p=0.0001) (Table 7). In the control area, Nematoda dominated before the construction, whereas Cirripedia were the most occurring taxon after the construction (AC-BC) (p=0.01)

SIMPER-analyses also revealed the most characteristic species/taxa for each combination of factors which are described in Table 7. The most structuring taxa are also represented in the PCA-plot (Figure 13).

Table 7. p-values for the different factors and their interaction effect on the diet composition of dab (*L. limanda*) and the characteristic prey species/taxa for each group (AI= after impact; AC= after control; BI= before impact; BC = before control) based on SIMPER-analyses.

FACTOR	**Pair-wise tests**	**p**	**Group**	**Characteristic species/taxa**
Time (B/A)		0.0001	AI	*N. swammerdamei*, *L. conchilega*, *J. herdmani*
Area (C/I)		0.0001	AC	Cirripedia sp.
BA x CI		0.0001	BI	Brachyura sp., Paguridae sp., Copepoda sp.
	B/A within I	0.01	BC	Nematoda sp.
	B/A within C	0.0001		
	C/I within A	0.0005		
	C/I within B	0.005		

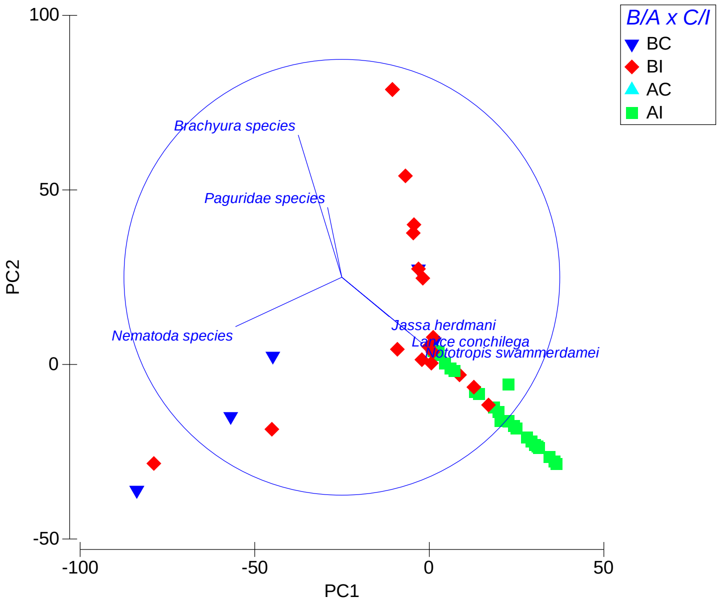

Figure 13. PCA (Principal Component Analysis) plot based on numerical prey abundances of dab (L. limanda) with indication of the most important prey taxa. Axes 1 and 2 explain 15% and 12.2 % of the total variation, respectively.

10.4. DISCUSSION

In this study the feeding behaviour of two demersal fish species (dab *Limanda limanda* and lesser weever *Echiichthys vipera*) was examined in and around the C-Power wind farm located on the Thorntonbank. To investigate whether the wind farm had an effect on the diet of the fishes, stomach content analyses were performed and changes in stomach fullness or diet composition were investigated in a Before After Control Impact (BACI) design.

LESSER WEEVER (*ECHIICHTHYS VIPERA*)

We encountered a relatively low percentage of **empty stomachs** for lesser weever (*E. vipera*). Quiniou (1978), Dauvin (1988), Creutzbert & Witte (1980), Vasconcelos *et al.* (2004) and Derweduwen *et al.* (2012) all found higher percentages of empty stomachs. Different authors noted that the time of sampling may play a role, since the nocturnal activity of lesser weever leads to fuller stomachs towards the morning (Lewis, 1976; Wheeler, 1978 in Vasconcelos *et al.*, 2004). Also the type of prey may influence the stomach fullness (Derweduwen *et al.*, 2012).

No significant differences in **fullness index** could be denoted between the wind farm area and the control area, neither before nor after the construction of the wind farm. This may partially be due to the use of a conversion formula to obtain the ash free dry weight (AFDW) of a number of fishes, leading to a rough estimation of the real AFDW.

The number of prey species and the **prey diversity** in the diet of lesser weever seemed not to be influenced by the wind farm nor by the individual factors ‘time’ and ‘area’. Notable however is that the long clawed porcelain crab *Pisidia longicornis* was found for the first time in the stomachs of lesser weever. This species is a common inhabitant of hard substratum communities (Ingle, 1980; Zintzen *et al.*, 2006; Zintzen *et al.*, 2008b) and abundantly present on the wind turbines almost directly after construction (Kerckhof *et al.*, 2010; De Mesel *et al.*, 2013; De Mesel *et al.*, 2015),

The **diet** of lesser weever in the control and impact area mainly consisted of Brachyura and Mysida, both numerically and concerning the frequency of occurrence. In a previous study, Derweduwen *et al.* (2012) found that lesser weever mostly foraged on Mysida, which was also found by Vasconcelos *et al.* (2004). Other studies also reported that the diet of lesser weever mainly consists of Crustacea, including Decapoda (Creutzberg and Witte, 1989; Quiniou, 1978; Sorbe, 1981; Dauvin, 1988; Collignon and Aloncle, 1960).

After the construction of the wind farm, the importance of Brachyura and Mysida decreased, while Amphipoda emerged in the diet and became the most important prey taxon in the impact area. The species, responsible for this was *Jassa herdmani*. This is the first observation of *J. herdmani* in the diet of any soft substrate demersal fish in and around the wind turbines. In a previous study, this amphipod species was not yet encountered in the fish diet (Derweduwen *et al.*, 2012). *Jassa herdmani* is a dominant species of the epifaunal community on the foundation of wind farms (Kerckhof *et al,* 2010; Kerckhof *et al.*, 2012; De Mesel *et al*, 2013; De Mesel *et al.,* 2015) . Based on the studies of Bergström *et al.* (2012, 2013) and Wilhelmsson *et al*. (2006), Vandendriessche *et al.* (2015) hypothesized that increases or changes in density, biomass, diversity, or community structure of the soft sediment communities between the turbines would remain limited or that it would take a long time before the reef effect expands into the sandy space between the turbine rows. Given the dominance of *J. herdmani* both on the hard substrate and in the diet of lesser weever (and dab, see further) and the relatively high abundance (up to 809 ind/m^2) of the species on the soft sediment near the turbines (Coates *et al.*, 2013), we can conclude that - at least to some extent - the reef effects already did expand into the soft sediments between and beyond the wind mills, *circa* 200 m from the turbines.

There was a substantial wind farm effect on the **prey species composition** of lesser weever which was mainly caused by two species: *J. herdmani* and *Gastrosaccus spinifer*. The Mysida, including *G. spinifer*, were dominant in the fish stomachs from both areas before the wind farm was constructed. Coates *et al.* (2016) also found high abundances (up to 42 ind/m² after construction) of *G. spinifer* in the soft sediment near the turbines, but only in the wind farm area and not in the control area.

The high number of *J. herdmani* encountered in the fish stomachs from the wind farm after construction was responsible for the community differences between the control and wind farm area after construction. The species is a highly available prey item in the wind farm with densities of more than 10 000 ind/m^2 on the turbines (De Mesel *et al.*, 2015) and up to 809 ind/m² in the soft sediment near the turbines (Coates *et al.*, 2013). Hyslop *et al.* (1980) and De Crespin *et al.* (2000) stated that the total abundance of prey items in a stomach depends on food availability and the prey digestion rate, but also that the hierarchical interactions among predators should be kept in mind. So, it is very likely that the dominance of *J. herdmani* in the diet of lesser weever originating from the wind farm, is due to the fact that this amphipod is a very abundant and thus easily accessible prey species in the soft sediment around the turbines. Furthermore, Lindeboom *et al.* (2011) and Vandendriessche *et al.* (2015) observed a decrease of lesser weever in the wind farm compared to the control area which may have led to a decrease in intraspecific competition and hence a relatively higher food availability. *Jassa herdmani* was also regularly found in the fish stomachs from the control area which might indicate that the wind farm effect is expanding into the surrounding area.

DAB (*LIMANDA LIMANDA*)

The percentage of **empty stomachs** of dab (*L. limanda*) was relatively low.

The **fullness index** showed some differences between control and impact area, before and after the construction of the wind farm. The fullness index was higher in the impact area - which is in accordance to our previous diet study (Derweduwen *et al.,* 2012) - and lower after the construction, both in impact and control area. However, none of these differences were significant. So, it can be stated that the observed differences in fullness index are not caused by the presence of the wind farm and the altered surrounding habitat.

The number of prey species and the **prey diversity** in the diet of dab was significantly higher in the impact area compared to the control area. Since this higher diversity in the impact area was already present before the wind farm was constructed, we may expect no overall wind farm effect. However, the non-parallelism in number of prey species - with a higher number of prey species in the impact area after construction and a lower number in the control area after construction - indicates that the wind farm does have an effect on species richness.

The following studies may help explain this increased diversity of the diet of dab. As a well-known ecosystem-engineer, the tube-building polychaete *Lanice conchilega* has the potential to increase habitat complexity and heterogeneity (Rabaut *et al.*, 2007; Van Hoey *et al.*, 2008). Both authors also indicated a significant and positive correlation between the macrobenthic abundance, diversity and biomass with increasing densities of *L. conchilega.* Furthermore, Coates *et al.* (2013) found increased densities of *L. conchilega* in the C-Power wind farm and demonstrated its dominance close to the turbines. So, the combination of the ecosystem-engineer capacities of *L. conchilega* and its increased density in the wind farm, might explain the higher diversity of the diet of dab observed in the wind farm.

A few other encountered taxa/species of which the distribution was limited to the wind farm area, were Hydrozoa and the dwarf swimming crab *Liocarcinus pusillus*, the former is a typical hard-substratum taxon (Kerckhof *et al.*, 2010; De Mesel *et al.*, 2013; De Mesel *et al.*, 2015) and the latter likes coarser sediments (Froglia and Manning, 1982) and was already found in the wind farm area (Derweduwen *et al.*, 2012). Two other typical hard-substratum species, *Phtisica marina* and *P. longicornis* (Kerckhof *et al.*, 2010; De Mesel *et al.*, 2013; De Mesel *et al.*, 2015) were hardly or not encountered in the stomachs, although they have been found previously (Derweduwen *et al.*, 2012).

The **diet** of dab strongly varied between areas (control and impact) and before and after construction. Most dietary studies of dab have classified the species as a general feeder with a relatively wide prey spectrum (Hinz *et al.*, 2005).

The **prey species composition** in the diet was significantly affected by the wind farm. Before the construction of the wind farm, a similar taxon composition in control and impact area could be observed but with different proportions. The appearance of Nematoda is notable and the taxon did not yet occur in our previous diet study (Derweduwen *et al.,* 2012). It is not clear whether the Nematoda are ingested preys or free-living parasites in the stomachs of dab. Significant differences between control and impact area could mainly be attributed to Nematoda and Brachyura. Also Mysida - only present in the control area - and Amphipoda - only present in the impact area - contributed to these differences. After the construction of the wind farm, a completely different picture emerged for the taxon composition in the control area, which mainly can be explained by the higher abundance of Nematoda before construction and of Cirripedia after construction. The taxon composition in the impact area after and before construction differed, particularly in altered proportions, i.e. more Brachyura before and more Amphipoda after construction. Differences between control and impact were particularly caused by higher abundances of Cirripedia - which also did not occur in our previous study (Derweduwen *et al.,* 2012) - in the control area and higher abundances of the Amphipoda *Nototropis swammerdamei* in the impact area. The presence of Amphipoda in the wind farm area was already described in Derweduwen *et al.* (2012). However, its presence has evolved into a dominance in the impact area after the construction of the wind farm.

Coates *et al.* (2013) already revealed altered macrofaunal communities in close vicinity to a wind turbine foundation. These altered macrofaunal communities can partly elucidate the observed changes in prey species composition in the diet of dab. However, the most abundant prey species in the stomachs from the wind farm area – *N. swammerdamei* - only occurred on the hard substratum of the foundations (De Mesel *et al.*, 2013) and did not on the soft sediment (Coates *et al.*, 2013). The second and third most important species in the wind farm area – *L. conchilega* and *J. herdmani* – did occur on the soft sediment (Coates *et al.*, 2013), the latter to a lesser extent more than 15 m away from the turbines. Due to the opportunistic feeding strategy of dab, this species can be expected to be highly adaptable in respect to habitat and ecosystem change (Hinz *et al.*, 2005) and may profit from the wind farm area as a new habitat with its associated fauna. It is most likely that dab not only was foraging on the soft sediment but also actively foraged on the hard substratum. Since several hard substratum species (i.e. *L. pusillus* and *N. swammerdamei*) were already found in the stomachs from the impact area before the wind farm was constructed, the occurrence of those hard-substratum species cannot exclusively be explained by the wind farm.

So, it seems that the feeding behaviour of dab is not only influenced by the presence of the wind farm. There are other factors playing in the impact area and in the control area, independently of the construction of the wind farm. However, the construction of the wind farm probably has enlarged the observed effects.

FUTURE RESEARCH

For future research, it is recommended to analyse a larger number of stomachs to increase the statistical power. For example, the wind farm effect for dab might have been proven statistically significant if the number of 'After Control' samples was higher. Secondly, we could not yet analyse the diet of whiting, another commercially important fish species, due to the limited number of individuals in the control area. The main reasons for these limitations were the limited number of beam trawl samples that could be taken within the foreseen ship time and the logistic problems encountered during sampling in and around the wind concession zones.

10.5. CONCLUSION

Lesser weever showed no significant differences in fullness index, between control and impact areas, nor before and after construction. Also the diversity of the diet was not affected by the presence of the wind farm. However, the diet composition did change: lesser weever consumed significantly more of the species *Jassa herdmani* both in the control and to a greater extent in the impact area and less of the mysid *Gastrosaccus spinifer* and Brachyura. This amphipod species *J. herdmani* is typically associated with hard substrates and was highly available in the wind farm (De Mesel *et al.*, 2015). This was the first record of *J. herdmani* to be found in the diet of a demersal fish species in this area.

The fullness index of dab also displayed no significant differences. Although the impact values were slightly higher, both before and after construction, the fullness index decreased after construction, both in impact and control areas. The diversity and the composition of the diet of dab were influenced by the presence of the wind farm. The number of prey species was higher in the impact than in the control area, after construction. Species that were responsible for these differences were *Nototropis swammerdamei, J. herdmani* and *Lanice conchilega.* The latter species is a well-known ecosystem-engineer with the potential to enhance habitat complexity and heterogeneity. Its presence might have led to a significant higher prey diversity in the wind farm and hence in the stomachs of dab. The prey species composition of dab was variable since dab is known as an opportunistic feeder (Hinz *et al.*, 2005). Still, some differences were observed that could be related to the presence of the wind farms. However, since some of the hard-substratum prey species found in the impact area did already occur before the wind farm was constructed, other factors must play a role.

To summarise, we can state that the observed differences in prey diversity and diet composition of the fish in the wind farm and the direct vicinity of the wind concession zone are clearly induced by the presence of the hard substrates of the wind farm and its associated fauna. Consequently, it can be stated that the reef effect, related to the introduction of hard substrates (and its attraction for associated hard sub fauna), is expanding into the surrounding soft sediments.

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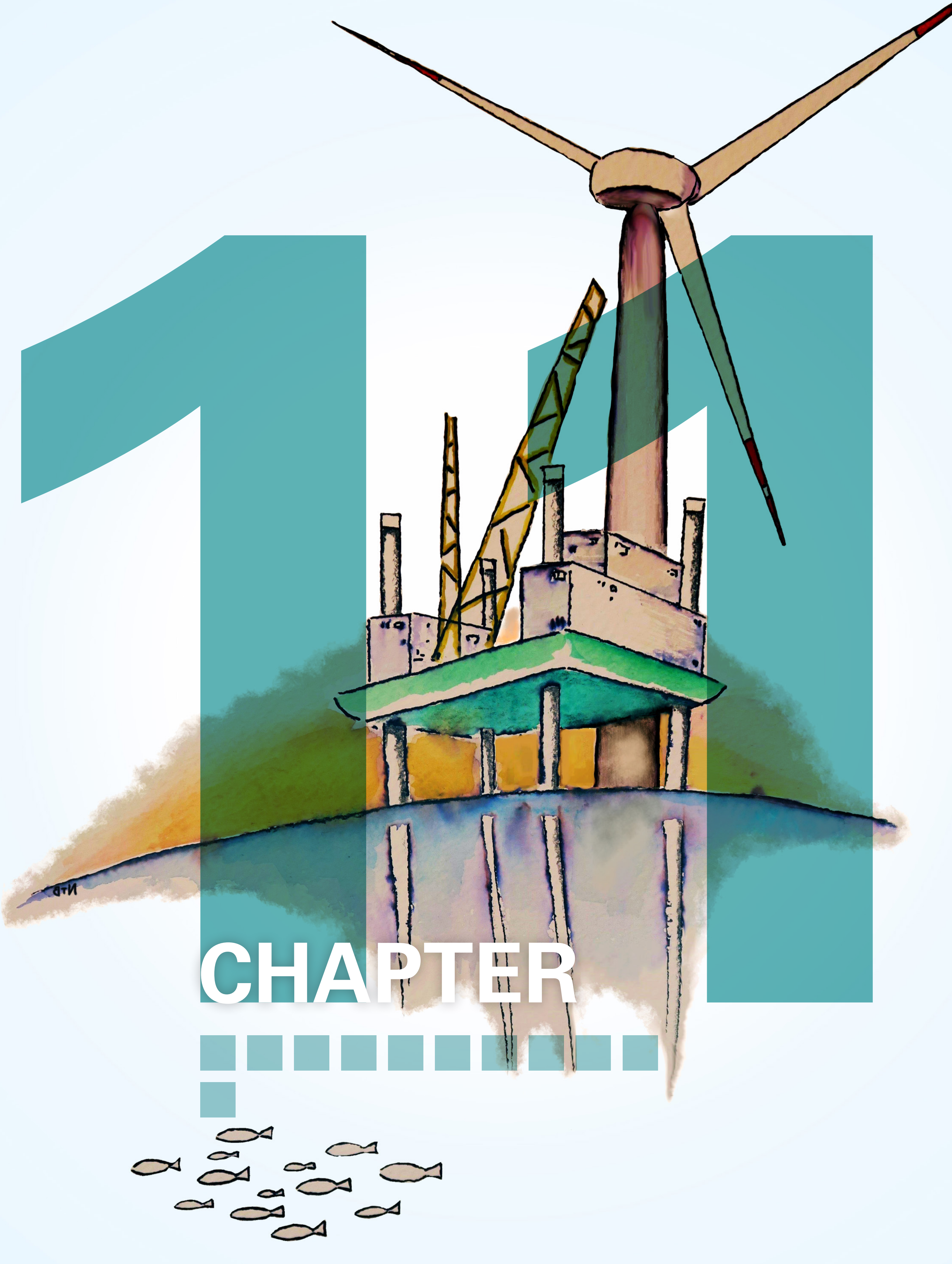

11

CHAPTER

CHAPTER 11

THE EFFECTS OF HIGH INTENSITY IMPULSIVE SOUND ON YOUNG EUROPEAN SEA BASS *DICENTRARCHUS LABRAX*, WITH SPECIAL ATTENTION TO PILE DRIVING

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*E. Debusschere[*1,2], K. Hostens[1], S. Vandendriessche[1], D. Botteldooren[3], M. Vincx[2] & S. Degraer[4]*

[1]Flanders Research Institute for Agriculture, Fisheries and Food (ILVO), Animal Sciences, Aquatic Environment and Quality, Ankerstraat 1, 8400 Oostende, Belgium.

[2]Ghent University, Biology Department, Marine Biology Section, Krijgslaan 281 S8, 9000 Ghent, Belgium.

[3]Ghent University, Department of Information Technology, Research Group Acoustics, Sint-Pietersnieuwstraat 41, 9000 Ghent, Belgium.

[4]Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management (MARECO), Gulledelle 100, 1200 Brussels, Belgium.

[*]Corresponding author: Elisabeth.Debusschere@vliz.be

ABSTRACT

Throughout the North Sea, a new anthropogenic sound source, pile driving, was recently introduced. It is the main method to install offshore wind farms (OWFs) and will regularly be used during the next couple of years. Pile driving generates strong impulsive noise that can affect the health and wellbeing of marine life. However, the exact impact, the underlying mechanisms and the ecological consequences of anthropogenic sound on marine life are not yet understood, especially for fish. This study investigated the impact of pile driving on young sea bass *Dicentrarchus labrax*. More specifically, the acute and delayed mortality, acute and chronic physiological stress responses and the impact of lower intensity impulsive sound on the fish behaviour were assessed through field and laboratory experiments. A field experiment at 45 m from the pile driving activity revealed no acute or delayed mortality but the fish showed strong acute secondary stress responses, a 50% decrease in oxygen consumption rate. This result could not be completely reproduced by two laboratory studies, indicating the importance of the frequency content in addition to the standard sound metrics for the physiological stress responses. Furthermore, juvenile fish reduced their swimming activity and ceased all aggressive attacks on conspecifics at the onset of the impulsive sound exposure, but showed behavioural recovery within 25 minutes. The results also showed that the initial response can change under repeated exposure. Based on these acute short-term effects, the ecological consequences of pile driving sound on fish health are probably subtle. More research on multiple species and at population level are required as well as long-term data, especially on behavioural responses, in order to decide on the ecological relevance of pile driving on young fish.

11.1. INTRODUCTION

More than 25 years ago, a relation between man-made (anthropogenic) sound and its negative effects on marine mammals was established (Simmonds and Lopezjurado, 1991). Since then, marine mammals have dominated the bioacoustics research, although recently the focus has widened to fish, and to a lesser extent, also to invertebrates (Southall et al., 2007, Slabbekoorn et al., 2010, Williams et al., 2015). Sound plays an essential role in conveying environmental information to marine fauna. Particularly in marine mammals, sound is known to play a key role in social and foraging behaviour. But of all vertebrates, fish exhibit the greatest diversity in hearing sensitivity and hearing structures and are a vital component in most ecosystem food webs (Popper and Fay, 2011). The main contributors to the anthropogenic sound energy in the North Sea are shipping, seismic surveys, underwater explosions and pile driving (Ainslie et al., 2009). The frequency range of man-made sound often overlaps with the hearing range of the fish. Consequently, underwater sound has the potential to cause auditory injuries, physiological stress and behavioural disturbance, and to mask biologically relevant sounds (Slabbekoorn et al., 2010). In addition, sound pressure can influence the swim bladder volume which can result in (mortal) internal injuries (Halvorsen et al., 2012b). So, the impact of anthropogenic sound on fish can range from immediate death to no impact at all. However, the exact impact, the underlying mechanisms and the ecological consequences of anthropogenic sound on marine life are not yet understood, especially for fish. In Europe, anthropogenic underwater noise was labelled as a pollutant within the Marine Strategy Framework Directive of the European Commission (Directive 2008/56/EC). Consequently, the impact of underwater sound on marine life, generated by various anthropogenic sound sources, need to be evaluated in order to take appropriate measures. Throughout the North Sea, a new anthropogenic sound source, pile driving, was recently introduced. It is the main method to install offshore wind farms (OWFs) and will regularly be used during the next couple of years. As OWFs are one of the options EU member states choose in order to achieve the renewable targets set by the Europe (Renewables Directive 2009/28/EC). Therefore, this PhD study took pile driving as the source of high intensity impulsive sound to study its impact on marine fish. Pile driving effects were assessed for young individuals of European sea bass *Dicentrarchus labrax*, a fish species with a closed swim bladder, so-called physoclists (Debusschere, 2016). The PhD started from the assumption made by a Dutch report in 2009 (Prins et al., 2009), which hypothesized a 100% mortality in fish eggs and larvae up to 1 km around a pile driving source. This assumption was based on modelled fish larvae distributions, mortality rate due to underwater explosions and back-calculated energy levels of underwater sound related to pile driving activities in the Dutch part of the North Sea.

This study had a multidisciplinary approach, aiming to disentangle the effects of impulsive sound (produced by pile driving) on young fish, thereby focusing on the following research questions:

(I) Are young fish (larvae and juveniles) affected by impulsive sound, what are the effects, and at what level do they manifest, *e.g.* mortality, stress responses or behavioural responses?

(II) Can the effects on young fish be linked to a specific sound-related metric or biological parameter? Can sound thresholds at which underwater sound negatively affects young fish be identified?

(III) What is the ecological significance of the observed effects?

(IV) How will the results from this PhD add to management and policy regulations in Belgium (and Europe), *i.e.* in order to minimise the environmental impact of pile driving activities in future offshore wind farms, and to achieve Good Environmental Status (GES) for Marine Strategy Framework Directive (MSFD) descriptor 11?

The study was published as a doctoral thesis and this report corresponds to the executive summary (Debusschere, 2016). Within the PhD framework, field and lab experiments were carried out in order to assess the impact of high intensity pile driving sound on acute and delayed mortality, acute and chronic physiological stress responses and the impact of lower intensity impulsive sound on the behaviour of young European sea bass. In addition, the critical sound parameters of physiological stress responses were studied in detail.

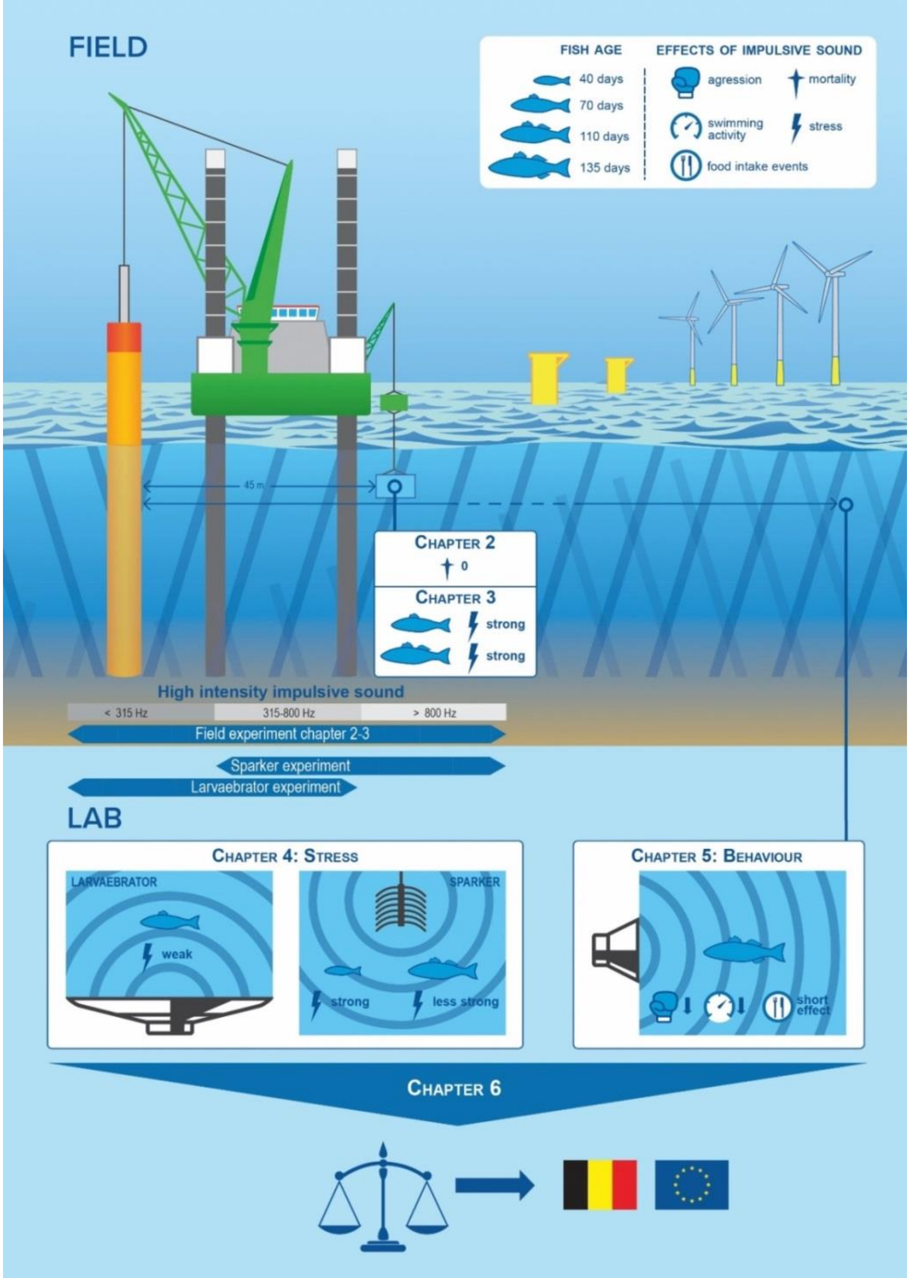

Figure 1. Schematic overview of the chapters of the PhD study comprising field and lab experiments with juvenile European sea bass, preceded by a general introduction (Chapter 1) and completed with a general discussion (Chapter 6) (Debusschere, 2016). Chapter 2 and 3 comprise the *in situ* experiment performed on board of the pile driving vessel assessing respectively, the impact on mortality and stress responses of young juvenile fish. Chapter 4 discussed the stress responses of the fish in two lab experiments with two high intensity sound sources whereas Chapter 5 used a lower intensity sound source in the lab to study fish behaviour. Figure taken from Debusschere (2016; availabe via www.vliz.be/nl/imis).

11.2. RESULTS

FIELD EXPERIMENTS: MORTALITY AND STRESS RESPONSES

An *in situ* experiment on board of a pile driving vessel was performed, addressing acute and delayed mortality of juvenile (68 and 115 days old) European sea bass (Debusschere et al., 2014). It was the first field study to assess fish mortality as close as 45 m from an offshore pile driving source over a complete pile driving session (Figure 1 - Chapter 2). Fish were exposed to 1739 up to 3067 pile driving strikes with a single strike sound exposure level (SEL_{ss}) between 181 and 188 dB re 1 µPa²·s, and a cumulative sound exposure level (SEL_{cum}) between 215 and 222 dB re 1 µPa²·s. No increased acute mortality was observed when we compared European sea bass (68 and 115 days old) exposed to pile driving with a control group exposed to ambient background sound levels in between the pile driving sessions. This study validates the results provided by other studies inside acoustically controlled chambers in the laboratory (Bolle et al., 2012, Halvorsen et al., 2012a, Casper et al., 2013a). Fish survival was further monitored in the lab for two weeks. At least under optimal laboratory conditions, we observed no delayed mortality caused by pile driving. This study rejected the 100% mortality hypothesis as stated by a Dutch report in 2009 (Prins et al., 2009). Moreover, if internal injuries were present, they were shown not to be mortal.

A second aim of the *in situ* experiment was to assess the physiological stress response of juvenile sea bass (68 and 115 days old) to high intensity sound produced by pile driving (Figure 1 - Chapter 3). So far, this was not yet studied. The primary, secondary and tertiary stress responses were investigated during and after exposure to a complete pile driving session (Debusschere et al., 2016). As a primary stress response proxy, whole-body cortisol seemed to be too sensitive to 'handling' bias (Figure 2). However, a strong secondary stress response to pile driving was detected as significant reductions in oxygen consumption rate (49 – 55%) and low whole-body lactate concentrations. In contrast to fish used on the first day of the trip (monopile 1), the fish used on the second day (monopile 2) had already been indirectly exposed to pile driving. Fish in the control group of that second day reduced their respiration by 34 to 40% compared to the control group on the first day. This may be indicative of a prolonged stress response or increased sensitivity towards new stressors. A tertiary stress response only manifests when homeostasis cannot be re-established. After 30 days in the laboratory, specific growth rate and condition of the exposed fish were not affected compared to unhandled fish, so a tertiary stress response was absent. Only a short-term reduction in metabolic rate was demonstrated while the long-term consequences of repeated impulsive sound exposure for fish in the field are yet to be determined.

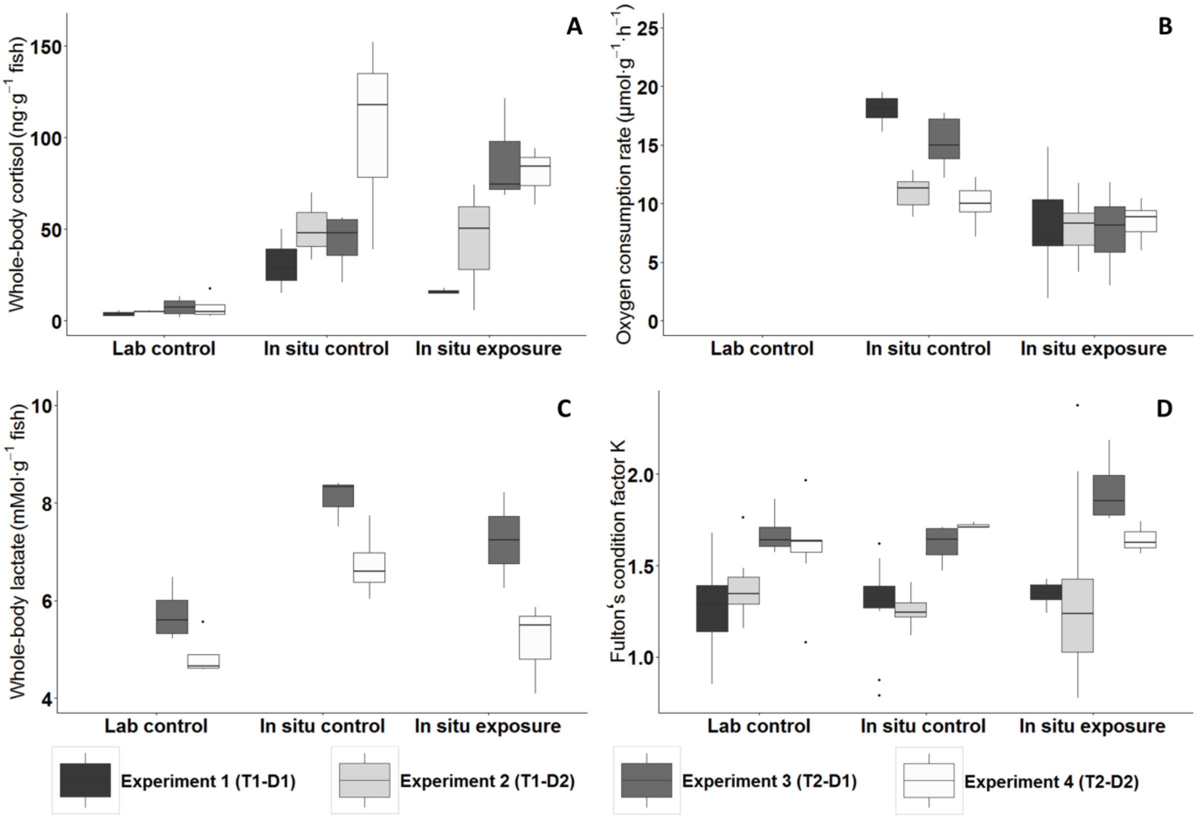

Figure 2. Stress responses of juvenile European sea bass *Dicentrarchus labrax*, based on four experiments (trip 1-2; day 1-2) and three treatments each: no handling/no exposure (lab control), exposed to ambient sound (*in situ* control), and exposed to impulsive sounds during a complete pile driving session at 45 m from the pile driving activity (*in situ* exposed). Fish were 68 days old (dph) at the start of experiments 1 and 2 and 115 dph in experiments 3 and 4. (A) Whole-body cortisol (ng·g^{-1} fish); (B) Oxygen consumption rate (µmol·g^{-1}·h^{-1}); (C) Whole-body lactate (mMol·g^{-1} fish, no data for experiments 1 and 2); (D) Fulton's condition factor K measured after 30 days. Figure taken from Debusschere et al. (2016).

LINKING LABORATORY TO FIELD OBSERVATIONS: STRESS RESPONSES

The critical sound parameters responsible for the acoustic physiological stress response observed in the field experiment needed to be explored further. Therefore, the primary and secondary stress responses of larval and juvenile European sea bass to strong impulsive sound were compared between two lab experiments using different sound sources (SIG sparker and larvaebrator) (Debusschere et al., submitted). These results were then compared with the stress responses measured during an *in situ* pile driving study (Figure 1 - Chapter 4) (Debusschere et al., 2016). Both lab sound sources produced similar levels at maximum energy for the standard sound pressure metrics as the *in situ* pile driving, being zero-to-peak sound pressure level ($L_{z\text{-}p}$) of 208 dB re 1 µPa), SEL_{ss} of 181 dB re 1 µPa²·s and SEL_{cum} of 214 dB re 1 µPa²·s. However, the three sources differed in their sound frequency spectra (Figure 3). The whole-body cortisol results (a proxy for primary stress responses) confirmed the susceptibility of both juvenile and larval fish to handling stress. Still, the increased (or altered) whole-body cortisol levels indicated that high intensity impulsive sound evoked an acoustic primary stress response (Figure 4A-C). Common ground between the field and two lab experiments was found at the high energy levels (SEL_{ss}) produced between the 315 and 630 Hz 1/3 octave bands (Figure 3). This frequency range covers the responsiveness range of European sea bass to sound, relating the primary stress response in juvenile sea bass to hearing. Reduced oxygen consumption rates of ~50% were observed in the juveniles in the field experiment and larvae in the sparker experiment (max. exposure), and to a lesser extent in the juveniles of the sparker experiment (Figure 4D-F). Consequently, the secondary stress response can most likely be linked to high intensity sound produced at higher frequencies (>800 Hz), above the responsiveness range of European sea bass. This secondary stress response may be associated with the pressure induced swim bladder oscillations. It may be clear that high intensity impulsive sound must cover a broad frequency range (similar to a real *in situ* pile driving) to evoke strong secondary stress responses, such as reduced oxygen consumption rate and reduced whole-body lactate levels in juvenile sea bass (Figure 4G-I). This implies that lab results can not directly be translated to the real world, as some known (like frequency content) and unknown parameters may not be comparable. More studies on different life stages and on the role of non-standard sound parameters - such as particle motion - are needed to further clarify the triggering parameters and sound thresholds of the stress response of fish.

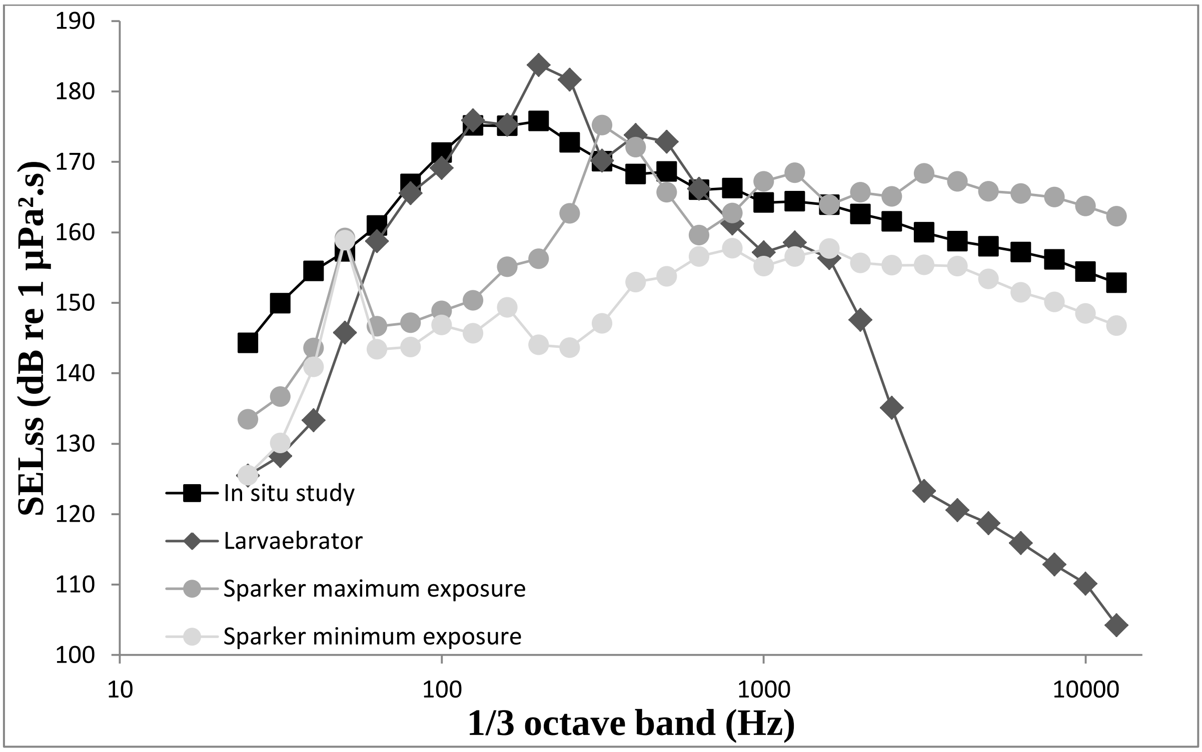

Figure 3. The measured frequency spectra of the sparker and larvaebrator experiments compared to the *in situ* experiment, showing the mean single strike sound exposure levels (SEL_{ss}) in the 1/3 octave bands (SD not shown for reasons of comparison). Figure taken from Debusschere et al. (submitted).

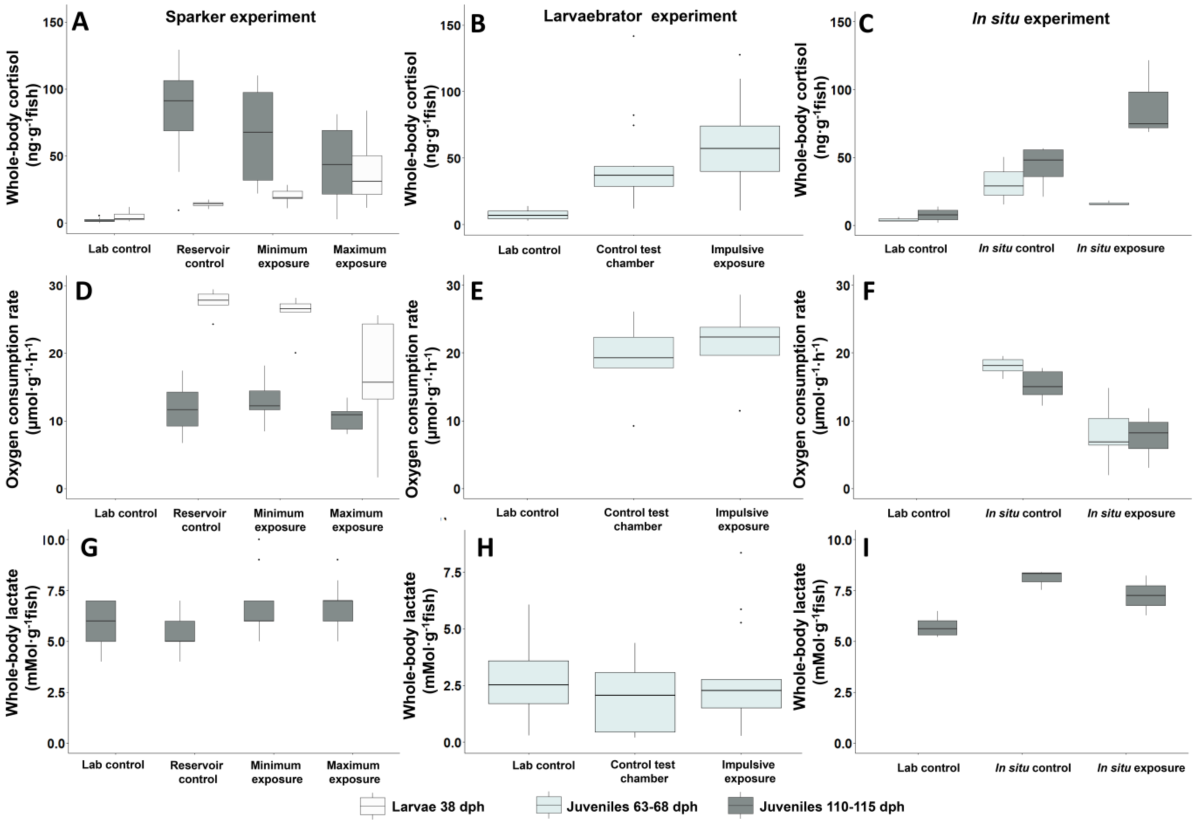

Figure 4. Biochemical and physiological stress responses of fish to high intensity impulsive sound in two lab experiments (sparker and larvaebrator experiment) and one field experiment (*in situ* experiment adapted from Debusschere et al. (2016). The experiments were performed with European sea bass larvae (38 dph, sparker experiment) and juveniles (110 dph in sparker experiment, 63 dph in larvaebrator experiment, 68 and 115 dph in *in situ* experiment). (A-C) whole-body cortisol levels, (D-F) Oxygen consumption rate, (G-I) whole-body lactate levels. The Box-and Whisker plots represent the median between the 25 and 75% percentiles of the box, outliers are plotted as individual points. Figure taken from Debusschere et al. (submitted).

BEHAVIOURAL RESPONSES

At a larger scale, underwater sound has the potential to disturb the behaviour of fish even at lower sound pressure levels, resulting in a much wider impact range around the pile driving source than high sound pressure levels (Slabbekoorn et al., 2010). Since functionally important behaviour, such as social interactions and foraging, can contribute significantly to the survival and reproduction of fish, any impact on functional traits can directly be translated into fitness consequences. However, so far only a couple of studies have tested the acute impact of anthropogenic sound exposure on fish behaviour (Purser and Radford, 2011, Voellmy, 2013, Voellmy et al., 2014a, Voellmy et al., 2014b, Shafei Sabet et al., 2015). Consequently, fish behaviour was studied in response to impulsive sound on three consecutive days in a laboratory set-up (Figure 1 - Chapter 5) (Debusschere et al., in prep.). In this laboratory study, we tested the influence of pile driving sound on the swimming activity and aggressive behaviour of young juvenile European sea bass *Dicentrarchus labrax* before, during and immediately after the 25 min sound exposure period (1000 strikes, SEL_{ss} =146 dB re 1 µPa²·s, L_{z-p} = 165 dB re 1 µPa; SEL_{cum} = 176 dB re 1 µPa²·s). We also tested the impact on feeding tendency and efficiency of fish when they were already exposed to the impulsive sound for 15 minutes. Juvenile sea bass interrupted their swimming activities and ceased any aggressive actions to conspecifics at the onset of the impulsive sound exposure. These behavioural effects returned to the pre-exposure baseline within the 25 minute exposure period. On the first day, a slightly reduced number of food intake events were observed during and after the sound exposure, which can indicate an attention shift induced by the sound exposure (Figure 5). This attention shift was no longer clearly observed during the two following days of the experiment. Feeding efficiency was not affected by the sound exposure and illustrated that sea bass were alert to external stimuli under impulsive sound exposure. These findings indicate that the initial response does not persist but can progress over time or under repeated exposure. It remains to be tested whether this also applies to wild-ranging fish.

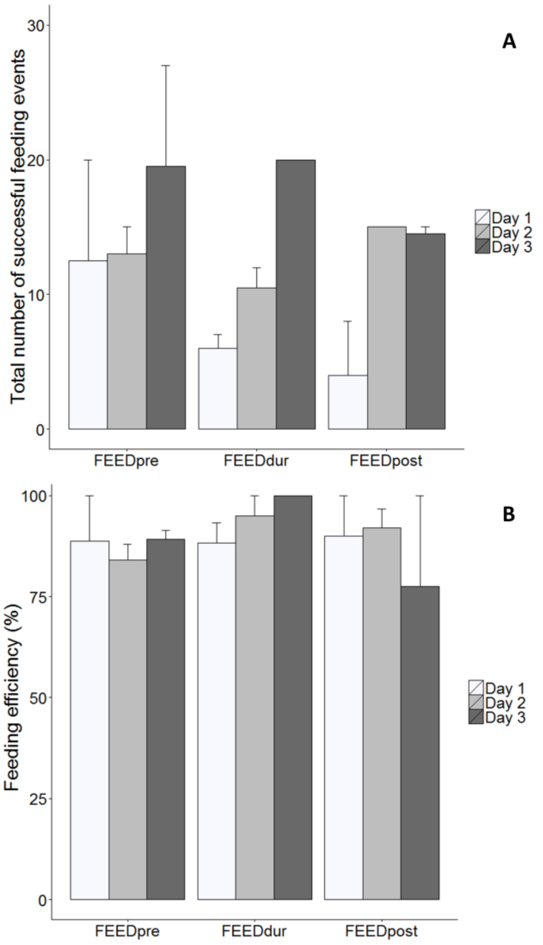

Figure 5. Feeding behaviour of the focal fish for each feeding moment, i.e. before (FEEDpre), during (FEEDdur) and after (FEEDpost) sound exposure on three consecutive days. (A) Total number (sum ± SE) of successful feeding events in 10 minutes after food was offered. (B) Feeding efficiency (%) during 10 minutes after food was offered (mean % ± SE). Figure taken from Debusschere (2016).

11.3. DISCUSSION

THE EFFECTS OF IMPULSIVE (PILE DRIVING) SOUND ON YOUNG FISH (LARVAE AND JUVENILES)

The results of both field and lab experiments allowed to answer research question I on the specific impact of high intensity or strong impulsive pile driving sound on European sea bass. Exposure to a complete pile driving session as close as 45 m from a pile driving activity did not result in acute or delayed mortality of juvenile European sea bass. Both our data and the laboratory results performed by other researchers in acoustically controlled chambers (*i.e.* the larvaebrator and the High Intensity Controlled Impedance Fluid filled wave Tube, HICI-ft) strongly contest the assumption of 100% mortality of fish larvae in a range of 1 km around the pile driving activity (Prins et al., 2009, Bolle et al., 2012, Casper et al., 2012, Halvorsen et al., 2012a, Halvorsen et al., 2012b, Casper et al., 2013a, Debusschere et al., 2014, Bolle et al., submitted). It did lead to a strong physiological stress response limited to a relative short period of time, which can be extended by multiple sound exposures (Debusschere et al., 2016). Based on the field and lab results, the physiological stress responses found in larvae and juveniles could be related to the standard sound metrics (SEL_{ss}, SEL_{cum} and L_{z-p}) and the frequency range in which the highest energy was found. Furthermore, the primary and secondary stress response could be related to hearing and swim bladder oscillations, respectively (Debusschere et al., submitted). The studies mentioned above involve high intensity underwater sound found at close range from the pile driving source. At larger distances from the pile driving source, the impulsive sounds contain less energy but can still induce a behavioural response in juvenile European sea bass at the onset of the sound exposure. During the sound exposure, European sea bass were able to recover from the initial stress response, and repeated exposure had no clear effect on feeding (Debusschere et al., submitted). Combining these results with other data from literature reveals the interspecific variability of fish in their behavioural response to the same type of stressor (Voellmy et al., 2014a, Shafei Sabet et al., 2015). More species with varying life history strategies need to be studied before the results can be generalised with confidence.

PROPOSING SOUND THRESHOLDS FOR THE EFFECTS ON YOUNG FISH

To provide an answer on **research question II**, the study results are integrated with current knowledge (Popper et al., 2014). This allows us to make suggestions regarding sound thresholds for mortality, physiological stress and behavioural changes of young physoclistous fish. Since mortality was absent in our field study, the mortality threshold must lie above the measured sound parameters (SEL_{ss}> 188 dB re 1 µPa²·s; SEL_{cum}> 222 dB re 1 µPa²·s.; L_{z-p} > 210 dB re 1 µPa) (Debusschere et al., 2014). This study is the first to propose a sound threshold range at which physiological stress responses in juvenile fish are evoked: high-intensity impulsive sound need to have at least a SEL_{ss}

of 170 to 181 dB re 1 µPa²·s at frequencies higher than 315 Hz to evoke physiological stress (Debusschere et al., submitted). A threshold for behavioural disturbance linked to pile driving cannot yet be determined.

THE ECOLOGICAL SIGNIFICANCE OF THE OBSERVED EFFECTS

Additionally, consequences on an ecological level need to be evaluated (research question III). In other words, effects on an individual level need to be scaled up to population level, since individual effects in fish are subordinate to population effects from an ecological point of view (Bejder et al., 2009). In order to do so, data on the presence of sound sources, sound propagation, individual impact, population size, distribution, and affected (sub)population are needed before the individual effect can be modelled into a population effect. This is not yet possible for fish, but given the results about the effects found on individual fish, it can be assumed that the ecological consequences of pile driving sound on fish health are subtle (Bolle et al., 2012, Halvorsen et al., 2012a, Casper et al., 2013b).

REVIEWING EUROPEAN MANAGEMENT AND POLICY REGULATIONS WITH RESPECT TO THE STUDY RESULTS

Evaluating the European and national legislation on man-made underwater sound is necessary to provide adequate advice to minimise the impact of pile driving activities on the marine environment (research question IV). In Europe, the Marine Strategy Framework Directive (MSFD) defined a Good Environmental Status (GES) in which underwater sound needs to be at levels that do not adversely affect the ecosystem (Directive 2008/56/EC; Descriptor 11). A Technical Subgroup Noise (TSG Noise) has been commissioned to further develop the descriptor on underwater noise (Van der Graaf et al., 2012, Dekeling et al., 2014). This subgroup proposed the establishment of a sound register, to log all sound producing human activities. The subgroup also identified 'considerable displacement' of marine organisms as the most relevant impact of impulsive sound. Finally, an inventory of the pulse-block days in the EU regional seas can be modelled. This is based on the presence of anthropogenic sound sources that are producing sound levels above the threshold linked to the 'considerable displacement' in ¼ ICES rectangles, which are intervals of 30' (longitude) and 1° (latitude) over an area between 36°N and 85°30'N and 44°W and 68°30'E. A GES should be applicable to all marine organisms, while the TSG Noise mainly based its advice on marine mammals, whereas 'considerable displacement' may not be the most relevant impact on fish. Fish are also neglected in the national legislation of the EU Member States (JNCC, 2010, Betke, 2014, Dähne et al., 2014, Rumes et al., 2015, RWS, 2015). Based on this PhD, our management advice is that the effects of pile driving sound on fish are considered to be more subtle than anticipated and no stringent measures are needed ad hoc in Belgium or in other member states (Rumes et al., 2015). However, more research is needed to support or reject the decision to exclude fish from management, thereby still ensuring GES for all marine fauna.

FUTURE PERSPECTIVES

Finally, future research targets were identified to further unravel the impact of pile driving sound on fish which are needed to progress towards an acoustically sound approach. The lack of particle motion data remains a big gap and needs to be addressed by future studies. The underlying critical sound parameters that evoke physiological stress and behavioural responses in fish need to be unravelled further. Furthermore, data is needed on the long-term impact of acoustic stressors in order to model the ecological consequences of pile driving at population level. Studying the fish in their natural environment with new technologies is a promising strategy. Finally, the impact of continuous sound that will be produced for the next 20 years of the operational OWFs on fish health need to be addressed.

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12

CHAPTER

CHAPTER 12

SEABIRD MONITORING AT OFFSHORE WIND FARMS IN THE BELGIAN PART OF THE NORTH SEA

UPDATED RESULTS FOR THE BLIGH BANK & FIRST RESULTS FOR THE THORNTONBANK

N. Vanermen[*], W. Courtens, M. Van de walle, H. Verstraete & E.W.M. Stienen

Research Institute for Nature and Forest, Kliniekstraat 25, 1070 Brussels (Anderlecht), Belgium.

[*] *Corresponding author: Nicolas.Vanermen@inbo.be*

ABSTRACT

Since 2005, the Research Institute for Nature and Forest (INBO) performs monthly BACI-designed surveys to study seabird displacement following the construction of offshore wind farms in the Belgian part of the North Sea. For the first time since its completion in 2013 we report our findings for the C-Power wind farm at the Thorntonbank, and we also give an update of the results for

the Bligh Bank wind farm after five years of post-impact monitoring.

Compared to earlier reports and publications, we introduced some improvements in our modelling strategy.

To correct for decreasing detectability with distance, the seabird numbers observed were now distance-corrected, and by allowing the detection functions to vary with wind force or wave height, temporal variation due to observation conditions was further reduced. We also included a fishery factor in the model, allowing to correct for the presence of beam trawlers in the vicinity of our survey tracks. As expected, this factor often explained a significant part of the variation in the counted numbers of gulls and northern fulmars.

Based on the resulting impact models, we found significant avoidance by northern gannet and common guillemot at both sites. Common guillemot decreased in densities by 68% and 75% at the Thorntonbank and Bligh Bank respectively, and northern gannet by 99% and 82%. Razorbill decreased in numbers at the two sites, this decrease being significant at the Bligh Bank only (67%). Both sites attracted great black-backed gulls, this species having increased in numbers significantly by a factor 6.4 and 3.6 at the Thorntonbank and Bligh Bank respectively. The previously reported attraction effects of lesser black-backed gull and herring gull at the Bligh Bank were confirmed after two more years of monitoring, but no such effect was observed at the Thorntonbank. Finally, Sandwich tern appeared to be attracted to the offshore wind farm at the Thorntonbank, this effect being significant only for the buffer zone. This is in line with the results for the phase I of the C-Power wind farm when we also found attraction of Sandwich tern to the immediate surroundings of the six turbine wind farm.

While the avoidance of common guillemot and northern gannet seems readily interpretable from a disturbance perspective, it is still difficult to pinpoint the observed increases in seabird numbers, even more so because these are not always consistent between both sites under study. Gaining more insight in the diurnal and tidal-dependent variation in numbers and behaviour of birds occurring inside the offshore wind farms seems indispensable for understanding the observed patterns and learning whether birds come to the wind farms merely for roosting and the related stepping stone function, or whether offshore wind farms also offer increased food availability. This should be investigated through oriented research making use of bird radar data, GPS tracking data of tagged gulls, fixed cameras and/or visual observations from a fixed location inside the wind farm.

12.1. INTRODUCTION

In order to meet the targets set by the European Directive 2009/28/EG on renewable energy, the European Union is aiming at a total offshore wind farm (OWF) capacity of 43 GW by the year 2020. Meanwhile, the offshore wind industry is growing fast and at the end of 2015, 3,230 offshore wind turbines were fully grid-connected in European waters, totalling 11.0 GW (EWEA 2016). Currently, three offshore wind farms are operational in the Belgian part of the North Sea (BPNS). In 2008, C-Power installed the first six wind turbines (30 MW) at the Thorntonbank, located 27 km offshore, followed by the construction of 48 more turbines in 2012 and 2013 (295 MW). In 2009-2010, Belwind constructed 55 turbines (165 MW) at the Bligh Bank, 46 km offshore. Located in between these two wind farms, in 2013 Northwind NV built 72 more turbines at the Lodewijckbank, 37 km offshore.

Since 2005, the Research Institute for Nature and Forest (INBO) performs seabird counts specifically aimed at studying seabird displacement caused by the presence of offshore wind turbines. Due to logistic constraints, the study effort was concentrated on the Thorntonbank and Bligh Bank wind farms only. Here we present the results of our seabird displacement study at the respective OWFs after 3 and 5 years of operation.

12.2. METHODS

SEABIRD COUNTING

Ship-based seabird counts were conducted according to a standardized and internationally applied method, combining a '**transect count**' for birds on the water and repeated '**snapshot counts**' for flying birds (Tasker *et al*. 1984). The focus is on a 300 m wide transect along one side of the ship's track. While steaming, all birds in touch with the water (swimming, dipping, diving) located within this transect are counted ('transect count'). Importantly, the distance of each observed bird (group) to the ship is estimated, allowing to correct for decreasing detectability with increasing distance ('distance correction') afterwards. The transect is therefore divided in four distance categories (A = 0-50 m, B = 50-100 m, C = 100-200 m & D = 200-300 m). Counting all flying birds crossing this same transect, however, would cause an overestimation and would be a measure of bird flux rather than actual bird density. The birds' flying speed is significantly higher than the ship's movement, and therefore more birds will be flying through the surveyed area in the course of any observation period, compared to numbers present at any one instance (Tasker *et al.* 1984). Flying birds are therefore counted by performing instantaneous counts in one minute intervals ('snapshot counts') within a quadrant of 300 by 300 m inside the transect. As the ship covers a distance of approximately 300 m per minute (when sailing the prescribed speed of 10 knots), the full transect length is covered by means of these subsequent 'snapshots'. Afterwards, observation time is linked to the corresponding GPS-coordinates saved by the ship's board computer. Taking in account the transect width and distance travelled, the combined result of a transect and snapshot count can be transformed to a number observed per km^2, i.e. a seabird density at a specified location. Up to 2012, observations were aggregated in ten-minute bouts, which were cut off to the nearest minute at waypoints. Since 2013, resolution is increased and seabird observations are pooled in two-minute bouts, again cut off to the nearest minute at waypoints.

In practice, we count all birds observed, but those not satisfying above conditions (i.e. not occurring in the transect nor during snapshots) are given another code and are not included in the density analyses afterwards.

We also record as much information as possible regarding the birds' age, plumage, behaviour, flight direction and association with objects, vessels or other birds.

MONITORING SET-UP

Monitoring was performed according to a **Before-After Control-Impact** (**BACI**) set-up. Both wind farm areas were surrounded by a buffer zone of 3 km to define the 'impact area', being the zone where effects of the wind farm on the presence of seabirds can be expected. Next, a comparably large control area was delineated, harbouring comparable numbers of seabirds before OWF construction, and showing a similar range in water depth and distance to the coast. The distance between control and impact areas was kept small enough to be able to survey both on the same day by means of a research vessel (RV).

Following fixed monitoring tracks, the Thorntonbank study area was counted on a highly regular basis from 2005 until present, while the Bligh Bank study area was studied from April 2008 to April 2015 (Figures 1, 2 & 3). During this dedicated monitoring program both sites should have been visited monthly, but research vessels were not always available and planned trips were sometimes cancelled due to adverse weather conditions (significant wave heights above 2 m and/or poor visibility). Before this dedicated monitoring program, the sites were counted on a much more irregular basis, but we did include surveys dating back to 1993 provided that the control and impact area were visited on the same day.

Table 1. Definition of the reference, construction and impact periods at the Thorntonbank and Bligh Bank study areas as applied in the impact analyses.

OWF	Phase	Period
Thorntonbank	Reference period	< 04/2008
	1st construction period	04/2008 –> 05/2009 (highly restricted access)
	Impact period (phase I)	06/2009 –> 04/2011 (6 turbines)
	2nd construction period	05/2011 –> 09/2012 (variable access)
	Impact period (phase I, II & III)	10/2012 -> present (54 turbines)
Bligh Bank	Reference period	< 09/2009
	1st construction period	09/2009 –> 09/2010 (highly restricted access)
	Impact period (phase I)	10/2010 –> present (55 turbines)

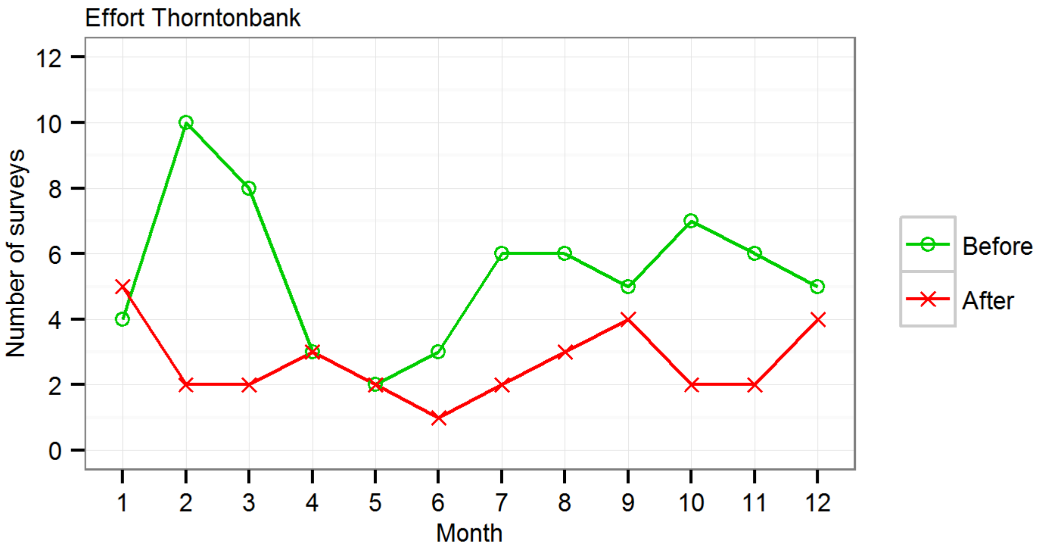

Figure 1. Count effort in the Thorntonbank study area with indication of the number of surveys performed before the construction of the phase I turbines (<04/2008), and after the construction of the phase II & III turbines (>09/2012).

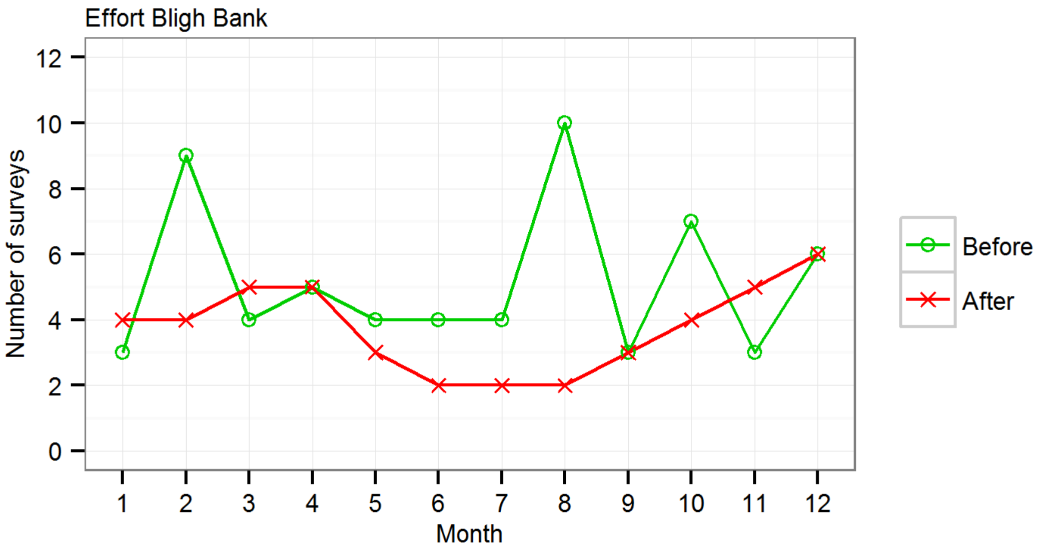

Figure 2. Count effort in the Bligh Bank study area with indication of the number of surveys performed before (<09/2009) and after (>09/2010) the construction of the turbines.

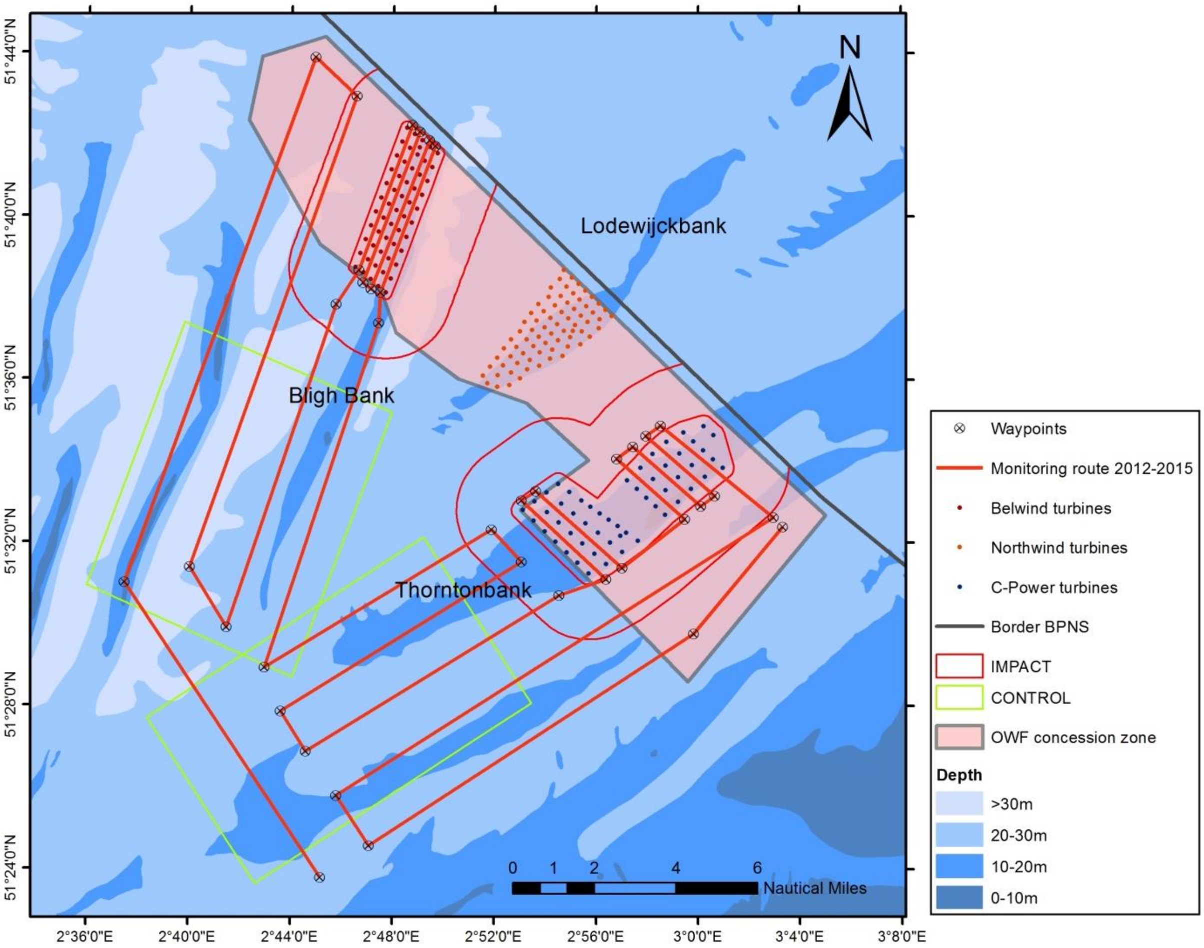

Figure 3. Monitoring route through the OWF study area in the period 2012-2015.

OFFSHORE WIND FARMS

The two wind farms under study were the C-Power wind farm at the Thorntonbank and the Belwind wind farm at the Bligh Bank (Figure 3).

The Thorntonbank wind farm is located 27 km off the coast of Zeebrugge, and consists of 2 subareas of respectively 24 and 30 wind turbines (see Figure 3), measuring 10.7 and 9.2 km² and with a water depth between 12 and 27.5 m (C-Power 2016). The distance between the turbines ranges from 500 up to 800 m.

The wind farm was built in three phases:

- Phase 1: 6 x 5 MW turbines (gravity-based foundations), operational since May 2009
- Phase 2: 30 x 6.15 MW turbines (jacket foundations), operational since October 2012
- Phase 3: 18 x 6.15 MW turbines (jacket foundations), operational since September 2013

The wind farm at the Bligh Bank is located 46 km off the Belgian coast. It has an area of 17 km² with a water depth range of 15 to 37 m. The farm consists of 5 rows of eleven 3 MW turbines (with 500 – 650 m distance in between) and a transformation platform, all of which were installed on steel monopile foundations (Belwind 2016). The first construction activities took place in September 2009, and the wind farm became fully operational in December 2010.

DISTANCE ANALYSIS

Before performing impact analyses we corrected the numbers of seabirds observed on the water for decreasing detection probability with distance to the ship (Buckland *et al*. 2001, Thomas *et al*. 2010). Detection probability is further likely to depend on group size and observation conditions (Marques & Buckland 2003). Observation conditions were included in the detection models as ‘wind force’ (beaufort scale) or ‘wave height’ (categorized as 0-0.5m / 0.5-1.0m / 1.0m-2.0m / 2.0-3.0m, …), both being estimated at the time of observation.

We fitted half-normal and hazard-rate detection functions to our data. Adding cosine or polynomial adjustments in the presence of group size as a covariate often resulted in non-monotonic detection functions (implying that detection probability would increase with increasing distance which is assumed not very plausible) and these adjustments were therefore no longer considered. We thus fitted following ‘full models’ with a non-adjusted half-normal and hazard-rate detection function:

- group size + wind force
- group size + wave height
- log(group size) + wind force
- log(group size) + wave height

The best fitting full model was chosen based on the ‘Akaike Information Criterion’ (AIC), and backward model selection was applied to refine the detection function. In the end, this distance analysis resulted in species-specific detection probabilities varying with the selected covariates, and observed numbers were corrected accordingly.

BACI ANALYSIS

For the BACI analysis we aggregated our count data per area (control / impact) and per monitoring day, resulting in day totals for both zones, thus avoiding auto-correlation between subsequent counts and minimizing overall variance. We only selected days on which both the control and impact area were visited, minimizing variation resulting from short-term temporal changes in seabird abundance. When a counted subject is randomly dispersed, count results tend to be Poisson-distributed, in which the mean equals the variance (McCullagh & Nelder 1989). Seabirds, however, often occur strongly aggregated in (multi-species) flocks, typically resulting in count data with a high proportion of zeros, relatively few but sometimes very large positive numbers and a high variance exceeding the mean, resulting in high over-dispersion. Such count data can be analyzed through a generalized linear model with a negative binomial (NB) distribution (Ver Hoef & Boveng 2007, Zuur *et al*. 2009). When data appeared to exhibit (much) more zeros than can be predicted by a Poisson or NB distribution, zero-inflated (ZI) models were used (Potts & Elith 2006, Zeileis *et al.* 2008), which consists of two parts: (1) a ‘count component’ modelling the data according to a Poisson or NB distribution and (2) a ‘zero component' modelling the excess in zero counts. In ZI models, the zero-component was limited to an intercept.

Our **response variable** equals the number of birds observed (inside the transect and during snapshot counts) per survey in the control or impact area. To correct for varying monitoring effort, the number of km² counted was included in the model as an offset-variable. The **explanatory variables** used were (i) an area factor CI (Control / Impact area), (ii) a time factor BA (Before / After construction), (iii) an offshore wind farm factor OWF (wind farm present / absent) and (iv) a fishery factor (fishing vessels present / absent). For the latter we only considered fishing vessels observed within a distance of 3 km from the monitoring track. Finally, the continuous variable 'month' was used to model seasonal fluctuations by fitting a cyclic smoother or a cyclic sine curve, the latter described by a linear sum of sine and cosine terms (Stewart-Oaten & Bence 2001, Onkelinx *et al*. 2008). Seasonal patterns can often be modelled applying a single sine curve with a period of 12 months, but sometimes even better by adding another sine curve with a period of 6 or 4 months, thus allowing to model more than one peak in density per year or an asymmetric seasonal pattern. During the process we considered five different possibilities for explaining seasonal variation in numbers:

1. Intercept model (no seasonal variation)
2. 12 month period sine curve
3. 12 + 6 month period sine curve
4. 12 + 4 month period sine curve
5. Cyclic smoother

At first, all 5 full models (above sine curves and smoother added with the aforementioned factors, but without interactions) were fitted using different distributions (Poisson, NB, ZI Poisson, ZI NB). Based on the resulting AIC values, the best fitting distribution was selected. Next, all possible models nested within the 5 full models were fitted applying the selected distribution. Based on the resulting AIC matrix the most likely factor-seasonality combination was chosen. Note that for each species and each OWF, three different analyses were performed based on three different impact datasets (impact + 0.5 km, impact + 3 km, buffer 0.5-3 km, see Figures 4 & 5). In most cases, the same covariate combination resulted in the lowest AIC for all 3 data selections, and in all cases, at least 2 out of 3 datasets favoured the same factor combination. Whatever the outcome, the most favoured covariate combination was applied over all 3 datasets to estimate the OWF displacement effect. When the best-fitting model did not contain the OWF factor, this was added to the model afterwards in order to estimate its effect.

In the results section (§3) we often refer to (i) the OWF coefficient, being the model coefficient for the OWF factor variable and an estimator of the displacement effect, and (ii) the estimated density, being the model prediction for a specific month and BA / CI factor combination, with the offset variable set to 1 km².

At the Thorntonbank we encountered a specific situation. The corridors between the C-Power turbines used for seabird monitoring vary in width between 650 and 850 m. For security reasons, the research vessels aim to sail right in the middle of these corridors, implying that the turbines and associated birds are always just outside our 300 m wide count transect, and are not included in the impact analysis. Therefore, we also analysed an adjusted response variable for species very often observed roosting on the jacket foundations (herring, lesser black-backed and great black-backed gull). This response variable is calculated by adding (i) the number of birds that should have been counted inside the transect if the turbine-associated birds would have occurred homogenously spread across the area to (ii) the actual number of birds counted inside the transect (assuming this number is representative for the whole area). This is best illustrated with an example: at 28/08/2015 we counted no less than 161 great black-backed gulls resting on the jacket foundations, and merely 1 bird was observed inside our transect, despite a survey effort of 7.4 km² inside the impact area. As we checked 43 turbines out of a total of 54 turbines, we estimate the number of great black-backed gulls associated with turbines in the Thorntonbank OWF as a whole at 202 birds. The wind farm area surrounded by a 500 m wide buffer zone measures 36 km², and the density of turbine-associated great black-backed gulls in this area is thus 5.6 birds/km². Assuming these birds would have occurred homogenously spread across the area, and knowing we counted 7.4 km², we thus recalculate the number of birds inside the transect as: 1 + (5.6*7.4) ≈ 42. The original and recalculated response variable are always analysed both, and the difference is clearly indicated in the graphs and tables.

BACI modelling was performed for thirteen seabird species occurring regularly in the wind farm areas, i.e. northern fulmar (*Fulmarus glacialis*), northern gannet (*Morus bassanus*), great skua (*Stercorarius skua*), little gull (*Hydrocoloeus minutus*), common gull (*Larus canus*), lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), black-legged kittiwake (*Rissa tridactyla*), Sandwich tern (*Thalasseus sandvicensis*), common tern (*Sterna hirundo*), common guillemot (*Uria aalge*) and razorbill (*Alca torda*). Both tern species are largely absent at the Bligh Bank and therefore tern data were only analysed for the Thorntonbank study area.

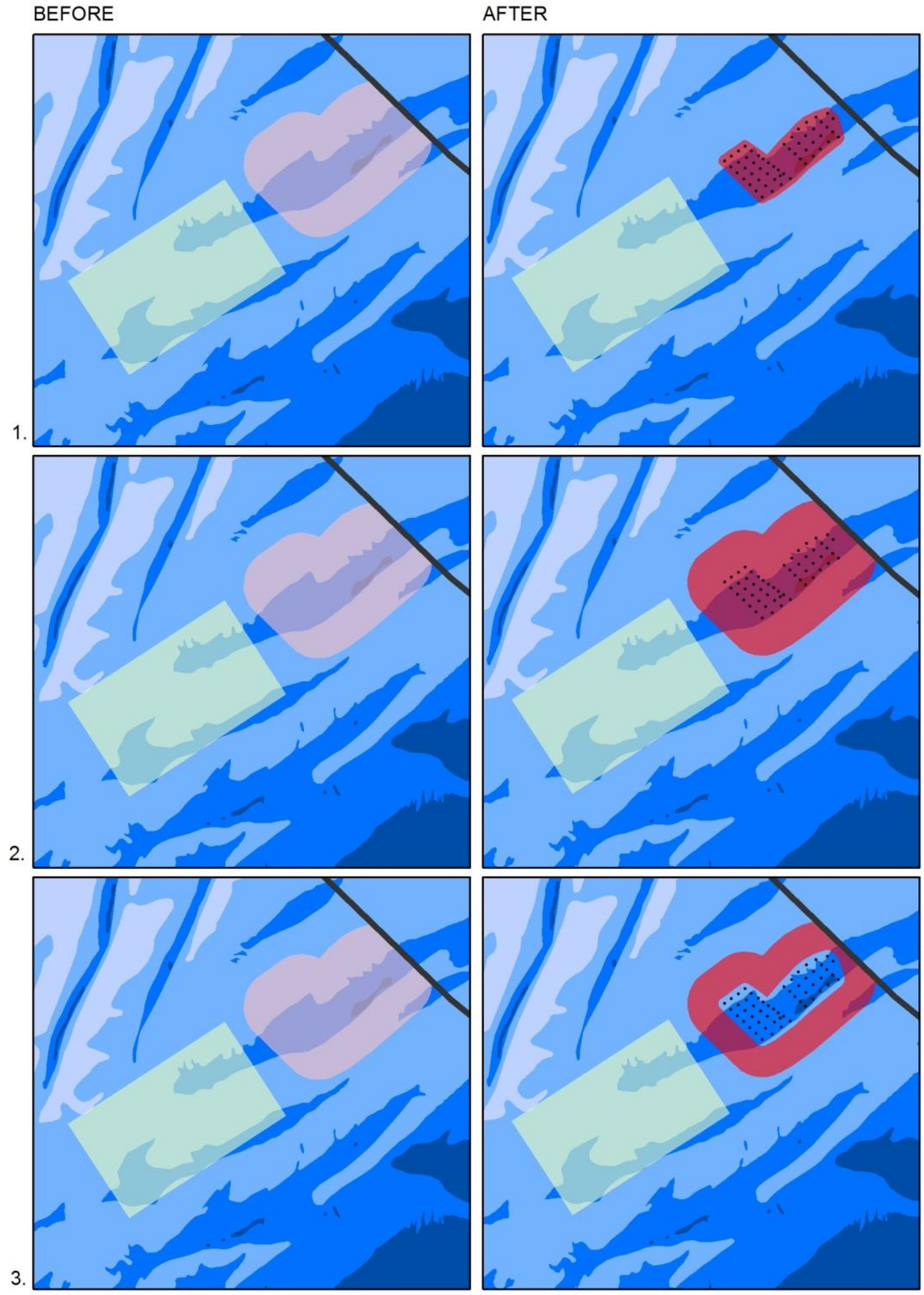

Figure 4. Overview of the BACI polygons used to study OWF induced seabird displacement at the Thorntonbank (green = control area / red = impact area).

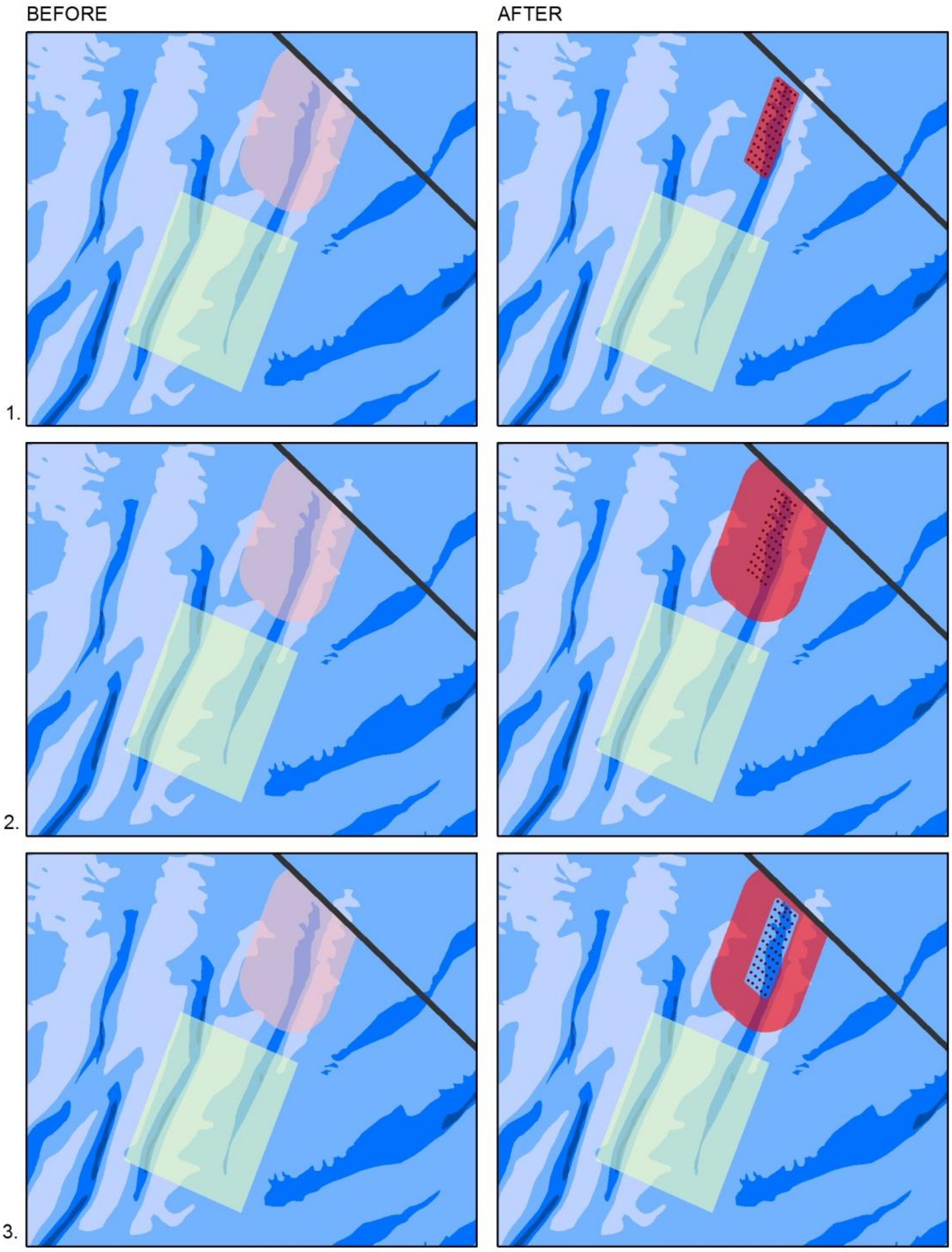

Figure 5. Overview of the BACI polygons used to study OWF induced seabird displacement at the Bligh Bank (green = control area / red = impact area).

STATISTICS

All data handling and modelling was performed in R.3.2.1 (R Core Team 2015a), making use of the following packages:

RODBC (Ripley & Lapsley 2013), foreign (R Core Team 2015b), date (Therneau 2014), ggplot2 (Wickham 2009), compare (Murrell 2014), reshape (Wickham 2007), plyr (Wickham 2011), MASS (Venables & Ripley 2002), mgcv (Wood 2011), glmmADMB (Skaug *et al.* 2014), Distance (Miller 2015) & mrds (Laake *et al.* 2015).

12.3. RESULTS

GENERAL OBSERVATIONS

By far the most commonly observed bird species in both OWFs during operation are gulls, making up a highly similar percentage of 93.0 and 93.4% of all non-passerine birds observed in the Thorntonbank & Bligh Bank OWF respectively (Table 2). Gulls were observed roosting on the turbine (jacket) foundations or transformation platforms in relatively large numbers, which is particularly true for great black-backed gull at the Thorntonbank (670 out of 840 birds in total). Clearly, jacket foundations offer much more roosting possibilities compared to monopiles, and a resulting 62.8% of the large gull species observed at the Thorntonbank were associated with man-made structures, compared to 18.0% at the Bligh Bank. Despite the reported avoidance effects (Vanermen *et al*. 2015a), auks (common guillemot and razorbill) are relatively often observed inside the OWF boundaries, totaling 188 and 102 individuals at the Bligh Bank and Thorntonbank respectively. Quite unexpected were the regular observations of shag (in total 17 individuals seen), a species which is otherwise rare in the BPNS.

Also worth mentioning is the regular occurrence of sea mammals inside the OWFs. In total, 45 harbour porpoises and 5 white-beaked dolphins were observed inside the Bligh Bank wind farm.

Table 2. Number of birds and sea mammals observed inside the Thorntonbank (526 km of surveying) and Bligh Bank (714 km of surveying) OWFs during operation.

		Bligh Bank		Thorntonbank	
		Total	Roosting on constructions	Total	Roosting on constructions
BIRDS					
Northern fulmar	*Fulmarus glacialis*	1	0	1	0
Northern gannet	*Morus bassanus*	27	0	10	0
Great cormorant	*Phalacrocorax carbo*	2	2	30	25
European shag	*Phalacrocorax aristotelis*	8	3	9	9
Unidentified cormorant	*Phalacrocorax* sp.	0	0	2	1
Barnacle goose	*Branta leucopsis*	4	0	0	0
Brent goose	*Branta bernicla*	11	0	0	0
Eurasian sparrowhawk	*Accipiter nisus*	0	0	1	0
Bar-tailed godwit	*Limosa lapponica*	0	0	1	0
Whimbrel	*Numenius phaeopus*	1	0	0	0
Eurasian curlew	*Numenius arquata*	23	0	0	0
Pomarine skua	*Stercorarius pomarinus*	1	0	0	0
Mediterranean gull	*Ichthyaetus melanocephalus*	1	0	0	0
Little gull	*Hydrocoloeus minutus*	0	0	10	0
Black-headed gull	*Chroicocephalus ridibundus*	45	0	16	0
Common gull	*Larus canus*	1689	0	100	2
Lesser black-backed gull	*Larus fuscus*	538	38	592	128
Herring gull	*Larus argentatus*	210	4	67	18
Yellow-legged gull	*Larus michahellis*	5	0	0	0
Great black-backed gull	*Larus marinus*	434	182	840	670
Unidentified large gull	*Larus sp.*	60	0	472	421
Black-legged kittiwake	*Rissa tridactyla*	884	0	235	1
Unidentified gull		34	0	0	0
Sandwich tern	*Thalasseus sandvicensis*	4	0	17	0
Common tern	*Sterna hirundo*	0	0	1	0
Common guillemot	*Uria aalge*	80	0	59	0
Unidentified auk	*Uria aalge* or *Alca torda*	20	0	11	0
Razorbill	*Alca torda*	88	0	32	0
Atlantic puffin	*Fratercula arctica*	1	0	0	0
Domestic pigeon	*Columba* sp.	3	0	1	0
Short-eared owl	*Asio flammeus*	1	0	0	0
Common starling	*Sturnus vulgaris*	382	2	122	3
Other passerines		72	2	27	4
SEA MAMMALS					
White-beaked dolphin	*Lagenorhynchus albirostris*	5	0	0	0
Harbour porpoise	*Phocoena phocoena*	45	0	4	0
Grey seal	*Halichoerus grypus*	1	0	1	0

DISTANCE ANALYSIS

For every species except for great skua, hazard-rate detection models fitted our data better than half-normal detection functions (Table 3). Observation conditions proved to affect detectability of seabirds significantly and either wave height or wind force was retained in all species except for great skua and both tern species. The natural logarithm of group size was retained for most species except for northern gannet and great skua, while for common guillemot group size was preferred over log(group size). Cluster detection probabilities were highest (>80%) for conspicuous species like great skua and northern gannet, and lowest (<60%) for northern fulmar, common gull, black-legged kittiwake and common guillemot.

Table 3. Results of distance analysis.

Species	Detection function	Covariates	Cluster detection probability
Northern fulmar	Hazard-rate	log(group size) + wave	0.57
Northern gannet	Hazard-rate	wave height	0.80
Great skua	Half-normal	/	0.83
Little gull	Hazard-rate	log(group size) + wind force	0.64
Common gull	Hazard-rate	log(group size) + wave	0.52
Lesser black-backed	Hazard-rate	log(group size) + wind force	0.67
Herring gull	Hazard-rate	log(group size) + wind force	0.66
Great black-backed	Hazard-rate	log(group size) + wind force	0.72
Black-legged kittiwake	Hazard-rate	log(group size) + wave	0.56
Sandwich tern	Hazard-rate	log(group size)	0.73
Common tern	Hazard-rate	log(group size)	0.60
Common guillemot	Hazard-rate	group size + wind force	0.56
Razorbill	Hazard-rate	log(group size) + wind force	0.63

BACI MODELLING RESULTS

Northern fulmar

In both study areas, northern fulmars showed a strong overall decrease in densities. After impact, only two positive observations occurred in the impact areas, one in the Thorntonbank OWF buffer zone and one inside the Bligh Bank OWF. No observations were thus made in the 'impact + 0.5 km' area at the Thorntonbank and the 'buffer 0.5-3 km' area at the Bligh Bank. In these cases meaningful statistics are no longer possible (see Tables 4 & 5: p=0.999, implying almost 100% unreliability), explaining the empty spaces in the left panels of Figures 6 & 7. Apart from these absences, other results also suggest avoidance by northern fulmars. However, due to the very low number of positive observations, confidence intervals are broad and effects are only significant for the 'impact + 3 km' area at the Bligh Bank, for which our models estimate a negative coefficient of -3.13, corresponding to a decrease in numbers of 96%.

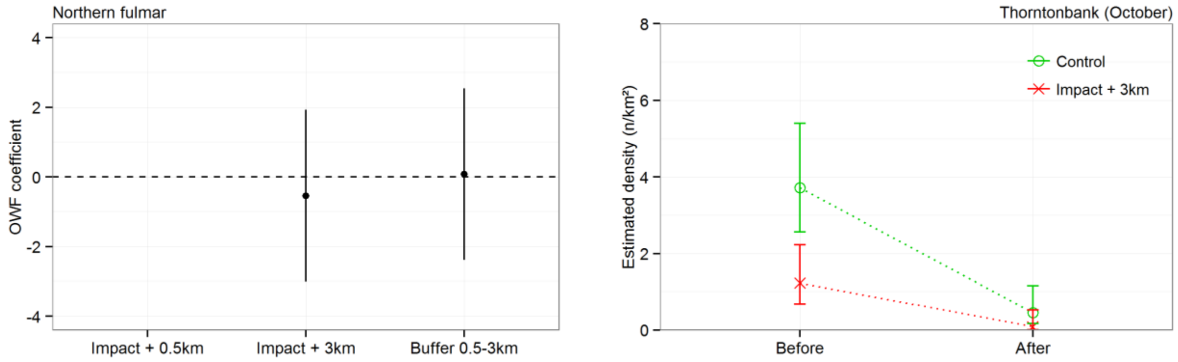

Figure 6. Modelling results for northern fulmar in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right (but note that zero-inflation equals 75%).

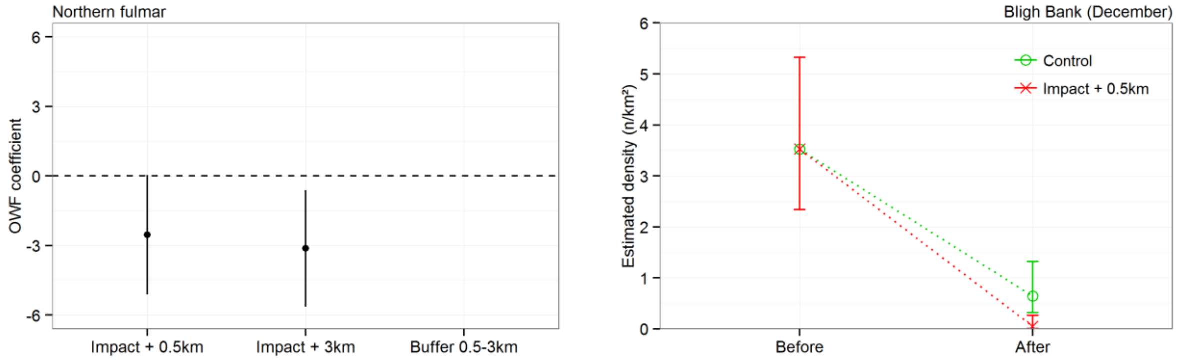

Figure 7. Modelling results for northern fulmar in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right (but note that zero-inflation equals 68%).

Northern gannet

Northern gannets avoided both the Thorntonbank and the Bligh Bank OWF. At the Thorntonbank there was only one positive count inside the OWF after impact, while in the Bligh Bank OWF northern gannets were observed inside the transect on six surveys, totaling 15 birds. Transforming the resulting negative OWF coefficients learns that gannet numbers significantly decreased with 99% & 82% in the 'impact + 0.5 km' areas at the Thorntonbank and Bligh Bank respectively. These results are quite consistent with the estimate obtained after three years of post-impact monitoring at the Bligh Bank when a decrease of northern gannets by 85% was reported (Vanermen *et al*. 2015a). In the buffer zones, decrease in densities was more moderate with 60% & 26% for the Thorntonbank and Bligh Bank respectively, the effect being no longer significant at the latter.

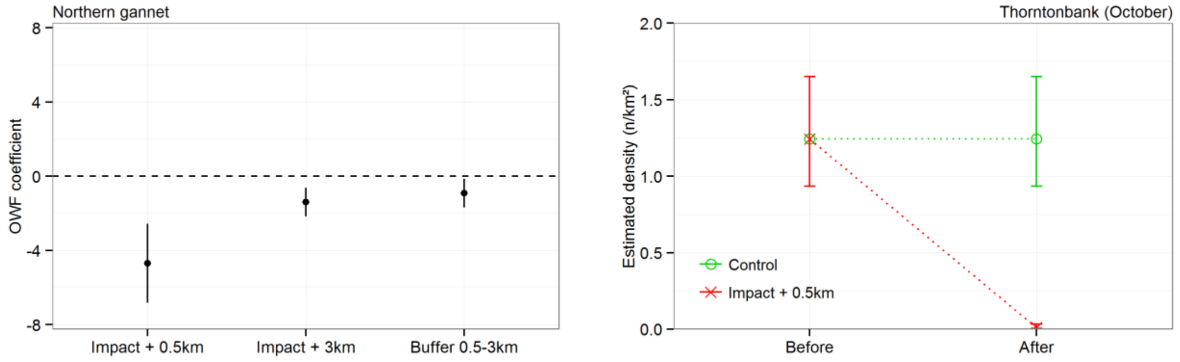

Figure 8. Modelling results for northern gannet in the Thorntonbank study area with OWF coefficients and their 95% CI's on the left and BACI density estimates for the month with maximum numbers on the right.

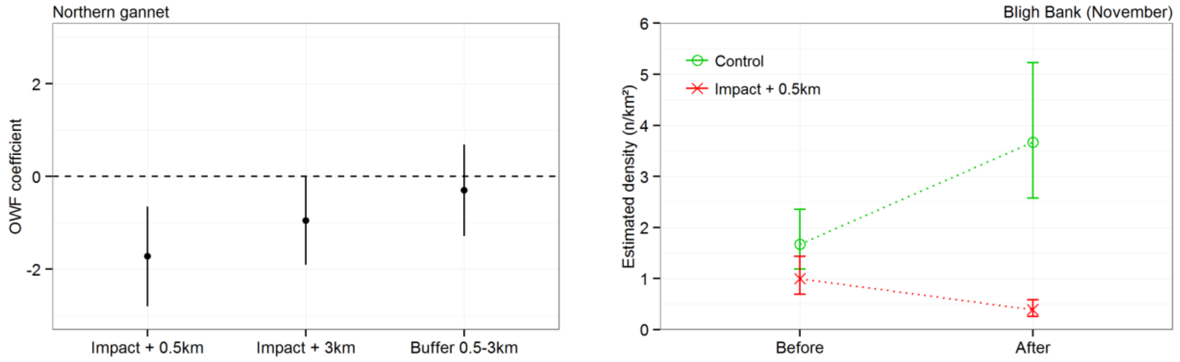

Figure 9. Modelling results for northern gannet in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Great skua

Great skua showed contradictory results with slightly positive OWF coefficients at the Thorntonbank study area and negative coefficients at the Bligh Bank. Due to the low number of positive observations after impact (no positive observations inside the OWFs and only one positive count in each of the buffer zones) and resulting broad 95% confidence intervals, none of these effects are statistically significant.

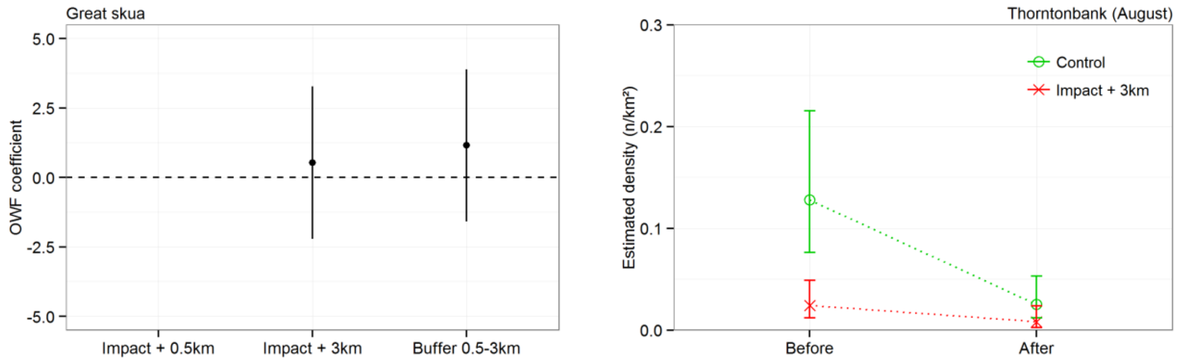

Figure 10. Modelling results for great skua in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

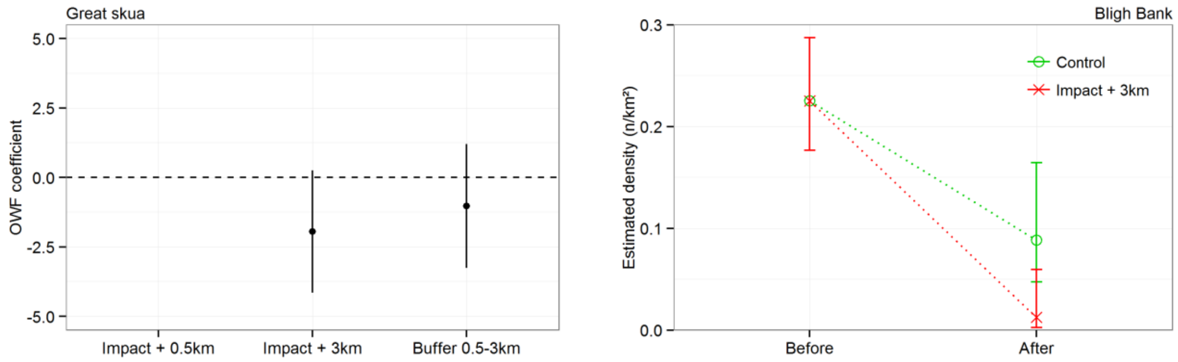

Figure 11. Modelling results for great skua in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and year-round BACI density estimates on the right (but note that zero-inflation equals 79%).

Little gull

Our BACI analysis detected a significant decrease of little gull density by 87% in the 'impact + 0.5 km' area at the Thorntonbank. Interestingly, OWF coefficients show a similar pattern in both study areas, being negative for the OWF area itself and positive in the buffer zone, suggesting local displacement out of the turbine-built area towards the near surroundings. However, only the aforementioned decrease in the Thorntonbank OWF proved statistically significant.

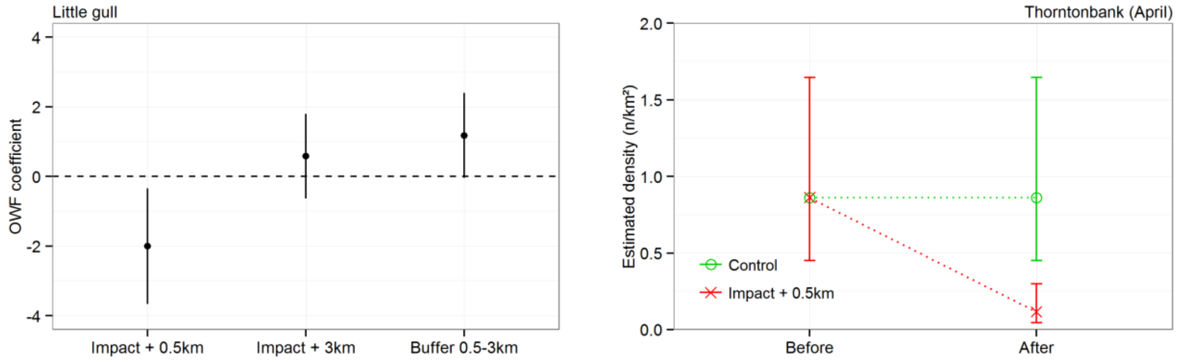

Figure 12. Modelling results for little gull in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

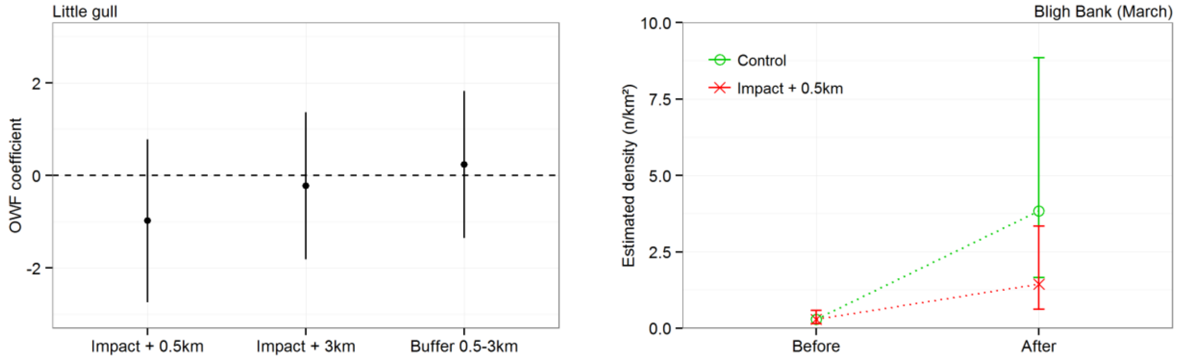

Figure 13. Modelling results for little gull in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Common gull

Common gull showed contradictory results with negative OWF coefficients at the Thorntonbank study area and positive coefficients at the Bligh Bank, however, none of these coefficients significantly differed from zero due to broad 95% confidence intervals. Importantly, the strongly positive coefficient (1.79) found for the Bligh Bank OWF is fully determined by the survey of 20/12/2010 when no less than 1,071 common gulls were observed between the turbines and inside the transect! This high number is very exceptional, as positive counts in the Bligh Bank OWF occurred in only 10 out of the 41 remaining surveys, totaling 64 birds. Hence, over a period of 5 years we counted 94% of the birds on one single day. Leaving out the count of 20/12/2010 results in a completely different coefficient estimate of -0.67, being much more similar to the -0.98 coefficient found for the Thorntonbank.

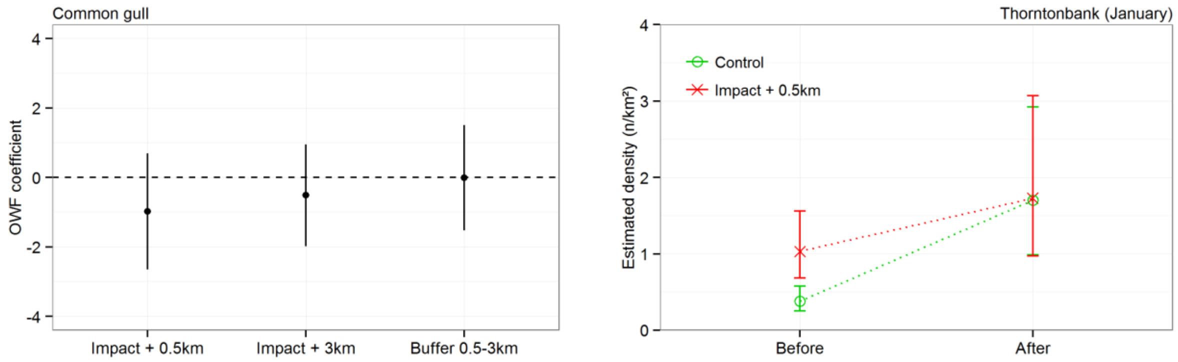

Figure 14. Modelling results for common gull in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

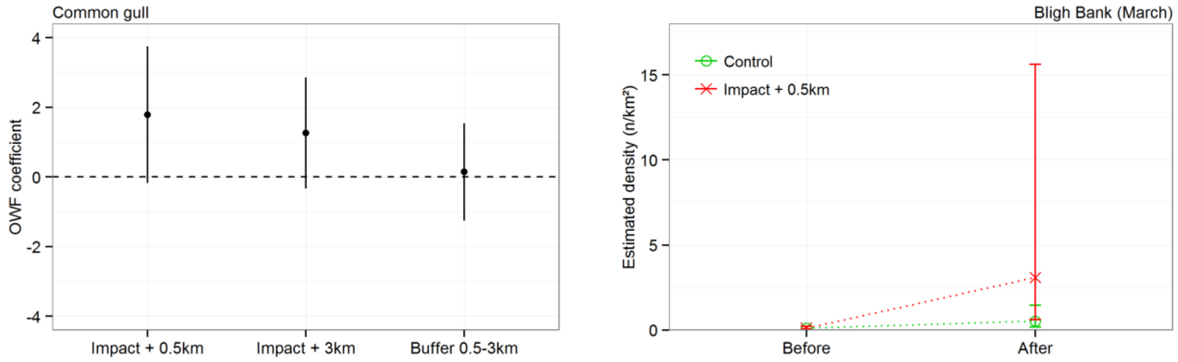

Figure 15. Modelling results for common gull in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Lesser black-backed gull

The highly positive OWF coefficients found for the Bligh Bank three years after impact (Vanermen *et al*. 2015a) still prevailed after 5 years of post-impact monitoring, and the increase in numbers is now estimated at a factor 8.1 for the 'impact + 0.5 km' area and a factor 7.7 for the buffer area, illustrating a strong attraction effect. At the Thorntonbank, however, no such effect was observed and densities remained at a high level of almost 6 birds/km² throughout the study area. Adjusting for birds associated with the turbines did not result in major changes in the outcome. Interestingly, there is a clear onshore-offshore gradient in the occurrence of lesser black-backed gulls in the BPNS with numbers dropping quickly beyond 20 nautical miles offshore (Vanermen *et al.* 2013). This is also illustrated by the background densities as measured in both study areas with almost 6 birds/km² at the Thorntonbank and only about 1 bird/km² at the Bligh Bank. The marked difference in response of lesser black-backed gulls towards the presence of an OWF between these two locations seems to support the stepping stone theory, in which the presence of OWFs with its numerous roosting possibilities allow birds to extend their natural distribution further offshore.

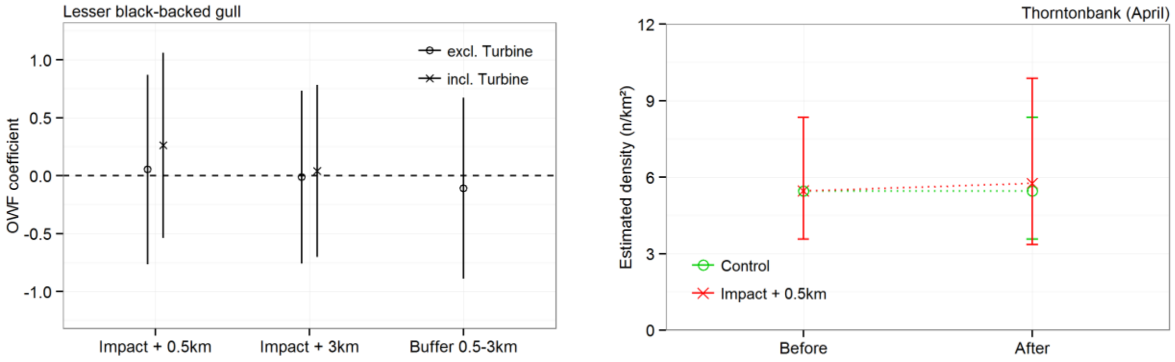

Figure 16. Modelling results for lesser black-backed gull in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers (exclusive turbine associated birds) on the right.

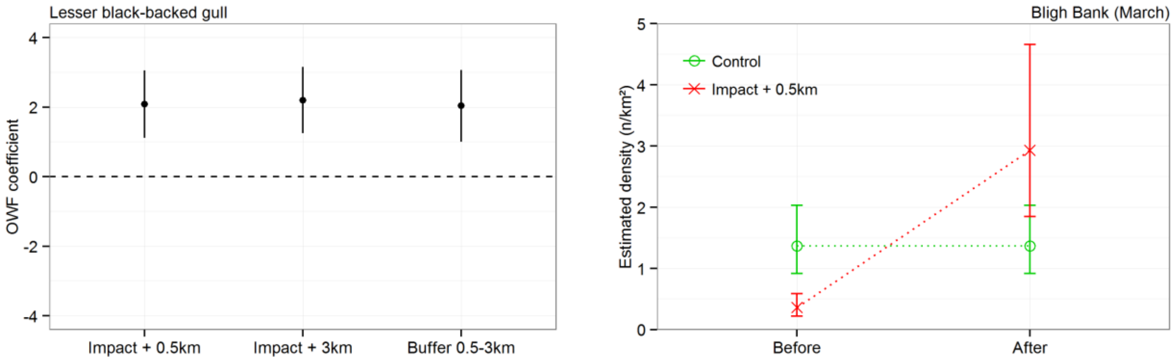

Figure 17. Modelling results for lesser black-backed gull in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Herring gull

At the Thorntonbank, fairly constant spring densities (≈0.4 birds/km²) of herring gull were observed throughout the study period. The OWF coefficient for the wind farm area itself ('impact + 0.5 km') is about zero, and accounting for birds associated with the turbines did not result in major changes in the estimated OWF coefficients. For the buffer area, we found a significant negative effect of -1.66, corresponding to a drop in numbers of 81%. From an ecological point of view, however, this drop in density is hard to explain.

The highly positive OWF coefficient found for herring gull densities at the Bligh Bank after 3 years of impact monitoring (Vanermen *et al.* 2015a) did not fully withstand the test of time. After 2 more years of post-impact monitoring the OWF coefficient dropped from 2.25 to 1.47, and is now only borderline significant. This drop in effect is fairly easy explained by the fact that only one high count is responsible for the positive coefficients obtained at the Bligh Bank (see also common gull). On 20/12/2010, 139 herring gulls were observed inside the transect and inside the wind farm. Later on, herring gulls were observed on 7 occasions only. When dropping this single survey from the analysis the OWF coefficient drops from 1.47 to 0.05.

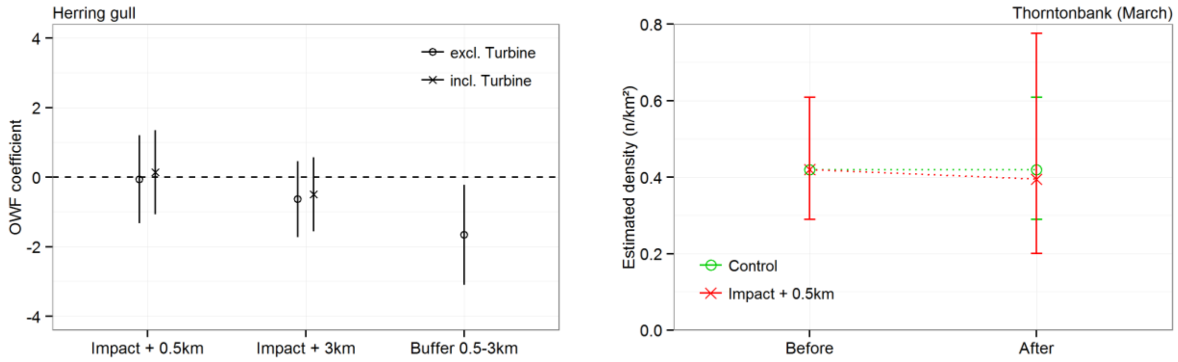

Figure 18. Modelling results for herring gull in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers (exclusive turbine associated birds) on the right.

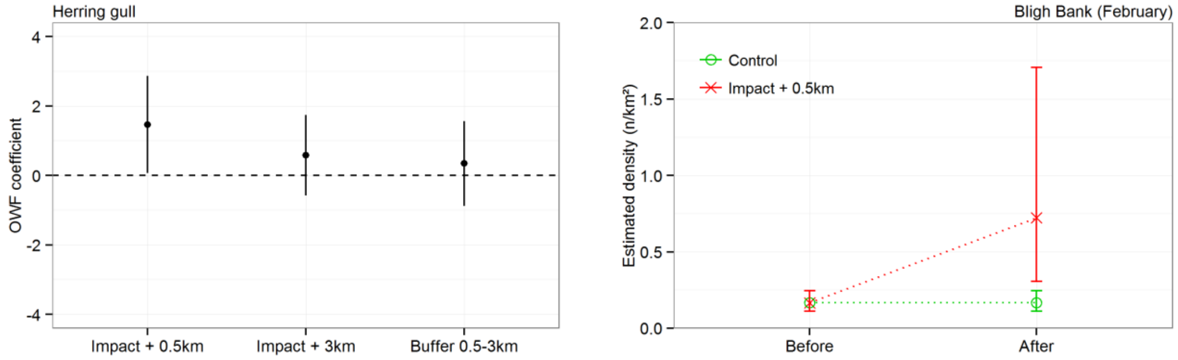

Figure 19. Modelling results for herring gull in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Great black-backed gull

In contrast to the two previous species, great black-backed gull does show some consistency in results between both investigated sites. At the Thorntonbank, the standard analysis results in OWF coefficients close to zero. But when taking in account the numerous birds observed roosting on the jacket foundations, OWF coefficients become highly positive, with e.g. a value of 1.86 for the 'impact + 0.5 km' area, corresponding to an increase in numbers by a factor 6.4. At the Bligh Bank too, strongly positive and significant OWF coefficients were found, i.e. 1.29 for the 'impact + 0.5 km' area (~ factor 3.6 increase), the positive effect of 0.61 in the buffer area being no longer significant. The effect at the Bligh Bank has thus become much stronger than the previously reported 0.38 OWF coefficient after three years of post-impact monitoring (Vanermen *et al.* 2015a).

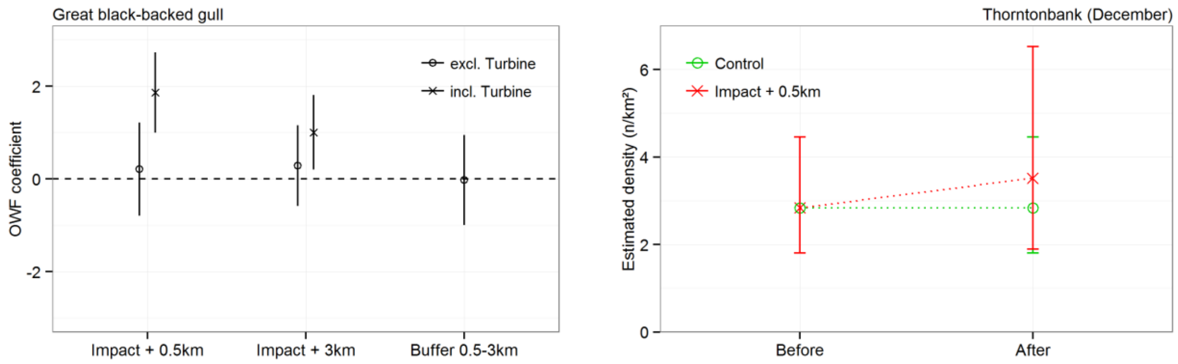

Figure 20. Modelling results for great black-backed gull in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers (exclusive turbine associated birds) on the right.

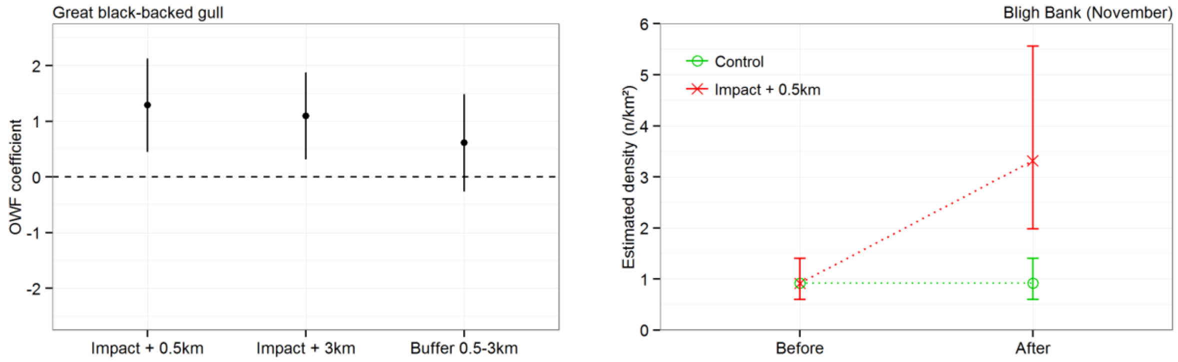

Figure 21. Modelling results for great black-backed gull in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Black-legged kittiwake

Results for black-legged kittiwake strongly differed between locations, with slightly positive non-significant coefficients at the Bligh Bank (0.26-0.43) compared to significantly negative coefficients at the Thorntonbank. According to our BACI models, black-legged kittiwakes decreased in numbers by 86% and 57% in the Thorntonbank 'impact + 0.5 km' and 'buffer 0.5 - 3 km' areas respectively.

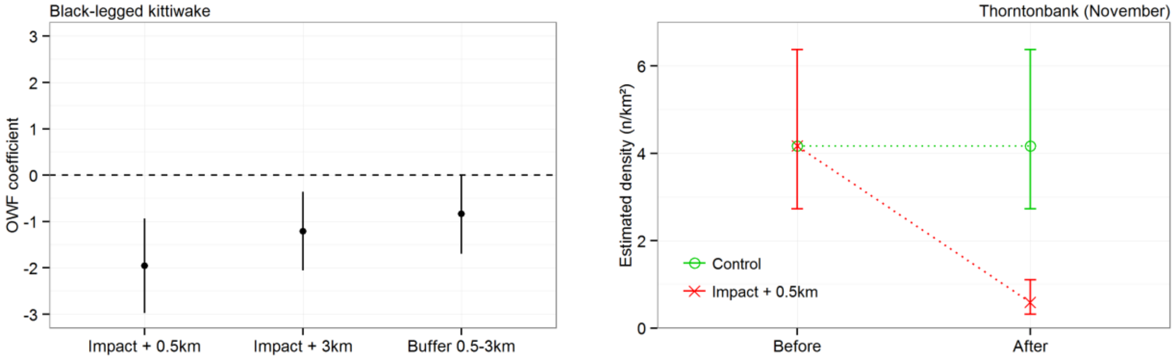

Figure 22. Modelling results for black-legged kittiwake in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

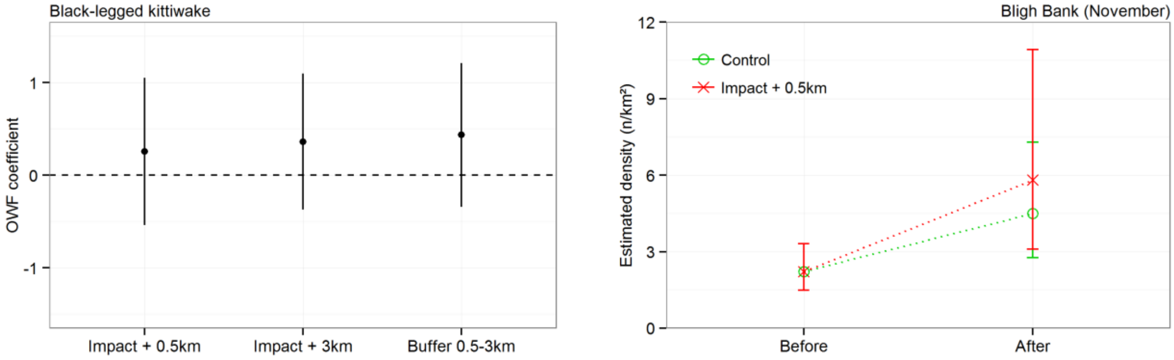

Figure 23. Modelling results for black-legged kittiwake in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Sandwich tern

In the Thorntonbank study area, numbers of Sandwich tern show a less marked decrease in the impact area as opposed to the control area, resulting in positive OWF coefficients. In the buffer zone, the model predicts a significant increase in numbers by a factor 5.6. Despite statistical significance, results should be interpreted with care due to the very low number of positive observations after impact (2 observations inside the OWF and 4 in the buffer zone). On the other hand, when only 6 turbines were present (phase I – see Table 1) we also found a significantly positive OWF coefficient for the 3 km buffer zone (Vanermen *et al.*, 2013).

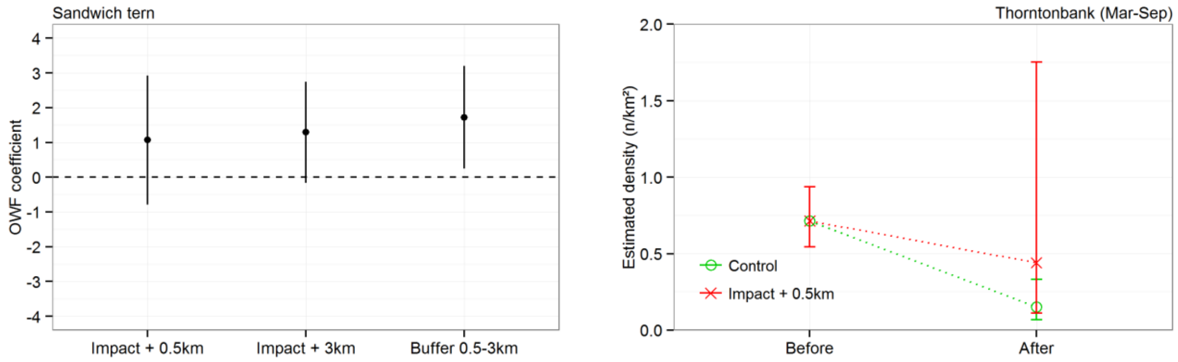

Figure 24. Modelling results for Sandwich tern in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the period March to September on the right (but note that zero-inflation equals 74%).

Common tern

Before the construction of the OWF at the Thorntonbank, positive observations of common tern were already few (2 in the control area & 5 in the impact area). After impact, however, no positive observations were made at all, neither in the impact nor in the control area (see Figure 25). As a 100% decrease in numbers occurred in both areas, there can be no demonstrable effect of the presence of the wind farm.

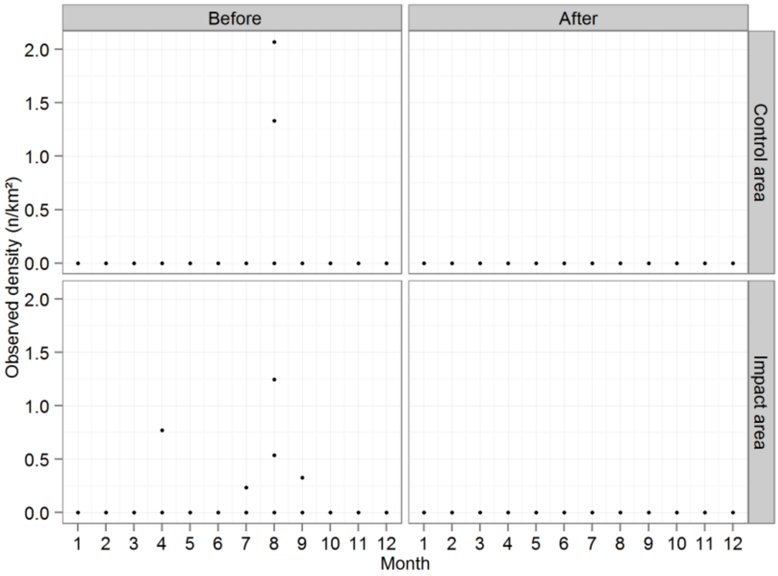

Figure 25. Observed densities of common tern in the control and impact area before and after the construction of the OWF at the Thorntonbank.

Common guillemot

Our BACI study showed common guillemots to avoid both wind farms under study. The significantly negative OWF coefficients of -1.13 and -1.39 correspond to a decrease in numbers of 68% and 75% respectively. In the buffer area coefficients are still negative with -0.27 at the Thorntonbank and -0.68 at Bligh Bank, corresponding to a decrease of 24 and 49% respectively. In case of the former, however, the decrease in the buffer area proved not statistically significant. These results are highly comparable to the decrease of 71% reported three years after turbine construction at the Bligh Bank (Vanermen *et al*. 2015a).

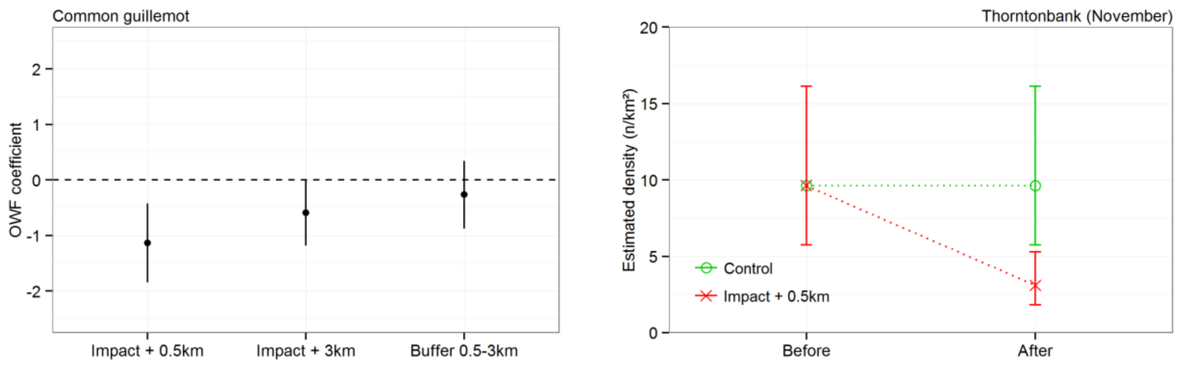

Figure 26. Modelling results for common guillemot in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right (but note that zero-inflation equals 10%).

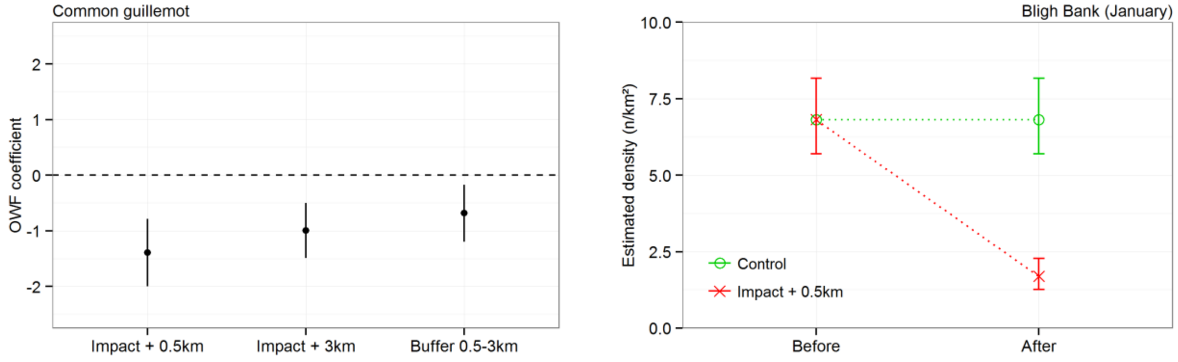

Figure 27. Modelling results for common guillemot in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

Razorbill

Results for razorbill suggest avoidance of offshore wind farm areas. At the Bligh Bank study area, the significantly negative OWF coefficient found for the ‘impact + 0.5 km’ area equals -1.12 and corresponds to a decrease in numbers by 67%. This result is very similar to the OWF coefficient of -1.01 reported in Vanermen *et al.* (2015a). On the other hand, the OWF coefficient calculated for the ‘buffer 0.5 - 3 km’ area is limited to -0.39 (~ 32% decrease), and does not differ significantly from zero. At the Thorntonbank, none of the OWF coefficients proved to be statistically significant, but a negative coefficient of -0.80 (~ 55% decrease) was found for the ‘impact + 0.5 km’ area.

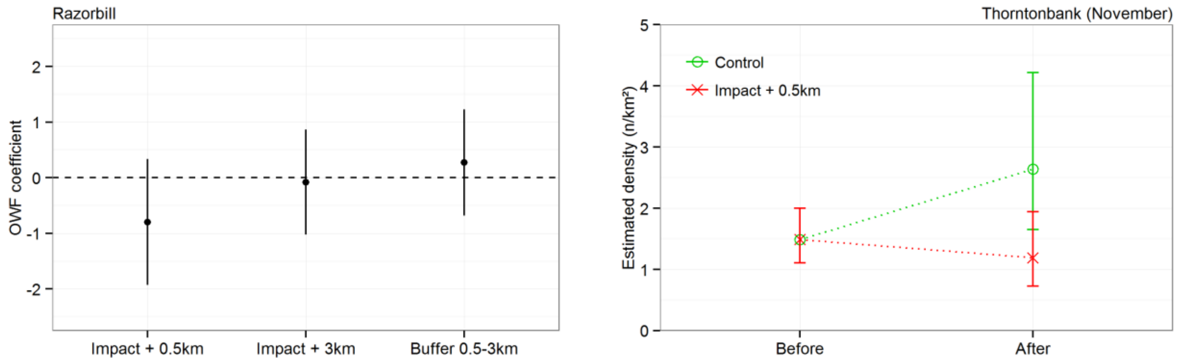

Figure 28. Modelling results for razorbill in the Thorntonbank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

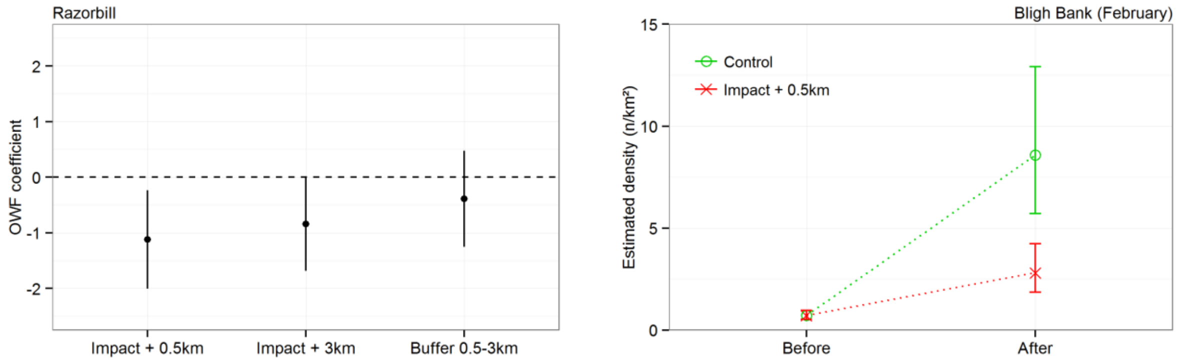

Figure 29. Modelling results for razorbill in the Bligh Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.

SUMMARIZING TABLES

Our BACI results are summarized in Table 4 & 5, which list all OWF coefficients and corresponding P values as estimated during the modelling process. All impact model coefficients are displayed in the Tables 6 & 7 in annex 3.

Table 4. BACI modelling results for the C-Power wind farm at the **Thorntonbank** after 3 years of operation, with indication of the displacement-related OWF model coefficients and their respective P values; model results based on an adjusted response variable including turbine-associated birds are indicated by “(T)” in the species column (P<0.10., P<0.05*, P<0.01**, P<0.001***; red cells indicate significant avoidance, green cells indicate significant attraction).

	Impact + 0.5 km		Impact + 3 km		Buffer 0.5-3 km	
	OWF Coefficient	P-Value	OWF Coefficient	P-Value	OWF Coefficient	P-Value
Northern fulmar	-20.98	0.999	-0.54	0.669	0.08	0.949
Northern gannet	-4.70	0.000***	-1.40	0.000***	-0.92	0.020*
Great skua	-23.08	1.000	0.54	0.701	1.15	0.409
Little gull	-2.01	0.018*	0.59	0.345	1.18	0.058.
Common gull	-0.98	0.252	-0.51	0.493	-0.01	0.989
Lesser black-backed gull	0.05	0.899	-0.01	0.972	-0.11	0.786
Lesser black-backed gull (T)	0.26	0.519	0.04	0.914		
Herring gull	-0.06	0.923	-0.63	0.258	-1.66	0.024*
Herring gull (T)	0.14	0.818	-0.49	0.365		
Great black-backed gull	0.21	0.676	0.28	0.522	-0.02	0.960
Great black-backed gull (T)	1.86	0.000***	1.00	0.014*		
Black-legged kittiwake	-1.95	0.000***	-1.21	0.005**	-0.84	0.055.
Sandwich tern	1.07	0.258	1.29	0.082.	1.72	0.022*
Common guillemot	-1.13	0.002**	-0.59	0.048*	-0.27	0.392
Razorbill	-0.80	0.167	-0.08	0.869	0.27	0.577

Table 5. BACI modelling results for the Belwind wind farm at the **Bligh Bank** after 5 years of operation, with indication of the displacement-related OWF model coefficients and their respective P values (P<0.10., P<0.05*, P<0.01**, P<0.001***; red cells indicate significant avoidance, green cells indicate significant attraction).

	Impact + 0.5 km		Impact + 3 km		Buffer 0.5-3 km	
	OWF Coefficient	P-Value	OWF Coefficient	P-Value	OWF Coefficient	P-Value
Northern fulmar	-2.54	0.053.	-3.13	0.015*	-22.93	0.999
Northern gannet	-1.72	0.002**	-0.95	0.051.	-0.30	0.551
Great skua	-19.45	0.998	-1.95	0.083.	-1.03	0.364
Little gull	-0.98	0.277	-0.22	0.784	0.23	0.773
Common gull	1.79	0.074.	1.26	0.122	0.14	0.842
Lesser black-backed gull	2.09	0.000***	2.20	0.000***	2.04	0.000
Herring gull	1.47	0.040*	0.58	0.326	0.35	0.578
Great black-backed gull	1.29	0.003**	1.09	0.006**	0.61	0.168
Black-legged kittiwake	0.26	0.525	0.36	0.332	0.43	0.273
Common guillemot	-1.39	0.000***	-0.99	0.000***	-0.68	0.009**
Razorbill	-1.12	0.013*	-0.84	0.049*	-0.39	0.376

12.4. DISCUSSION

In this report we presented the results of our monitoring study on seabird displacement effects following the construction of offshore wind farms in the BPNS. For the first time after its completion in 2013 we did so for the C-Power wind farm at the Thorntonbank, and we also gave an update of the results for the Bligh Bank wind farm after five years of post-impact monitoring. Monitoring at the Bligh Bank has now been temporarily put on hold and the program is to be resumed during post-impact years 10 to 12, to study whether earlier observed effects still prevail or otherwise if some form of habituation towards the wind farm presence has occurred among residing seabirds.

In order to further increase the reliability of our data analyses, we introduced some adjustments to our methodology. In the first place we performed multi-covariate distance sampling to correct the observed numbers of seabirds for decreasing detectability with distance, allowing the species-specific detection functions to vary with observation conditions and group size (Buckland *et al*. 2001, Thomas *et al*. 2010, Marques & Buckland 2003). Typically, detection probability decreased with wave height or wind force and increased with group size. Correcting the observed seabird numbers according to the estimated detection probabilities thus reduced temporal variation resulting from varying observation conditions.

Secondly, we applied a different model selection approach compared to earlier reports (e.g. Vanermen *et al*. 2013), moving away from a step by step model selection strategy. Instead we identified a relatively large set of candidate models and chose a single best model based on the 'Akaike Information Criterion' (AIC). While the resulting model will mostly be the same as the one obtained through step by step model selection, a major advantage of this so-called information-theoretic approach is that listing all AIC values in one matrix gives a good and instantaneous overview of how different candidate models relate to one another in terms of likelihood (AIC being a log-likelihood based criterion). Using this strategy clarifies that differences in AIC are sometimes very small (<1), implying there is more than one 'good' model, each of them estimating the wind farm effect somewhat differently. The differences in AIC values among a set of models can be recalculated to relative model probabilities ('Akaike weights'), and the ratio between two of these model probabilities can be regarded as the odds. For example, when two models differ in AIC by 1 unit, the model with the lowest value is only 1.6 times more likely to be the best of both. On the other hand, the relation between difference in AIC and model probability is highly non-linear and when models differ in AIC by 10 units, the odds are already 148 to 1 in favour of the model with the lowest AIC. Knowing all this, it was tempting to perform multi-model inference (MMI, Burnham & Anderson 2002), in which 'Akaike weights' are calculated for a set of candidate models, which in turn can be used to calculate a weighted average of their coefficient estimates. When performing an exploratory MMI for several species (at least for those showing marked OWF effects), the multi-model inferred OWF coefficient estimate was always very close to the value estimated by the single best model strategy. Coefficients of the single best models of common guillemot for example were -1.13 and -1.39 for the Thorntonbank and Bligh

Bank, while through MMI, values of respectively -1.16 and -1.36 were obtained. For northern gannet, single best model coefficients were -4.70 and -1.72, compared to MMI coefficients of -4.68 and -1.86. Great black-backed gull at last showed OWF coefficients of 1.86 and 1.29 for the Thorntonbank and Bligh Bank, with the MMI exercise resulting again in highly comparable coefficients of 1.80 and 1.35. These results show that our modelling strategy leads to quite balanced and robust results, emphasizing the qualitative and quantitative value of the OWF coefficients found and listed in Tables 4 & 5.

A third and last optimization in our modelling strategy was the inclusion of a fishery factor in the models. As expected, the presence of fishery activity in or in the vicinity of the study area greatly influenced the number of scavenging seabirds present and often explained a significant part of the variation in our count data (Tables 6 & 7 in annex 3). On the other hand, we should emphasize that a simple true-false covariate based on the observation of one or more beam trawlers within 3 km of the monitoring track is a very raw measure of fishing activity and it would be much better to obtain a quantitative measure of actual trawling activity in the hours preceding the seabird surveys based for example on AIS vessel tracking information.

In the context of seabird displacement monitoring and offshore wind farming, a before-after gradient (BAG) design has recently been recommended as a preferred alternative to the classic BACI design (JNCC 2015). In a BACI framework, the impact effect is calculated based on the assumption that without the impact a parallel trend in numbers as observed in the control area(s) would have occurred in the impact area. A reliable BACI analysis thus largely depends on the possibility of being able to delineate one or more suitable control areas, which might not always be the case. A BAG approach on the other hand assumes any pre- and post-impact changes to be a function of distance and that any impact-related effects are the same in all directions from the impact source (Oedekoven *et al.* 2013). When abundance and distribution of animals would change over time in an area without the introduction of any anthropogenic impact, one would expect such post-impact changes to be distributed without major reference to the impacted location. On the contrary, impact-related changes are most likely to occur in and around the impacted site and significant changes centered around the impact site therefore provide compelling evidence for impact-related effects (MacKenzie *et al*. 2013). In preliminary analyses we tested whether our BACI designed monitoring data could in fact be processed applying a BAG analysis, and for some species this appeared to work out beautifully. However, a well-designed BAG study is supposed to generate data of a wide area with the wind farm located in the middle, allowing to test the aforementioned assumption that a potential OWF effect declines with distance in all directions. Our survey tracks on the other hand were designed in a way that the study area has a rectangular shape with the OWF located in the corner, implying we can test the ‘gradient’ assumption sufficiently in only one direction. More problematic is the fact that since both our OWFs are located at the edge of the study area polygon, spatial smoothers suffer from edge effects exactly at our points of interest. At the moment, we feel that pushing our BACI designed data in a BAG analysis can provide nice visual presentation of OWF related impact effects, but can never match the potential additive value in terms of statistical evidence of an a priori BAG designed monitoring study.

With five years of post-impact monitoring at the Bligh Bank and three years at the Thorntonbank, there are now two relatively well-studied offshore wind farms in the Belgian part of the North Sea. Ideally, both sites could be regarded as 'replicates', but this is clearly not the case. On the contrary, both sites differ strongly in background densities of seabirds, environmental variables, wind farm layout and turbine characteristics, and each of these factors may influence displacement effects in their own way. It is therefore very interesting to compare the results obtained at both sites, and we see that for some species there is a striking consistency, while for others we observed opposite effects.

Northern gannet and common guillemot avoid both the Thorntonbank and Bligh Bank OWF, while great black-backed gull is attracted to both. Razorbill decreased in numbers at the two sites, this decrease being significant at the Bligh Bank only. As shown through power analyses, it might be a simple matter of time before the observed decrease of razorbill at the Thorntonbank proves to be statistically significant as well (Vanermen *et al.* 2015b). Interestingly, the previously reported significant effects after three years of post-impact monitoring at the Bligh Bank (Vanermen *et al*. 2015a) were all confirmed after five years, illustrating the robustness in results.

Other more or less consistent results, yet not necessarily significant, were obtained for northern fulmar and little gull. Numbers of northern fulmar significantly decreased at the Thorntonbank, while the species was not observed once inside the Bligh Bank OWF boundaries after impact. Little gull showed an interesting combination of negative coefficients in the OWF areas itself, opposed to positive coefficients in the surrounding buffer zones. This pattern is most marked at the Thorntonbank and accordingly, we reported attraction effects of little gull to the immediate surroundings of the phase I of the C-Power wind farm (Vanermen *et al*. 2013). Sandwich tern was not studied at the Bligh Bank because the species is largely absent there, but appeared to be attracted to the OWF at the Thorntonbank, this effect being significant for the buffer zone. As for little gull, this is in line with the results for the phase I of the C-Power wind farm during which we also found attraction of Sandwich tern to the surroundings of the six turbine row (Vanermen *et al*. 2013). The results for the latter two species correspond to findings in Denmark and the Netherlands where terns and little gulls were also observed to be attracted to the wind farm edges rather than to the OWF area itself (Petersen *et al*. 2006, Krijgsveld *et al*. 2011).

For other species, however, results appeared more inconsistent. Black-legged kittiwake avoided the Thorntonbank wind farm area while an opposite (yet non-significant) effect was observed at the Bligh Bank. The previously reported attraction effects of lesser black-backed and herring gull at the Bligh Bank were confirmed after two more years of monitoring, but no attraction seemed to occur in the more nearshore Thorntonbank wind farm. Interestingly, the Thorntonbank lies just within these two species' normal distribution range, while the Bligh Bank is located further offshore. With OWFs offering increased roosting possibilities, OWFs have been shown to serve as a stepping stone allowing birds to colonize areas that are otherwise off limit (Leopold *et al*. 2013). A stepping stone effect is likely to be much stronger outside compared to inside a bird's normal distribution and the marked difference in OWF effect between both sites therefore seems to support this theory. On the other hand it has also been hypothesized that seabirds may profit from increased food availability due to the so-called 'reef effect' following the introduction of turbine foundations as hard substrate in an otherwise sandy marine environment. But until this moment, this remains unproved and possibly also hard to detect based on ship-based seabird surveys. If birds would actually concentrate in OWFs for foraging purposes, this is likely to occur in a tidal-dependent way. Large gulls for example are now regularly observed feeding on mussels in the lower regions of the jacket foundations during low tide, and have also been observed foraging in the turbulent wake of the turbines during times of high tidal current. Unfortunately, during ship-based seabird surveys, the OWFs themselves are visited during limited time frames of about 1.5 hours. More ideally, repeated point-based observations are made over a full tidal cycle and the recently installed fixed camera at one of the turbines in the Thorntonbank OWF opens possibilities to do so without major logistical constraints. We therefore plan hourly counts of birds associated with the turbines, to look for possible tidal effects on their presence. At first sight, detecting birds on the water through this camera appears to be particularly challenging. Nevertheless, being able to do so seems indispensable to find out what birds are doing in the wind farms when they are not roosting on the foundations. Do they leave the area, thus supporting the stepping stone theory? Or do they remain within the OWF boundaries to look for food in the area itself? Analysing the GPS-data of lesser black-backed and herring gulls tagged in the colonies at Zeebrugge and Oostende may further help to understand patterns in the interaction between gulls and OWFs, provided of course that sufficient data of tagged birds coming to visit the OWFs can be gathered. If camera and GPS data would appear insufficient we still could go for full day observations from one of the turbine foundations or a transformation platform deck.

12.5. CONCLUSIONS

After five years of post-impact monitoring at the Bligh Bank OWF and three years at the Thorntonbank OWF we found significant avoidance by northern gannet and common guillemot at both sites. Common guillemot decreased in densities by 68% and 75% at the Thorntonbank and Bligh Bank respectively, and northern gannet by 99 and 82%. Razorbill decreased in numbers at the two sites, this decrease being significant at the Bligh Bank only (67%). Both sites attracted great black-backed gulls, this species having increased in numbers by a factor 6.4 and 3.6 at the Thorntonbank and Bligh Bank respectively. The previously reported attraction effects of lesser black-backed gull and herring gull at the Bligh Bank were confirmed after two more years of monitoring, but no such effect was observed at the Thorntonbank. Sandwich tern appeared to be attracted to the OWF at the Thorntonbank, this effect being significant for the buffer zone.

While the avoidance of common guillemot and northern gannet seems readily interpretable from a disturbance perspective, it is still difficult to pinpoint the observed increases in seabird numbers, even more so because these are not always consistent between both sites under study. Gaining more insight in the diurnal and tidal-dependent variation in numbers and behaviour of birds occurring inside the OWFs seems indispensable for understanding the observed patterns and learning whether birds come to the OWFs merely for roosting and the related stepping stone function, or whether OWFs also offer increased food availability.

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CHAPTER

13

CHAPTER 13

BIRD RADAR STUDY IN THE BELGIAN PART OF THE NORTH SEA: DEVELOPMENTS TO IMPROVE BIRD DETECTION

R. Brabant[*1], J. Vidao[2], A. Smith[2] and S. Degraer[1]*

[1]Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels, Belgium.

[2]DeTect, Inc. 1022 West 23rd St., Suite 620, Panama City, FL 32405, USA.

[*]Corresponding author: Robin.Brabant@naturalsciences.be

ABSTRACT

Dedicated bird radars have been used in ornithological studies for many years. This techniques has the advantage that it provides continuous data on a large scale. However, there are also several restrictions to this technique: the recorded radar data have a low taxonomic resolution and radars also records objects other than birds (e.g. sea surface, ships, rain). All unwanted detections are being referred to as clutter. The goal of this study is to develop a reliable filter, based on the differences in target characteristics as

recorded by the radar, to post-process the vertical radar data which removes as much clutter from the database as possible. This will result in a more accurate bird flux and therefore an improved outcome of the bird collision model.

The model tests showed very high scores for the criteria accuracy, sensitivity and specificity. The model precision is a lower in one of the two tests. This is caused by a relatively high number of false positives in the model results. This will be improved in the future by including variables in the decision tree analysis which are linked to the bird track level, instead of only using the variables recorded by the radar which describe the single targets, as was the case in the current model.

13.1. INTRODUCTION

Complementary to the seabird surveys, also a continuous monitoring of birds to study the impact of wind farms, making use of a bird radar, is performed (Brabant et al., 2012; Vanermen et al., 2013).

The goals of this study are:

(1) to assess to what extent wind farms are a barrier to local and migrating birds;
(2) to measure the flux of birds through the wind farm area and the temporal variation thereof (e.g. seasonal, diurnal);
(3) to estimate the number of birds colliding with the turbines based on the flux data, by using a mathematical bird collision risk model;
(4) to determine the temporal variation of bird intensity and direction of flight in the area to the south of the radar location and how this will change once the Norther wind farm is being built and operational.

These objectives will be achieved making use of a dedicated Merlin bird radar (DeTect-inc., Florida, USA) which is installed on the offshore platform in the C-Power wind farm on the Thorntonbank. The radar system consists of two radar antennas (Kelvin-Hughes Sharpeye solid state S-band antennas), one scanning in the horizontal pane and one in the vertical. The detection range of the radar antennas can be specified in the system's settings. For the horizontal scanning radar (HSR) the range is maximum seven nautical miles, but is usually set at four nautical miles. The range of the vertical scanning radar (VSR) is set to track to a height of two nautical miles. The radar operates continuously year-round and the system is remotely controlled. The system is operated by software called Merlin which is specifically designed to track individual birds.

The flight paths can be determined with the horizontal scanning radar. This radar registers targets 360° around its location. The Merlin software links consecutive registrations of a target, and thus registers the flight path of a moving target. This way it is possible to determine a bird's flight path, flight direction and changes in that direction (DeTect Inc., 2010; Brabant et al., 2012).

The flux of birds (birds/(km*hr)) can be deducted from the VSR-data. By rotating in the vertical pane the VSR is creating a 'radar screen' that registers all the targets moving through that screen. As this 'radar screen' is fairly narrow (opening angle 22°) every registration can be seen as one or a group of birds passing through that area. The flux of birds is expressed as migration traffic rate (MTR), i.e. number of birds that pass through a certain area during a certain time period (Krijgsveld et al., 2011).

The use of radar has several advantages and have been used in similar research for several years abroad, for instance in Denmark (Petersen *et al.*, 2006) and the Netherlands (Krijgsveld *et al.*, 2011). They provide continuous data on a large scale, also during conditions where it is very difficult to gather visual data (e.g. at night, during bad weather conditions, far offshore). However, there are also several restrictions to this technique: the recorded radar data have a low taxonomic resolution and radars also records objects other than birds (e.g. sea surface, ships, rain). These unwanted detections are being referred to as clutter.

The biggest problem offshore is the clutter caused by waves, i.e. seaclutter (figure 2). Waves and to a lesser extent rain result in large amounts of noise in the database. All this clutter needs to be filtered out before being able to study the bird movements in the area (HSR) and to reliably determine the real-time flux of birds in the wind farm area at different altitudes and to calculate a real-time collision risk (VSR).

In several studies in the past, filters were developed to classify radar data and to remove as much clutter as possible (Krijgsveld et al., 2011; Rosa et al., 2015; Vang et al, 2011). As our radar antennas are making use of the solid state technique compared to the more conventional magnetron antennas, and there are site specific circumstances (e.g. radar platform, turbines, bird community), it is necessary to develop these kind of data filters on a case-by-case base.

The first focus of this bird radar research is therefore to develop a clutter filter. The goal of this study is to develop a reliable filter which removes as much clutter from the vertical database as possible. This will result in a more accurate bird flux and therefore an improved outcome of the bird collision model.

13.2. METHOLOGY

To remove clutter from the vertical radar database as effective as possible, DeTect and RBINS developed a filtering model based on the differences in target characteristics as recorded by the radar.. This development consisted of four steps:

1. Develop a reference dataset;
2. Create a classification model based on the reference data;
3. Validate the model with test data;
4. Evaluation of the model.

STEP 1 - REFERENCE DATASET

We used MERLIN Editor, a Merlin software application which allows selecting individual targets and storing them in separate reference databases (e.g. weather, side lobes, birds). We classified targets as birds, rain, turbines and side lobes. This hence resulted in four reference datasets. This process was done through a remote connection with the radar system and not by visual observations at the radar site. To avoid bias in the reference datasets, we selected data from several periods in the year, each with its typical bird activity, i.e. spring migration, autumn migration and local bird movements.

Each target was stored in the database with a unique target identification code and over 40 variables describing the characteristics of the target (e.g. time of recordings, speed, heading, size, reflectivity). The variables in the Merlin vertical radar database most important for classification analysis are summarized by Rosa et al. (2015) (Table 1). The entire database table can be found in DeTect Inc. (2010).

Table 1. The target variables in the Merlin vertical radar database most important for classification analysis (Rosa et al., 2015).

Name	**Description**
Area	Number of pixels that create the target in the radar image
Ellipse Ratio	Ratio of the major axis of the equivalent ellipse to its minor axis
Ellipse Major and Minor	Total length of the major/minor axis of the ellipse that has the same area and same perimeter as the target
Hydro Radius	Ratio of target area to its perimeter
Maximum Segment	Length of longest horizontal line segment in a target
Perimeter	Length of the outer contour of a target in pixels
Target’s height and width	The maximum height/width of a bounding rectangle in pixels
Waddell’s disk	Diameter of a circle with the same area as the target
Average Reflectivity	Average (mean) reflectivity over the entire target area
Range	Distance or range away from the horizontal radar location to the target
Track length	Number of points belonging to the same bird track
Bearing	Orientation between the radar and the target (> 0 – 360 degrees)
Bearing fitness	Constrains the change in heading a track can make from scan to scan and still be correlated with a new plot. Value ranges from 0 to 1

Table 2 shows the number of reference targets which were selected in Merlin editor and which were used to develop the three DT models.

Table 2. Number of targets in the reference datasets used for the three step decision tree model.

Model 1	sidelobes	yes	64065
		no	67720
Model 2	weather	yes	160026
		no	68841
Model 3	birds	yes	67720
		no	78779

STEP 2 – MODEL BUILDING

The reference datasets were used to develop three decision tree (DT) models which uses discriminating variables to classify different target types (e.g. rain, birds). The first model extracts the sidelobe-interference (Figure 1). The second one filters out the clutter caused by weather (e.g. rain) and the third one extracts the birds from the remaining targets. The analysis was done with SQL server 2008 R2.

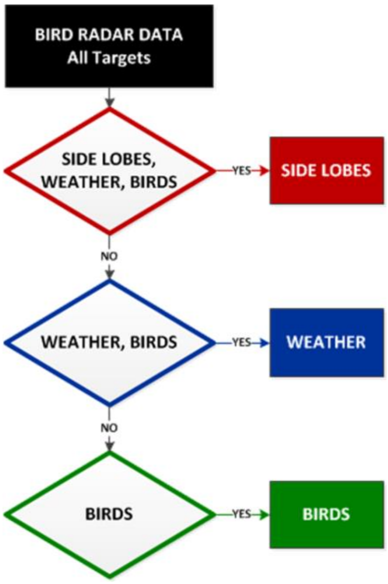


Figure 1. Schematic overview of the three decision tree models which were developed.

STEP 3 – MODEL VALIDATION

The SQL Data Mining models developed in Step 2 were tested with vertical bird radar data which were not used to build the model (i.e. validation dataset). These datasets were visually analysed and then analysed by the DT models, in the order shown in figure 1. The visual analysis was done by a radar expert and he classified the data in side lobes, weather, birds and unknown targets. We validated the model with two test datasets 13 and 17 April 2014. The test data of 17 April 2014 (test 2) contains a lot of rain. On the 13th there was no precipitation.

STEP 4 – MODEL EVALUATION

The results of both analyses (visual and classification models) were then compared to assess the performance and effectiveness of the model on non-reference data. The model performance was assessed based on four parameters: accuracy, sensitivity, specificity and precision (Table 2). These were calculated with a confusion matrix, using the Caret package in R.

Table 3. Model performance assessment parameter equations. TP = true positives, TN = true negatives, FP = false positives, FN = false negatives.

Accuracy	TP + TN / (TP+TN+FP+FN)
Sensitivity	TP/(TP+FN)
Specificity	TN/(TN+FP)
Precision	TP/(TP+FP)

13.3. RESULTS

The number of false positives (i.e. targets which are considered as birds by the model, but which are not) is considerably high (Table 4): 35.6% of the number of true positives in test 1 and 12.9% in test 2. However, the assessment criteria accuracy, sensitivity and specificity are all very high (between 89.4% and 99.2%), both for test 1 and 2 (Table 5). This means that the model effectively filters clutter from the vertical bird radar data, without losing significant numbers of bird targets. This is also shown in visualizations of the data of both tests, before and after application of the model (figure 2 & 3). Figure 3 clearly demonstrates that the model is very effective in removing rain from the data, revealing underlying bird tracks.

Table 4. Model validation test results: birds versus clutter. TP = true positives, TN = true negatives, FP = false positives, FN = false negatives.

	Test 1	Test 2
TP	1609	9294
TN	33831	151648
FP	573	1200
FN	182	133

Table 5. Model performance assessment parameter values.

	Test 1	Test 2
Accuracy	97.91%	99.18%
Sensitivity	89.84%	98.59%
Specificity	98.33%	99.21%
Precision	73.74%	88.56%

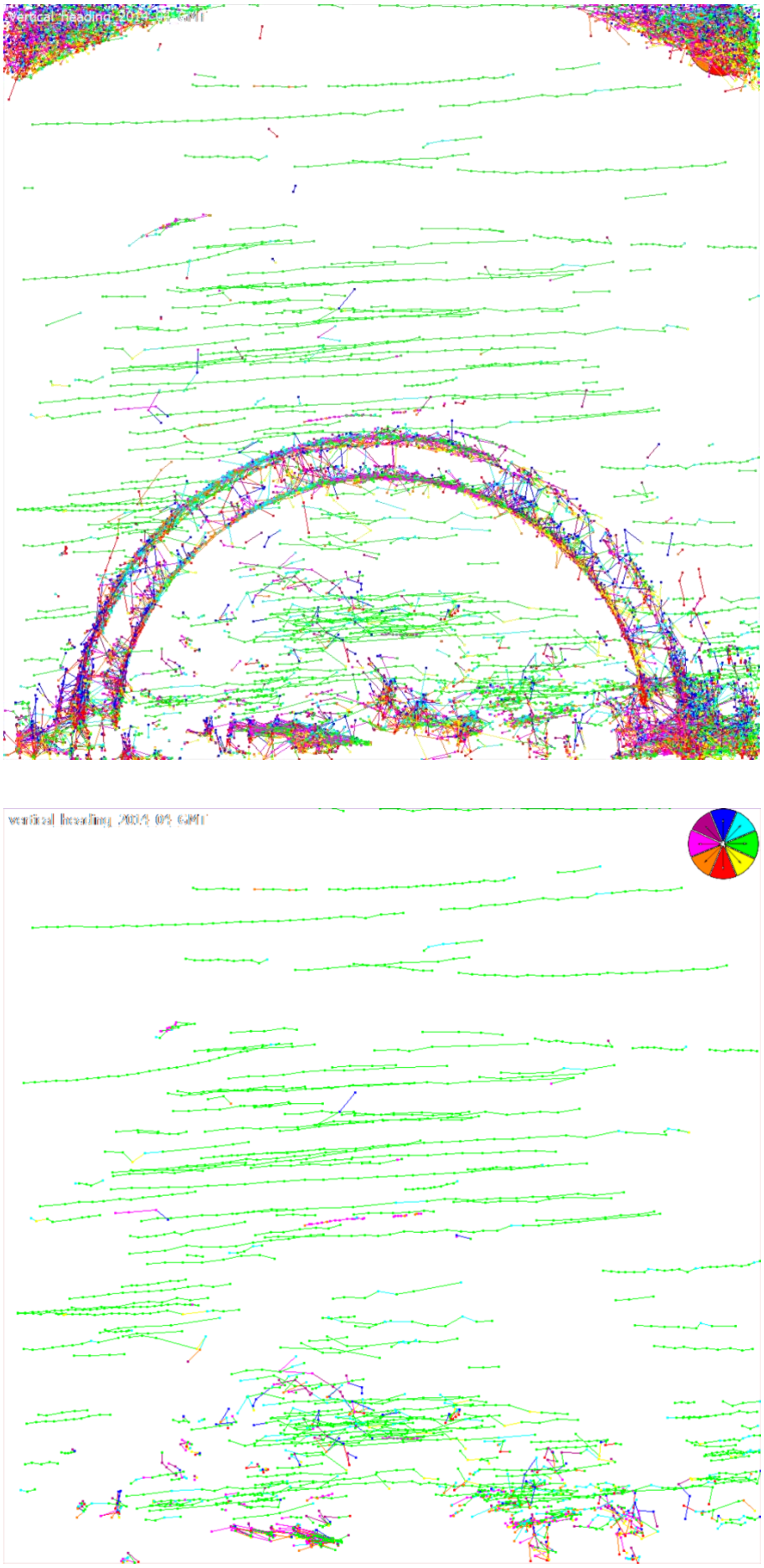
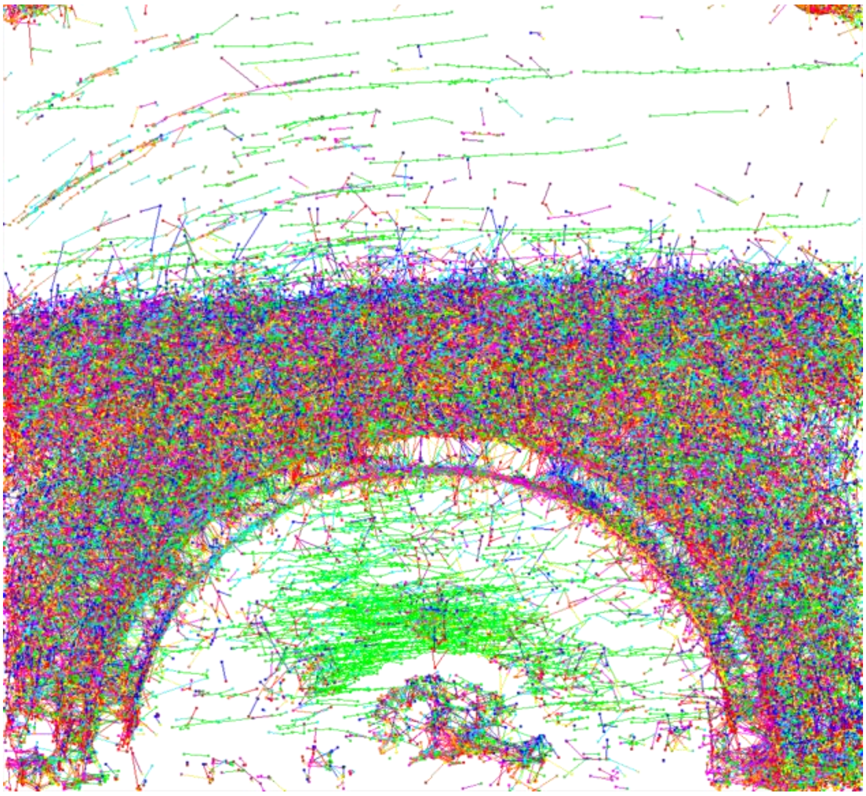


Figure 2. Visualisation of the vertical radar data of April 13th 2014, before and after the implementation of the decision tree models. Top image shows all tracks registered by the vertical radar, the image below shows the tracks which are classified by the model as birds.



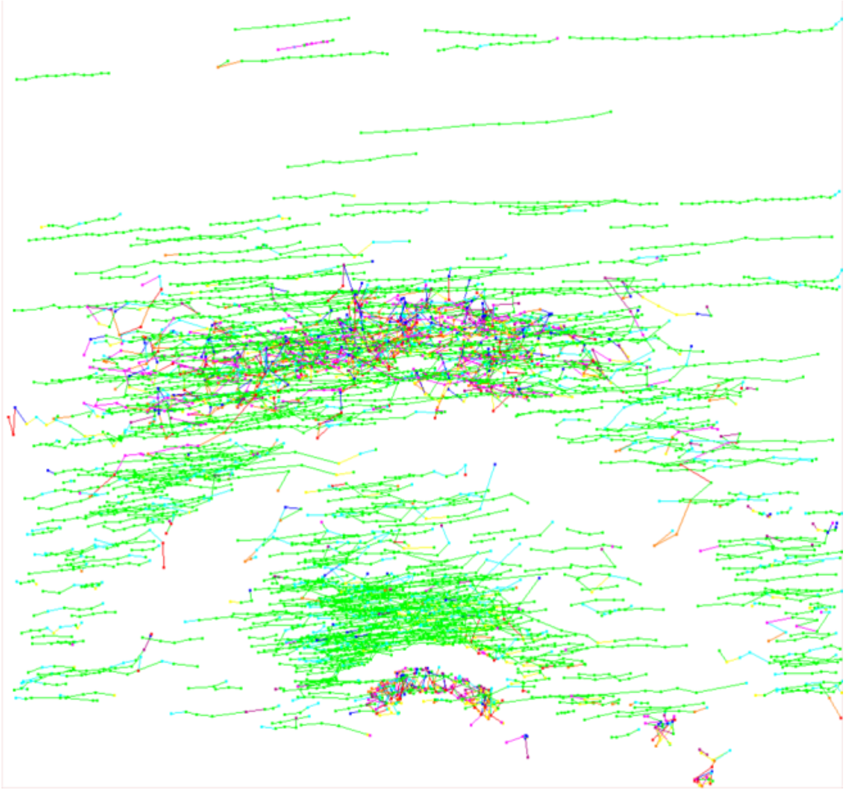


Figure 3. Visualisation of the vertical radar data of April 17th 2014, before and after the implementation of the decision tree models. Top image shows all tracks registered by the vertical radar, the image below shows the tracks which are classified by the model as birds.

13.4. DISCUSSION

Rosa et al. (2015) compared the vlassification success of bird radar data of six machine learning algorithms. The assessment criteria accuracy, sensitivity and specificity they found for the decision tree algoritm are comparable to the rates of this study. However, it is also interesting to assess the model precision as this is a measure for the number of true positives compared to the number of false positives (i.e. targets which are considered as birds by the model, but which are not). Compared to the other model assessment criteria, the precision rate is lower, especially in the first test (Table 5). This means that the model, at its current state is overestimating the number of birds with 35.6% and 12.9% in test 1 and test 2 respectively. As these bird data are used to measure the flux of birds in the wind farm and, in a next step, are then used to estimate the number of collisions of birds with wind turbines, it is important that the model is as precise as possible.

Therefore the model will be improved so the number of false positives is reduced to a minimum and thus the model precision will increase. Before the model, This will be done by including variables in the decision tree analysis which are linked to the track level, instead of only using the variables which describe the single targets, as was the case in the current model. As the heading and the speed of bird tracks is far more consistent compared to the erratic tracks of clutter, the standard deviation of speed and heading of the different targets within a track will be less. Therefore these variables at track level should help to further discriminate birds from clutter.

Once the model is final it will be applied to all historical data and in (near) real-time to the new data. This will result in an improved registration of the bird flux in the wind farm and therefore an improved assessment of the collision risk for birds, based on the bird flux at rotor swept height.

This current model is only applicable on VSR data. It is our aim to also develop a filter for the HSR data, based on a similar approach. The biggest challenge in this process will be to cope with seaclutter.

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14

CHAPTER

CHAPTER 14

BATS IN THE BELGIAN PART OF THE NORTH SEA AND POSSIBLE IMPACTS OF OFFSHORE WIND FARMS

R. Brabant[*1], Y. Laurent[2], L. Vigin[1], R.-M. Lafontaine[2] & S. Degraer[1]*

[1]Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels, Belgium.

[2]Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Conservation Biology, Rue de Vautier 29, 1000 Brussels, Belgium.

[*]Corresponding author: Robin.Brabant@naturalsciences.be

ABSTRACT

Several species of bats in northern Europe undertake seasonal migrations between their summer roosts and wintering areas. Doing so, they are known to cross open sea in some cases. Taking account of the increase of wind farms in the Belgian part of the North Sea and the entire North Sea, the lack of information on the spatio-temporal distribution of bats in Belgian waters and the results of some studies (onshore) demonstrating wind turbines can cause high mortalities in bats, a taxon in global decline, it

is important to quantify the risk of offshore wind farms in the North Sea to threaten bat populations.

To investigate bat distribution, we installed an automated acoustic recorder on the Belgian research vessel ‘Belgica’ to record bats while the vessel is at sea at night. The acoustic detector on the Belgica was operational during 93 nights in autumn 2014 and spring 2015, hence covering two full bat migration periods. In autumn 2014, 117 call sequences were registered in the BPNS, belonging to four different species. In spring 2015, only four sequences were registered, all during one night. The few recordings were all registered during only three nights. These results are not sufficient to solidly determine spatio-temporal patterns of bats in the BPNS, but allow drawing some preliminary conclusions on their frequency of occurrence and distribution at sea.

In 2015 and 2016, a network of nine Batcorders is collecting data in the Dutch and Belgian part of the North Sea and along the coastline. This detector network will increase our knowledge about the impact of offshore wind farms on bats as it will increase the number of detections of bats at sea and will allow direct comparison between data collected at the different locations, without seasonal or meteorological bias. This will allow addressing the question if bats are attracted to or avoid offshore wind farms. This may then lead to appropriate management or mitigation measures.

14.1. INTRODUCTION

Several species of bats in northern Europe undertake seasonal migrations between their summer roosts and wintering areas. Most species only travel short to moderate distances, up to several hundred kilometres per season. However, some species such as Nathusius’ pipistrelle (*Pipistrellus nathusii*), noctule (*Nyctalus noctula*), parti-coloured bat (*Vespertilio murinus*)and Leisler’s bat (*Nyctalus leisleri*) are known to migrate long distances of up to 2000 kilometres from Scandinavia and Central Europe to more temperate regions of western Europe, and back (Arthur & Lemaire, 2015; Hutterer *et al.*, 2005, Krapp & Niethammer, 2011; Dietz *et al.*, 2009).

The fact that bats forage at sea or cross the open sea during migration is well known. Bats have been found regularly in the southern North Sea, e.g. on oil rigs (Bekker & Boshamer, 2008; Russ, 2000; Skiba, 2009; Walter 2007; Brabant & Laurent *et al.*, 2016). Bats were also sighted during seabird surveys (INBO, unpublished data). In 2013, a Nathusius’ pipistrelle specimen banded in the UK, was found in the Netherlands (Leopold *et al.*, 2014). Lagerveld *et al.* (2014) report regular occurrences of bats in the Dutch offshore wind farms. Virtually all recordings of Lagerveld *et al.* (2014) concerned Nathusius’ pipistrelle. Noctules were recorded a few times. Both species are long-distance migrants but also occur as residents at the mainland near the coast. Most migratory activity of the Nathusius’ pipistrelle takes place from mid-August until the end of September (Lagerveld *et al.*, 2014).

Bats collect information about their surroundings by listening to the returning echoes of the sequences of high frequency echolocation calls they produce while flying. These echolocation calls are species-specific and can be used to identify bat species based on parameters from the individual calls (e.g. initial frequency, frequency of maximum energy, end frequency) and call sequence characteristics (e.g. time intervals between consecutive calls).

Ahlèn *et al.* (2009) showed that bats at sea use their echolocation and mostly fly at low altitudes (< 10 m). During migration they are often foraging and they adjust their flight height in response to the altitude of their prey. Moreover, other studies, in Sweden in particular (Ahlen, 2007), indicate that migratory bats regularly feed in the vicinity of offshore wind turbines because of the accumulation of flying insects around the turbines. Non-migratory species have also been reported to use wind farms as feeding sites. Doing so, they face an increased risk of colliding with the turbine blades or of barotrauma caused by rapid air pressure reduction near moving turbine blades (Kunz *et al.*, 2007; Dürr & Bach, 2004; Baerwald *et al.*, 2008).

Contrary to wind farms on land, the number of fatalities in offshore wind farms is very difficult to assess as it is impossible to search and collect carcasses. However, the number of collisions is likely to be lower than onshore (Leopold *et al.*, 2014): (1) at offshore wind farms, nearly all activity is limited to the migration period. At onshore wind farms, bat fatalities also occur outside of the migration period (although in relatively low numbers). (2) Bat activity offshore is generally limited to periods with calm weather suitable for long distance migration. Onshore, bats are recorded during a wider range of weather conditions. (3) Non-migratory bats, such as the common pipistrelle *Pipistrellus pipistrellus*, are virtually absent offshore. Onshore, common pipistrelle is one of the most common species.

Leopold *et al.* (2014) roughly estimate the number of collisions offshore, based on expert opinion, to be somewhere between 0 and 1 fatalities per turbine per year. This is a best educated guess based on the knowledge that fatalities in wind farms in large, open intensively used agricultural areas are typically around 1 fatality per turbine per year (Rydell *et al.*, 2010; Limpens *et al.*, 2013).

Taking account of the increase of wind farms in the Belgian part of the North Sea (BPNS) and the entire North Sea, the lack of information on the spatio-temporal distribution of bats in Belgian waters and the results of some studies (onshore) demonstrating wind turbines can cause high mortalities in bats (Voigt, 2012), a taxon in global decline, it is important to better quantify the risk of offshore wind farms in the North Sea to threaten bat populations.

Therefore this study aims at answering the following questions: (1) what is the distribution and density of the bat species observed at sea? (2) What is the spatial distribution (e.g. on – offshore gradient) and is this distribution species dependent? (3) What are the preferred meteorological conditions for (migrating) bats?

14.2. MATERIALS AND METHODS

To investigate bat distribution, we installed an automated acoustic SM3BAT recorder (wildlife acoustics Inc., Massachusetts, USA) on the Belgian research vessel ‘Belgica’ to record bats while the vessel is at sea at night. The device records the echolocation calls of bats (between 0 and 126 kHz) from shortly before sunset to shortly after sunrise, hence allowing studying the spatio-temporal distribution patterns of bats in BPNS. The Belgica is at sea more than 200 days a year to perform various research activities. During a normal campaign the vessel remains at sea during five days.

The recorder is triggered by the echolocation calls of bats and bat-like sounds. The recordings are saved as sound files on SD cards. . These recordings are used to identify the species present in the area. The results are presented as number of recorded bat call sequences per species.

To level of high numbers of recordings caused by one individual residing near the recorder, the recordings are also converted to detection positive ten minutes (DP10) meaning that a ten minute period is considered as positive if it contains at least one bat call (e.g. a specimen producing 100 calls in 10 minutes and a specimen only calling once are valued in the same way and render one DP10).

The recordings are processed with the software programs SonoChiro (version v3.3.2; Biotope, France) and Batsound (version v1.3.1; Pettersson Elektronik, Sweden) to extract the echolocation calls of bats and to aid the identification to the species level. The identifications were checked and evaluated following the identification criteria of Barataud (2012) and Arthur & Lemaire (2015).

Every registration has a timestamp which is linked to the time and GPS registration of the ship, allowing determining the exact time and location of observation.

To allow spatial analysis of the bat registrations, we calculated the sampling effort, i.e. how many minutes the Belgica was present in a certain area during the study period when the bat recorder was active. Therefore we divided the BPNS in grid cells of two by two kilometers. For each grid cell we calculated the number of minutes the Belgica was present within that cell while the bat recorder was active. The cells are being colour coded accordingly.

Wind speed and wind direction are being measured per ten minutes interval by the Flemish banks monitoring network. We used the data measured at the port of Zeebrugge (www.meetnetvlaamsebanken.be).

14.3. RESULTS

TEMPORAL DISTRIBUTION

The acoustic detector on the Belgica was operational during 48 nights in autumn 2014 (from 1st of September until 30th of November) and 45 nights in spring – summer 2015 (from March 16th until July 17th), hence covering two full bat migration periods. In autumn 2014, 117 call sequences were registered in the BPNS, belonging to four different species (Table 1 and Figure 1). 116 sequences from autumn 2014 were recorded during one single night (18 to 19 September). In spring 2015, only four sequences were registered, all during one night , i.e. 24 – 25 April (Table 1 and Figure 1).

Table 1. Number of bat call sequences per species in autumn 2014 and spring 2015. *Pipistrellus nathusii (*Pip nat*)*, *Pipistrellus pipistrellus (*Pip pip*)*, *Vespertilio murinus (*Ves mur*)*, *Myotis daubentonii (*Myo dau*)*, non-identified bat species (NI). The row 'DP10' indicates the number of 'detection positive 10 minutes', this is the number of 10 minute intervals wherein at least one call of a certain species was recorded.

Date	**Pip nat**	**Pip pip**	**Ves mur**	**Myo dau**	**NI**
18-19/09/2014	21	93	1	0	1
23/09/2014	0	0	0	1	0
24/4/2015	4	0	0	0	0
DP10	17	6	1	1	1

The 93 registered sequences of the common pipistrelle were all made during only 53 minutes (DP10 = 6) when the vessel was fairly close to the coast (ca. 5 km). The DP10 value for the Nathusius' pipistrelle was 17, meaning that the recordings for that species were more spread out over a longer period of time.

Besides the two pipistrelle species, we also registered call sequences of Daubenton's bat *Myotis daubentonii* and parti-coloured bat *Vespertilio murinus*.

The night of 18 to 19 September 2014 was a clear night with low wind speeds (average: 1.6 m/s; figure 1). During the night of 23 to 24 September 2014, wind speed was 5.3 m/s on average. In spring 2015, bat calls were registered during one night only with an average wind speed of 4.0 m/s. The average wind speed during the entire measuring period in autumn 2014 was 5.2 m/s, with a maximum and a minimum wind speed of respectively 20.6 m/s and 0.1 m/s. In spring 2015 the average was 7.0 m/s, with a maximum and minimum of 26.6 and 0.1 m/s.

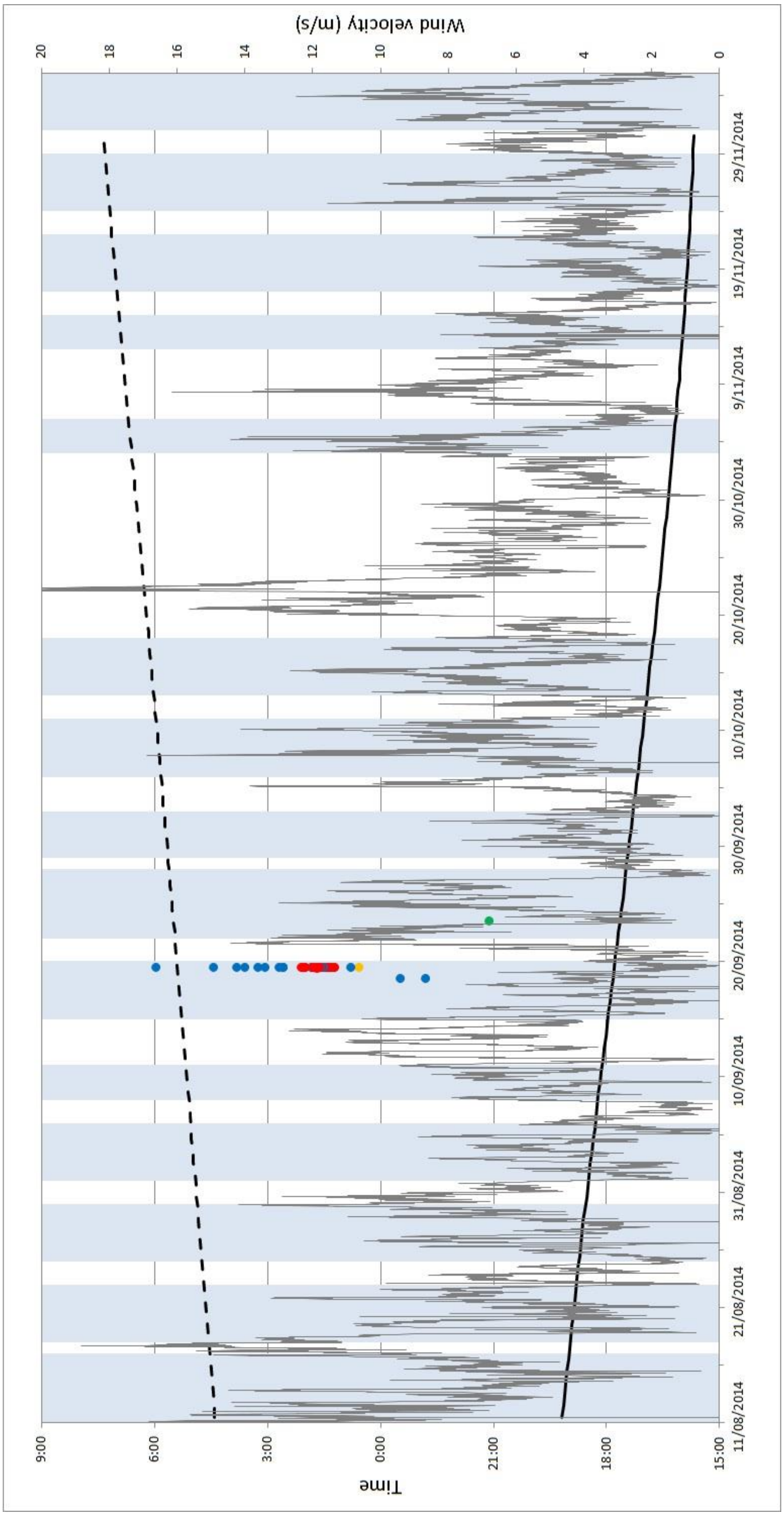

Figure 1: Number of call sequences per species (red: *Pipistrellus pipistrellus*; blue: *Pipistrellus nathusii*, yellow: *Vespertilio murinus*; green: *Myotis daubentonii*) registered by the SM3 songmeter, and wind speed (m/s; grey line) in autumn of 2014. The periods during which the Belgica was at sea are shown in grey. The time of sunset and sunrise is indicated by the black and dotted line, respectively.

SPATIAL DISTRIBUTION

Figure 2 indicates that the sampling effort was much larger in certain areas compared to others. This is especially the case in the coastal waters near the ports of Zeebrugge, Oostende and Nieuwpoort. The area around Zeebrugge is the area where most call sequences were registered. Furthermore, it is clear that the Belgica regularly visited the sand and gravel extraction zone (in the west of the BPNS) and the Thorntonbank (to the west of the area reserved for electricity production). Although the survey intensity was similar to the waters around Zeebrugge, no bats were registered in those areas.

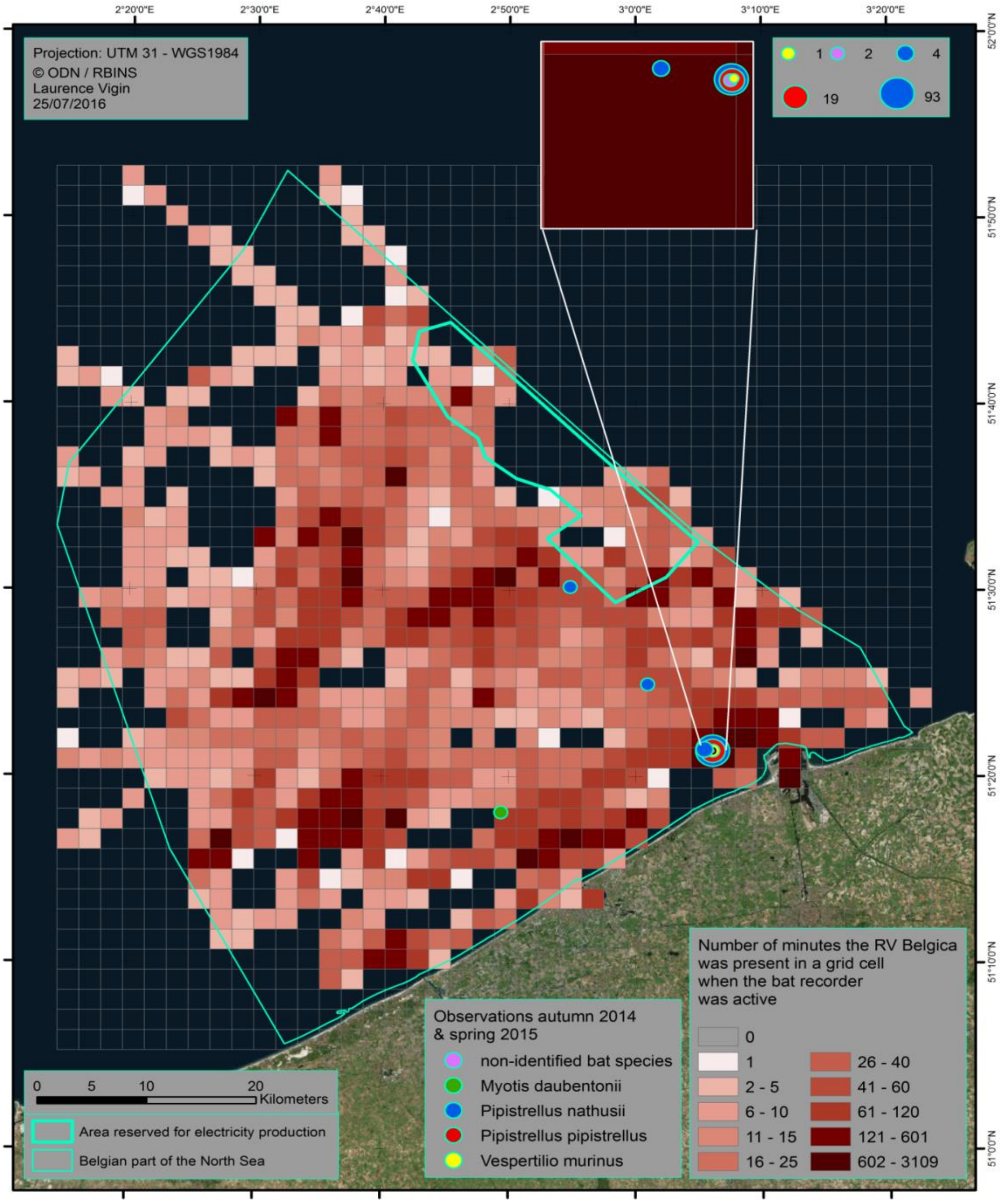

Figure 2. Location of the bat registrations in the Belgian part of the North Sea in autumn 2014 and spring 2015. The color code in the grid cells indicates the number of minutes the Belgica was present in that grid cell (2x2 km) when the bat detector was active in autumn 2014 and spring 2015.

14.4. DISCUSSION

GENERAL

Although the bat recorder was operational during 93 nights in autumn 2014 and spring 2015, we had very few recordings (121 sequences), which were all registered during only three nights. These results are not sufficient to solidly determine spatio-temporal patterns of bats in the BPNS, but allow drawing some preliminary conclusions on their frequency of occurrence and distribution at sea.

The Nathusius' pipistrelle was the most frequent species encountered during our study. We recorded 25 sequences during two nights (DP10 = 17). Nathusius' pipistrelles were recorded at 5, 12 and even 25 km from the coast. These findings of bats at sea correspond to the known fact that this species is a long distance migrant (Arthur & Lemaire, 2015; Hutterer *et al.*, 2005, Krapp & Niethammer, 2011; Dietz *et al.*, 2009) and to the bat registrations in the Dutch offshore wind farms where 98% of all sequences were identified as Nathusius' pipistrelle (Lagerveld *et al.*, 2014). Although the number of registered sequences of the common pipistrelle is 93, this was most likely only one specimen which was attracted by the ship and resided in its vicinity for about one hour when the vessel was fairly close to the coast (ca. 5 km). This is also refelected in the DP10 value for that species, which is only 6.

In certain areas in the BPNS where the sampling effort was high (figure 2), no bats were detected. Possibly, the weather conditions when the Belgica was present in those areas at night were not favorable for bat activity. During the night of 18–19 September 2014, when we recorder the most bat activity, the Belgica stayed at the same location so we do not know if there were bats present in other areas as well at that time. The locations of the bat registrations suggest there are more bats present near- versus offshore, but the data at hand are to scarce to allow to demonstrate this with certainty.

The average wind speed during the nights of the bat recordings was low, 1.6, 4.0 and 5.3 m/s, respectively coinciding with what was found in earlier studies, e.g. Lagerveld *et al.* (2014). Their findings of bats at sea in calm weather conditions resulted in the current mitigating measure for the Borssele offshore wind farm concessions (see chapter 2) in the Dutch part of the North Sea, stipulating that the cut-in wind speed[1] of wind turbines should be set at 5 m/s from August 15 until September 30 (i.e. main bat migration period). According to Eurobats (2014) the use of blade feathering[2], a higher turbine cut-in wind speed and shutting down turbines are the only mitigation measures which so far proved to be effective in reducing wind turbine-induced bat mortality.

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[1] The minimum wind speed at which the wind turbine will generate usable power

[2] Adjusting the angle of the rotor blade parallel to the wind, or turning the whole unit out of the wind, to slow or stop blade rotation

EVALUATION OF THE STUDY DESIGN:

The installation of the bat recorder on the vessel resulted in large amounts of noise in the sound files. This noise is generated by the vessel and saturated the batcorder in the low frequencies. Depending on the activity of the vessel (e.g. sailing or anchored) the frequency went up to 30kHz, which is already overlapping with the frequencies of the calls of certain bat species (e.g. Vespertilio murinus). So possibly, the noise generated by the vessel masks out certain bat registrations. In the future we will test different locations on the vessel to install the microphone, to minimize the noise in the data.

FUTURE RESEARCH:

Our preliminary results showed that an increased sampling effort is needed to get a representative view on the spatio-temporal distribution of bats at sea. The same holds true to study the impact of offshore wind farms on bats. To that extent, the recorder will remain installed on the research vessel Belgica from at least mid-March until the end of October. This will increase our general knowledge about the spatio-temporal distribution of bats at sea on the wider scale, i.e. the scale at which the Belgica operates (e.g. potential preferential routes, grouping sites, coastal migrations pathways).

Additionally, we recently started collaborating with the Dutch research institute IMARES and the Flanders Marine Institute (VLIZ). In 2015 and 2016, a network of nine identical IMARES recorders (Batcorder, EcoObs) collects data in the Dutch and Belgian part of the North Sea and along the coastline. This network is complemented with two Batcorders of VLIZ in the framework of the Lifewatch project (www.lifewatch.be). Hence, a total of eleven Batcorders are now operational in front of the Belgian and southern Dutch coastline (figure 3). They are installed on platforms inside wind farms, other platforms and along the coastline. The two recorders which are mounted offshore in the BPNS are on a turbine of C-Power (Lifewatch) and on Belwind's high voltage station (IMARES). The second Lifewatch Batcorder is installed along the Belgian coast, in Oostende. The recorders were configured identically to maximize the comparability of the data. These detectors will be active throughout the entire period when bats are active, i.e. from mid-March until the end of October 2015 and 2016.

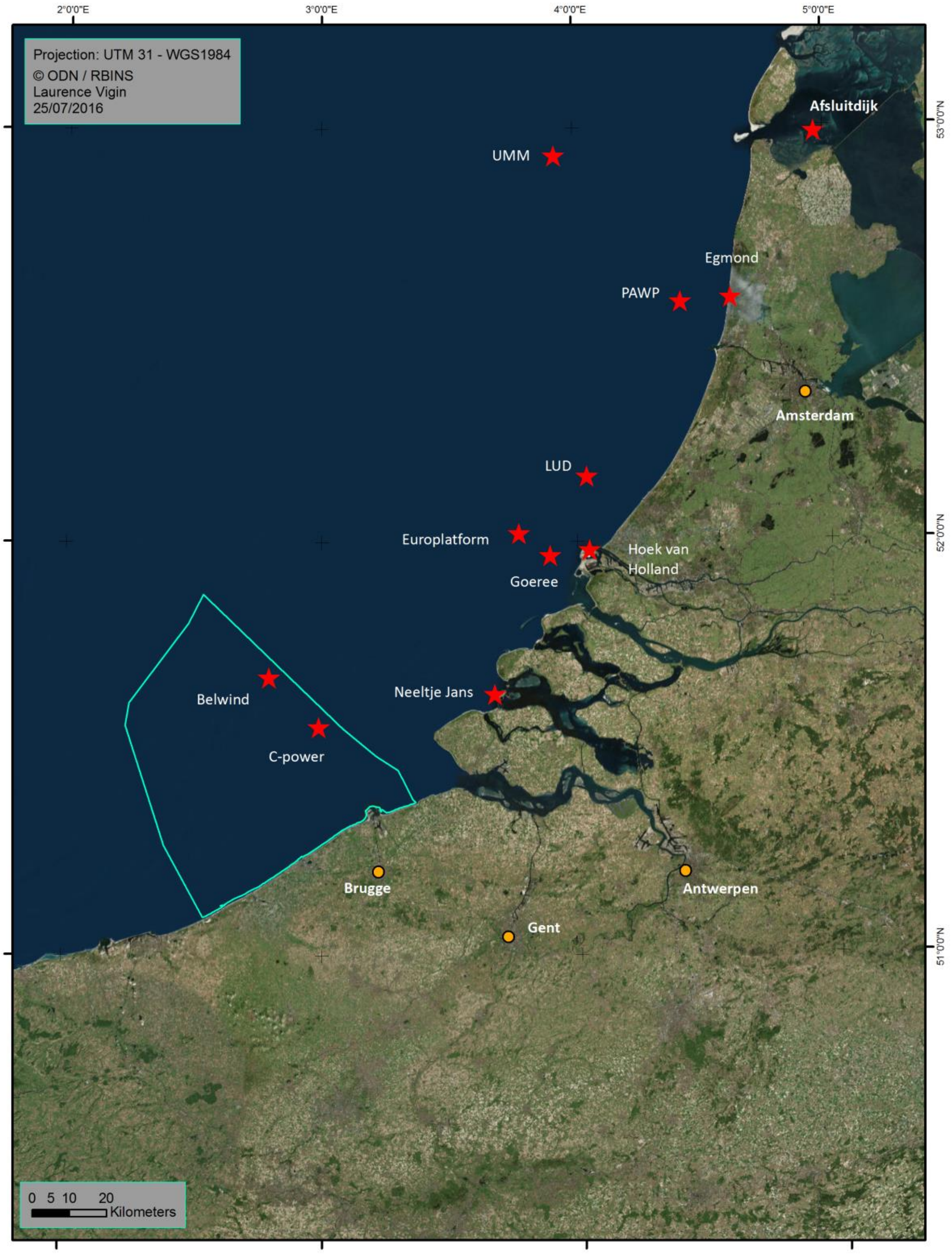

Figure 3. Batcorder network in the Belgian and Dutch part of the North Sea

This research mainly focuses on the presence of bats at sea, how the North – South and onshore – offshore gradients influence the density of bats and how this compares to the presence of bats in offshore wind farms. This detector network will also increase our knowledge about the impact of offshore wind farms on bats as it will increase the number of detections of bats at sea and will allow direct comparison between data collected at the different locations, without seasonal or meteorological bias. This will allow addressing the question if bats are attracted to or avoid offshore wind farms. This may then lead to appropriate management or mitigation measures.

In a later stage of the study (foreseen to start in 2017), we will also look into bat behaviour inside the wind farms. For such behaviour study, two bat recorders will be installed per wind turbine as measurements will have to be made at different altitudes, in order to determine the exact flying height of bats. This will give a better understanding of the activity of individuals, detect particular behaviour (e.g. display, foraging) and the risk associated with that behaviour (collision risk, barotrauma).

The use of other methodologies to investigate bat behaviour inside offshore wind farms and the associated risk (e.g. high resolution IR camera, radiotelemetry) will be investigated and considered in 2016 and 2017.

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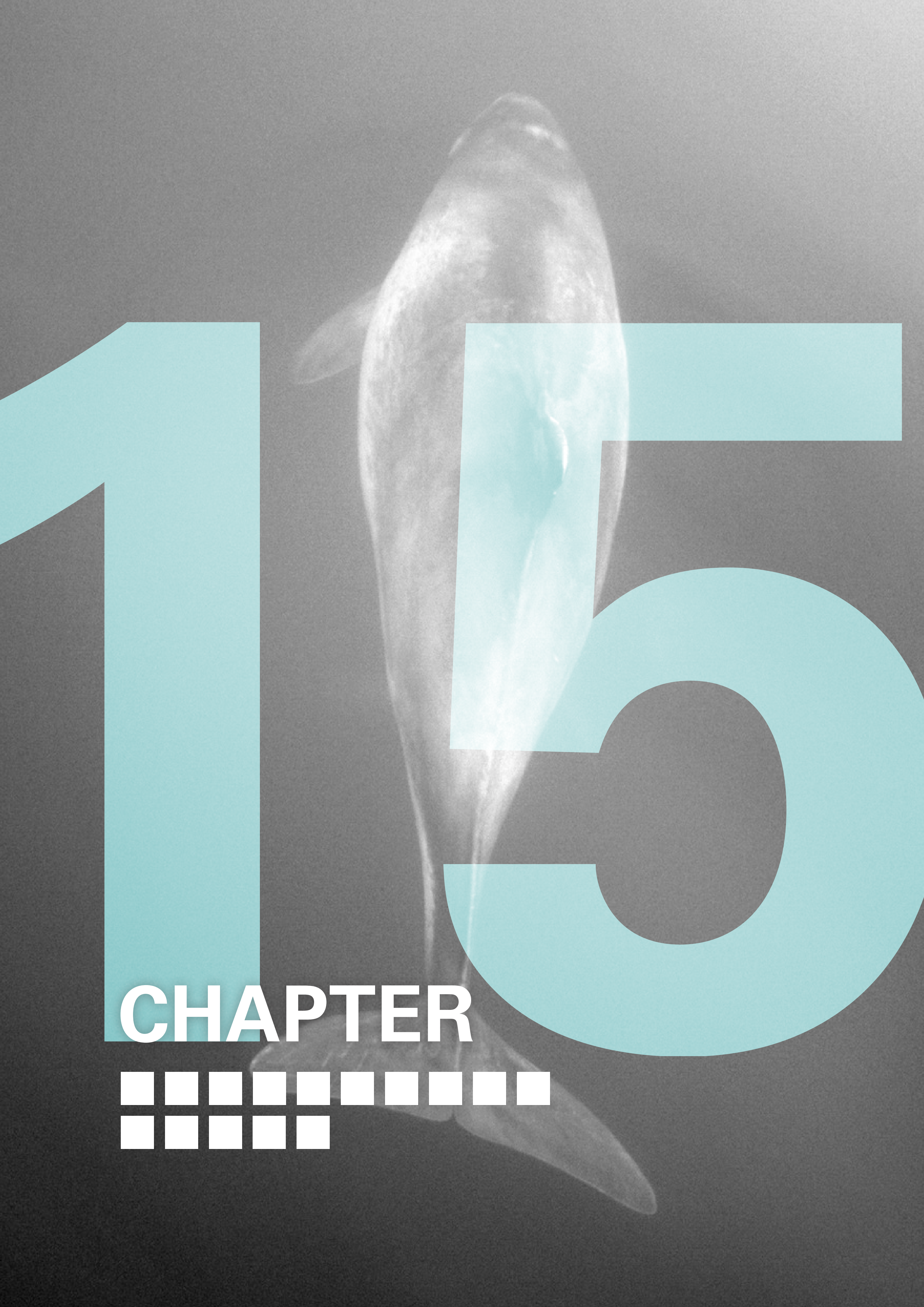

CHAPTER 15

CHAPTER 15

SEASONAL AND INTERANNUAL PATTERNS IN THE PRESENCE OF HARBOUR PORPOISES (*PHOCOENA PHOCOENA*) IN BELGIAN WATERS FROM 2010 TO 2015 AS DERIVED FROM PASSIVE ACOUSTIC MONITORING

J. Haelters[*1], B. Rumes[2], J. Vanaverbeke[2] & S. Degraer[2]*

[1]*Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment, Management Unit of the North Sea Mathematical Models (MUMM), 3de en 23ste Linieregimentsplein, 8400 Ostende, Belgium.*

[2]*Royal Belgian Institute of Natural Sciences (RBINS), Operational Directorate Natural Environment (OD Nature), Aquatic and Terrestrial Ecology (ATECO), Marine Ecology and Management Section (MARECO), Gulledelle 100, 1200 Brussels, Belgium.*

[*]*Corresponding author: Jan.Haelters@naturalsciences.be*

ABSTRACT

The harbour porpoise (*Phocoena phocoena*) is the most abundant cetacean in the Belgian part of the North Sea. We developed a mooring system for static passive acoustic monitoring (PAM) of this species using c-PoDs at locations of opportunity. Data of moorings between 2010 and 2015 at two locations were analysed. They revealed a significant seasonal trend in detections, assessed by month, with peaks in late winter - early spring and late summer, consistent with the results of aerial surveys and with strandings data. At one location there were significant differences in detections between years, with higher detection rates in every year between 2011 and 2014, and the highest detection rates in 2013 and 2014. The experiences gained are used to design a subsequent study strategy to monitor harbour porpoise presence in Belgian waters, including possible effects on their presence due to the construction of offshore windfarms.

15.1. INTRODUCTION

The elusive and highly mobile harbour porpoise (*Phocoena phocoena*) is the most abundant cetacean in the Belgian part of the North Sea (BPNS). Aerial surveys revealed that average densities in these waters range from 0.2 to 4 animals km^{-2} (Haelters et al., 2013; 2015; data RBINS, unpublished), totalling from a few hundred up to more than 10.000 porpoises (or in the latter case more than 3% of the best North Sea population estimate; Hammond et al., 2013). The harbour porpoise should thus be considered as a significant top of the food chain constituent in the BPNS.

Dedicated monitoring of harbour porpoises in Belgian waters started with aerial surveys (Haelters, 2009), with as their main goal to assess the reference situation prior to, and to study the impact of the construction and operation of offshore windfarms. Aerial surveys continue up to date, and demonstrated that porpoise density shows a seasonal pattern in Belgian waters and that concentration areas of porpoises occur (Haelters et al., 2011a; 2013).

As aerial surveys could only be performed with a low temporal resolution (five at the most per year), it is possible that changes in density and distribution in between surveys were missed. Also, due to short daylight time and frequent adverse weather conditions, as of yet no aerial surveys were undertaken between late autumn and late winter. Therefore, a project was set up to complement information generated through aerial surveys with data from continuous passive acoustic monitoring (PAM) as soon as a suitable and affordable PAM system was available. PAM, using autonomous devices that are placed at a fixed location for weeks to months generates data with a high temporal, but low spatial resolution (Au, 1993; Tregenza, 1999; Mellinger et al., 2007).

In this report we describe the results of the PAM study of harbour porpoises in Belgian waters between 2009 and 2015. We first developed and assessed suitable systems for mooring PAM devices on locations of opportunity. Using the data collected, we investigated whether temporal trends in harbour porpoise presence within and between years can be detected. The experiences gained are used to develop a subsequent strategy to monitor harbour porpoise presence in Belgian waters, including possible effects on their presence due to the construction of offshore windfarms.

15.2. MATERIAL AND METHODS

PODS

The only PAM device that was used between 2009 and 2015 was the Continuous Porpoise Detector (C-PoD, further indicated as PoD). PoDs consist of a hydrophone, a processor, batteries and a digital timing and logging system. They continuously monitor sounds between 20 kHz and 160 kHz, and can detect all odontocetes except sperm whales (*Physeter macrocephalus*). A PoD does not record sound itself, but compresses data, generating a raw file with for each click characteristics such as its time of occurrence, duration, dominant frequency, bandwidth and sound pressure level. Using dedicated software, the raw file can be objectively analysed to find click trains and to classify these into a.o. trains produced by odontocetes and trains that originate from other sources such as boat SONAR. Distinction can be made between harbour porpoises, a species producing narrow-band, high-frequency clicks, and dolphins, producing more broadband clicks with a lower frequency. The maximum detection range for porpoises is approximately 400 metres. PoDs have an autonomy of up to 200 days (www.chelonia.co.uk).

POD MOORING SYSTEMS

The moorings used in this study were mostly moorings of opportunity, using existing platforms: tripods and navigational buoys. Tripods are heavy structures moored on the seafloor. Their presence is indicated by a surface marker buoy, also used to retrieve it. Next to a PoD attached to the central (vertical) column at 1.5 m above the seafloor, the tripods mostly had also other oceanographic instruments attached to them (Van den Eynde et al., 2010) (Figure 1). A mooring system using existing navigational buoys was developed, leading to the concealment of the PoD in a lead-weighted stainless steel container (leaving the hydrophone exposed). This system was hung free from the buoy with a stainless steel chain at approximately 1.5 m below the water surface. The chain was protected with rubber hosing in order to limit chain rattling and prevent damage to the coating of the buoy (Figure 1). In two cases, a PoD was attached to a weight on the seafloor, where it hung free on a rope at around 1.5 m from the seafloor, using its positive buoyancy. These moorings were recovered using divers. Finally, a ‘stealth’ mooring system was tested; it consisted of a stone weight and a Danforth anchor separated by a 40 m long, stretched, bottom rope, and the PoD attached to the weight. While tests in shallow waters to recover the system using a grappling anchor were successful, the only time it was effectively used was unsuccessful, and the PoD was lost.

Figure 1. Prevailing mooring systems used: navigational buoy (left) and tripod, in combination with other oceanographic instruments (right) (images: RBINS).

POD MOORING LOCATIONS

Between 2009 and 2015 we performed 101 moorings of PoDs near the edge of territorial waters in the eastern (Thorntonbank, Gootebank, Bligh Bank) and western part of Belgian waters (Oostdyck Bank), and a few km off the coastal town of Blankenberge (MOW1; Table 1; Figure 2). The goal was to have, continuously, PoDs present at 2 to 3 locations. The locations were predominantly chosen as a function of the availability of a mooring of opportunity, and the distance to an offshore wind farm area. Between 2010 and 2015, mooring locations changed due to shifts in the position or presence of navigational buoys and the deployment of tripods dedicated to other research objectives.

Table 1. Mooring types and location of PoDs ; the locations are precise within a few hundred meters due to tides displacing buoys and the fact that the mooring of the tripods was made within that margin. Distance to the coast was measured to the beach, and does not take account of the harbour of Zeebrugge.

Location	**Type of mooring**	**Lat (°)**	**Lon (°)**	**Water depth vs. MLLWS (m)**	**Distance to the coast (km)**
MOW1	Tripod	51.356667	3.116667	7.3	3.7
Thorntonbank	Buoy	51.590333	3.005083	26.8	32
Thorntonbank	Buoy	51.566667	2.912917	26.7	31
Thorntonbank	Steel weight	51.543333	2.930000	21.5	28
Oostdyck W	Buoy	51.285833	2.438667	24.6	22.4
Gootebank	Buoy	51.449217	2.878717	23.8	21.3
Gootebank	Tripod	51.448100	2.876450	24.5	21.3
Bligh Bank	Stealth	51.711850	2.816533	29.6	49
Bligh Bank	Tripod	51.703333	2.813333	26.6	48

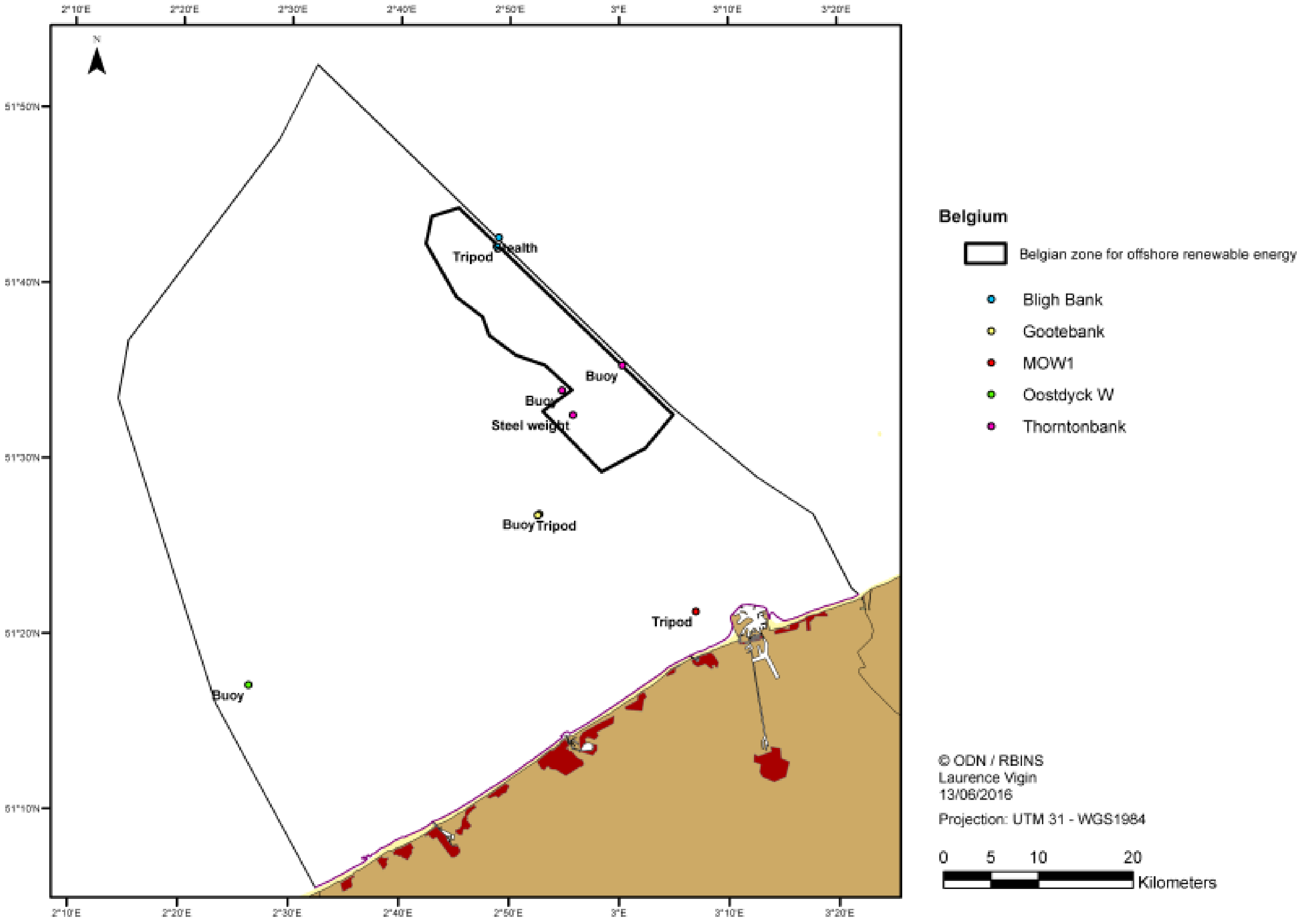

Figure 2. Location of PoD moorings

DATA ANALYSIS

The data were analysed using CPOD.exe software version 2.043. Of the four levels of confidence (quality) of the data, only high and moderate train quality was used, with the species filter set to harbour porpoises. Data were exported and further analysed using Excel and R-software (R Development Core Team, 2016). Different measures were initially used to describe harbour porpoise presence:

- Detection Positive Minutes per day (DPM day^{-1}): the number of minutes in a day in which harbour porpoises were detected;
- Time Present per day (TP day^{-1}; in seconds): cumulative duration of trains per day.

Both measures have their value: in case animals move quickly, and stay at one location for only a short time, more encounters (~DPM) would be recorded than if they would move slowly. The cumulative duration of trains (TP) would however remain more constant at different swimming speeds.

Data were treated per mooring, which lasted from two weeks to more than five months, yielding useful data for up to 143 days.

High levels of ambient noise interfere with the ability of a PoD to detect odontocetes in two ways: they mask clicks, and they use up the limited amount of data that can be stored per minute (resulting in % of time lost). In comparing data of 82 moorings, on average 95.2% of the minutes could be used for moorings on tripods, while 83.8% of the minutes could be used for moorings on buoys. This figure increased to 95.7%, respectively 87.4% when including minutes with up to 20% saturation. The minimum number of minutes that showed no saturation in a tripod system was 74.6%, while it was 55.3% in a PoD moored on a buoy (Figure 3).

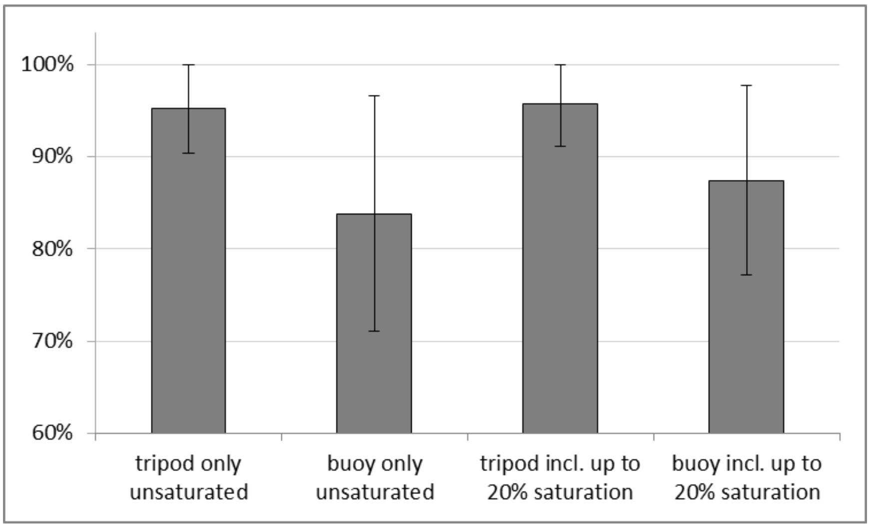

Figure 3. The percentage of minutes without saturation and those including saturation up to 20% that could be used differed between the two main mooring systems (including Standard Deviation).

Given the relatively high level of minutes showing saturation a a number of moorings, data were further treated as follows:

- All minutes with SONAR risk and/or continuous noise detected with the software, were omitted from the analysis.
- All minutes were included, except those with more than 20% time lost. While in theory not necessary to include minutes with up to 20% time lost for most of the files, this was done in order to treat all files in a standardised way.
- Days in which data for less than 50% of the total number of minutes/day were available, were omitted from the analysis.

When using minutes with time lost, the TP for each minute was corrected proportionally. Also when presenting DPM day^{-1} and TP day^{-1}, data were corrected proportionally with the minutes that were considered. As the temperature recording in the PoDs had not been calibrated, it was not used further. Instead, reliable sea surface temperatures were used for the Oostdyck W location and MOW1 (data extracted from http://marine.copernicus.eu).

For the two stations with data available from a sufficiently long period over multiple years (MOW1 and Oostdyck W), statistical modelling was performed on the DPM day^{-1} to describe seasonal trends in porpoise detection. TP day^{-1} was not used further for the statistical analysis, as there did not seem to be important deviations from a parallel track between TP day^{-1} and DPM day^{-1} (Annex 4). Preliminary data analyses revealed, as could be expected, strong autocorrelation when using total DPM day^{-1} as response variable. Therefore observations per day were pooled per month, providing a proxy for harbour porpoise detections per month at each station. Available predictors included 'year', 'month' and 'temperature'. As 'temperature' was strongly collinear with 'month', only month and year were used for the final analysis.

The continuous variable *month* was used to model seasonal fluctuations by fitting a cyclic sine curve, described by a linear sum of sinus and cosinus terms (Stewart-Oaten & Bence, 2001, Onkelinx et al. 2008, Vanermen et al. 2015). In order to allow multiple peaks in detections per year, several start formulations of the model were tested:

- TotalDPM~offset(days) + sin(2*pi*(Month/12)) + cos(2*pi*(Month/12)) + factor(Year)
- TotalDPM~offset(days) + sin(2*pi*(Month/12)) + cos(2*pi*(Month/12)) + sin(2*pi*(Month/6)) + cos(2*pi*(Month/6)) + factor(Year)
- TotalDPM~offset(days) + sin(2*pi*(Month/12)) + cos(2*pi*(Month/12)) + sin(2*pi*(Month/4)) + cos(2*pi*(Month/4)) + factor(Year)

The 'offset(days)' term takes into account the different length of the months and the number of mooring days per month. Based on AIC, the best model was determined, and further model selection was performed based on a backward selection with AIC as decision criterion. However, plots of residuals versus fitted values clearly indicated heterogeneity of variances. Therefore, we adopted a linear regression with generalized least-square extension (Zuur et al., 2009), which allows unequal variances among treatment combinations to be modeled as a variance-covariance matrix (West et al., 2006; Pinheiro & Bates, 2009).

Following West et al. (2006) and Zuur et al. (2009), the most appropriate variance-covariate matrix was determined using AIC scores in conjunction with plots of fitted values versus residuals with different variance-covariate terms relating to the independent variables, using restricted maximum-likelihood (ML) (REML, West et al., 2006). This procedure resulted in the use of a variance structure that allowed for different variances per stratum for 'year' or 'month' for the analysis of the data for MOW1 and Oostdyck W respectively (varIdent function, R package nlme). Once the appropriate random component had been determined, the fixed component of the model was refined by manual backwards stepwise selection using ML to remove insignificant variable terms.

No account was taken of windfarm construction activities during the period of the study. Effects on the presence of harbour porpoises during pile driving could have been present at all sites, and with a high level of certainty negative effects occurred at the mooring locations closest to the pile driving sites (Haelters et al., 2015). It has been demonstrated that piling can have effects on harbour porpoise presence up to distances of more than 20 km away from pile driving sites (Nedwell et al., 2003; Carstensen et al., 2006; Tougaard et al., 2009; Brandt et al., 2011; Murphy et al., 2012; Dähne et al., 2013; 2014; Haelters et al., 2015). However, possible negative or positive effects were not considered for Oostdyck W and MOW1 as these locations were respectively 40 and 23 km away from the nearest pile driving site and as piling was limited in time vs. the total PoD mooring time.

15.3. RESULTS

MOORING SYSTEMS AND DURATION

When only including periods yielding useful information (excluding lost PoDs or the periods with no data collected, e.g. due to batteries that ran out), PoDs yielded data for a total duration of 4,575 days between 2009 and 2015. The total number of days of moorings yielding useful information varied between locations (from 208 days at the Bligh Bank to 1,912 days at MOW1) and between years (Figure 4). Excluding 19 PoD moorings that did not yield data, the 47 PoD moorings on tripods yielded on average 46 days of data (10-143), while the 35 PoD moorings on buoys yielded on average 68 days of data (15-139).

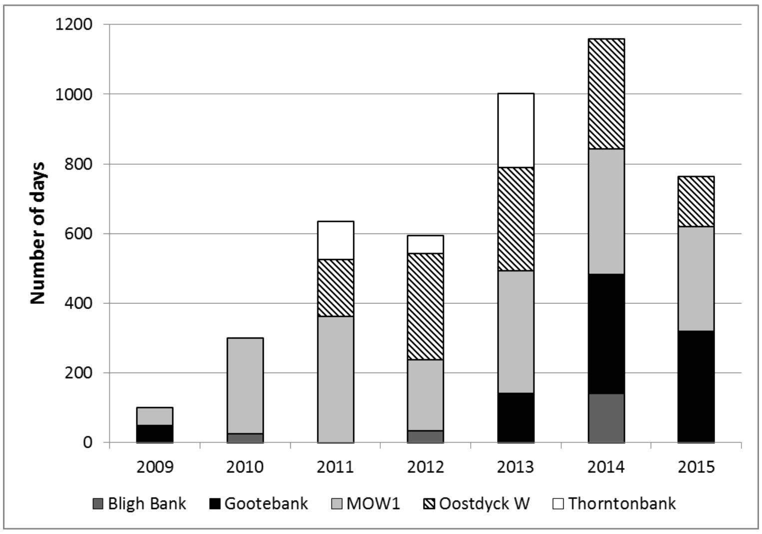

Figure 4. Moorings of pods (days) per location and per year; only days yielding useful information are included.

During the study, 7 PoDs were lost (including a buoy moored PoD that was later recovered in Denmark, and that still contained data) on a total of 101 moorings: 1 from a tripod (the whole tripod was lost), 5 from buoys and 1 from the stealth mooring system. The highest loss rate occurred in buoy moored PoDs: 5 losses out of 40 moorings (12.5%), vs. 1 out of 58 moorings (1.7%) in PoDs mounted on tripods. After the loss of PoDs that were fastened to buoys with a stainless steel wire, the wire was replaced by a stainless steel chain, but a few losses still occurred. Data collection without the PoD getting lost was unsuccessful in 13 moorings, including in the PoDs moored on the steel anchor weight (2).

TEMPORAL CHANGES IN DETECTION RATE

An overview of the raw data (average corrected DPM and TP per week and per month, and average DPM and TP per month split up into years are taken up in Annex 4 (Figures a-d).

For the statistical analysis, only DPM data from 2010 to 2015 were used, given the limited data available for 2009. At both mooring locations for which the PAM data were analysed (Oostdyck W and MOW1), there was a significant seasonal trend in DPM day^{-1}, assessed by month, with a peak in the detection rate in late winter – early spring and a smaller one in late summer (Figure 5). Only at MOW1 there were significant differences in DPM day^{-1} (aggregated per month) between years, compared to 2010, with higher detection rates in every year between 2011 and 2014 (Figure 6). The highest detection rates occurred in 2013 and 2014.

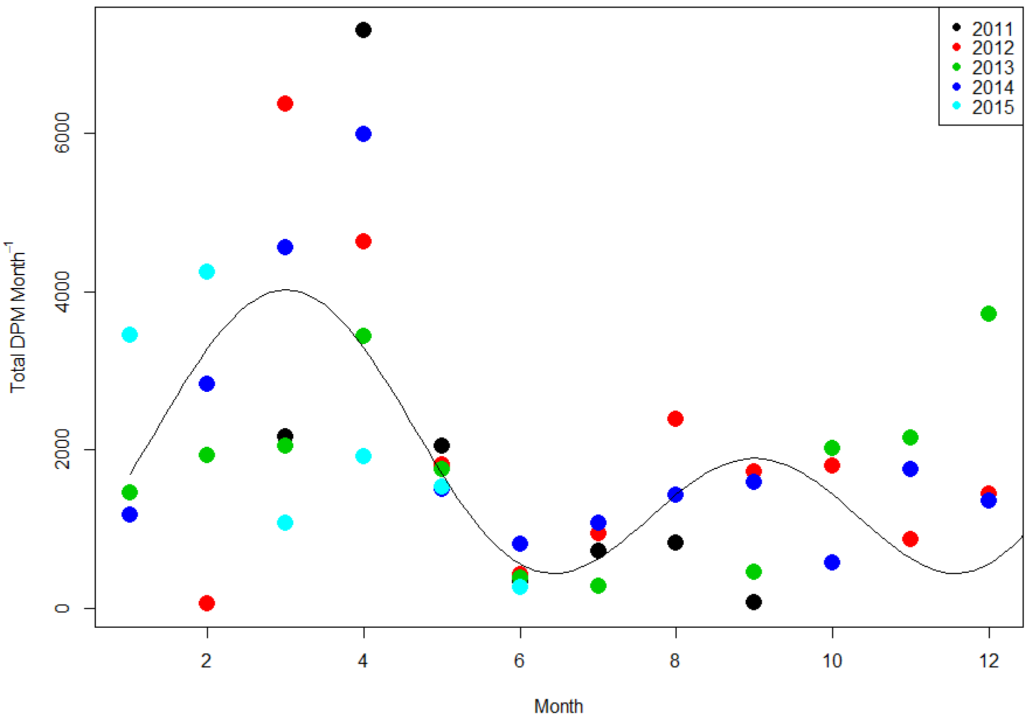

Figure 5. Model output (DPM month^{-1}) of seasonal trend at Oostdyck W.

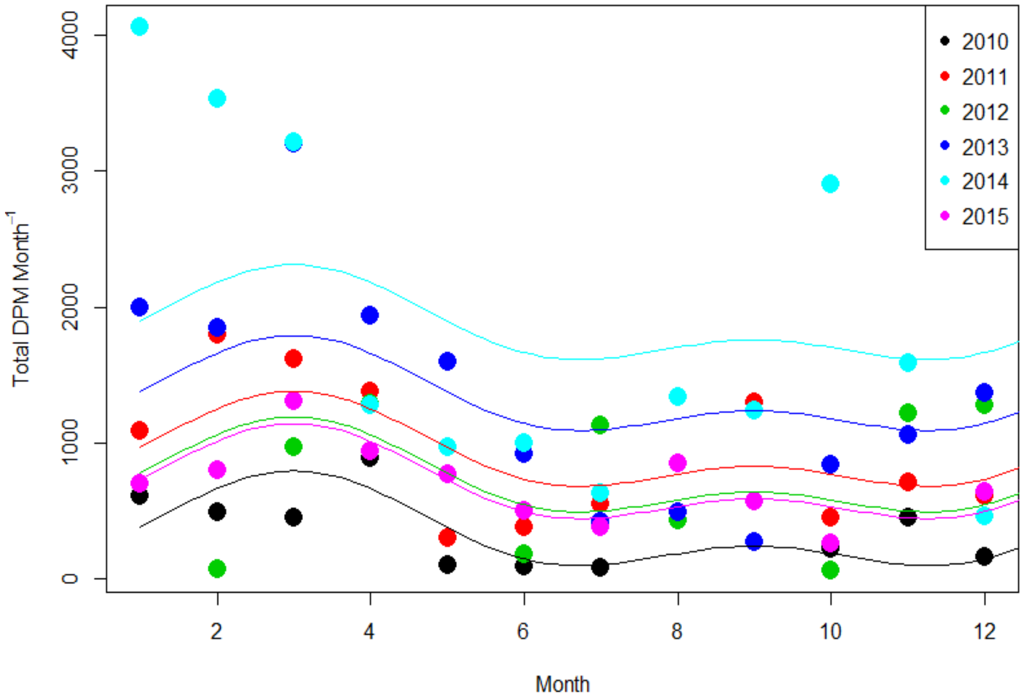

Figure 6. Model output (DPM month^{-1} year^{-1}) of seasonal trend at MOW1.

DIFFERENCES IN DETECTION RATE PER LOCATION

The detection rates at the Oostdyck W location were in general higher than at MOW1, with per month on average more than twice as many DPM day^{-1} and seconds TP day^{-1} (Figure 7; Annex 4).

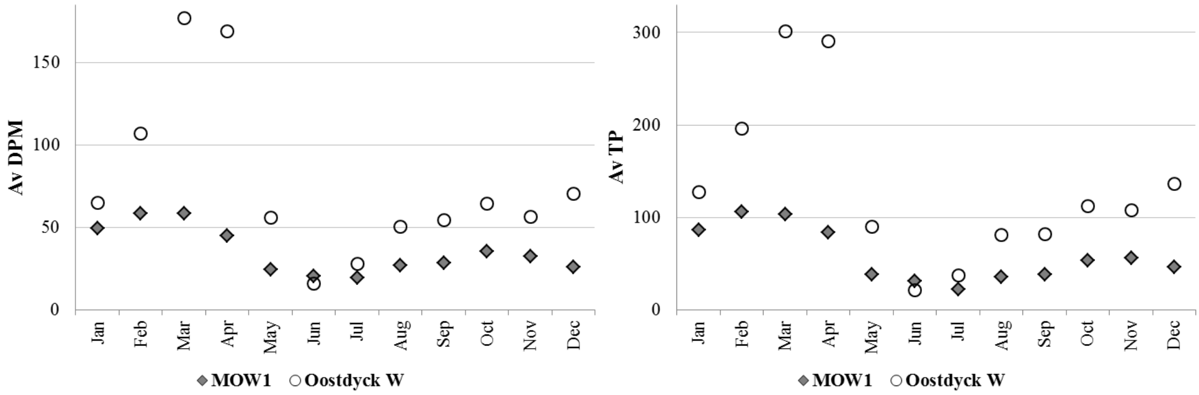

Figure 7. Average DPM/d (left) and TP/d (seconds; right) per month for MOW1 vs. Oostdyck W.

15.4. DISCUSSION

ISSUES IN MOORING PODS

As Belgian waters are characterized by predominantly soft sediments, experiencing a high level of bottom trawling, and given budgetary constraints, moorings were tested at locations of opportunity by developing mooring systems adapted to such locations. Both main used mooring techniques have their advantages and disadvantages. A large ship is needed for mooring (expensive) tripods, while for mooring PoDs on existing navigational buoys a small RHIB type vessel is sufficient. Especially with the PoDs hanging from the buoys, there were issues to be resolved with orientation (the PoD needs to be kept as vertical as possible) and robustness; this was eventually achieved using a relatively heavy system (with a lead weight at the bottom of the steel container).

PoD losses can occur even with robust mooring systems (e.g. Brasseur et al., 2004; Diederichs et al., 2009). In our study, losses in PoDs moored on buoys were higher than in tripod mounted PoDs. This is probably due to a combination of factors. Buoy moored PoDs are more vulnerable to damage during adverse weather conditions, as they are much more exposed than tripod mounted PoDs. After the replacement of stainless steel wire with a chain in buoy moored PoDs, a few losses still occurred due to the whole mooring system getting lost. In one case of a buoy moored PoD, the mooring system remained in place, while the PoD had disappeared, probably due to a broken 8 mm stainless steel screw keeping it fastened. Theft of buoy mooring systems or vandalism could not be excluded, as they were within easy reach. The loss of the tripod could have been the consequence of displacement to an unknown location by bottom trawl fisheries. In 2016 (not in this study) a tripod mounted PoD got detached (and lost) from a tipped tripod due to unknown reasons.

Saturation in PoDs hanging from buoys occurred on average more frequently than in PoDs mounted on tripod systems, as could be expected given higher underwater sound levels around buoys and the unavoidable continuous movement of the PoDs in this mooring system. Particularly in the data obtained from such moorings, broadband background noise can interfere with porpoise detections, by leading to an overload in the detection capabilities of PoDs, or by masking porpoise clicks. This is especially the case during periods with strong tidal currents and adverse weather conditions.

The unsuccessful data collection in a number of moorings was due to unknown reasons (3; including possibly a wrong initialisation of the PoD), loose SD cards (2) and the tipping over of tripods, automatically switching off the PoD (6). Data from the PoDs moored on the steel anchor weight (2) could not be used due to a pinger nearby, saturating the data with a 69 kHz sound.

A NEED FOR STANDARDISATION OF MOORING METHODS?

It is likely that the variation in the detection rate at different locations is not solely the consequence of a difference in the presence of porpoises, but also of the use of different mooring systems. It has been demonstrated for instance that detection rates can vary according to the deployment depth of C-PoDs (Sostres Alonso & Nuuttila, 2015). There could also be a different attraction of harbour porpoises to a tripod mounted PoD vs. a buoy moored PoD, resulting in a different detection rate, and there could be different false detection rates. Given the use of moorings of opportunity in our study, we could not assess the possible effects of this, but as the MOW1 (tripod) location was very shallow, we estimate that the effect of at least mooring depth would be minimal. However, possible differences in detections due to the use of different mooring systems should be avoided through a high level of standardisation, such as in the SAMBAH project (Static Acoustic Monitoring of the Baltic Sea Harbour Porpoise project; www.sambah.org). In this way, PAM data (generated by a similar PAM device) could be better compared over larger areas than is currently the case within the North Sea. This may however be difficult to achieve, given wide ranges in current velocity, depth, bottom type, etc.

STUDY DESIGN

In impact assessment of human activities, no firm conclusions can be drawn when using a small number of PoDs, as in this study. For a meaningful statistical analysis, more replicates and more locations with simultaneous PoD deployments are needed. For impact assessment of pile driving, PoDs should be placed along a gradient from the piling location, up to more than 20 km away (as in Brandt et al., 2011; 2012; Dähne et al., 2013), before piling starts up to weeks after the end of piling operations. For impact assessment of operational wind farms, PoDs need to be placed both within a windfarm, and at a location with similar environmental variables outside it, at a short distance (eg. at least two locations with 3 PoDs each) (Scheidat et al, 2011).

However mooring fewer PoDs, such as in this study, can yield useful information. They provide the basis for the analysis of technical aspects in the mooring of PAM devices, generate information about what PAM studies can achieve locally, and as such form the basis of further studies. Additionally, they provide information for the assessment of seasonal differences in harbour porpoise presence and migratory/foraging movements and in differences in the presence of porpoises in between years. The information obtained from a relatively small number of PoDs can thus contribute to other studies, such as of stranded animals and other studies providing information useful for managing activities possibly adversely affecting porpoises, such as piling and fisheries.

PAM VS. STRANDINGS AND AERIAL SURVEY DATA

In contrast to visual line transect methods (Buckland et al., 2001), PAM is a cue counting method, and it cannot usually directly provide an estimate of absolute density, a value often requested for in for instance environmental impact assessment studies. PoDs only measure the time during which animals are detected, and the number of clicks detected. Complicating factors in efforts to correlate detection rate with density of animals include the following:

- There may be a varying false positive detection rate in PoDs (although it is probably low), and it could be different between different mooring systems.
- The detection probability as a function of the distance around the PAM device is usually unknown.
- Vocalisations of harbour porpoises are directional, possibly leading to different detection rates in for instance benthic vs. pelagic feeding animals.
- Differences in group sizes, not detected through PAM, may be related to a combination of a seasonal variation in prey species and different social stages in the life cycle of harbour porpoises, with distinct periods of mating, breeding and lactation (Addink et al., 1995; Gaskin et al., 1984; Haelters et al., 2011b; Lockyer, 2003).
- While porpoises echolocate almost continuously (Verfuß et al., 2005; Akamatsu et al., 2007), there are diurnal rhythms (likely to reflect differences in prey choice and hunting behaviour) and perhaps also seasonal differences in echolocation (Stedt et al., 2015; Brandt et al., 2016).
- Tidal noise and noise originating from adverse meteorological conditions could affect the echolocation capabilities of harbour porpoises, which may during running tides adapt their echolocation activities.

All these factors lead to the conclusion that there is no straightforward correlation between detection rate, as a result from acoustic activity, and the density of porpoises (Brandt et al., 2016; Kyhn et al., 2008; Kyhn & Tougaard, 2009). Specific scaling factors would be needed to convert PAM data into absolute densities of animals over a given area and time period. Estimating such multipliers constitutes a complex and challenging analytical problem that has been approached through tracking individual animals in the proximity of PAM devices (Kyhn et al., 2012a; 2012b; Tougaard, 2008; Thomas & Marques, 2012; Marques et al., 2013). Tougaard (2008) converted 2.7 detection positive minutes per hour in a T-POD type PAM device into a density estimate of 0.69 porpoises/km². A more pragmatic way to provide an empirical estimate of absolute density from PAM data would be to correlate density estimates from aerial line transect surveys to PAM data (Haelters et al., 2013).

The results of the PAM at Oostdyck W and MOW1 are consistent with the results of aerial surveys (Haelters et al., 2013; 2015) and with strandings data (Haelters et al., 2016), both revealing a seasonal pattern, with in general the highest detection rates in late winter and early spring. Strandings also showed a peak in late summer and early autumn, consistent with a peak in PAM detections. However, strandings data are heavily biased due to meteorological conditions and changes in mortality throughout the year. PAM yielded in general higher detection rates at the Oostdyck W location than at MOW1, which would also be consistent with the results of aerial surveys, although the use of a different mooring system might have some influence. Significant year-to year differences in detection rate were apparent in one of the mooring locations; the lowest detection rates in PAM at MOW1 occurred in 2010 and 2015, also the years with the lowest number of stranded animals (Haelters et al., 2016).

Erratic peaks in the detection rate, possibly due to erratic invasions of harbour porpoises in the BPNS, were present. Peaks in harbour porpoise density are probably the consequence of changes in local prey availability in combination with higher density areas nearby (Haelters et al., 2011a; Gilles et al., 2016; Haelters & Geelhoed, 2015), and the fact that only a small part of the distribution area of the North Sea harbour porpoise population is covered in this study (Hammond et al., 2015; ICES, 2014).

15.5. CONCLUSION

For this PAM study, the detection rate was analysed at the locations MOW1 and Oostdyck W. At both locations it showed a peak in late winter - early spring, and a smaller peak in late summer - early autumn. This is consistent with data obtained from aerial surveys and strandings. At MOW1, there were significant differences from year to year.

The research conducted until now should be considered as a trial phase: mooring systems needed to be developed and tested, moorings were not possible at any location, there were only a limited number of locations and no replicates. Issues encountered during this study are, however, considered in the monitoring programme starting in 2016.

In order to avoid different detection rates due to the use of different mooring systems, such systems should be standardised. In general, the number of saturated minutes, leading to time lost, was higher in buoy moored PoDs than in PoDs mounted on tripods. PoDs moored on buoys had a higher loss rate than those mounted on tripods. Therefore, it is advised to use a system that places PoDs at a reference height from the seafloor (eg. at around 1.5 m above the seafloor), by using tripods or weights equipped with an acoustic release and no surface marker buoy. The number and placement of the PoDs should not be at random, but should be chosen as a function of the objectives of the study.

While keeping in mind that there are inherent issues in PAM (as is the case in other cetacean monitoring methods) that cannot be resolved, PAM has demonstrated its potential to add to the information obtained through aerial surveys. Although many difficulties and uncertainties remain, it provides useful data, certainly if combined with data originating from other research. Density estimation from PAM will gain importance in the future. The use of PAM is increasingly popular for short- to long-term (i.e. weeks to years) monitoring of cetaceans, both for basic ecological research and for impact assessment of human activities and will become a standard way of monitoring cetaceans (Marques et al., 2011).

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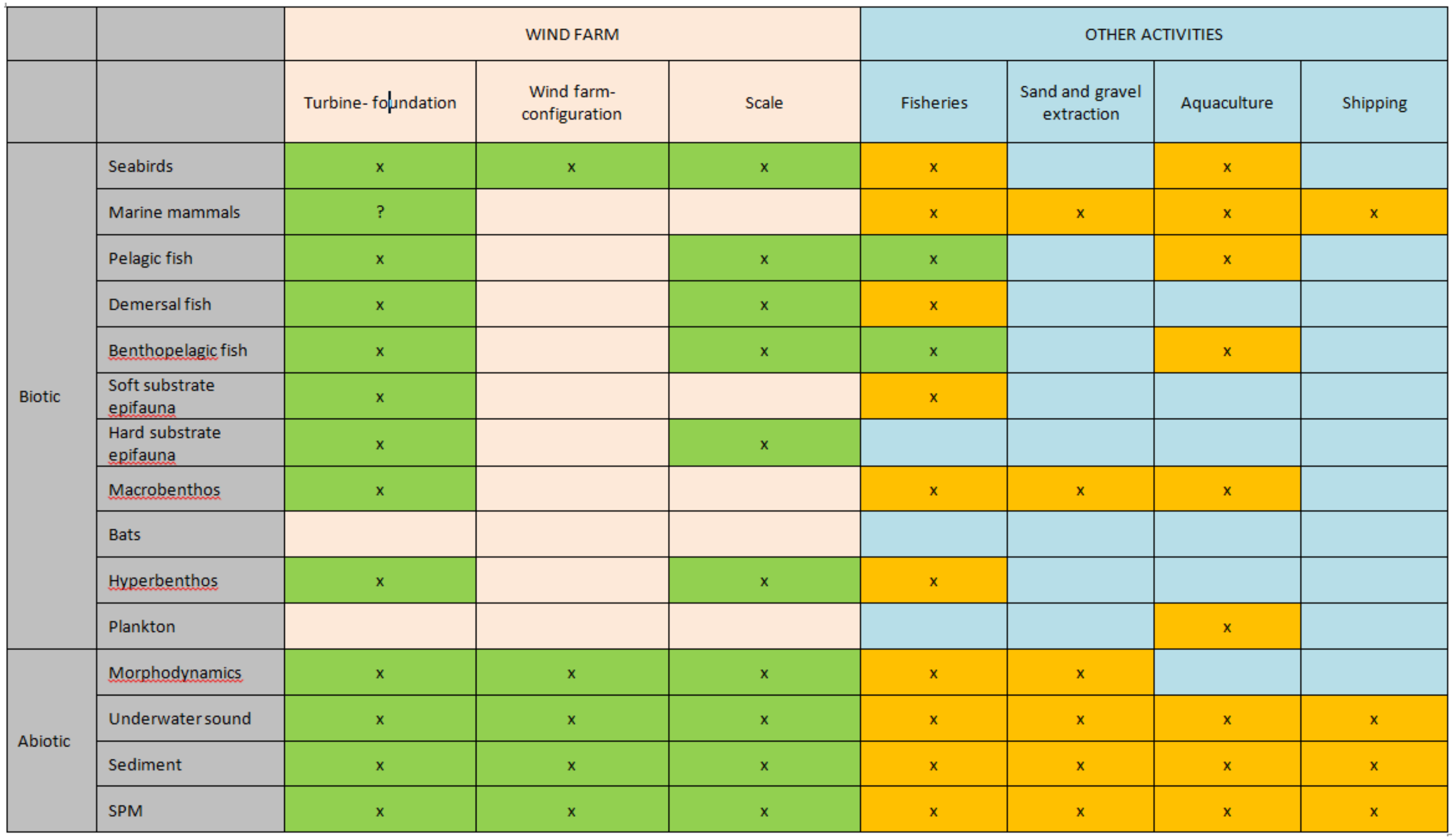
ANNEX 1

Overview of the variables (temporal, spatial, associated with the wind farm and associated with other human activities) influencing the impact assessment of offshore wind farms. If a variable is relevant for an ecosystem component it is marked with an ‘x’ and then color coded depending on the form of variation (red = variables in which we are not interested and which we can exclude with an appropriate sampling design; orange = variables in which we are not interested, but which cannot be excluded; green = variables which are interesting in function of rendering advice in the framework of future wind farms and which should be covered by the basic monitoring programme).

		TEMPORAL							
		Long term trend	year-to-year variability	Seasonal variability	Effect of the tide	Diurnal variability	Effect of meteorological conditions	Time x Effect interaction	Development phase of the wind farm
Biotic	Seabirds	🟧 x	🟧 x	🟩 x	🟧 x	🟩 x	🟧 x	🟩 x	🟩 x
	Marine mammals	🟧 x	🟧 x	🟩 x	🟧 x			🟩 x	🟩 x
	Pelagic fish	🟧 x	🟧 x	🟩 x	🟧 x	🟥 x	🟧 x	🟩 x	
	Demersal fish	🟧 x	🟧 x	🟥 x		🟥 x		🟩 x	
	Benthopelagic fish	🟧 x	🟧 x	🟩 x	🟧 x	🟥 x	🟧 x	🟩 x	
	Soft substrate epifauna	🟧 x	🟧 x	🟥 x				🟩 x	
	Hard substrate epifauna	🟧 x	🟧 x	🟥 x				🟩 x	
	Macrobenthos	🟧 x	🟧 x	🟥 x		🟥 x		🟩 x	
	Bats	🟧 x	🟧 x	🟩 x		🟩 x	🟧 x	🟩 x	
	Hyperbenthos	🟧 x	🟧 x	🟥 x	🟧 x	🟥 x	🟥 x	🟩 x	
	Plankton	🟧 x	🟧 x	🟩 x		🟥 x		🟩 x	
Abiotic	Morphodynamics	🟧 x	🟧 x	🟥 x	🟧 x		🟥 x	🟩 x	🟩 x
	Underwater sound	🟧 x	🟧 x				🟩 x		🟩 x
	Sediment	🟧 x	🟧 x	🟥 x	🟧 x		🟥 x	🟩 x	🟩 x
	SPM	🟧 x	🟧 x	🟩 x	🟧 x		🟧 x	🟩 x	🟩 x

		SPATIAL							
		Inshore-offshore	Depth (Vertical distribution along the foundation)	Community	Sediment	Hydro-dynamics (Current)	SPM	Edge-effects	Distance from turbine
Biotic	Seabirds	x	?			x		?	x
	Marine mammals	x	?			x		?	x
	Pelagic fish	x	?			x			x
	Demersal fish	x		x	x			?	x
	Benthopelagic fish	x	x	x	x	x		?	x
	Soft substrate epifauna	x		x	x	x	x		x
	Hard substrate epifauna	x	x	x		x	x		
	Macrobenthos	x		x	x	x	x		x
	Bats	x						?	x
	Hyperbenthos	x		x	x	x	x	?	x
	Plankton	x	?			x			x
Abiotic	Morphodynamics	x			x	x			x
	Underwater sound								x
	Sediment	x			x	x		?	x
	SPM	x	x		x	x		?	x



		WIND FARM			OTHER ACTIVITIES			
		Turbine- foundation	Wind farm- configuration	Scale	Fisheries	Sand and gravel extraction	Aquaculture	Shipping
Biotic	Seabirds	x	x	x	x		x	
	Marine mammals	?			x	x	x	x
	Pelagic fish	x		x	x		x	
	Demersal fish	x		x	x			
	Benthopelagic fish	x		x	x		x	
	Soft substrate epifauna	x			x			
	Hard substrate epifauna	x		x				
	Macrobenthos	x			x	x	x	
	Bats							
	Hyperbenthos	x		x	x			
	Plankton						x	
Abiotic	Morphodynamics	x	x	x	x	x		
	Underwater sound	x	x	x	x	x	x	x
	Sediment	x	x	x	x	x	x	x
	SPM	x	x	x	x	x	x	x

ANNEX 2

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
1990	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1991	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1992	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1993	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1994	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1995	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1996	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1997	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1998	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
1999	28/09/2010	WT2	B	C	*Echiichthys vipera*	EMPTY
2000	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2001	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2002	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2003	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2004	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2005	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2006	28/09/2010	WT2	B	C	*Echiichthys vipera*	EMPTY
2007	28/09/2010	WT2	B	C	*Echiichthys vipera*	EMPTY
2008	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
2009	28/09/2010	WT2	B	C	*Echiichthys vipera*	FULL
208	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
209	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
210	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
211	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
212	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
213	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
214	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
215	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
216	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
217	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
218	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
219	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
220	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
221	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
222	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
223	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
224	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
225	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
226	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
227	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
228	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
229	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
230	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
231	19/08/2010	WT5s	B	I	*Echiichthys vipera*	EMPTY
232	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
233	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
234	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
235	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
236	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
237	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
238	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
239	19/08/2010	WT5s	B	I	*Echiichthys vipera*	FULL
2023	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2024	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2025	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
2026	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2027	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
2028	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2029	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2030	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2031	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2032	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2033	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2034	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
2035	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2036	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2037	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2038	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2039	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2040	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
2041	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
2042	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2043	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2044	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2045	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
2046	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	FULL
2047	28/09/2010	WT8=TRACK 5	B	I	*Echiichthys vipera*	EMPTY
1943	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1944	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1945	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1946	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1947	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1948	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1949	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1950	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
1951	27/09/2012	WT2	A	C	*Echiichthys vipera*	EMPTY
1952	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1953	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1954	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1955	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1956	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1957	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1958	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1959	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1960	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1961	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1962	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1963	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1964	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1965	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1966	27/09/2012	WT2	A	C	*Echiichthys vipera*	FULL
1967	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1968	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1969	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1970	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1971	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1972	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1973	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1974	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1975	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1976	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1977	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1978	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1979	5/10/2013	WT2	A	C	*Echiichthys vipera*	EMPTY
1980	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1981	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1982	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1983	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1984	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1985	5/10/2013	WT2	A	C	*Echiichthys vipera*	EMPTY
1986	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1987	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1988	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1989	5/10/2013	WT2	A	C	*Echiichthys vipera*	FULL
1905	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1897	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1898	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1899	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1900	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
1901	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1902	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1903	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1904	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1906	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1907	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1908	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1909	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1910	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1911	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1912	27/09/2012	TRACK 2	A	I	*Echiichthys vipera*	FULL
1928	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1929	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1930	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1931	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1932	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1933	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1934	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1935	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1936	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1937	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1938	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1939	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1940	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1941	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1942	27/09/2012	TRACK 4	A	I	*Echiichthys vipera*	FULL
1753	27/09/2012	WT5=TRACK 3	A	I	*Echiichthys vipera*	FULL
1754	27/09/2012	WT5=TRACK 3	A	I	*Echiichthys vipera*	FULL
1755	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1756	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	EMPTY
1757	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	EMPTY
1758	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1760	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1761	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1762	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1763	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	EMPTY
1764	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1765	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	EMPTY
1766	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1865	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1866	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	EMPTY
1867	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1868	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1869	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
1870	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	EMPTY
1871	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1872	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1873	27/09/2012	WT5=TRACK3	A	I	*Echiichthys vipera*	FULL
1878	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1879	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1880	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1881	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	EMPTY
1882	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1883	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1885	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1886	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	EMPTY
1887	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	EMPTY
1888	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1889	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1890	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	EMPTY
1891	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1892	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1893	27/09/2012	WT8=TRACK5	A	I	*Echiichthys vipera*	FULL
1913	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	EMPTY
1914	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1915	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1916	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1917	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	EMPTY
1918	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1919	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1920	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1921	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1922	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1923	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1924	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1925	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	EMPTY
1926	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
1927	25/09/2013	TRACK 2	A	I	*Echiichthys vipera*	FULL
139	28/09/2010	WT2bis	B	C	*Limanda limanda*	EMPTY
140	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
141	28/09/2010	WT2bis	B	C	*Limanda limanda*	EMPTY
142	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
143	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
144	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
145	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
146	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
147	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
148	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
149	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
150	28/09/2010	WT2bis	B	C	*Limanda limanda*	EMPTY
151	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
152	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
153	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
154	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
155	28/09/2010	WT2bis	B	C	*Limanda limanda*	FULL
193	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
194	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
195	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
196	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
197	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
198	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
199	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
200	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
201	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
202	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
203	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
204	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
205	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
206	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
207	19/08/2010	WT5s	B	I	*Limanda limanda*	FULL
119	28/09/2010	WT5	B	I	*Limanda limanda*	EMPTY
120	28/09/2010	WT5	B	I	*Limanda limanda*	EMPTY
121	28/09/2010	WT5	B	I	*Limanda limanda*	EMPTY
122	28/09/2010	WT5	B	I	*Limanda limanda*	FULL
123	28/09/2010	WT5	B	I	*Limanda limanda*	FULL
124	28/09/2010	WT5	B	I	*Limanda limanda*	FULL
2017	28/09/2010	WT8=TRACK 5	B	I	*Limanda limanda*	FULL
2018	28/09/2010	WT8=TRACK 5	B	I	*Limanda limanda*	FULL
2019	28/09/2010	WT8=TRACK 5	B	I	*Limanda limanda*	FULL
2020	28/09/2010	WT8=TRACK 5	B	I	*Limanda limanda*	FULL
2021	28/09/2010	WT8=TRACK 5	B	I	*Limanda limanda*	FULL
2022	28/09/2010	WT8=TRACK 5	B	I	*Limanda limanda*	FULL
2010	27/09/2012	WT2	A	C	*Limanda limanda*	EMPTY
2011	27/09/2012	WT2	A	C	*Limanda limanda*	FULL
2012	27/09/2012	WT2	A	C	*Limanda limanda*	FULL
2013	27/09/2012	WT2	A	C	*Limanda limanda*	FULL
2014	27/09/2012	WT2	A	C	*Limanda limanda*	FULL
2015	27/09/2012	WT2	A	C	*Limanda limanda*	FULL
2016	27/09/2012	WT2	A	C	*Limanda limanda*	FULL
1767	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1768	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1769	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL

fish_ID	Date	Location	B/A	C/I/F	scientific_name	Stomach E/F
1770	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1771	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1772	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1773	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1774	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1775	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1850	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1851	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1852	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1853	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1854	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1855	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1856	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1857	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1858	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1859	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1860	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1861	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1862	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1863	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1864	27/09/2012	WT5=TRACK3	A	I	*Limanda limanda*	FULL
1778	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	EMPTY
1781	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	EMPTY
1874	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	EMPTY
1875	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	FULL
1876	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	FULL
1877	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	EMPTY
1884	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	EMPTY
1895	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	FULL
1896	27/09/2012	WT8=TRACK5	A	I	*Limanda limanda*	FULL

ANNEX 3

Table 6. Impact model coefficients for all species studied at the Thorntonbank OWF study area.

Species	Impact polygon	Intercept (Count)	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	Sin (1/4yr)	Cos (1/4yr)	BA	CI	Fishery	OWF	Theta	Intercept (Zero)
Northern fulmar	Impact + 0.5 km	0.49	-0.70	0.53					-1.97	-1.14		-20.98	0.94	0.75
	Impact + 3 km	0.47	-0.77	0.35					-2.11	-1.11		-0.54	0.83	0.76
	Buffer 0.5-3 km	0.47	-0.78	0.34					-2.11	-1.10		0.08	0.82	0.76
Northern gannet	Impact + 0.5 km	-0.46	s(month)									-4.70	0.29	
	Impact + 3 km	-0.47	s(month)									-1.40	0.30	
	Buffer 0.5-3 km	-0.47	s(month)									-0.92	0.30	
Great skua	Impact + 0.5 km	-4.22	-2.59	0.49	0.82	1.02			-1.51	-1.68		-23.08	0.50	
	Impact + 3 km	-4.07	-2.25	0.61	0.96	0.89			-1.62	-1.66		0.54	0.50	
	Buffer 0.5-3 km	-4.08	-2.25	0.61	0.96	0.88			-1.62	-1.66		1.15	0.50	
Little gull	Impact + 0.5 km	-2.22	s(month)									-2.01	0.12	
	Impact + 3 km	-2.34	s(month)									0.59	0.13	
	Buffer 0.5-3 km	-2.33	s(month)									1.18	0.13	
Common gull	Impact + 0.5 km	-3.95	2.32	2.11					1.49	0.99	1.38	-0.98	0.26	
	Impact + 3 km	-3.84	2.21	2.04					1.30	1.11	1.05	-0.51	0.28	
	Buffer 0.5-3 km	-3.77	2.22	2.00					1.27	1.08	0.80	-0.01	0.26	
Lesser black-backed gull	Impact + 0.5 km	-0.51	s(month)								1.09	0.05	0.31	
	Impact + 0.5 km (T)	-0.51	s(month)								1.06	0.26	0.32	
	Impact + 3 km	-0.44	s(month)								0.77	-0.01	0.31	
	Impact + 3 km (T)	-0.44	s(month)								0.78	0.04	0.31	
	Buffer 0.5-3 km	-0.43	s(month)								0.67	-0.11	0.29	

Species	Impact polygon	Intercept (Count)	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	Sin (1/4yr)	Cos (1/4yr)	BA	CI	Fishery	OWF	Theta	Intercept (Zero)
Herring gull	Impact + 0.5 km	-2.48	1.62	0.03							0.96	-0.06	0.16	
	Impact + 0.5 km (T)	-2.49	1.61	-0.02							1.04	0.14	0.17	
	Impact + 3 km	-2.48	1.56	0.06							0.98	-0.63	0.19	
	Impact + 3 km (T)	-2.48	1.54	0.02							1.04	-0.49	0.19	
	Buffer 0.5-3 km	-2.58	1.59	0.13							1.32	-1.66	0.17	
Great black-backed gull	Impact + 0.5 km	-1.86	-0.05	2.18			0.10	0.72			1.33	0.21	0.22	
	Impact + 0.5 km (T)	-1.81	-0.02	1.83			0.12	0.48			1.51	1.86	0.26	
	Impact + 3 km	-1.82	s(month)								1.65	0.28	0.24	
	Impact + 3 km (T)	-1.65	s(month)								1.66	1.00	0.26	
	Buffer 0.5-3 km	-2.38	-0.17	2.58	0.88	-0.73					2.15	-0.02	0.21	
Black-legged kittiwake	Impact + 0.5 km	-0.76	s(month)								1.14	-1.95	0.26	
	Impact + 3 km	-0.80	s(month)								1.26	-1.21	0.28	
	Buffer 0.5-3 km	-0.80	s(month)								1.32	-0.84	0.27	
Sandwich tern	Impact + 0.5 km	-0.34							-1.55			1.07	1.55	0.74
	Impact + 3 km	-0.42							-1.63			1.29	1.17	0.69
	Buffer 0.5-3 km	-0.38							-1.62			1.72	1.29	0.71
Common guillemot	Impact + 0.5 km	-2.55	1.39	6.21	-1.13	-1.69						-1.13	0.97	0.10
	Impact + 3 km	-2.76	1.57	6.48	-1.30	-1.87						-0.59	0.94	0.12
	Buffer 0.5-3 km	-2.86	1.69	6.61	-1.39	-1.91						-0.27	0.89	0.11
Razorbill	Impact + 0.5 km	-6.33	0.99	9.43	-0.94	-3.51			0.57			-0.80	0.47	
	Impact + 3 km	-6.50	1.25	9.67	-1.27	-3.61			0.56			-0.08	0.50	
	Buffer 0.5-3 km	-6.43	1.25	9.56	-1.28	-3.60			0.55			0.27	0.49	

Table 7. Impact model coefficients for all species studied at the Bligh Bank OWF study area.

Species	Impact polygon	Intercept (Count)	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	Sin (1/4yr)	Cos (1/4yr)	BA	CI	Fishery	OWF	Theta	Intercept (Zero)
Northern fulmar	Impact + 0.5 km	-0.66	0.26	0.83	-0.27	1.09			-1.69		1.14	-2.54	1.13	0.68
	Impact + 3 km	-0.65	0.26	0.81	-0.27	1.08			-1.74		1.19	-3.13	1.13	0.68
	Buffer 0.5-3 km	-0.69	0.41	0.79	-0.31	1.26			-1.87		1.47	-22.93	1.26	0.68
Northern gannet	Impact + 0.5 km	-1.33	-0.07	1.10			-0.86	-0.22	0.79	-0.52		-1.72	0.47	
	Impact + 3 km	-1.31	0.03	1.07			-0.71	-0.26	0.73	-0.50		-0.95	0.49	
	Buffer 0.5-3 km	-1.31	0.06	1.09			-0.64	-0.30	0.74	-0.51		-0.30	0.45	
Great skua	Impact + 0.5 km	-1.48							-1.05			-19.45		0.80
	Impact + 3 km	-1.49							-0.93			-1.95		0.79
	Buffer 0.5-3 km	-1.49							-0.93			-1.03		0.79
Little gull	Impact + 0.5 km	-9.82	8.48	6.66	-5.11	-0.10			2.58			-0.98	0.10	
	Impact + 3 km	-9.13	7.62	5.96	-4.59	-0.01			2.45			-0.22	0.10	
	Buffer 0.5-3 km	-8.97	7.34	5.85	-4.64	-0.16			2.38			0.23	0.10	
Common gull	Impact + 0.5 km	-6.13	2.41	4.70	-1.04	-1.60			1.47		2.46	1.79	0.14	
	Impact + 3 km	-6.11	2.35	4.69	-1.05	-1.65			1.43		2.39	1.26	0.16	
	Buffer 0.5-3 km	-6.81	3.00	5.85	-1.79	-1.95			1.41		1.62	0.14	0.18	
Lesser black-backed gull	Impact + 0.5 km	-1.33	s(month)							-1.33	1.07	2.09	0.29	
	Impact + 3 km	-1.36	s(month)							-1.38	1.07	2.20	0.29	
	Buffer 0.5-3 km	-1.38	s(month)							-1.42	1.09	2.04	0.25	
Herring gull	Impact + 0.5 km	-4.04	1.64	1.65							1.79	1.47	0.13	
	Impact + 3 km	-3.72	1.65	2.25			0.24	0.64			1.00	0.58	0.16	
	Buffer 0.5-3 km	-3.91	2.10	2.24			-0.18	0.69			0.88	0.35	0.16	
Great black-backed gull	Impact + 0.5 km	-2.07	s(month)									1.29	0.20	
	Impact + 3 km	-2.02	s(month)									1.09	0.22	
	Buffer 0.5-3 km	-2.01	s(month)									0.61	0.19	

Species	Impact polygon	Intercept (Count)	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	Sin (1/4yr)	Cos (1/4yr)	BA	CI	Fishery	OWF	Theta	Intercept (Zero)
Black-legged kittiwake	Impact + 0.5 km	-1.55	0.81	2.46			-0.62	-0.26	0.70		0.92	0.26	0.34	
	Impact + 3 km	-1.59	0.91	2.52			-0.76	-0.28	0.54		1.24	0.36	0.35	
	Buffer 0.5-3 km	-1.58	0.96	2.53			-0.84	-0.21	0.41		1.47	0.43	0.33	
Common guillemot	Impact + 0.5 km	-1.24	1.34	2.87								-1.39	0.61	
	Impact + 3 km	-1.21	1.26	2.80								-0.99	0.71	
	Buffer 0.5-3 km	-1.18	1.19	2.74								-0.68	0.74	
Razorbill	Impact + 0.5 km	-6.40	2.92	7.13	-1.23	-2.06			2.49			-1.12	0.39	
	Impact + 3 km	-6.68	3.02	7.57	-1.36	-2.29			2.40			-0.84	0.37	
	Buffer 0.5-3 km	-6.78	3.08	7.77	-1.52	-2.41			2.34			-0.39	0.37	

ANNEX 4

Raw data (unprocessed, corrected per day) of TP day^{-1} and DPM day^{-1} aggregated and proportionally corrected by month and week, and TP day^{-1} and DPM day^{-1} aggregated and proportionally corrected by month by year for stations Oostdyck W and MOW1

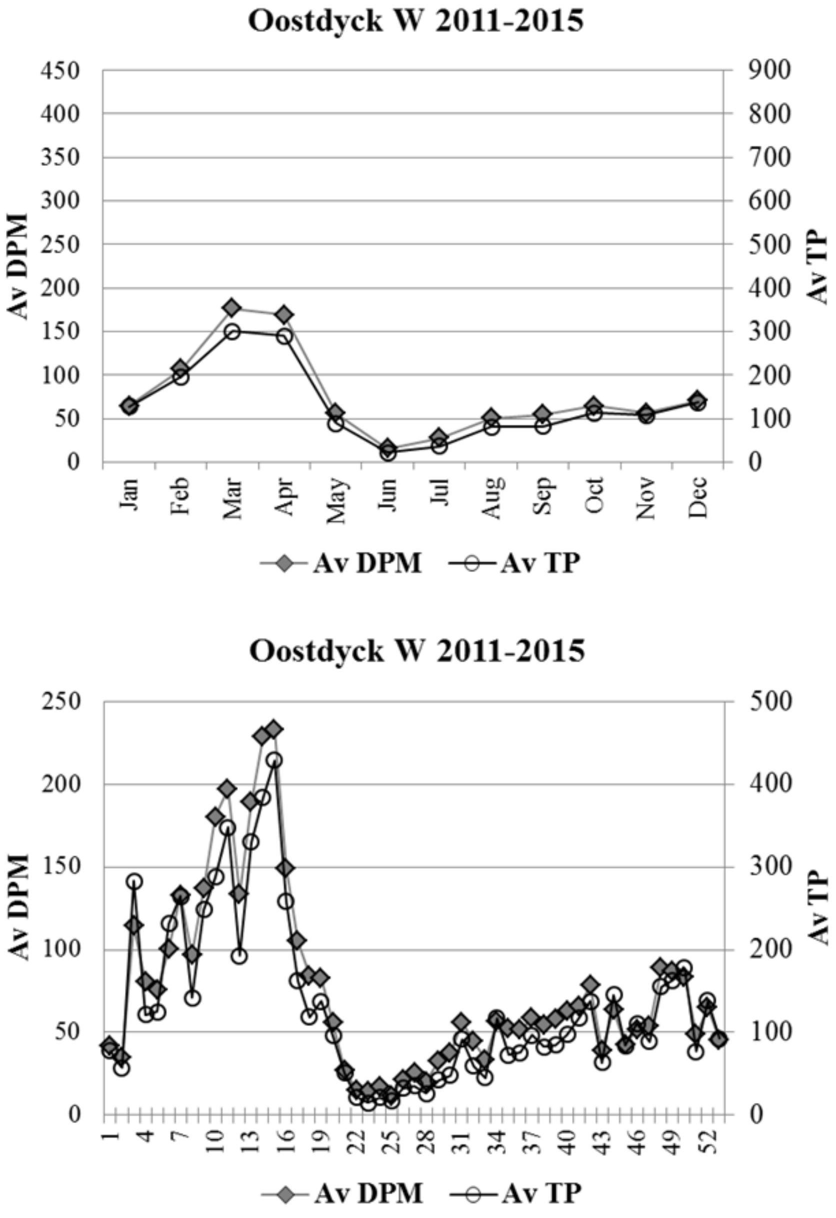


Figure a. TP day^{-1} and DPM day^{-1} aggregated by month (top) and week (bottom) for Oostdyck W

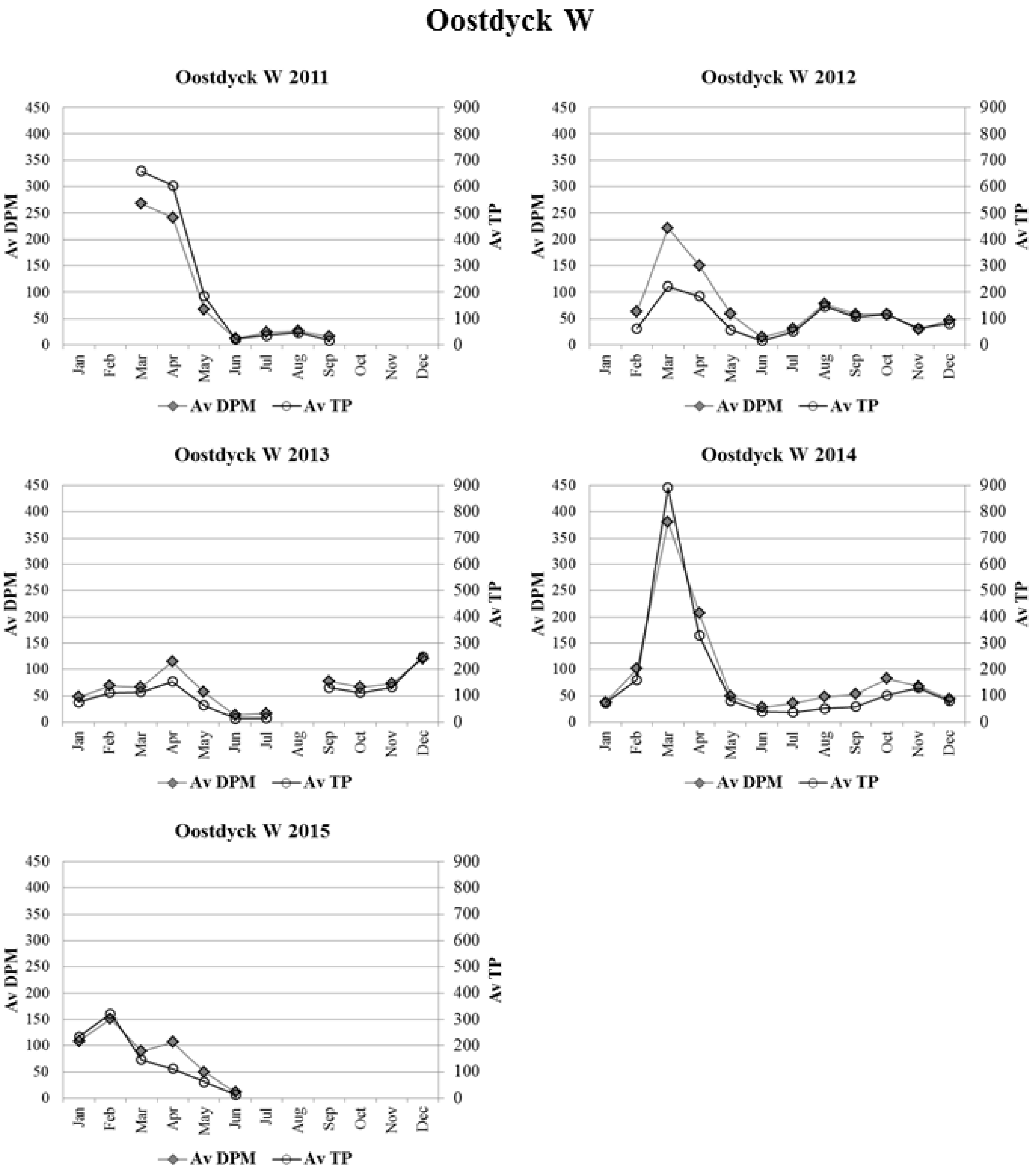
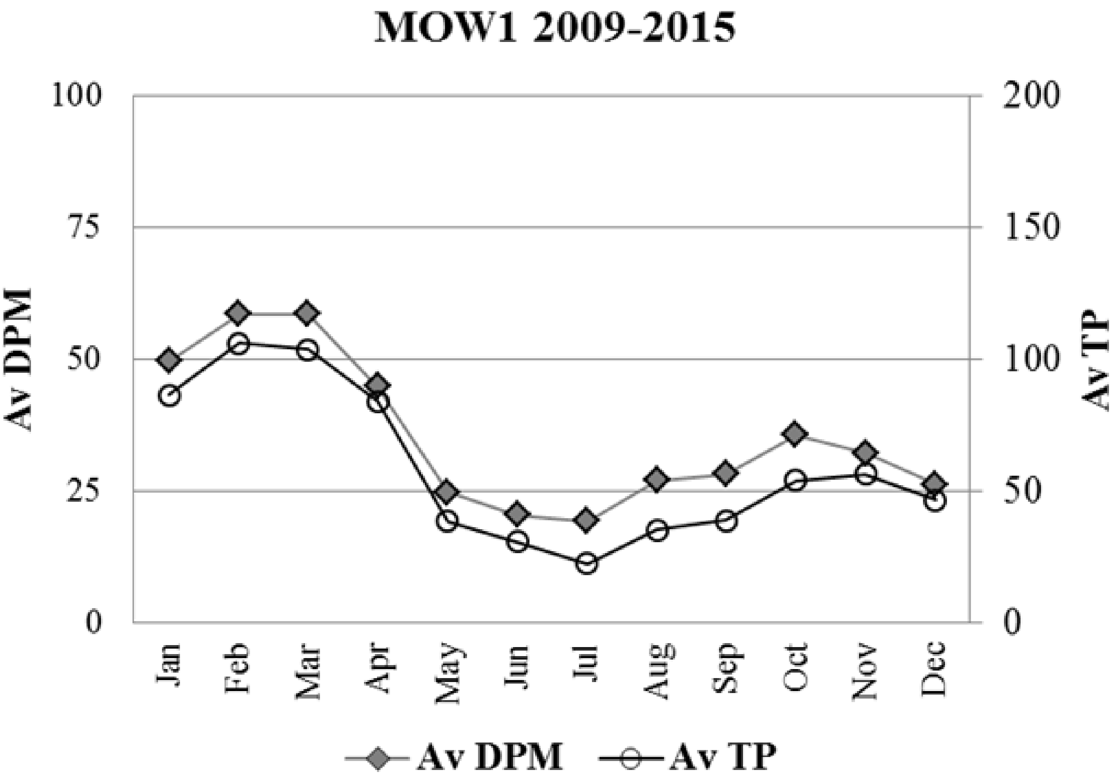


Figure b. TP day^{-1} and DPM day^{-1} aggregated by month and year for Oostdyck W



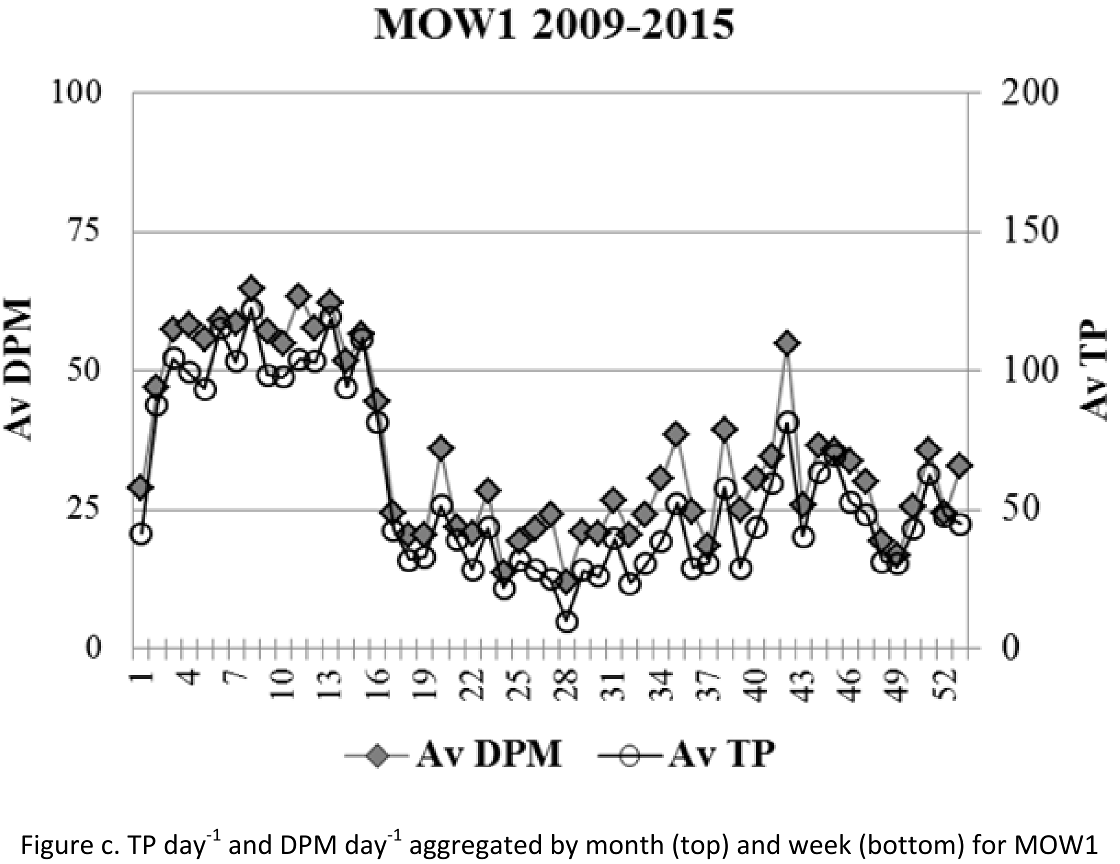


Figure c. TP day^{-1} and DPM day^{-1} aggregated by month (top) and week (bottom) for MOW1

MOW1

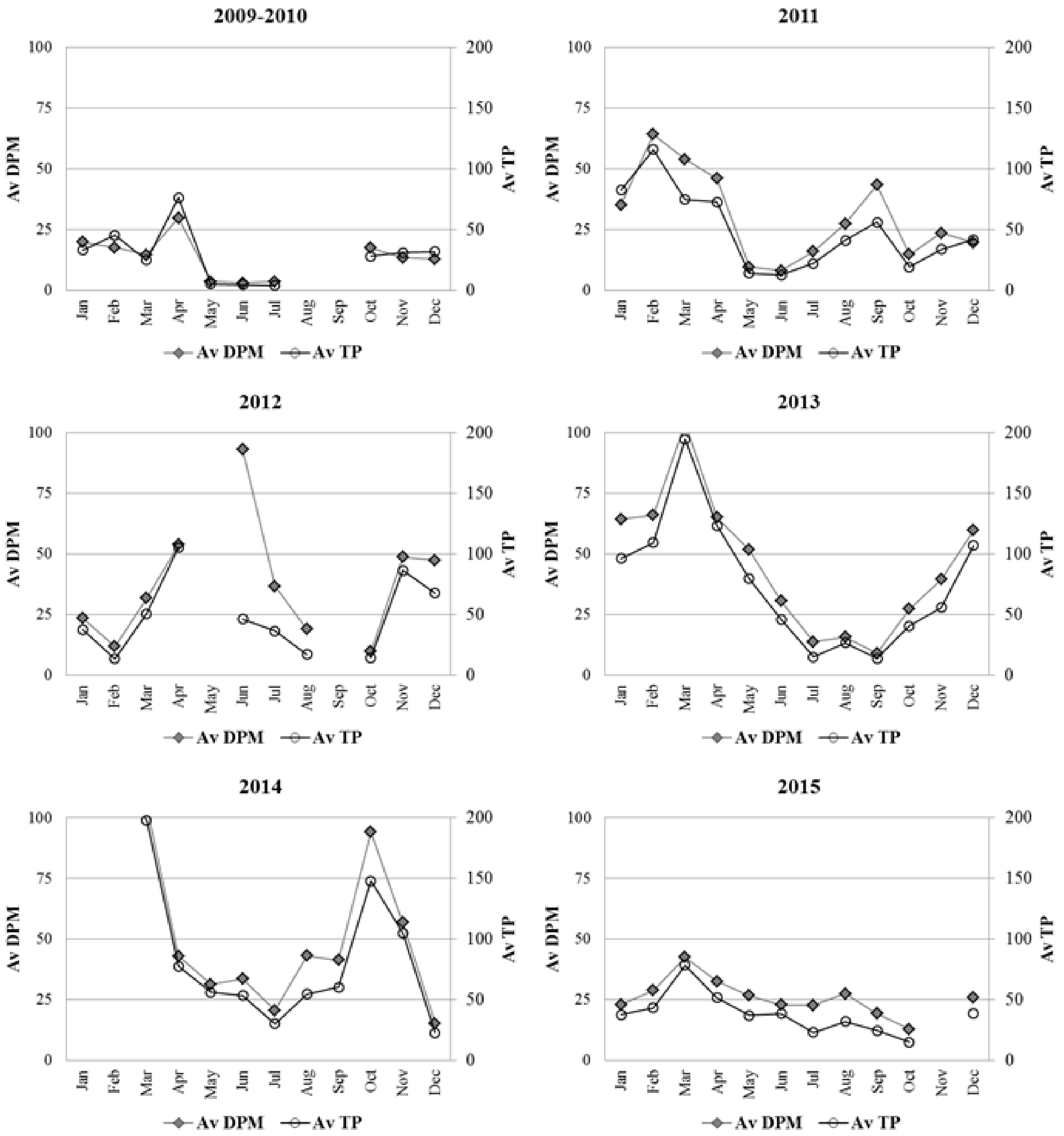


Figure d. TP day^{-1} and DPM day^{-1} aggregated by month and year for MOW1

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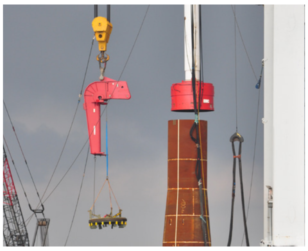
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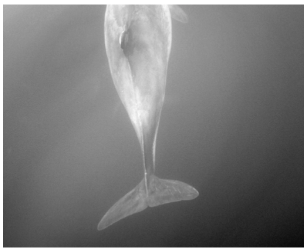
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