MEMOIRS
on the Marine Environment

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

**A CONTINUED MOVE TOWARDS
INTEGRATION AND QUANTIFICATION**

Edited by
Steven Degraer
Robin Brabant
Bob Rumes
Laurence Vigin

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2017

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This report should be cited as:

Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds). 2017. *Environmental impacts of offshore wind farms in the Belgian part of the North Sea: A continued move towards integration and quantification*. Brussels: Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management Section.

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This publication has been peer-reviewed.

Acknowledgements

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PREFACE

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 are expected to make an important contribution (ca. 43%, assuming 2000 MW installed capacity by 2020) to achieve that goal.

Within the BPNS, a zone of 238 km² is reserved for the production of electricity from water, currents or wind. Four wind farms are already operational. With five more to come, major ecological changes may however be expected.

Prior to installing a wind farm, a developer must obtain a domain concession and an environmental permit. 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 onto the marine environment.

Within the monitoring programme, the Royal Belgian Institute of Natural Sciences and its partners assess the extent of the anticipated impacts on the marine ecosystem and aim at revealing the processes behind these impacts. The first objective is basically tackled through the baseline monitoring, focusing on the *a posteriori*, resultant impact quantification, while the second monitoring objective is covered by targeted or process monitoring, focusing on cause-effect relationships of *a priori* selected impacts. As such, baseline monitoring deals with observing rather than understanding impacts and hence leads to area-specific results, which might form a basis for halting activities.

This report, targeting marine scientists, marine managers, policy makers and offshore wind farm developers, presents an overview of the scientific findings of the Belgian offshore wind farm monitoring programme, based on data collected unto 2016.

DEGRAER Steven, BRABANT Robin, RUMES Bob and VIGIN Laurence

EXECUTIVE SUMMARY

A CONTINUED MOVE TOWARDS INTEGRATION AND QUANTIFICATION

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The Belgian target figure for the contribution of electricity production from renewable energy sources is 13% of the total energy consumption, which is to 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 this goal. When all Belgian wind farms are built, there will be almost 500 wind turbines in the BPNS. The 9 wind farms will have a capacity of 2200 MW and will cover up to 10% of the total electricity needs of Belgium or nearly 50% of the electricity needs of all Belgian households. As of 2016, an installed capacity of 870 MW, consisting of 232 offshore wind turbines, is operational in the BPNS. With 238 km² reserved for offshore wind farms in Belgium and 344 km² in the adjacent Dutch Borssele offshore wind farm (cumulative), ecological impacts are inevitable, which is why an extensive environmental impact monitoring programme was set up. This monitoring programme started with an explorational phase in 2005 and has been fully operational since 2008.

The monitoring programme targets physical (*i.e.*, hydro-geomorphology and underwater sound), biological (*i.e.*, hard substrate epifouling and fish communities, soft substrate macrobenthos, epibenthos and demersal-benthopelagic fish, seabirds and marine mammals), as well as socio-economic (*i.e.*, seascape perception and offshore renewables appreciation) aspects of the marine environment although not all components are studied every year. The Operational Directorate Natural Environment (OD Nature) of the Royal Belgian Institute of Natural Sciences (RBINS) coordinates the monitoring and specifically covers hydro-geomorphology, underwater noise, hard substrate epifauna, radar detection of seabirds, marine mammals and socio-economic aspects. In 2016, 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), and soft substrate macrobenthos (Marine Biology Section, Ghent University). For details on the specific research, strategies followed and methodologies used, one is referred to the individual chapters.

This executive summary summarises the individual report chapters targeting the ecosystem components under consideration in the monitoring programme, *i.e.*, hard substrate epifouling organisms, soft sediment macrobenthos, epibenthos and demersal-benthopelagic fish, underwater sound, seabirds and marine mammals. We particularly emphasise the progress made in our continuous move towards increased levels of integration and quantification, as such moving away from data-rich, yet information-poor monitoring programmes (*sensu* Wilding *et al.* 2017) towards information-rich monitoring.

The knowledge and expertise in relation to sampling technicalities and designs for offshore wind farm monitoring gained from the first phase of basic monitoring in Belgian waters (2005, 2008-2016) was revisited in 2015 (Degraer *et al.* 2016). The workshop concluded 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, and (3) how to plan the most appropriate sampling design for the basic monitoring program. The revised monitoring program for the benthic and the pelagic realm excludes sources of noise in the data by means of an adaptation of the monitoring design as far as possible. 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) in contrary were targeted for and are to be used as explicit drivers for restructuring the monitoring programme.

The revised basic monitoring programme was first implemented in 2016. In its attempt to exclude unwanted variability in the data collected, this revision targets *e.g.*, a stratified rather than a randomly distributed sampling design. For the soft sediment macrobenthos for example, samples were taken at two distances from the turbines, *i.e.*, 350-500 m and 50 m (Chapter 4). The aim

was to investigate whether the macrobenthic community continues to shift away from the *Nephtys cirrosa* and *Ophelia limacina-Glycera lapidum* communities that used to dominate the offshore wind farm zone, towards the richer *Abra alba-Kurtiella bidentata* community, typical for muddy sands (Coates *et al.* 2014). Differences in community composition could indeed be detected at the Thornton Bank, with richer macrobenthic communities further away from the turbine. This difference could not statistically be related to differences in environmental conditions (*i.e.*, grain size distribution and total organic matter). On the Bligh Bank however, higher organic matter contents were indeed found further from the turbines, but these did not coincide with significantly different communities. No clear differences in community composition were detected between foundation types (jacket versus gravity based foundations). While this may be linked to the low number of samples available for the gravity-based foundation (n = 3), the effect of turbine presence and foundation type might manifest itself mainly or only in close vicinity of the turbines (< 50 m) and as such remain unconcealed by the current sampling design. Sediment refinement and organic enrichment may indeed be restricted to the immediate proximity of turbines, and hence out of reach of the current monitoring design. Future monitoring of the macrobenthic community structure may hence need to be refocused on closer distances to the turbines as to reveal turbine impacts.

Since 2005, the potential effects of wind farms on soft sediment epibenthos, and demersal-benthopelagic fish are investigated by means of a basic beam trawl monitoring programme targeting the Thornton Bank and Bligh Bank wind farms (Chapter 5). For both wind farms, the number of epibenthic and demersal-benthopelagic fish species remained similar over the years and was not affected by the construction of the wind farms. Epibenthic density and biomass showed a similar trend in both wind farms,

with an increase in the first two years after construction (mainly because of higher densities and biomasses of the common star fish *Asterias rubens*, the hermit crab *Pagurus bernhardus*, the flying crab *Liocarcinus holsatus* and the serpents' table brittle star *Ophiura albida*; year- and wind farm-dependent). In both wind farms, these higher values however levelled off three years after construction. As for epibenthos, demersal-benthopelagic fish seemed to show more variance in densities only in the first few years after construction. These results indicate that the soft sediment ecosystem in between the turbines (at distances > 200 m) has not really changed five to six years after construction and that species assemblages within the offshore wind farms seem to be mainly structured by temporal variability playing at larger spatial scales (*e.g.*, temperature fluctuations, hydrodynamic changes, plankton blooms). One species, plaice *Pleuronectes platessa*, however seems to be positively affected by the offshore wind farms. Plaice densities steadily increased after construction, possibly linked to (locally) increased food availability and/or fisheries exclusion inside the wind farms.

Given the uncertainty about the impact of pile-driving sound on (commercial) fish health, a field experiment was designed to determine the direct effect of pile driving on the health status of Atlantic cod *Gadus morhua* (Chapter 3). Large netted cages with one year old cod individuals (length: 31 ± 4 cm) were positioned at various distances (75 m, 400 m, 1400 m and 1700 m) from a pile driving location and exposed to the pile driving sound for about 16 hours. Average single strike sound exposure levels decreased from 175 dB re 1 µPa² s at 400 m distance to 168 dB re 1 µPa² s at 1700 m distance. A steep increase in swim bladder barotrauma was detected with decreasing distance from the pile driving source, with no swim bladders ruptured at 1700 m and up to 90% of swim bladders ruptured at 75 m distance. Although most fishes in the cages close to the sound source survived the experiment, they all showed many haemorrhages and a high degree of abnormal swimming behaviour. Possibly, some of the abnormal swimming behaviour could be related to inner ear damage (not investigated here). Both internal bleeding and abnormal swimming behaviour however hint towards a reduced longer term survival rate for those fish hit by the impulsive pile driving sound at short distance. These results indicate that with the current sound limits applicable to Belgian waters (*i.e.*, zero to peak level L_{z-p} up to 185 dB re 1 µPa at 750 m), swim bladder barotrauma can occur in fish within a radius of 750 m from the pile driving location. Interpretation of these results in relation to optimal sound limits however remains challenging as this field experiment represents a worst-case scenario with fish caged and no chance to escape, and cod having a closed swim bladder, which is most sensitive to swim bladder injuries.

As an example of maximal exploitation of the data available, the hard substrate epifauna data was explored based on biological trait composition rather than the species composition of the epifouling communities. We were particularly interested in qualifying the differences of natural (*e.g.*, gravel beds) versus artificial (*e.g.*, turbine foundations and scour protection) hard substrates and if the latter could be put forward as surrogate for the threatened and declining natural hard substrata. Both habitats harbour a rich species diversity and share a number of species. The initial results show that natural hard substrata harbour a much higher species number and also more unique species than the artificial ones and there are also some differences in life traits. Therefore, it seems that artificial hard substrata cannot act as alternatives to the loss of natural hard substrata.

The influence of offshore wind farms on seabirds and marine mammals remains a major concern during licensing, construction and operation. For this reason, two

extensive monitoring programmes were set up in Belgian waters. Within the framework of the revised basic monitoring programme, both programmes are exploring new ways of investigation. Examples presented in this report are mainly focused on fine-scale distribution patterns of seabirds and marine mammals in space and time as a response to the presence of offshore wind farms (seabirds) and pile driving activities (marine mammals). These quantitative approaches (*e.g.*, seabird tele metry in relation to seabird behaviour and passive acoustic monitoring in relation to short-term spatial distribution changes in marine mammals) represent new ways towards a full understanding of the ecological impacts of offshore wind farms and hence bridge basic and targeted monitoring.

With over 1000 individuals observed, bird counts at the Thornton Bank (wind farm and control area) showed great black-backed gull to be by far the most numerous species (Chapter 7). The seabird displacement surveys demonstrated the Thornton Bank wind farm to be avoided by 4 species (*i.e.*, northern gannet *Morus bassanus* [-97%], little gull *Hydrocoloeus minutus* [-89%], black-legged kittiwake *Rissa tridactyla* [-75%) and common guillemot *Uria aalge* [-69%]) compared to the control area and the period before impact. In contrary, the wind farm attracted great black-backed gull *Larus marinus* (x 6.6), Sandwich tern *Thalasseus sandvicensis* (x 5.7; buffer zone only) and herring gull *Larus argentatus* (x 2.9). When zooming into the behaviour of some species making use of transect count data, GPS tracking data and observations with a fixed camera installed on turbine I5 of the Thornton Bank OWF, great black-backed gulls tend to favour outer turbines for roosting, suggesting a partial barrier effect. Lesser black-backed gulls on the other hand seemed to spend half of the time inside the wind farm area roosting on the jacket foundations, and to spend relatively less time (15%) flying inside compared to outside the wind farm (44%

for the wider BPNS; 20% for the nearby control area). Telemetry data showed this species' presence in the study area to be highest between 6 am to 12 am with the proportion of non-flying birds mostly above 70% during the full diurnal cycle. 11% of the large gulls observed on the jacket foundation of turbine I5 was found foraging on its intertidal. A continued study of this behavioural shift (*e.g.*, decrease in relative time period flying) may shed a new light onto the anticipated collision mortality among large gulls.

Not only seabirds are potentially impacted by offshore wind farms. They are also of concern for other bird species like passerines (*i.e.*, non-seabird species). Large numbers of non-seabirds are indeed known to migrate at sea and over-seas mass migration events frequently occur (mostly blackbird *Turdus merula*, song thrush *Turdus philomelos*, redwing *Turdus iliacus*, robin *Erithacus rubecula* during night time, and meadow pipits *Anthus pratensis*, European starling *Sturnus vulgaris* and chaffinch *Fringilla coelebs* during day time). The development of offshore wind farms in the North Sea might impact these migrating birds as they can collide with the turbines. As to investigate the spatial and temporal patterns of bird migration at a large spatial scale and at high altitudes (in this study restricted to 1.8 km), we made use of a bird radar (Chapter 8). Bird migration traffic rates (MTR, $birds.km^{-1}.hr^{-1}$) showed that migration at sea was most intense during the nights of October and early November (up to ~800 $birds.km^{-1}.hr^{-1}$). Especially in October a clear peak in MTR values occurs at dusk. A second smaller peak is noticeable at dawn. The altitude profile suggests migration at night to happen at higher altitudes compared to daytime movements (maximum MTR at 100-150 m altitude during daytime and 200 to 300 m at night; note: radar data less reliable below 150 m altitude). While passerines tend to dominate night time migration, daytime migration tends to be a mixture of seabird and non-seabird species. Although

no clear correlation with weather conditions could be revealed, MTR values seemed higher when the wind blew from the N, NE, E and SE and when wind speed was lower than 13 m/s. In the future, the recorded bird fluxes will be analysed with an explanatory model approach to identify the variables driving the observed migration at sea (*e.g.*, wind direction and speed, hour of day, Julian day, bird flux at the previous day).

From May to September 2016 pile driving was taking place at the Bligh Bank. The investigation of 5 complete piling events of five steel monopiles of 5 m diameter (no sound mitigation measures in place) revealed a maximum sound exposure level (single strike) ranging between 166 and 174 dB re $1\mu Pa^2$ s at 750 m distance and a cumulative sound exposure level (full piling of a monopile) ranging between 201 and 209 dB re 1 μPa^2 s at 750 m distance from the source (Chapter 2). Applying these data to the pile driving activities foreseen for 2018 and 2019, the behavioural response zone for harbour porpoises *Phocoena phocoena* could reach some 2800 km², in the worst case scenario presented in this report.

During piling, porpoise detections, as detection positive minutes per 10 minutes interval, decreased by up to 75% at stations located up to 20 km from the location of the piling event. Inside the work area, detections decreased well before the start of piling works. At larger distances (20-55 km) porpoise detections nearly doubled during piling events, which may be due to displaced porpoises entering the area. Pile driving sound levels at the furthest distance where reductions in porpoise detections were observed were ~159 dB re 1 µPa (L_{z-p}), which is close to the threshold level for major disturbance for harbour porpoise proposed in literature.

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CHAPTER 1

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

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Abstract

Offshore wind farms are expected to contribute significantly to the Belgian 2020 targets for renewable energy. As of 2016, an installed capacity of 870 Megawatt (MW), consisting of 232 offshore wind turbines, is operational in the Belgian part of the North Sea. In 2017 and 2018, an additional capacity of respectively 275 and 320 MW will be added (fig. 1), with three other projects scheduled for the next few years after that. 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 chapter provides an overview of the offshore renewable energy development in the Belgian part of the North Sea.

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
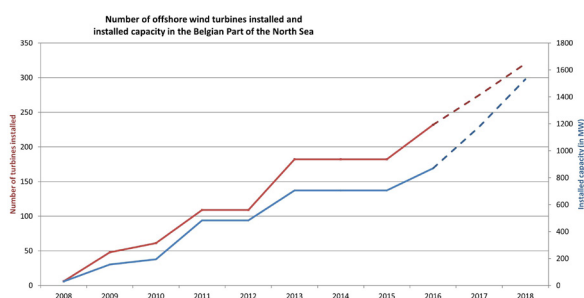
Figure 1. Number of offshore wind turbines installed and installed capacity in the Belgian part of the North Sea since 2008.

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 (fig. 3).

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.

Table 1. Overview of wind farms in the Belgian part of the North Sea (situation on 20 October 2016)

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 since 2013
Nobelwind		50	3.3	165	YES	YES	Operational since 2017
Northwind		72	3	216	YES	YES	Operational since 2014
Norther		44	8	320	YES	YES	Construction foreseen to start in 2018
Rentel		42	7.35	275	YES	YES	Construction started in July 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 II		22-32*	3-10	217-224	YES	YES	Construction foreseen to start in 2019

*number of turbines and/or total capacity still to be decided; **including 5 MW of wave energy.

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 and 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.

On 13 March 2017, NEMOS received an environmental permit for the construction and exploitation of a temporary research structure for wave energy conversion, at a distance of about 500 meters north of the eastern harbour wall in Ostend. A monitoring programme focusing on underwater sound and the impact on soft substrate benthos was imposed. After an operational test phase that ends in 2020, the installation will be dismantled and removed.

2. Marine spatial plan and aquaculture

On 20 March 2014, Belgium approved a new marine spatial plan for the BPNS 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 (fig. 3). 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. On 22 May 2017, the resultant *Edulis*-project deployed a mussel longline in the C-Power windfarm in order to test the practical viability of mussel longline aquaculture in the wind farms (fig. 2)

Figure 2. Project Edulis mussel longline in C-power windfarm (Nancy Nevejan, UGent).

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 ready in 2018.
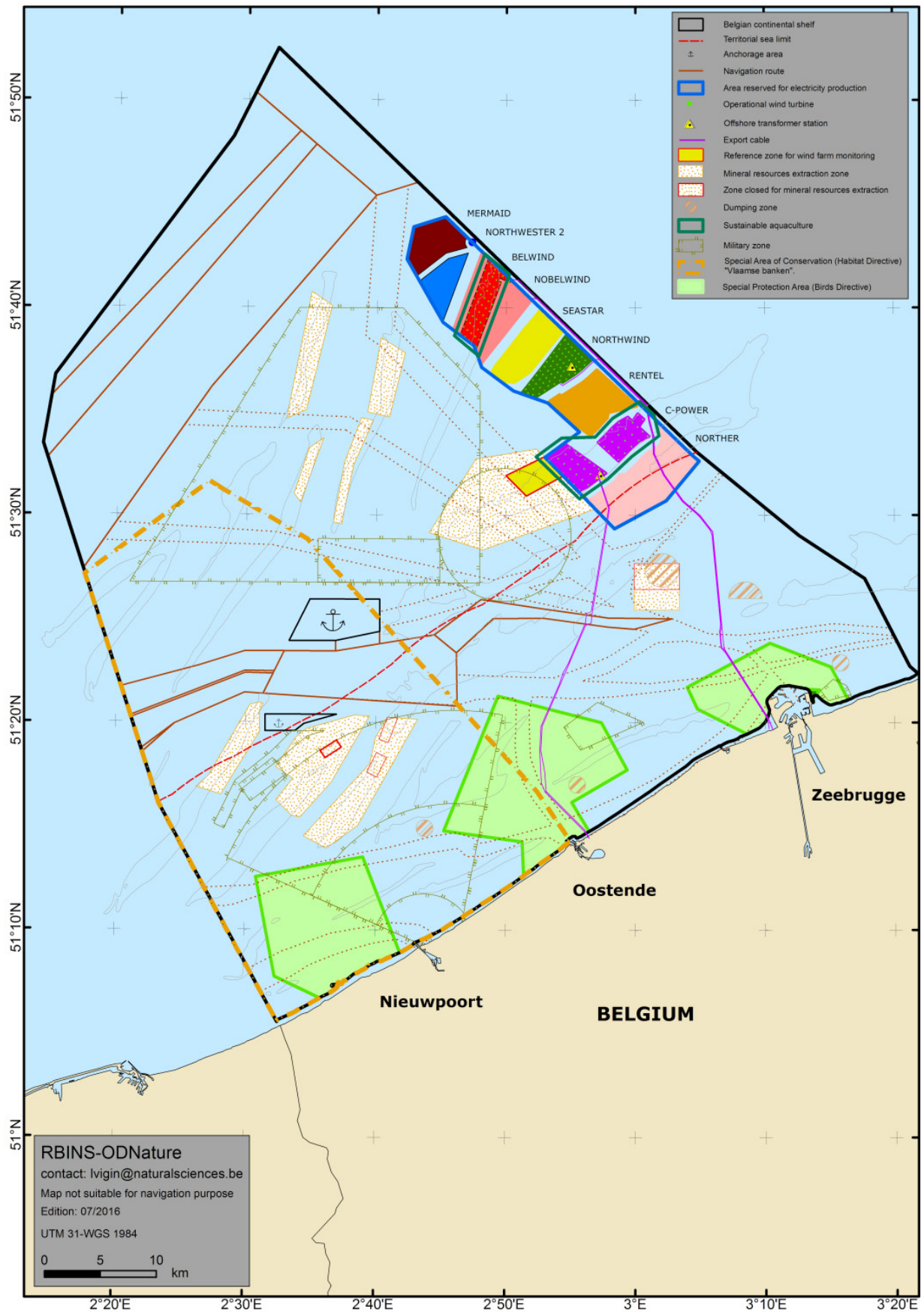
Figure 3. Marine spatial plan of the Belgian part of the North Sea.

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. In its current iteration, the Modular Offshore Grid (MOG), consisting of a single Offshore Switch Yard (OSY) located near the Rentel concession and four export- and/or interconnection cables, would connect four of the remaining wind farms to the grid (fig. 4). Construction of the MOG is expected to start at the end of 2018.
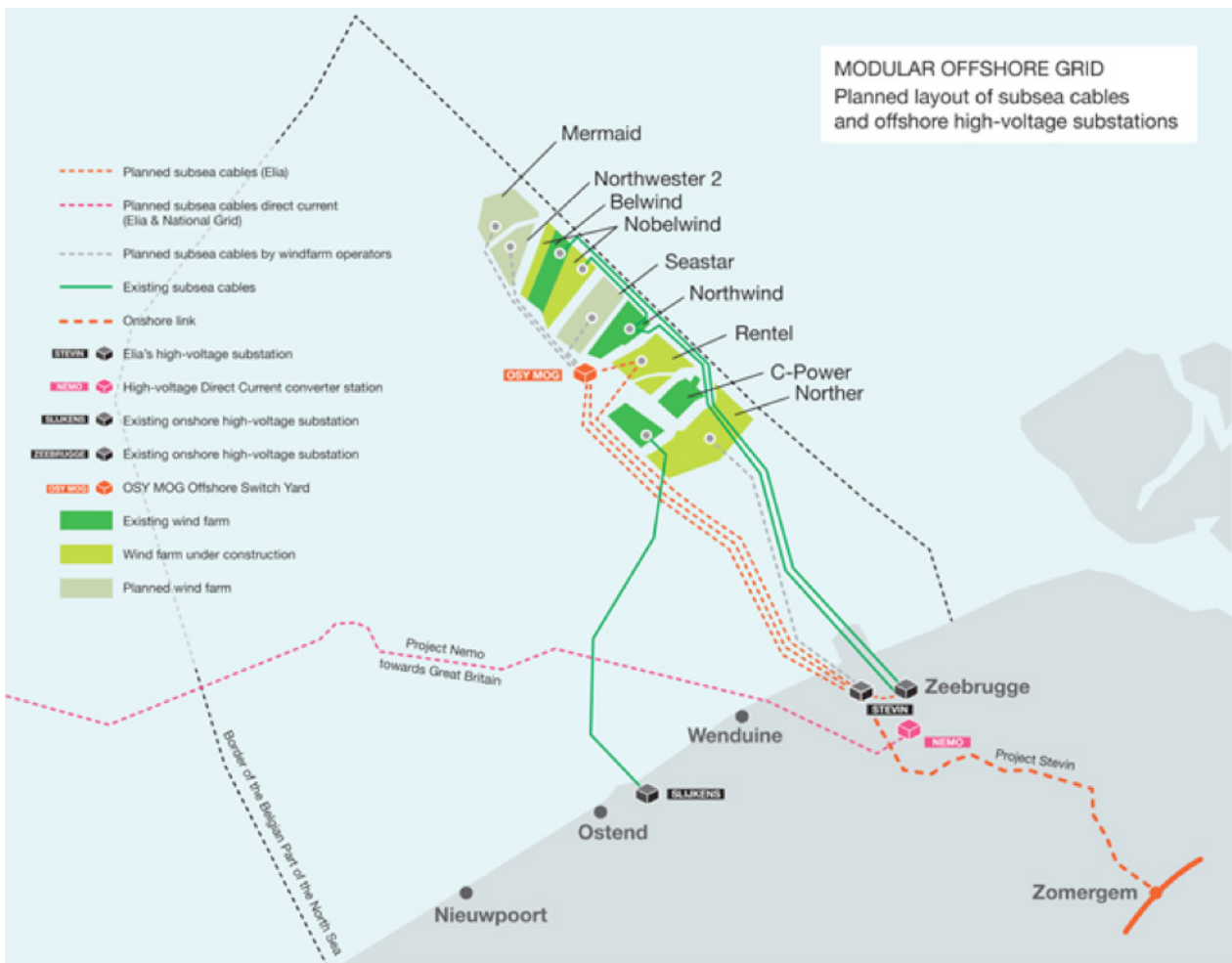
Figure 4. Design for the Modular Offshore Grid (MOG) (Source: http://www.elia.be).

CHAPTER 2

CHARACTERIZATION OF THE UNDERWATER SOUND EMITTED DURING THE INSTALLATION OF MONOPILE STEEL FOUNDATIONS AT THE NOBELWIND OFFSHORE WINDFARM AND CUMULATIVE EFFECTS

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Abstract

The construction works of the Nobelwind wind farm offshore on the Belgian coast was monitored for the emission of energy into the sea by means of underwater sound (pressure). Five complete piling events which cover the driving to full depth of 5 steel monopiles of 5 m diameter using a hydraulic hammer with a maximum power of 1400 kJ are described. No direct mitigation of the produced sound pressure was used and at 750 m distance, measured maximum SEL_{ss} ranged from 166 to 174 dB re 1µ Pa^2 s. Different metrics are proposed for the assessment of the cumulative effect of the piling works. SEL_{cum} ranged from 201 to 209 dB re 1µ Pa^2 s. Number of strokes used ranged from 2582 to 3696 while the energy used ranged respectively from 1764080 to 4048143 kJ. During piling work foreseen for 2018 and 2019, the behavioural response zone for the harbour porpoise could reach some 2800 km^2.

1. Introduction

High intensity impulsive sound like the one produced by underwater explosions, pile driving or seismic surveys using air-guns are known to affect marine life adversely (Hawkins & Popper 2016). Most of the available studies characterize local exposition to impulsive underwater sound (de Jong & Ainslie 2008; Bailey *et al.* 2010; Norro *et al.* 2013; Haelters *et al.* 2015; Popper & Hawkins 2012; 2016). Today, the development of marine renewable energy (MRE) in the North Sea shows numerous new projects. The risk exists that cumulative effects resulting from simultaneous construction or operation may affect marine life not anymore at the individual level but at a population level. Underwater sounds propagate at about 1500 ms^{-1} over large distances and are not stopped by national boundaries making the assessment of cumulative sound pressure a regional matter.

The purpose of this report was (1) to characterize the emitted underwater sound during piling events, (2) to evaluate the emitted sound during the piling of the foundation of the Nobelwind offshore high voltage station (OHVS) that is, currently the largest pile ever piled in the Belgian part of the North Sea (BPNS), and (3) to quantify cumulative effects and define the spatial extent of behavioural response zone for the harbour porpoise.

2. Material and methods

2.1. Study area

The Nobelwind wind farm is situated offshore the Belgian North Sea coast on the Bligh Bank (fig. 1). The wind farm actually represents the second phase of construction extending the Belwind wind farm operational since January 2011. This second phase of the wind farm construction requires the installation of 50 steel monopiles of 5 m diameter (lower end) and of lengths ranging from 54 m to 76 m. One additional monopile for the offshore high voltage station (OHVS) was installed in the concession zone. This steel monopile of 6.8 m diameter (lower end) and a length of 72 m represents the largest monopile ever driven into the Belgian seabed.

The first steel monopile of the Nobelwind offshore wind farm was installed on 16 May 2016 (BBK01) and the last one was piled on 22 September 2016 (BBI04). During the piling construction works underwater sound emitted during five complete pile driving events was recorded by means
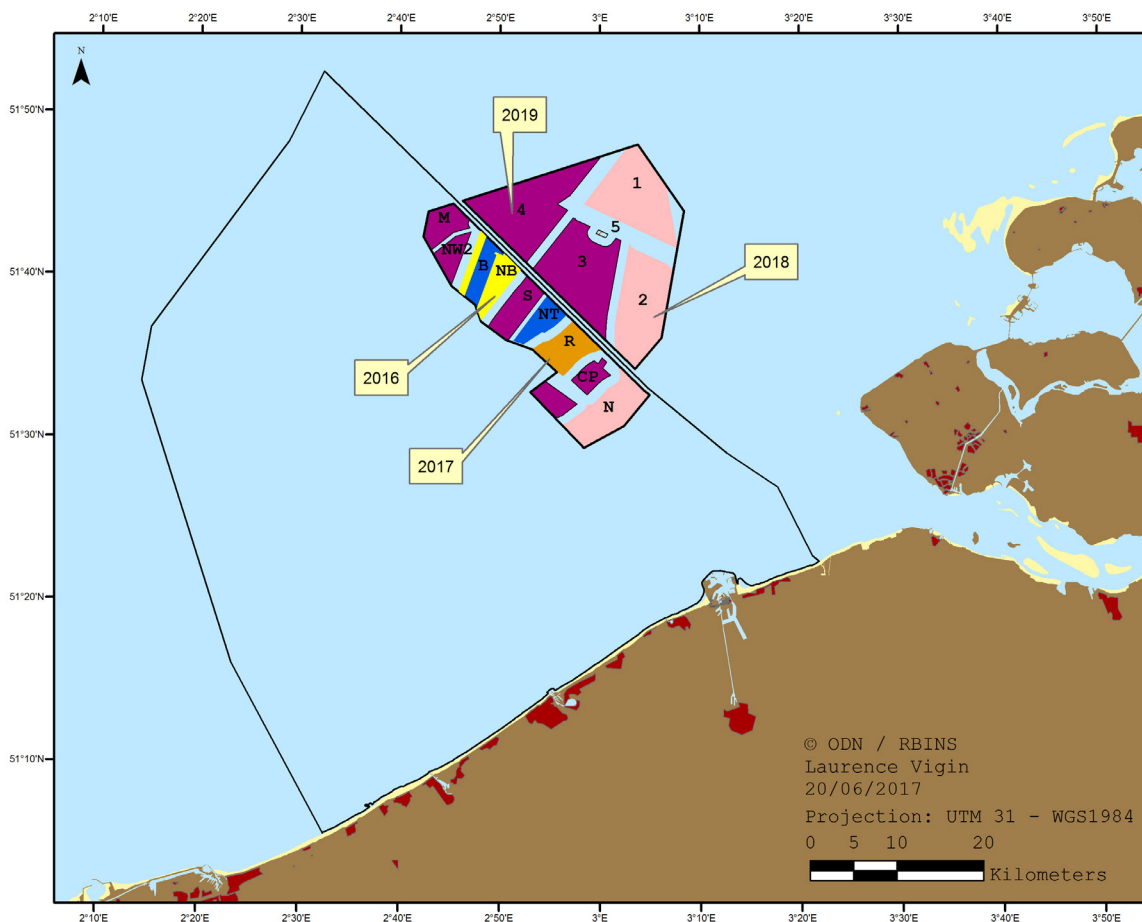
Figure 1. Implementation of the Nobelwind windfarm (NB on the map). In pink what is foreseen for construction in 2018 and in purple what is foreseen for 2019.

of a moored station. A 1400 kJ hammer operated from a jacking-up platform was used for the piling job. No directly emitted sound mitigation was in place but the construction permit (granted in 2008 and extended in 2015) required the use of an acoustic deterrent (Lofitech seal scarer), to be deployed one hour before piling starts and for a "ramp up" or "soft start" procedure to be used at the beginning of every piling event.

2.2. Underwater sound measurement equipment

Underwater sound was recorded from a moored station consisting of an instrumented tripod (fig. 2). The tripod was equipped with a complete measurement chain including a recorder RTsys EA-SDA14 and one hydrophone B&K 8104. RTsys calibrated the complete measurement chain prior to shipping from the factory. The calibration was verified using a calibrator B&K 4229 (piston-phone) prior to every deployment.

The instrumented tripod was deployed on 14 August 2016 using an acoustic release to lower it to the seabed in the vicinity of the planned piling location (WGS84 N 51° 39,875; E 002° 50,590 by 38 m depth relative to mean sea level). As such, the distance between the measuring equipment and the piling locations ranged from 850 to 3600 m. No surface marker was left on site to reduce navigation risk inside the construction zone as well as to avoid any perturbing sound originating from a line linking a surface buoy to the tripod. Scientific divers serviced the instrument on 27 August 2016 and retrieved the recording instrument on 25 October 2016.

2.3. Underwater sound measurements and post-treatment

Sound pressure was recorded continuously at a sampling rate of 78125 Hz and stored on hard drive coded in WAVE format.

During the period of deployment, the following piling events (table 1) were fully recorded.

Homemade routines in MATLAB were used for the post treatment of the records.
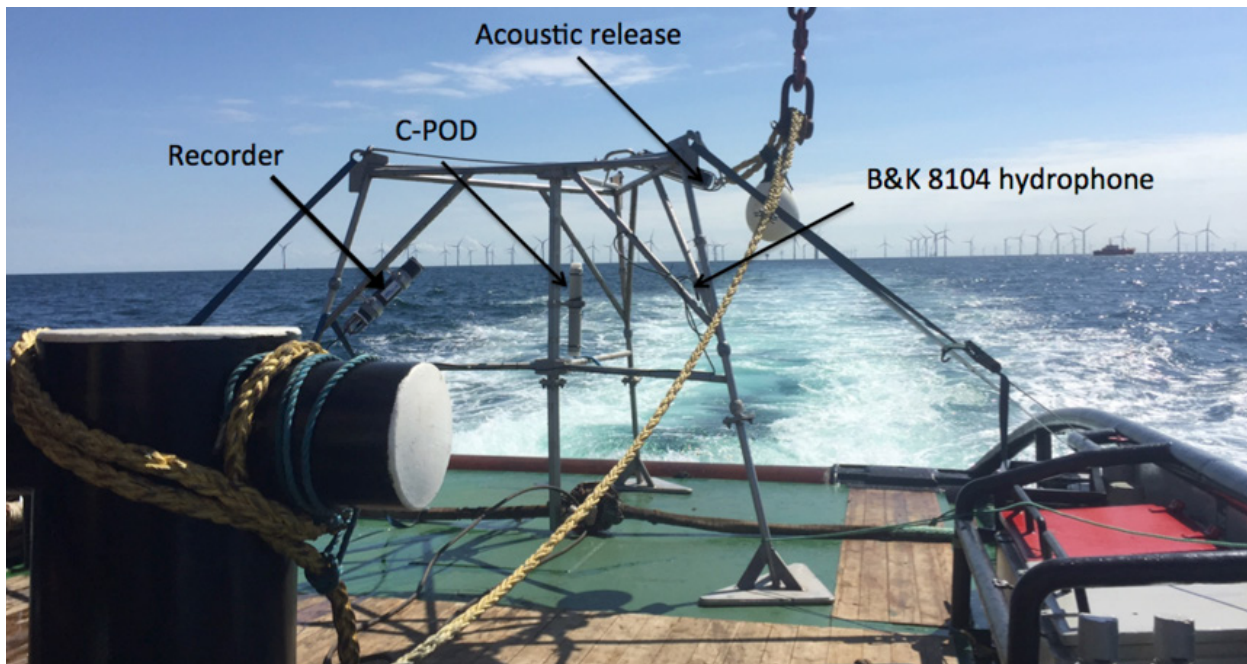
Figure 2. Tripod with RTsys sound recorder and B&K 8104 hydrophone mounted, C-POD and acoustic release ready for deployment on the rear deck of the MTS Valour (© Alain Norro/RBINS).

Table 1. Data available for the Nobelwind windpark piling phase

Pile name	Distance from recorder (m)	Start	End	Total duration
BBH01	1600	18/8/16	18/8/16	2h09
BBH03	860	17/8/16	17/8/16	4h32
BBH05	2100	30/8/16	30/8/16	4h24
BBH06	2000	27/8/16	27/8/16	2h46
BBH07	3400	31/8/16	31/8/16	2h10

Sound exposure level for a single strike and for a full piling event (SEL_{ss} and SEL_{cum}) as well as the normalization of levels to the reference distance of 750 m were computed according to Norro *et al.* 2013.

Because the intensity of the sound emitted depends on the size of the sound source, the intensity of the sound increases with the pile diameter. ITAP in Germany proposes an experimental model (Bellmann *et al.* 2017) that permits estimation of both SEL and L_{z-p} from the diameter of the monopile to be driven into the sediment. The ITAP model will be used to estimate level generated by the piling of the OHVS.

2.4. Cumulative effects

Cumulative effects characterize the effects resulting from the total number of strokes required for the installation of a single monopile or for a complete wind farm or even for the construction of a cluster of wind farms forming a zone of energy production. Metrics are elaborated to try to provide such integrated information. The cumulative sound exposure level (SEL_{cum}) is one of those metrics. In such context, the total energy spent during the complete operation of the piling of one monopile or wind farm or cluster of wind farms is another metric.

Worst-case scenarios for cumulative assessment occur here when construction in both Belgian and Netherland waters are simultaneously conducted (fig. 1). In this case large zones around these works may impact the behaviour of harbour porpoises. De Jong *et al.* (2017) estimated that a SEL_{ss} level of 140 dB re $1\mu Pa^2$ s should not be exceeded to guarantee absence of behavioural response for the harbour porpoises. Based on such threshold and the propagation model proposed by Norro *et al.* (2013), impact area size are estimated by drawing circles centered on every new project foreseen for a given year.

3. Results

Not considering the OHVS, the measured L_{z-p} ranged from 190 to 198 dB re $1\mu Pa$ while SEL_{ss} ranged respectively from 166 to 174 dB re 1 μPa^2 s. Piling of the 6.8 m diameter monopile used for the OHSV is modelled to increase the maximum reported L_{z-p} by 5 dB and the SEL_{ss} by 4 dB.

The installation of the OHVS required about twice the number of strokes compared to BBH03 and about three times the energy spent for piling BBH03. When considering the sound levels to be similar to the one measured for other monopiles ($OHSV_{similar}$), a SEL_{cum} reaching 211 dB re 1 μPa^2 s is estimated for piling the OHVS pile. When applying the ITAP model ($OHSV_{mod\ Itap}$), a L_{z-p} of 203 dB re 1 μPa and a maximum SEL_{ss} of 178 dB re 1 μPa^2 s was estimated.

The propagation model proposed by Norro *et al.* (2013) estimated a 20 km radius impact zone centered on the sound emission point (OHVS not considered here).

Based on this and on the location of future piling sites presented at fig. 1, one can estimate a 2800 km² behavioural response zone for harbour porpoise in the worst case scenario presented in this paper.

4. Discussion

There is not only one adequate metric that could be used to translate the loudness of the

Table 2. Nobelwind construction phase

Pile name	$L_{z\text{-}p}$ in dB re 1µ Pa	Max SEL_{ss} in dB re 1µ Pa^2 s	SEL_{cum} in dB re 1µ Pa^2 s	Total energy (kJ)	Number of strokes
BBH01	197	174	209	2977919	3297
BBH03	198	174	205	1764080	2582
BBH05	196	171	206	2892379	3123
BBH06	190	166	201	2229876	2753
BBH07	191	169	205	4048143	3696
OHVS$_{similar}$	≈	≈	≈ 211	5180744	5157
OHVS$_{mod\ Itap}$	203	178	215		

Measured parameters Lz-p , Max SELss and SELCUM normalized at 750m distance from the pile. Total energy & number of strokes provide from the hammer log (Nobelwind data). Offshore High Voltage Station (OHVS) not monitored *in situ* but estimation proposed here in two options (see text).

produced sound to an effect on the marine biota. Hawkins and Popper (2016) showed that the cumulative sound exposure level (SEL_{cum}) introduced for marine mammals is not the appropriate metric to be used for fishes and invertebrates. Hawkins and Popper (2016) propose characterizing the emitted sound using other metrics such as the sound exposure level of a single stroke (SEL_{ss}) together with the total time of piling and the total number of strokes.

The figures proposed here for a limited number of monopile installations can be extrapolated to the complete construction resulting in very high levels (above 185 dB re 1 µPa $L_{z\text{-}p}$) of underwater sound in the vicinity of the construction site.

Moreover, because of the concentration of the zones reserved for energy production (fig. 1), a cross-border strategy on cumulative sound emissions needs to be encouraged should a reduction of excessive underwater sound be strived for in the near future.

The 20 km circle radius of behavioural disturbance for harbour porpoise (*Phocoena phocoena*) confirms the radius of 16 km that was already proposed by Norro

et al. (2013) for the major behavioural disturbance zone based on a level peak uto peak ($L_{p\text{-}p}$) of 155 dB re 1 µPa. That radius was further confirmed by Haelters *et al.* (2015) investigating harbour porpoise distribution changes during piling activities. There is a need for more research and standardization from the bio-acoustician in the development of behavioural response thresholds for marine mammals as well as for other animals like fishes and invertebrates.

In 2018, construction of three new wind farms in the zone is planned. Two of them will be installed in Dutch waters and one in Belgian waters. For 2019, the construction of another five wind farms is planned for, with three inside Belgian and two inside Dutch waters. From the above it is clear that any construction inside the Belgian zone will impact Dutch waters and any construction inside most of the Dutch Borssele zone will impact Belgian waters.

5. Conclusion

For a monopile of 5 m diameter and a hydro-hammer of 1400 kJ without direct underwater sound mitigation, a $L_{z\text{-}p}$ ranging from 190 to 198 db re 1 µPa at 750 m

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from the source was detected, while at the same distance the SEL_{ss} ranged from 166 to 174 dB re 1 μPa^2 s. An estimation of the emitted underwater sound resulting from the installation of a 6.8 m monopile used for the OHVS gave a L_{z-p} of about 200 dB re 1μPa at 750 m and a SEL_{ss} of about 180 dB re 1 μPa^2 s.

A zone of 20 km radius was confirmed as a behavioural response zone for harbour porpoise and concerns are highlighted for the coming year 2018 and 2019 when construction of seven new windfarm projects is scheduled in the Belgian and adjacent Dutch (Borssele) offshore energy zones. During the year 2018 and 2019 the zone of behavioural response for harbour porpoises may reach some 2800 km^2.

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CHAPTER 3

SWIM BLADDER BAROTRAUMA IN ATLANTIC COD WHEN *IN SITU* EXPOSED TO PILE DRIVING

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Abstract

In view of the rapid increase of offshore wind farms in the North Sea, and in order to further determine sound thresholds to be used in international guidelines, it is needed to acquire more knowledge on the effects of pile driving sounds on fish health. Therefore, in the summer of 2016, a field experiment was undertaken in the Nobelwind OWF on the BPNS to determine the direct effect of pile driving on the health status of Atlantic cod (*Gadus morhua*). Large netted cages, each holding 9 to 12 cod individuals (avg. size 31 cm), were submerged at 8 m under the water surface. The cages were placed at increasing distances (75 m, 400 m, 1400 m and 1700 m) from the sound source, being the offshore installation vessel *Vole au vent*. All cages were submerged for on average 16 hours before pile driving, after which all fish were exposed to one pile driving event (lasting on average 2 hours). A similar control experiment was repeated in the same period when no pile driving took place. Underwater sound levels were measured at different distances during pile driving, while background measurements were made to determine ambient sound levels. Average single strike sound exposure levels (SEL_{ss}) decreased from 175 dB re 1μPa²s at 400 m distance to 168 dB re 1μPa²s at 1700 m

distance. Ambient sound pressure levels (SPL) varied between 114 and 138 dB re 1μPa. After retrieval of the cages onboard RV Simon Stevin, all cod individuals were evaluated for buoyancy in water tanks. Shortly afterwards, all fish were euthanized and examined for swim bladder barotrauma and internal bleeding. Overall, 11% cod were retrieved dead, most probably due to handling stress, as no direct relation could be found with distance to the sound source. On the other hand, a steep increase in swim bladder barotrauma was detected with decreasing distance to the pile driving source: no swim bladders were ruptured at 1700 m nor at the control treatments, 20% were ruptured at 1400 m distance, 40% at 400 m distance and up to 90% of the swim bladders were ruptured at 75 m distance. Although most fishes in the cages in the direct vicinity of the piling source (100 m distance) did survive this short term experiment, they all showed many multiple instances of internal bleeding and a high degree of abnormal swimming behavior, hinting towards a reduced survival rate on the longer term. However, these immediate detrimental effects seem to occur only locally, close to the high impulsive sound source, as swim bladder injuries rapidly decreased with increasing distance from the pile

driving source. Results of this *in situ* experiment provide valuable information to scientifically evaluate the current "critical sound limits" implemented in Belgium in the setting of the Marine Strategy Framework Directive.

1. Introduction

Underwater sound related to human activities is an increasing source of pollution in the marine environment (Hildebrand 2009). Although offshore wind farms (OWFs) do create green energy, they alter temporarily and permanently the marine ecosystem by introducing different types of underwater sound. Especially during the construction phase, high impulsive sound is generated when the steel foundation piles are driven into the sea bottom. Impulsive underwater sound can be detrimental to marine life. Several laboratory experiments on fish and marine mammals showed disturbance of behaviour, physiological stress, internal and external injuries, sometimes leading to mortality (Popper & Hastings 2009; Hawkins & Popper 2016). However, a recent *in situ* study in the Belgian

part of the North Sea (BPNS) only showed short term physiological effects in larval and juvenile seabass (*Dicentrarchus labrax*) after exposure to high impulsive pile driving sound in the direct vicinity (< 50 m) of a real pile driving event (Debusschere *et al.* 2014; 2016). Still, during that field experiment adult whiting (*Merlangus merlangus*) was seen floating at the surface at the moment of pile driving. Next to the need to further determine solid sound thresholds to be used in international guidelines, this anecdotal observation was the immediate reason for the current *in situ* experiment with Atlantic cod (*Gadus morhua*).

2. Material and methods

2.1. Study area

To examine the impact of pile driving on Atlantic cod, a field experiment was undertaken during construction of the Nobelwind wind farm, situated on the Bligh Bank (fig. 1). In total, 50 monopiles have been installed and each one was designed for its specific position in the wind farm, and varied in length, diameter and steel thickness. The monopiles

Table 1. Characteristics, date, pile driving time, number of strikes, energy and depth for each monopile

MP number	J05	J08	J07	I06
Measurement type	Cod exposure 1	Cod exposure 2	Sound measurement 1	Sound measurement 2
Date	5/07/2016	13/07/2016	12/07/2016	13/09/2016
Time of day (h)	01:35	10:18	08:56	15:50
Diameter (m)	5	5	5	4.5
Steel thickness (mm)	70	70	70	78
Length (m)	66.4	67.7	65.3	67.2
Depth in seafloor (m)	30	30	32	30
Total strikes	2985	2888	3606	3123
Total energy (kJ)	2488771	2380981	3020305	2017849
Total pile driving time (h)	2:18	2:03	3:11	1:52
Net hammering time (h)	1:14	1:17	1:38	1:12

were installed by using a hydraulic piling hammer (IHC Hydrohammer B.V.). During our exposure experiments, monopiles J05 (lat. 51.67223°, long. 2.86620°) and J08 (lat. 51.67255°, long. 2.84803°) were driven into the seabed. When sound was measured monopiles J07 (lat. 51.67005°, long. 2.85506) and I06 (lat. 51.65195, long. 2.84043) were installed (table 1, fig. 1).

2.2. Characteristics, catching and housing of Atlantic cod

Atlantic cod (*Gadus morhua*) is an important commercial species but due to overexploitation, it is classified as a vulnerable species on the IUCN list (http://www.iucnredlist.org/). Age I and II-group cod are known to aggregate around OWFs in the BNS (Reubens *et al.* 2013; 2014). Atlantic cod is a round fish with a closed swim bladder (physoclist), which makes it more vulnerable to swim bladder injuries. Physoclistous fish cannot rapidly change the volume of their swim bladder, but depend on gas secretion and absorption to regulate their buoyancy. Consequently, when exposed to high impulsive sound such as pile driving, the swim bladder acts as an air bubble which vibrates. These vibrations can cause damage to the swim bladder itself or to neighbouring organs (Halvorsen *et al.* 2012a and references herein).

The Atlantic cod used for this experiment were caught using hook and line gear (bait: *Arenicola marina*) near the gravity-based foundations of the C-Power wind farm (51°33′N, 2°56′E, WGS84) from RHIB Zeekat on 23 June 2016 and 7 July 2016. Depth around the foundations is around 23 m at mean low water spring (MLWS). In order to minimize the risk of barotraumas, fish were hauled very slowly to allow them to release excess gas and prevent swim bladder rupture. Fish ranged between 21 and 42 cm in total length, and were on average 31 cm (± SD 4 cm) (age I-group). 85% of the fish survived the angling, the other 15% died almost immediately because of barotrauma of the swim bladder. In total, 87 individuals (70 on 23 June, 17 on 7 July) survived to be used as test animal. After capture, the fish were kept on board RV Simon Stevin in an aerated flow-through seawater tank covered with wet blankets to create a shaded environment during transit to the land-based facilities. Transit at sea took between 6 and 8 hours.

Back on land in Ostend, the water tanks were immediately transported (5 minutes) to the Marine Station Ostend (MSO) of VLIZ, where the cod individuals were housed in two large, circular water tanks (4000 l, 2.5 m Ø and 1.2 m depth) for acclimatization. Each tank contained a maximum of 45 individuals. Both tanks were completely separated and were provided with aeration by a flow through of ozone sterilized seawater in a closed circulation system. Furthermore, each tank had its own filtration system: a biological filter tank and mechanical filters (drum filter and protein skimmer). The tanks were located in a climatic room (100 m²) with adjustable light and temperature regime. Conditions in the tank were kept as similar as possible to the natural conditions, which were a sea water temperature around 15.5° C, salinity of 30 PSU and a light regime (with dimmed light) of 16 h light/8 h dark. Each tank was equipped with a temperature sensor and a redox sensor. Mortality was checked every day: only one dead fish was observed and removed. Water quality (pH, NH_4^+/NH_3, NO_2^-, NO_3^-, O_2 % and kH) was checked on a near daily basis and when needed, part of the seawater in the circulation system was replaced by fresh sea water to restore the water quality. After an acclimatization period of 5 days, the cod were fed every two to three days with frozen fish, shrimps or lugworms (± 2% of estimated body weight/day).

The cod were kept in these aquarium units between 5 and 19 days before the experiment took place. At the day of the experiment, the cod were transferred to a mobile water tank, transported to RV Simon Stevin (5 minutes) and put on board. During

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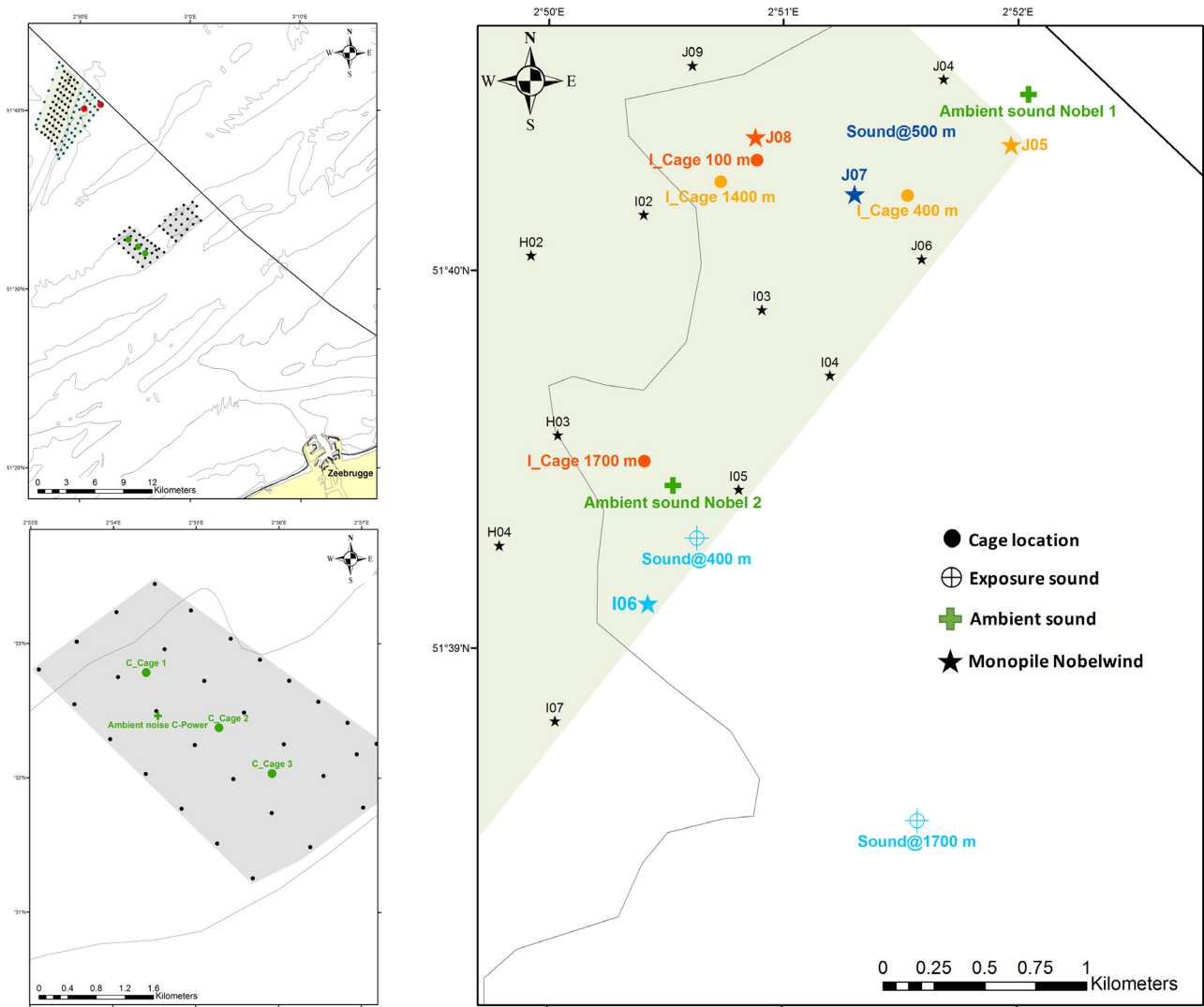
Figure 1. Top left: Overview of study area with indication of control locations (green dots) and exposure locations (red dots) in under construction wind farm Nobelwind (in blue). Below left: Locations of the three control cages in the C-Power wind farm and location of the background sound measurement for the control locations. Right: Zoom in showing 1) the monopiles driven in the seabed during cod exposure (orange J08 and yellow J05) with position of the exposure cages (same colour as mp) relative to the monopile; 2) the monopiles driven in during sound measurements (I06 and J07 in blue) with position of measurement locations (same colour as mp); 3) locations of the background sound measurements in the Nobelwind concession area when no pile driving took place.

transit to the *in situ* experimental location, tanks were provided with aeration and seawater flow through, and covered with wet blankets to create a shaded environment.

The experimental protocol was approved by the ethical committee of the Institute for Agricultural and Fisheries research (ILVO) (Permit Number: EC 2016/275 and recognition LA1300512 for temporal storage in MSO).

2.3. Cage design and experimental set-up

To expose cod, large netted cages (mesh size 2 x 2 cm, 1.5 m Ø and 6 m height), were submerged at an average depth between 7 and 14 m (fig. 2). The cages were kept in place by a weight of 600 kg at the seabed. Subsurface buoys were used to keep the cage open and a surface buoy was put in place to be able to relocate the cage position for pick-up (fig. 2). Cages were put in position and
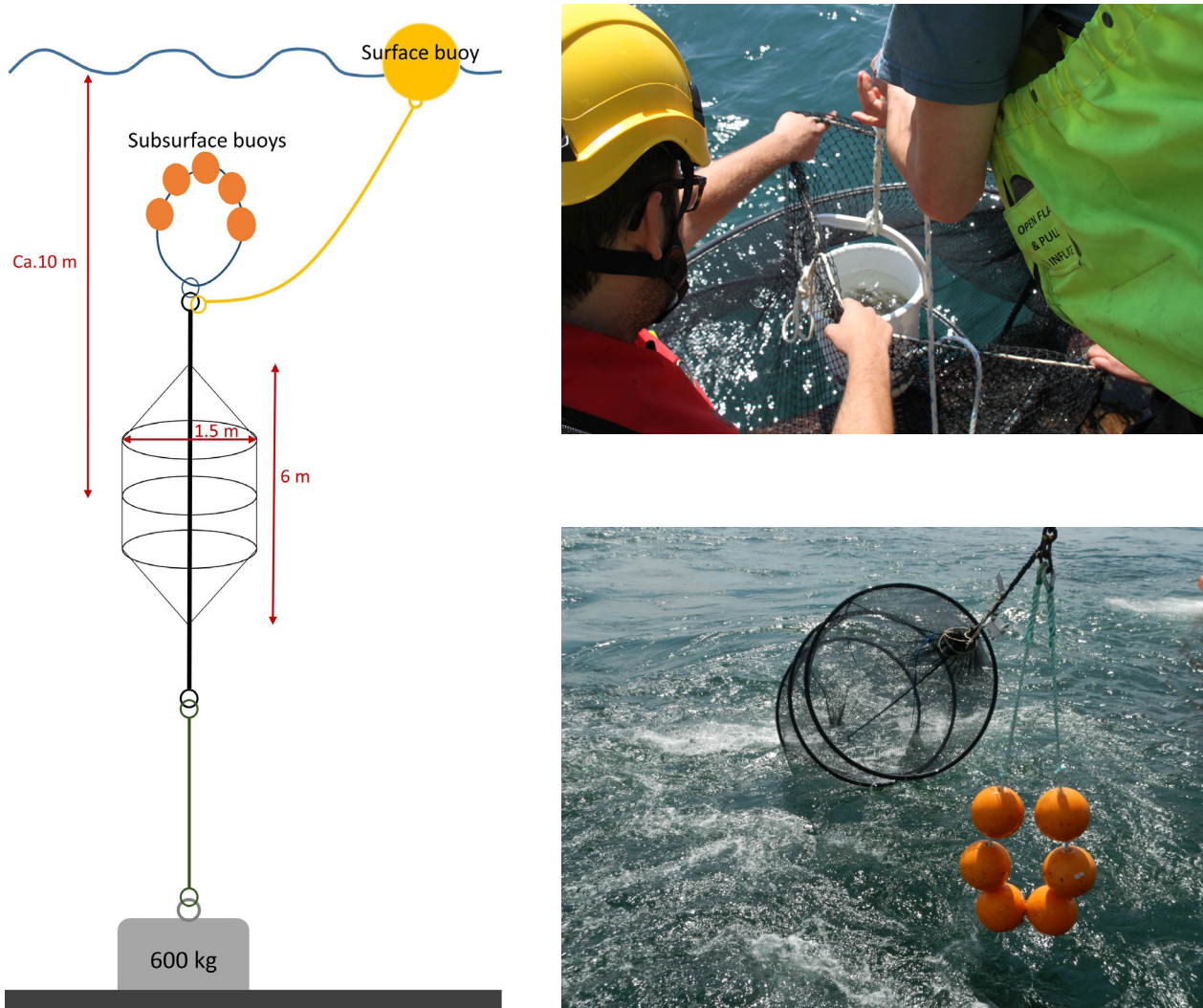
Figure 2. Left: schematic drawing of the set-up of the cages used for the exposure and control treatments. Right top: picture of cage. Right down: putting cod in the cage.

picked up by RV Simon Stevin at moments with sea state ≤ 3.

Lowering of the cages was executed with the winch of the ship's A-frame. Once the cage was partly submerged, fish were gently put in the cage with a bucket filled with seawater. After closing the cage, it was lowered further until the mooring weight touched the bottom. Length of the ropes (cage to weight; cage to surface buoy) was adjusted to the depth of the seabed at location to make sure that cage depth was similar for all cages.

The study set-up consisted of two treatments: exposure and control. Exposure cages, each holding 10 or 12 age I-group cod, were placed at increasing distances (100, 500, 1400 and 1700 m) from the sound source, being the offshore installation vessel *Vole au vent* (fig. 1). This was done at two different moments in time, when pile driving was predicted to occur. At each date, two cages were put out (table 2, fig. 1). Cages were put in place at least 12 hours before the start of pile driving in order for the fish to adapt to the pressure at depth and acquire neutral buoyancy. It was intended to retrieve cages after 24 hours (cf. control treatment), but due to bad weather conditions at both exposure occasions, this was not possible. Cages were only retrieved after 45 hours (table 2).

Table 2. Metadata on timing, depth, fish length and exposure time for each experimental set-up (Exp = exposure)

Experiment	Exposure 1 (Nobelwind)		Exposure 2 (Nobelwind)		Control (C-Power)		
Treatment	Cage@500 m	Cage@1400 m	Cage@100 m	Cage@1700 m	Cage 1	Cage 2	Cage 3
Monopile	J05	J05	J08	J08	/	/	/
Number of cod individuals	10	10	12	12	9	9	9
Average length ± SD (cm)	30 ± 4	30 ± 3	30 ± 5	31 ± 5	31 ± 4	33 ± 5	30 ± 3
Date in	4/07/2016	4/07/2016	12/07/2016	12/07/2016	6/07/2016	6/07/2016	6/07/2016
Time in (h)	12:30	13:00	15:02	14:15	14:30	14:55	15:15
Date out	6/07/2016	6/07/2016	14/07/2016	14/07/2016	7/07/2016	7/07/2016	7/07/2016
Time out (h)	12:05	12:55	12:52	12:20	11:40	12:15	13:11
Date piling	5/07/2016	5/07/2016	13/07/2016	13/07/2016			
Total time in H$_2$O (h)	47:35	47:55	45:50	45:55	21:10	21:30	21:56
Time in H$_2$O before exp (h)	13:05	12:35	19:03	18:15			
Time in water after exp (h)	32:12	33:02	24:31	23:59			
Avg depth cage (m)	10	12	8	8	14	7	11
Depth location (m)	33	35	33	33	24	22	21

For the control treatment, a similar experiment was repeated in C-Power (located at 15 km from Nobelwind), when no pile driving took place on the BPNS (fig. 1). Three replicate cages, each holding 9 cod individuals, were put out for 21 to 22 hours under similar conditions as the exposure treatments in order to be able to measure the cage effect (table 2).

Retrieval of the cages occurred with RHIB Zeekat and RV Simon Stevin. The RHIB could get close to the surface buoy to attach the winch rope of RV Simon Stevin onto the cage. When attached to the vessel, winching started very slowly in order to allow the cod to release excess gas, and not to rupture the swim bladder. Both cages of exposure 1, control cage 2 and exposure 2 cage@100 m were not retrieved ideally due to collapse of the circular hoops of the cages whilst winching up. Because of these

retrieval issues, these fish got probably extra stress.

2.4. Cod necropsy

Upon retrieval of the cages, the cod individuals were, with help of a hand net, fished out of the cages when these surfaced, and placed onboard in water tanks with flow through and aeration. Necropsy was started two to three hours after retrieval of the fish from the cages, so they had time to adjust their buoyancy to atmospheric pressure.

Just before necropsy, buoyancy status of each fish was judged and noted by two persons. Buoyancy status of a fish was evaluated by observing its swimming behaviour during 5 minutes. Behaviours identified were: normal swimming near the bottom, on side swimming, belly-up swimming, abnormal swimming which is all behaviour different from the above (*e.g.*, struggling to

get down or up, very passive…) and dead. Afterwards, the fish was euthanized in an overdose anesthetic (5 g benzocaine dissolved in 25 ml acetone and 1 l sea water), total length and wet weight were measured, the fish was coded and a picture taken. Fish were taken randomly from the different cages at each experimental day, and handed over to a person performing the necropsy, who was unaware of the cage treatment. All necropsies were done by the same person.

The necropsy was focused on potential swim bladder (SB) injuries. For each fish, inflation or deflation of the SB was noted; the presence of ruptures or small holes in the SB; was noted as well; and it was also written down whether air was trapped in the body wall and if so, what the volume of the air bubble was. Each necropsy was documented with at least one photograph. The protocol followed was outlined after personal communication with Michele Halvorsen (CSA Ocean Science).

2.5. Acoustic equipment and sound recordings

Sound pressure was measured using two Brüel & Kjaer hydrophones (type 8104, voltage sensitivity 47.7 µV.Pa^{-1}, charge sensitivity 0.391 pC.Pa^{-1}, 10 m cable and 50 m cable).

The 10 m cable hydrophone was connected to the charge channel of a Brüel & Kjaer portable amplifier (Nexus type 2690-0S). The 50 m cable hydrophone was connected to a Brüel & Kjær amplifier (Nexus type 2692-0S4) The measurement chain was completed resp. with a multi-channel portable recorder (Tascam DR-680) and an audio MARANTZ Solid State Recorder (type PMD671). The signal was recorded in 1-channel WAVE format (.wav) on Compact Flash cards of resp. 16 GB (SanDisk Ultra) and 2 GB (Sandisk Ultra II) with a sampling rate of 44,100 Hz at 24 bit. To standardize the recorded signals, a reference signal together with the output sensitivity was used. Batteries powered all equipment. Hydrophones were deployed at 10 m depth for all sound recordings.

Recordings of pile driving sound were performed at two occasions (MP J07 on 12 July 2016 and I06 on 13 September 2016). The nearby measurements were made from a drifting RHIB (Zeekat) with motor turned off and the further away measurement from the anchored RV Simon Stevin (table 3, fig. 1). Background measurements were made to measure ambient sound at two occasions and at both the exposure location and the control location (fig. 1). See table 3 for details on the sound recordings.

Table 3. Detailed information (date, duration, ship, sensitivity) of all sound recordings performed

	Pile driving 1	Pile driving 2		Background		
	Sound@500 m	Sound@400 m	Sound@1700 m	Nobel 1	Nobel 2	C-Power
Date	12/07/2016	13/09/2016	13/09/2016	12/07/2016	13/07/2016	13/07/2013
Ship	RHIB Zeekat	RHIB Zeekat	RV Simon Stevin	RHIB Zeekat	Geosurveyor X	Geosurveyor X
Moving/Anchored	drift	drift	anchored	drift	anchored	attached to turbine
Recording duration	7'	21'	53'	36'	38'	35'
Number of strikes	283	945	1228	0	0	0
Sensitivity	100 µV Pa^{-1}	100 µV Pa^{-1}	1 mV Pa^{-1}	31.6 mV Pa^{-1}	10 mV Pa^{-1}	10 mV Pa^{-1}

The sound pressure metrics, zero-to-peak sound pressure level (L_{z-p}), average sound pressure level (SPL), single strike and cumulative sound exposure level (SEL_{ss} and $SEL_{cum,p}$) were calculated using Matlab R2012b (version 8.0). In addition, the sound pressure metrics were calculated per 1/3 octave band, resulting in the highest energy over the frequencies. More details on the sound pressure parameters and how these were calculated can be found in Debusschere *et al.* (2014).

3. Results

3.1. Sound parameters

The pile driving sound levels that were measured at two occasions at 10 m water depth reached an average SEL_{ss} of 175-176 dB re1 µPa².s at 400-500 m distance and 168 dB re1 µPa².s at 1700 m distance from the sound source (table 4). The L_{z-p} rose to 196-199 dB re1 µPa at 400-500 m distance and 188 dB re1 µPa at 1700 m distance, while $SEL_{cum,p}$ reached resp. 210-212 dB re1 µPa².s and 203 dB re1 µPa².s (table 4). The dominant energy during exposure (SEL_{ss}) was present

in the range 125-200 Hz, although no steep decline was recorded towards the higher frequencies (fig. 3). The ambient SPL during the background sound measurements varied between 114 dB re1 µPa (in Nobelwind) and 138 dB re1 µPa (in C-Power) (table 4).

3.2. Buoyancy status

In total, 8 out of 71 (11%) cod individuals died during the field study. Dead fish occurred in both the control and the exposure treatments (fig. 4). These fish probably died due to handling stress, as no direct relation could be found with distance to the sound source.

In the control treatments, on average 81% of all fish were evaluated with normal swimming behaviour versus 55% in the exposure treatments. The lowest percentage (33%) of normal swimming behaviour was noted for the cage@100 m, which consequently had also the highest percentage of swimming behaviour deviating from normal (42%). The other exposure cages showed a normal swimming behaviour between 50-60% (cage up to 1400 m) and 75%

Table 4. Sound pressure metrics measured at different distance from the sound source during pile driving and background metrics when no pile driving took place

| Exposure sound metrics	Sound@500 m	Sound@400 m	Sound@1700 m
Total number of strikes	3606	3123	3123
Total strikes measured	283	945	1228
SEL_{ss} mean (dB re1 µPa².s)	176	175	168
L_{z-p} (dB re1 µPa)	199	196	188
$SEL_{cum, p}$ (dB re1 µPa².s)	212	210	203
1/3 octave band with most energy (Hz)	125	160	200
Background sound metrics	**C-Power**	**Nobel 1**	**Nobel 2**
SPL (dB re1 µPa)	138	120	114
1/3 octave bands with highest energy (Hz)	25	50	125-200
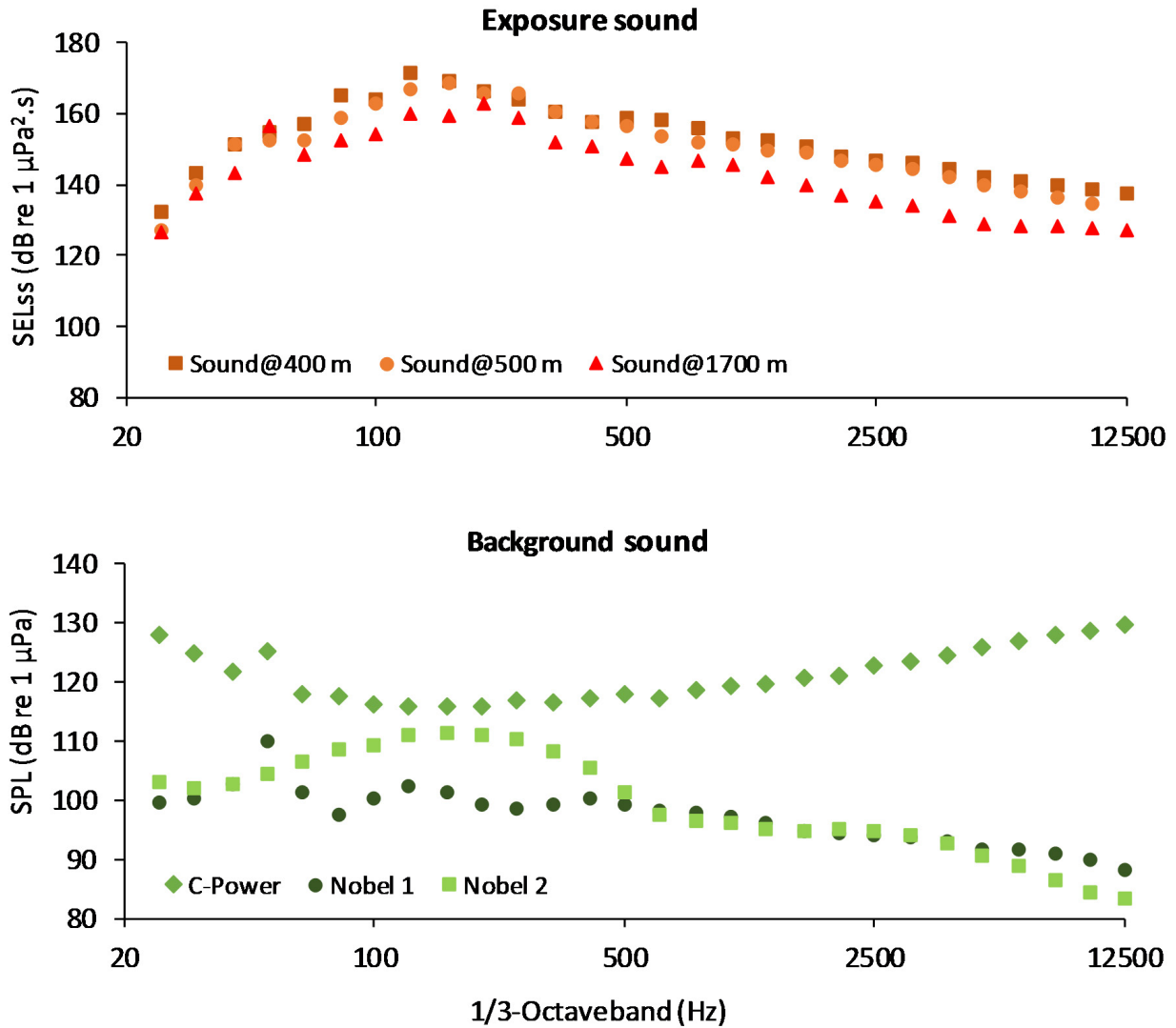
Figure 3. Measured frequency spectra in the presence (upper graph) and absence (lower graph) of pile driving. Mean SELss of the total recorded piling strikes versus 1/3 octave bands for exposure sound and SPL versus 1/3 octave bands for the background sound.
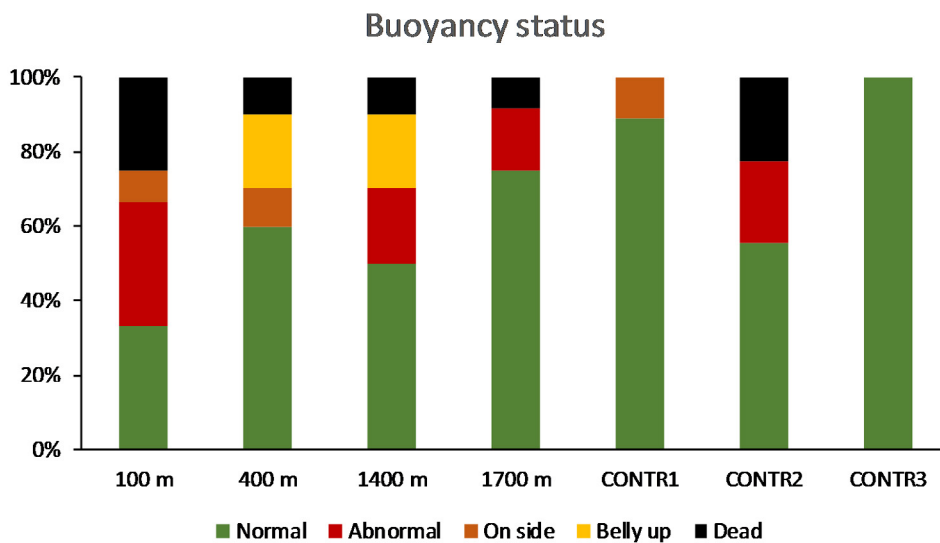
Figure 4. Relative occurrence of swimming behaviour for each control and exposure cage.

(cage@1700 m). Only 4 belly-up swimmers and 3 side swimmers were observed in all cages, mainly in individuals from exposure cages. For the control cages, only control cage @2 (which was not retrieved in the best circumstances) showed a slightly higher percentage of abnormal swimming behaviour (22%).

3.3. Swim bladder injuries and internal bleeding

A steep increase in swim bladder barotrauma was detected with decreasing distance to the pile driving source: no swim bladders were ruptured at 1700 m nor at the control treatments, 20% were ruptured at 1400 m distance, 40% at 400 m distance and more than 90% of the swim bladders were ruptured at 100 m distance (fig. 5). At most cages SB inflation was mostly 100%, while at the cage@100 m, a high percentage of deflated SBs (75%) was observed (fig. 5).

Concerning internal bleeding, the highest percentage of fish with multiple instances of internal bleeding (92%) was again detected for the cage@100 m, while at the exposure cages further away from the pile driving source, percentage of fish with internal bleeding still ranged between 20 and 50%. At control cages, on average 7% of fish with internal bleeding was observed (fig. 5).

4. Discussion

Pile driving for offshore wind farm construction causes ruptured swim bladders and internal bleeding in age I-group cod (avg. total length 31 ± 4 cm). However, these internal injuries decreased rapidly with increasing distance from the pile driving source, and consequently with decreasing sound level. The immediate detrimental effects seem to be restricted, occurring only close to the high impulsive sound source. At 100 m distance of the pile driving source, over 90% of the swim bladders were ruptured while at a distance of 1700 m, no ruptured swim bladders were found anymore, only a few
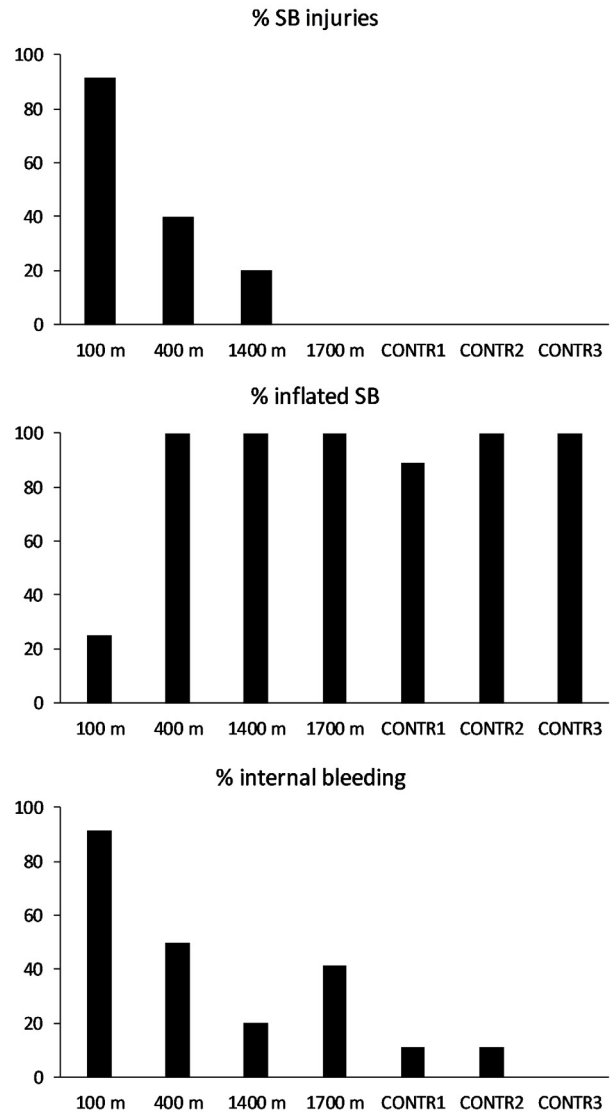
Figure 5. Percentage of swim bladders (SB) with barotrauma (upper graph), of inflated/deflated SBs (middle graph) and of fish with internal bleeding (lower graph) for each control and exposure cage.

internal bleeding and most fish showed normal swimming behaviour. Furthermore, this field experiment represents a "worst-case" scenario: fish were caged and had no chance to swim away if they would have wanted; and cod is a representative for fish with a closed swim bladder (*i.e.*, physoclist), which are most sensitive to swim bladder injuries (Halvorsen *et al.* 2012b).

Most cod survived on the short term, but since they all showed numerous multiple

instances of internal bleeding and a high degree of abnormal swimming behaviour, their survival chances on the longer term would probably be reduced. Most of these fish at 100 m had deflated swim bladders due to the large ruptures in the swim bladder, and although it is shown that these injuries might heal over time (Casper *et al.* 2013), the time needed for healing makes them more vulnerable to predation and other threats in the wild.

Although, we had the intention to measure pile driving sound simultaneously with cod exposure, we did not succeed due to weather and logistical issues. For similar exposure experiments in the future, we strongly recommend to use smart digital autonomous hydrophones which can be deployed together with the cages. This would increase the robustness of the results, and reduce the demanding logistical organization. Nevertheless, we were able to measure underwater sound during pile driving from two different monopiles and at three different distances. The sound metrics presented here are serving as proxies. However, Debusschere *et al.* (2014) has shown that sound metrics during pile driving do not differ a lot between different monopiles with similar characteristics, driven in the seafloor to a depth of 30-33 m and in a similar sandy environment. So, we trust our sound measurements to be valuable, and not to diverge too much from what the fish experienced during exposure.

The sound parameters measured at 400 and 500 m distance were very similar, with values a bit higher at 500 m, but this could probably be explained by the larger pile diameter (5 m compared to 4.5 m), since sound pressure level increases with increasing diameter (Nehls *et al.* 2007 and references therein). Additionally, at 500 m distance we also measured the last strokes of the pile driving event which contain more energy.

Pile driving sound showed a frequency peak between 125 and 200 Hz which is right in the middle of the hearing range of cod

that is between 30 and 470 Hz with greatest sensitivity in the range between 60 to 310 Hz (Chapman & Hawkins 1973). Since this was a short-term study, we chose to focus on swim bladder barotraumas, and did as such not look for injuries at the inner ear. Possibly, some of the abnormal swimming behaviour observed at the further distance cages could be related to potential inner ear damage. For future studies, it would be interesting to investigate whether potential inner ear injuries at further distances of the sound source influence fish behaviour on the short- and/or the long-term.

Halvorsen *et al.* (2012a) showed that the severity of injuries is not only owing to the total energy level of exposure (SEL_{cum}); the energy level of exposure of one single impulse (SEL_{ss}), and the number of impulses are as important. Therefore, it is important to include these parameters in measures to manage the activities generating impulsive sounds (Halvorsen *et al.* 2012a). Based on the results of our *in situ* exposure experiment, where we observed no ruptured swim bladders at 1700 m distance from the sound source for Atlantic cod, swim bladder barotrauma in Atlantic cod could be prevented at SEL_{ss} values of 165 dB re1 μPa^2.s and SEL_{cum} values of 200 dB re1 μPa^2.s or lower. Zero-to-peak levels (SPL_{z-p}) should not exceed 185 dB re1 μPa in order to prevent swim bladder barotrauma. The current "critical sound limit" implemented in Belgium in the setting of the Marine Strategy Framework Directive is 185 dB re1 μPa at 750 m (Rumes *et al.* 2015), no sound thresholds for SEL_{ss} or SEL_{cum} are in place at the moment. Our results indicate that with the current sound limits, swim bladder barotrauma can occur in physoclistous fish like Atlantic cod when they are within a radius of 750 m distance around the sound source during pile driving. This is, however, a small-scale effect, and it seems unlikely to cause significant effects at the population level. Nevertheless, in order to investigate what the observed effect means on a wider scale, the individual

impact can provide the basis for a population impact assessment. This is outside the scope of this manuscript, but it is important to consider when deciding on management or mitigation measures. The information gathered during our *in situ* exposure experiment contributes to the knowledge base on effects of impulsive sound, and can be used to scientifically evaluate and potentially modify existing sound limits.

5. Conclusion

This field experiment was a logistic and organizational challenge, and although the design could be criticized as no replicate cages were submerged at the different distances and sound measurements were not taken simultaneously with exposure of the fish, the obtained results are valuable because they increase the available knowledge of sound pressure effects on physoclistous fishes and help to evaluate current sound thresholds. To our knowledge, this is the first *in situ* experiment in which age I-group cod is exposed to pile driving in the field, and it scientifically underpins the anecdotic observation of whiting floating at the surface during pile driving, which was the immediate motivation of our experiment. This experiment proved that it should be repeated to answer further research questions relating inner ear injuries, long-term survival rate, etc.; this time, however, with small, autonomous digital hydrophones (*e.g.*, icListen HF-X2) that can be deployed together with the cages. Ideally, particle motion is also measured, since this is an important second component of sound, and its role in the effects of impulsive sound on fish needs further investigation.

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CHAPTER 4

EFFECT OF TURBINE PRESENCE AND TYPE ON MACROBENTHIC COMMUNITIES INSIDE AN OFFSHORE WINDFARM?

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Abstract

At present, three offshore wind farms are operational in the Belgian part of the North Sea (BPNS) and five more will be built in the near future to meet Belgium's 2020 targets for renewable energy. Introducing these artificial hard substrates in a soft sediment habitat (*i.e.*, reef effect) is believed to cause the largest impact on the marine environment and at different scale. Many studies already demonstrated the reef effects on macrobenthos in the immediate vicinity of wind turbines. In this report we studied whether there is an effect of turbine presence on macrobenthic community structure and if so, if this effect differs between different types of foundations. Samples were taken at two distances from the turbines: far (350-500 m) and close (50 m). Our results show that the installation of offshore wind turbines can induce changes in the macrobenthos. This is mainly seen at the Thornton Bank, where communities of the far sites differ significantly from the close sites, with a higher Shannon-Wiener diversity and evenness at the far sites (respectively H' close: 1.62 ± 0.14; far: 1.93 ± 0.06; and J' close: 0.72 ± 0.04; far: 0.81 ± 0.02). These community changes occurred independently of the abiotic environment (measured variables: grainsize [µm], total organic matter [%] TOM and 2 mm sediment fraction), for which no differences were detected with respect to turbine presence. However, on the Bligh Bank, a higher organic matter content was found further from the turbines, but this did not result in differences between the communities of the two distances to the turbine. No differences were observed for both the abiotic and the biotic variables between jacket and gravity based foundations. This might be due to an unrepresentative sample size at the gravity based foundations. Alternatively, the effect of turbine presence and foundation type might manifest itself within close vicinity of the turbines (< 50 m) and as such remain unconcealed by the current sampling design. To tackle this, it is recommended to perform a targeted monitoring study to investigate potential changes in sedimentology and organic enrichment in the close vicinity (7-100 m) of the three turbine types present in the BPNS.

1. Introduction

At present, nine Belgian projects representing a total capacity of 2.2 GW were granted both a domain concession and an environmental permit to meet Belgium's 2020 targets for renewable energy: three projects are operational, one is under construction and at least five will be constructed in the near future in the Belgian part of the North Sea (BPNS) (Degraer *et al.* 2016). Introducing these artificial hard substrates in a soft sediment habitat (*i.e.*, reef effect) is believed to cause the largest impact on the marine environment and at different scale (Petersen & Malm, 2006) due to, for example, changes in hydrodynamics and presence of epifaunal coverage along the turbine. Additionally, fisheries exclusion in windmill parks may alter the marine environment at different scales (De Mesel *et al.* 2013; 2015; Reubens *et al.* 2013; 2014). Only when a monitoring program is conducted to assess the effects of the installation of artificial hard substrates on the marine environment, an environmental permit is received by the project developer (Brabant *et al.* 2013). In Belgium, this monitoring program is coordinated by the Operational Directorate Natural Environment (OD Nature) of the Royal Belgian Institute of Natural Sciences and targets physical, biological and socio-economical aspects of the marine environment (Degraer *et al.* 2016). In this report, we focus on the possible effects on the macrobenthic community in offshore windfarms (OWF). Many studies have already demonstrated reef effects on macrobenthos in the immediate vicinity of wind turbines (Barros *et al.* 2001; Coates 2013; 2014a; 2014b).

Sediment type and food supply are two of the main natural factors that structure macrobenthic communities, next to temperature and the influence of different water masses (Pearson & Rosenberg 1978; Wilhelmsson & Malm 2008; Kröncke 2011; Kröncke *et al.* 2011). Coates *et al.* (2013; 2014a) revealed changes in sedimentology up to a distance of 50 m from the turbines: grain size reduced significantly due to a decreased current flow in the wake of the turbines (15-50 m "behind" the turbines in comparison with larger distances of 100-200 m). In addition, organic matter content increased close to the turbines primarily as a result of the deposited faeces, pseudo-faeces and dead individuals of epifauna on the foundations (Barros *et al.*, 2001; Maar *et al.*, 2009; Kerckhof *et al.*, 2010, De Mesel *et al.*, 2013). These changes can trigger changes in macrobenthic community structure (Coates *et al.*, 2011, 2013; Ysebaert *et al.*, 2009). Coates *et al.* (2014a) revealed an increased macrobenthos density along with an enhanced diversity close to the windmill. At 1 and 7 m distance from the foundation, the dominance of two hard-substrate amphipods, *Monocorophium acherusicum* and *Jassa herdmani*, highlighted the direct effect of the presence of the wind turbine. At distances of 15-50 m, shifts in species dominance were detected, with an increased dominance of the amphipod *Urothoe brevicornis* and the tube building polychaetes *Lanice conchilega* and *Spiophanes bombyx* close to the foundation (Coates *et al.* 2013). As many macrobenthos species are an important food source for organisms higher in the food web (Vandendriessche *et al.* 2015), changes in macrobenthic communities have the potential to alter food web energy flows (Dannheim *et al.* 2014). Hence, effects of windmills can also be found higher up in the food web, resulting, for example, in the attraction of pouting *Trisopterus luscus* and Atlantic cod *Gadus morhua* inside the OWF (Vandendriessche & Reubens *et al.* 2013; Reubens *et al.* 2013).

Reubens *et al.* (2016) also revealed changes in macrobenthos community structure in the offshore windfarms. Differences were observed between samples close to (50 m) and further away from the turbine (350-500 m). However, the results of Reubens *et al.* (2016) were not consistent with those of Coates *et al.* (2014a), who found higher densities and species numbers in 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 the far samples. As Reubens *et al.* (2016) did not observe differences in sedimentology between the close and far samples (in contrast with Coates *et al.* 2014a), it remains unclear which underlying ecological processes were responsible for the observed community changes. Reubens *et al.* (2016) suggested that this might be related to the turbine type used. Foundation types are mainly selected according to the environmental conditions (*e.g.*, water depth and sediment type) together with production and installation costs. With a water depth that ranges from 20-40 m at the BPNS, offshore windfarm (OWF) developers have hitherto used three different foundation types: gravity based, jacket and monopile foundations (fig. 1), each with different (pre-)construction-related activities such as dredging and pile driving (Coates 2014c). For a detailed description, see Coates (2014c) and Rumes *et al.* (2013).

The study of Reubens *et al.* (2016) was performed on a windmill farm dominated by jacket foundations, while the study of Coates *et al.* (2014a) focused on effects near a gravity based foundation. Jackets hav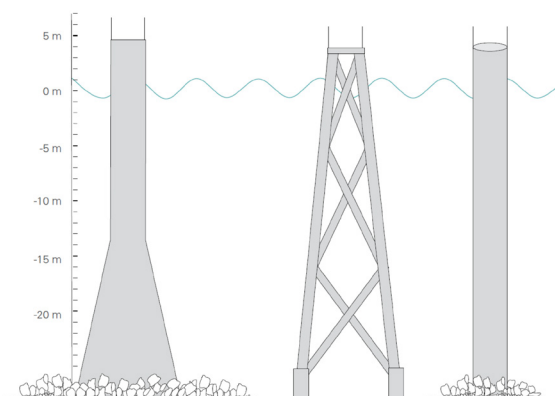e an open structure, allowing the main current flow to pass through. Gravity-based foundations, on the other hand, obstruct currents and areas with a lower current flow are generated in the wake of the turbine. These differences in flow velocity influence colonization potential of epifaunal species as well as sediment and TOM resuspension (Reubens *et al.* 2016).

In this report, we investigate whether there is an effect of turbine presence on macrobenthic community structure and if so, if this effect differs between different types of foundations.

2. Material and methods

2.1. Study area

Three projects are operational in the 238 km² area in the BPNS that was allocated to offshore renewable energy production (fig. 2). The current study was conducted in the concession area of two offshore wind farms: "C-Power", which is located on the Thornton bank (TB) sandbank, and Belwind, located at the Bligh Bank (BB). The C-Power 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). The 55 Belwind turbines are monopiles which were constructed in 2009-2010 and are operational since 2011.

2.2. Sample design, collection and treatment

A systematic stratified sampling design was adopted (fig. 3). Samples were collected in autumn 2016 at two distances, consistent with the sampling design of 2015 (Reubens *et al.* 2016), so a one-way spatial (close vs. far) comparison of samples can be conducted. Close samples were taken at approximately 50 m from the turbines on the South-West side. This is the smallest distance which is easily reached by a small vessel. If sampling in the South-West direction was not possible (to comply with a minimum distance of 50 m

Figure 1. Three foundation types present in the Belgian part of the North Sea, from left to right: gravity based, jacket and monopile foundation (Rumes *et al.* 2013).
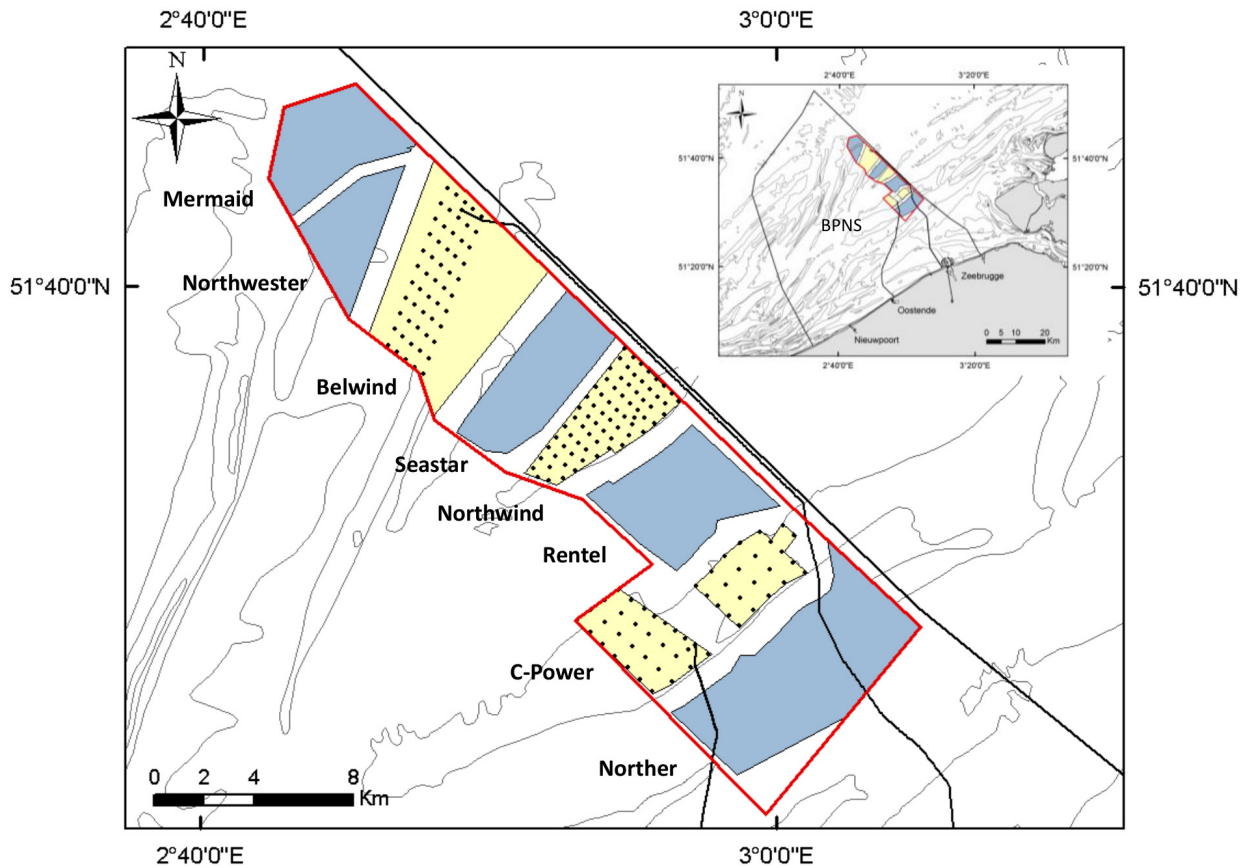
Figure 2. Wind farm concession area (red area) in the Belgian part of the North Sea. Three offshore wind farms have been constructed (yellow areas) on the Thorntonbank (C-Power), Bligh Bank (Belwind phase I) and Lodewijkbank (Northwind). Two power cables from C-Power and Belwind run to the port of Ostend and Zeebrugge, respectively (black lines). Five additional domain concessions have been granted to Norther, Rentel, Seastar, Northwester and Mermaid (blue areas). Wind turbines are marked as black dots (Coates 2014c).

from infield electricity cables), samples were taken at the North-East side of the turbine. The far samples were gathered in the middle between the four surrounding wind turbines (*i.e.*, the farthest possible distance). These distances ranged between 350 and 500 m from the turbines (fig. 3). Samples were collected on board the *RV* Simon Stevin and Aquatrot on 24, 25 and 29 October 2016. Table 1 shows when the different stations were sampled with which vessel and the number of samples.

Samples were obtained by means of a 0.1 m² Van Veen grab, sieved alive onboard over a 1 mm mesh-sized sieve and preserved in a 4% formaldehyde-seawater solution. In the laboratory, samples were stained with rose Bengal. After rinsing over a 1 mm sieve and sorting, organisms were identified to species level whenever possible. Some organisms were identified at a higher taxa level because of the difficulty of identification or small size. Individuals were counted and biomass (blotted wet weight, mg) was determined for every species per sample.

Table 1. Overview of the number of samples taken at each location and sampling date

Date	Vessel	Station	# Samples
24/10/2016	Simon Stevin	TB_far	32
25/10/2016	Simon Stevin	BB_far	24
25/10/2016	Aquatrot	BB_close	15
25 and 29/10/2016	Aquatrot	TB_close	16
29/10/2016	Aquatrot	GB	2
25/10/2016	Simon Stevin	GB	14

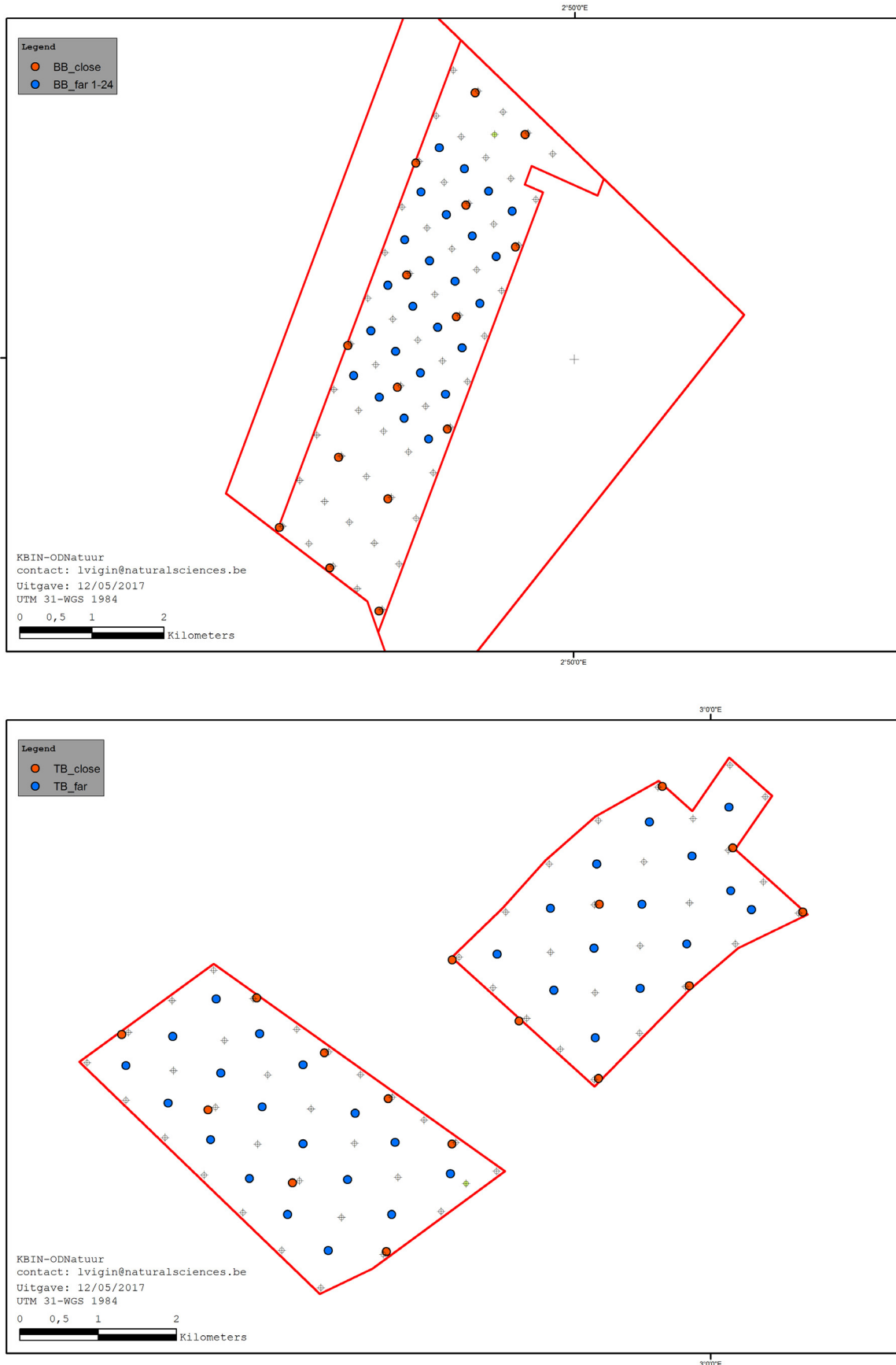**Figure 3.** Overview of close and far samples at the Bligh Bank (up) and Thornton Bank (low). Black dots represent foundations, red and blue dots are sampling positions.

Environmental data such as grain size distribution (GS) and total organic matter content (TOM) were sampled parallel with the macrobenthos samples by means of a core (Ø 3.6 cm) taken from the Van Veen grab samples. After drying at 60 °C the grain size distribution was measured using laser diffraction on a Malvern Mastersizer 2000G, hydro version 5.40. Grain size fractions are given as volume percentages w ith a range from fine clay (max. 4 μm) to coarse gravel/shell material (max. 2 mm). Sediment fractions larger than 2 mm were quantified using a 2-mm sieve. The total organic matter (TOM) content was determined per sample from the difference between the dry weight (48 h at 60 °C) and the ash-free dry weight (2 h at 500 °C).

2.3. Data analysis

The three close samples at the Thornton Bank that were taken at gravity based foundations were removed from the analyses to test the effect of distance from the turbine, so that only samples at jacket foundations were included for the Thornton Bank. Also, two samples at the Thornton Bank were removed as they proved to be outliers: TB6_far with a much higher species number (36) and abundance (5070 ind. m⁻²), and TB6_close with a very low species number (3) and abundance (30 ind. m⁻²). Rare species were not removed from the dataset, as the presence of these species might be a first indication of changes in the macrobenthic community (not evaluated in this report). The total abundance (ind. m⁻²), biomass (mg WW m⁻²), number of species (S), Shannon-Wiener diversity index (H') and Pielou's evenness (J') were calculated. One-way Anova (1 factor: position; two levels: close vs far) was performed to statistically investigate differences between the distances. Levene's test was used to verify homogeneity of variances, while the Shapiro-Wilk test was used to check for normality. In case assumptions were not met, data were (double) logarithmic transformed. If after transformation the assumptions

were still not fulfilled, an assumption-free PERMANOVA (Permutational Analysis of Variance [Anderson *et al.* 2008] with the same design [1 factor: "position"]) was used, based on a Euclidean distance matrix.

Permutational Anova (PERMANOVA) with a fixed one-factor design (position) was also 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). As the design was unbalanced, it was decided to use Type III sums of squares. The number of permutations was set to 9999 and unrestricted permutation of raw data was performed as there was only one factor in the design. The multivariate analysis was based on a Bray-Curtis resemblance matrix and performed on fourth-root transformed abundance data. Homogeneity of multivariate dispersions was tested using the PERMDISP routine, using distances among centroids. Principal Coordinates Analysis (PCO) was run to visualize the data. Vector overlay was based on multiple correlations and only species with Spearman correlation R > 0.6 are shown. In addition, a similarity percentages (SIMPER) routine analysis was done to specify the contributions of individual species to the distinction between groups of samples and/or to the similarity of samples within a group (Clarke & Gorley 2006).

Furthermore, a distance-based linear model (DistLM) based on Adjusted R² 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).

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 & Gorley 2006; Anderson *et al.* 2008) and in R (version 3.2.2) (Team 2015). A type I error significance

level of $p \leq 0.05$ was used in all tests. Results are expressed as means ± 1 standard error.

3. Results

The Thornton Bank (TB) and Bligh Bank (BB) contained a similar amount of TOM and had a comparable > 2 mm sediment fraction (1-way ANOVA, $p > 0.05$), but median grain size was significantly larger at the BB (1-way ANOVA, $p = 0.001$). Macrobenthic communities of both sandbanks differed strongly (1-way PERMANOVA $p = 0.0001$) (fig. 4), mainly due to higher macrobenthic densities at the TB than at the BB (1-way ANOVA, $p = 0.007$). For this reason, and to facilitate comparison with the results of Reubens *et al.* (2016), macrobenthic communities and the environment of both sandbanks were analyzed separately.
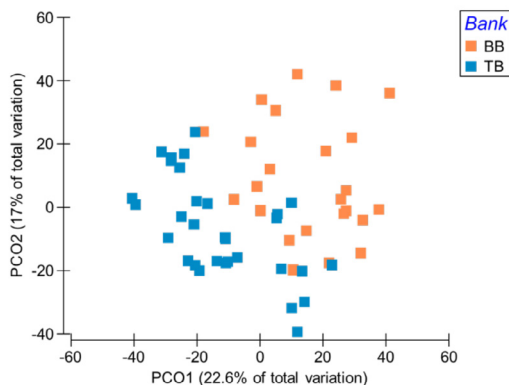

Figure 4. PCO (Principal Coordinates analysis) plot based on Bray-Curtis resemblance matrix of the fourth root transformed macrobenthic densities at the far sites from the two sandbanks (TB = Thornton Bank, BB = Bligh Bank).

3.1. Effect of distance from turbines

Almost all sediments consisted of coarse sands (median grain size between 300 and 500 μm) for both sandbanks (with the exception of 1 sample at TB_far [298 μm] and one at TB_close [649 μm]). TOM content remained low in all samples, around 0.5%, with slightly lower values at BB_close (0.37 ± 0.03%). The sediment fraction

> 2 mm at the Thornton Bank ranged from 0.12 to 10.54% and at the Bligh Bank from 0.12 to 18.31% for the far samples (fig. 5 and table 2). A univariate analysis on the abiotic data revealed that there were no significant differences in grain size and in the 2-mm fraction between the samples close to and far from the turbines at both sandbanks. Only the far samples at the Bligh Bank had a higher organic matter content than the close samples (1-way ANOVA, $p = 0.020$). This pattern was not observed at the Thornton Bank (table 3).

At the Thornton Bank, far samples displayed higher macrobenthos biomass, species richness, Shannon-Wiener diversity and evenness but somewhat lower densities than the close samples (fig. 6 and table 2). Except for Shannon-Wiener diversity and evenness, these differences were not significant (table 3). At the Bligh Bank, results were less consistent. At the far samples, there was a tendency for a higher biomass and evenness and a lower number of species, abundance and Shannon-Wiener diversity (fig. 6 and table 2). None of these differences were, however, significant (table 3). As *E. cordatum* influenced biomass substantially, this species was removed from the analysis, but even then no significant differences in biomass were observed.

The multivariate analysis on the macrobenthic community structure at both sandbanks showed significant differences between the far and close samples at the Thornton bank (PERMANOVA, $p = 0.011$) but not at the Bligh Bank (PERMANOVA, $p = 0.167$) (fig. 7). Permdisps were not significant (TB: $p = 0.114 -$ BB: $p = 0.349$), hence the significant differences between the two distances were not the result of a dispersion effect.

For the Thornton Bank, the dissimilarity between close and far sites was 54.41%. *Urothoe brevicornis* (13.91%), *Spiophanes bombyx* (7.35%) and *Bathyporeia elegans* (5.71%) together
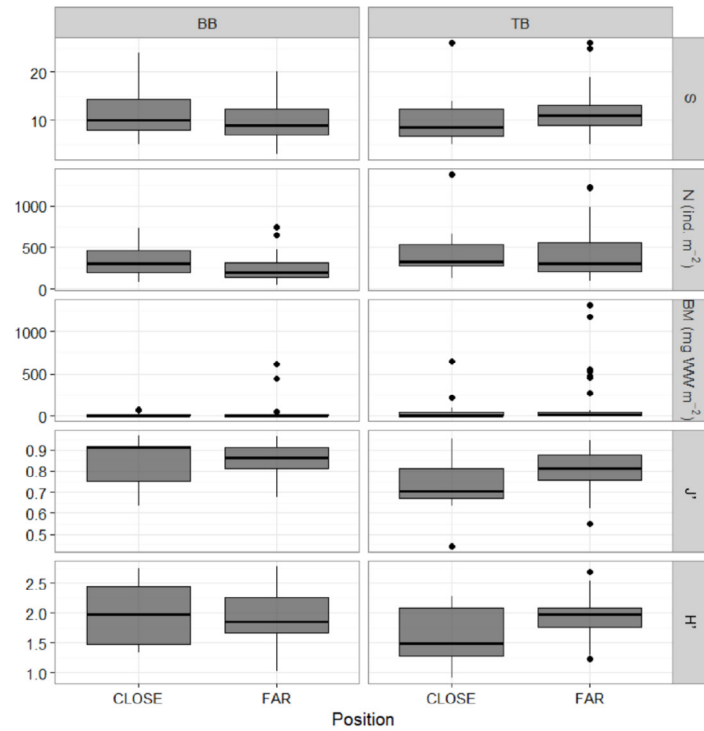
Figure 5. Box plots of the median grain size (GS [μm]), total organic matter (TOM [%]) and sediment fraction above 2 mm (> 2 mm %) per sampling site for the close and far samples (right: Thornton Bank, left: Bligh Bank). Black dots represent outliers.
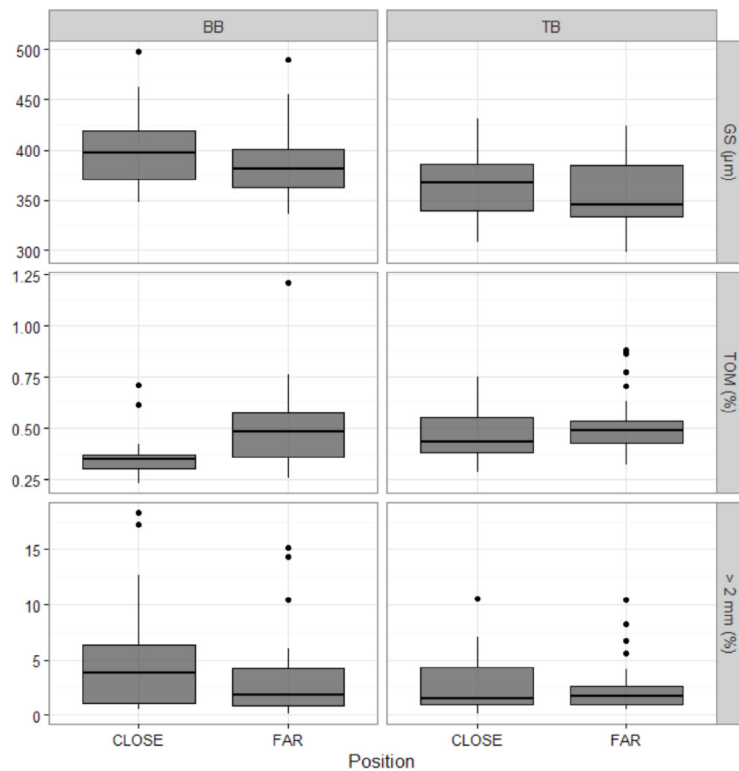
Figure 6. Box plots of the number of species (S), abundance (N), biomass (BM), evenness (J') and Shannon-Wiener diversity (H') per sampling site (right: Thornton Bank, left: Bligh Bank). Black dots represent outliers.

Table 2. Overview of number of stations and calculated community descriptors (mean ± SE) for the two distances (close-far) sampled at the Thornton Bank (TB – C-Power) and Bligh Bank (BB – Belwind) in 2016

	TB close	TB far	BB close	BB far
# Samples	12	31	15	24
Number of Species - S	10.25 ± 1.66	11.74 ± 0.91	11.60 ± 1.41	10.17 ± 0.94
Species abundance (ind. m^{-2}) - N	447.50 ± 96.05	440.65 ± 59.43	338.67 ± 50.09	260.42 ± 36.29
Biomass (mg WW m^{-2}) - BM	83.42 ± 53.80	162.88 ± 63.20	14.67 ± 5.95	53.18 ± 30.42
Evenness - J'	0.72 ± 0.04	0.81 ± 0.02	0.84 ± 0.03	0.86 ± 0.02
Shannon-Wiener - H'	1.62 ± 0.14	1.93 ± 0.06	1.98 ± 0.13	1.90 ± 0.09
Median grain size (μm) - GS	366.73 ± 11.39	355.48 ± 6.46	400.76 ± 10.66	387.96 ± 7.62
Total organic matter (%) - TOM	0.48 ± 0.04	0.51 ± 0.03	0.37 ± 0.03	0.50 ± 0.04
Sediment fraction > 2 mm (%) - > 2 mm	2.89 ± 0.94	2.48 ± 0.44	5.4 ± 1.54	3.41 ± 0.86

Table 3. Level of significance for all tests on the biotic and abiotic variables of the far versus the close samples at the two sandbanks (TB = Thornton Bank, BB = Bligh Bank)

BANK	GS	TOM	> 2 mm	S	N	BM	H'	J'
TB	0.359*	0.446*	0.662	0.200*	0.812*	0.486	0.020*	0.028*
BB	0.324*	0.020*	0.26	0.385*	0.186*	0.647*	0.603*	0.886*

* indicates that the analysis was performed with a one-way ANOVA, else PERMANOVA was used. Significant p-values are highlighted in red. GS = grain size, TOM = total organic matter content, > 2 mm is the sediment fraction larger than 2 mm, S = species richness, N = abundance, BM = biomass, H' = Shannon-Wiener diversity, J' = Pielou's evenness.
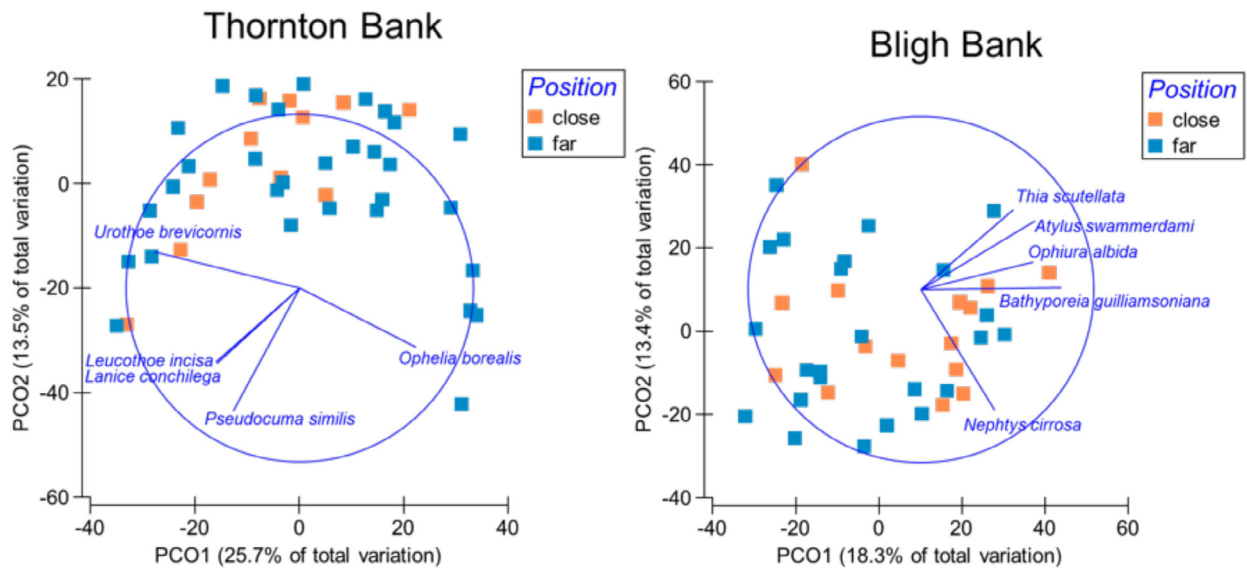

Figure 7. PCO (Principal Coordinates analysis) plots based on Bray-Curtis resemblance matrix of the fourth root transformed macrobenthic densities at the two sandbanks and at two distances from the wind turbines. Vector overlay is based on multiple correlations and only species with correlation > 0.6 are shown.

Table 4. Species that contributed to the difference in community composition between the close and far samples up to a cumulative value of ≥ 50%

	Group close	Group far		
Species	Average abundance	Average abundance	Contribution %	Cumulative contribution %
Urothoe brevicornis	13.15	8.73	13.91	13.91
Spiophanes bombyx	4.04	4.99	7.35	21.26
Bathyporeia elegans	2.54	3.95	5.71	26.97
Nemertea sp.	1.94	2.26	4.36	31.33
Nephtys juv.	3.59	5.02	4.32	35.65
Gastrosaccus spinifer	2.89	3.3	4.02	39.68
Nephtys cirrosa	7.29	8.23	3.96	43.64
Glycera sp.	0.79	1.70	3.59	47.23
Spio sp.	1.32	2.04	3.21	50.44

contributed more than 25% of this dissimilarity. *Urothoe brevicornis* was more abundant in the close samples, while *S. bombyx* and *B. elegans* were more abundant in the far samples. Many other species contributed to a lesser extent (table 4).

Figure 8. dbRDA plots based on Bray-Curtis resemblance matrix of the fourth root transformed macrobenthic densities at the two sandbanks and at two distances from the wind turbines.

A DistLM was carried out to investigate the relationship between the macrobenthic community and the environmental variables (fig. 8). The DistLM revealed that at the TB all three abiotic variables (grain size, total organic matter content and sediment fraction > 2 mm) have a significant relationship with the multivariate data, but together explained only 14.00% of the variation. At the BB, we also see a significant contribution of these three variables, but only 10.48% of the variation was explained.

3.2. Effect of foundation type

To reveal a possible foundation effect we studied the close samples of the Thornton Bank only, to exclude the "Bank" effect (the fact that the communities on both sandbanks are different). Within the Thornton Bank only three samples were taken at a gravity-based foundation and 13 at a jacket foundation. The PCO plot showed a large variation between the samples at the jacket foundations, which made it impossible to randomly select three samples for a balanced comparison (fig. 11). No significant differences were found between the two types, both for the abiotic and biotic variables (table 6). Additionally, no significant differences between the communities at jacket foundations and gravity-based foundations were observed (PERMANOVA, p = 0.810). These results suggest no foundation effect (gravity

Figure 9. Box plots of the median grain size (GS [μm]), total organic matter (TOM [%]) and sediment fraction larger than 2 mm (> 2 mm %) at two types of foundation at the Thornton Bank. Black dots represent outliers.
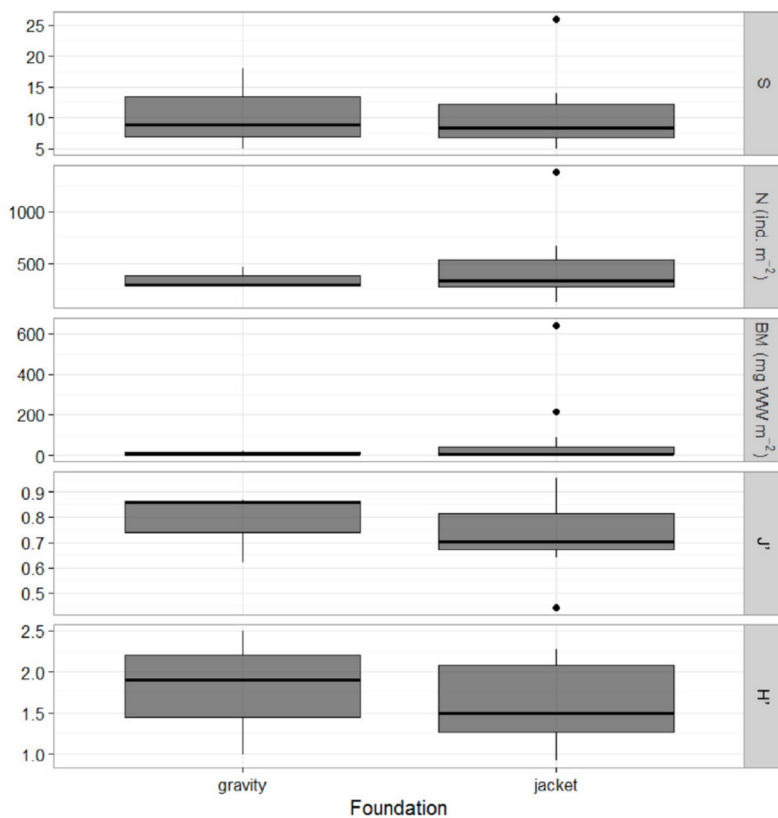
Figure 10. Box plots the number of species (S), abundance (N), biomass (BM), evenness (J') and Shannon-Wiener diversity (H') at two types of foundation at the Thornton Bank. Black dots represent the outliers.

Table 5. Overview of number of stations and calculated community descriptors (mean ± SE) at two types of foundation (gravity based and jacket) sampled at the Thornton Bank (TB - C-Power) in 2016

	TB gravity based	TB jacket
# Samples	3	12
Number of Species - S	10.67 ± 3.84	10.25 ± 1.66
Species abundance (ind. m^{-2}) - N	343.33 ± 63.60	447.50 ± 96.05
Biomass (mg WW m^{-2}) - BM	9.84 ± 5.72	83.42 ± 53.80
Evenness - J'	0.78 ± 0.08	0.72 ± 0.04
Shannon-Wiener - H'	1.80 ± 0.43	1.62 ± 0.14
Median grain size (µm) - GS	343.15 ± 14.18	366.73 ± 11.39
Total organic matter (%) - TOM	0.43 ± 0.08	0.48 ± 0.04
Sediment fraction > 2 mm (%) - > 2 mm	4.84 ± 3.81	2.89 ± 0.94

Table 6. Level of significance for all tests on the biotic and abiotic variables at the Thornton Bank, comparing the two types of foundation

	GS	TOM	> 2 mm	S	N	BM	H'	J'
TB gravityjacket	0.348	0.604*	0.492	0.964*	0.771*	0.543	0.600*	0.509*

* indicates that the analysis was performed with a one-way ANOVA, else PERMANOVA was used.
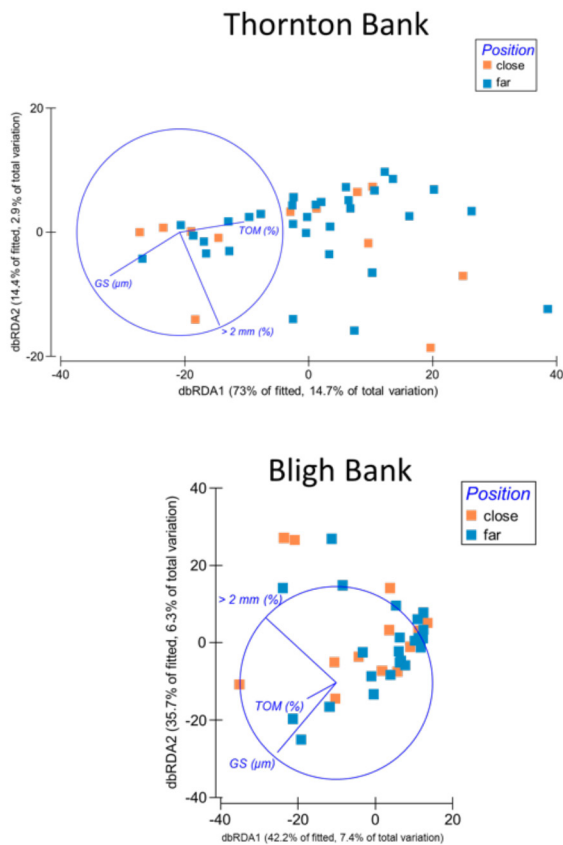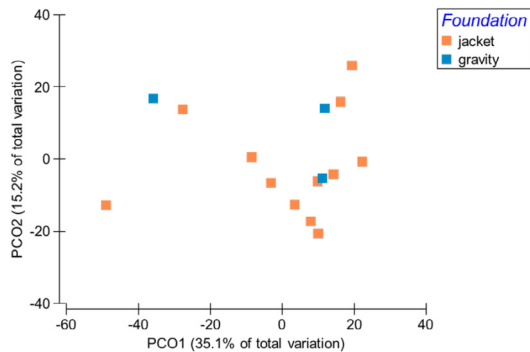

Figure 11. PCO (Principal Coordinates analysis) plots based on Bray-Curtis resemblance matrix of the fourth root transformed macrobenthic densities at the Thornton Bank for two foundation types.

vs. jacket) on the macrobenthic community on the Thornton Bank.

4. Discussion

In the BPNS, four subtidal macrobenthic communities have been distinguished, connected by transitional species assemblages (Degraer *et al.* 2003; 2008; Van Hoey *et al.* 2004). Situated in the eastern, offshore part of the BPNS, the macrofaunal communities in the OWF concession area are highly heterogeneous but primarily characterized by the *Nephtys cirrosa* and *Ophelia borealis-Glycera lapidum* communities (De Maersschalck *et al.* 2006). These communities are generally characterized by a low average species richness (5-7 species) and abundance (190-402 ind. m^{-2}), inhabiting medium sands with low organic matter content. The Thornton Bank was originally inhabited by these communities, but after windmill construction, a higher average species richness (10-30 species) and abundance (1390-18583 ind./m²) was observed, coinciding with a shift in dominant species (Coates *et al.* 2014a). As such, the community has evolved away from the *N. cirrosa* and *O. limacina-G. lapidum* communities. With an increased macrofauna abundance and a decreasing sediment grain size, it was suggested that the macrobenthic community is shifting towards a variation of the species-rich *Abra alba-Kurtiella bidentata* community (30 species and 6432 ind. m^{-2}), which is usually found in shallow and muddy sands (Van Hoey *et al.* 2004). It should be noted that these changes

were mainly observed in close vicinity of the windmills (< 50 m). Van Veen grab sampling is currently the best available method to sample/characterize macrobenthic communities. However, the sampling efficiency in communities poor in abundance and richness is rather low. This low sampling efficiency should be taken into consideration when interpreting the results. Our results also show a significantly lower median grain size at the Thornton Bank (298-423 μm) than at the Bligh Bank (336-490 μm) and a significantly higher macrofauna abundance (TB: 100 to 1220 ind. m^{-2}; BB: 50 to 750 ind. m^{-2}). The maximum abundance at the TB is lower in comparison with Coates *et al.* (2014a) due to the fact that these high values were reached at 25 m and 15 m from the turbine, while in this study only samples at 50 m were taken, missing out this increase. Although the characteristics of the observed community at the Thornton Bank are not within the range of the characteristics of the *Abra alba-Kurtiella bidentata* community, we do see a significantly different community than at the Bligh Bank (fig. 4). For this reason, and for easier comparison with the results of Reubens *et al.* (2016), both sandbanks were analysed separately.

4.1. Effect of distance from turbines

The effect of distance from the turbine foundation was not unambiguous for both sandbanks.

The measured environmental conditions (GS, > 2 mm fraction and TOM content) on the Thornton Bank were similar close to (*i.e.*, ca 50 m) and far from (*i.e.*, 350-500 m) the turbines. Despite the similarity in habitat type, the communities close to the turbines differed significantly from those further away from the turbines. These differences were observed in community structure, with a higher evenness and Shannon-Wiener diversity far from the turbines. More specifically, the communities in the close samples were characterized by higher

abundance of *Urothoe brevicornis*, whereas *Spiophanes bombyx* and *Bathyporeia elegans* were more dominant in the samples far from the turbines. To a certain extent, these results corroborate the study of Reubens *et al.* (2016), who also found differences in communities between areas near and far from the turbines. However, these differences were mainly present in species abundances and species richness, rather than in evenness and Shannon-Wiener diversity. On the other hand, the typifying species for the sampling sites close to and far from the turbines remained the same.

At the Bligh Bank, more TOM accumulated further away from the turbines than close to the turbines, but since this difference in TOM concentration was not observed in 2015 (Coomans 2017), this might also represent a temporary variation. Despite the potential difference in resource availability linked to different TOM concentrations, no significant differences were observed between the macrobenthic communities from the two distances. This agrees with the results of another study focusing on a wind farm with monopiles in Denmark, where no differences were found in benthic communities between sites at different distances (Leonhard & Pedersen 2005).

Sediment type and food supply are two of the main natural factors that structure macrobenthic communities. Grain size distribution can change in the immediate vicinity of an offshore wind turbine, inducing an important impact on the associated soft-sediment macrofauna, up to 50 m distance from the turbines (Leonhard & Pedersen 2005; Coates *et al.* 2014a). A significant refinement of the grain size close to (15-50 m) a gravity based turbine on the Thornton Bank (Coates *et al.* 2014a) and a tendency to finer sand close to monopiles in a Danish OWF (5-25 m) (Leonhard & Pedersen 2005) have been observed. In line with the study of Reubens *et al.* (2016), we did not observe such a refinement close to (50 m) the

turbines. This suggests that such refinement effects remain highly local in the immediate proximity of turbines, and do not extend beyond a maximum of a few tens of meters, 50 m being the limit of detection for changes in sediment granulometry.

It is generally accepted that the hard-substrate epifauna growing on foundations contribute to the organic matter input on the seabed by sedimentation of faeces and detritus (Barros *et al.* 2001; Maar *et al.* 2009; Kerckhof *et al.* 2010, De Mesel *et al.* 2013). Therefore, total organic matter content can be higher close to the turbines (Coates *et al.* 2014a). However, the sediment TOM content in this study was similar in samples close to and far from turbines on the TB and even lower in the samples closer to the turbines on the BB. Epifaunal communities appear to differ in composition between the monopiles of the BB and the gravity and jacket based foundations of the TB (De Mesel *et al.* 2013; pers. comm. Jan Reubens). For example, a 1-m mussel zone (*Mytilus edulis*) has developed on the concrete gravity based foundations of the TB, while this zone is only 0.5 m on the steel surface of the monopiles of the BB (De Mesel *et al.* 2013), and the jacket foundations of the TB are fully covered with mussels (Krone *et al.* 2013; pers. comm. Jan Reubens). In addition, the epifouling communities on the turbines may be in a different phase of succession as the monopiles from the BB are operational since 2011 and the jackets of the TB since 2013 (Degraer *et al.* 2016). A stable epifaunal community is generally reached after 5-6 years (or longer in case of storms and hard winters) (Leonhard & Pedersen 2005). Consequently, the macrobenthic communities thriving at the base of the foundations might also receive different quality and quantities of organic matter. Nevertheless, no increase in quantity of organic matter was observed in this study at the Thornton Bank, nor in Reubens *et al.* (2016), whereas the higher organic matter content far from the Bligh Bank turbines in

this study did not result in altered macrobenthic communities.

Although sediment characteristics are known to be an important factor structuring the macrobenthic community (Kröncke 2011; Kröncke *et al.* 2011), in this study, only a low proportion of the variation observed in the macrobenthic community structure was explained by the environmental variables (grain size, total organic matter and sediment fraction above 2 mm), and this for both sandbanks (TB: 14% and BB: 10%). This suggests that some other (abiotic and/or biotic) variables, which are key to explaining community differences, are missing in the current monitoring, of which a low sampling efficiency with Van Veen grab is one factor.

Other such factors potentially affecting macrobenthic communities are temperature and the influence of different water masses (Pearson & Rosenberg 1978; Wilhelmsson & Malm 2008; Kröncke 2011; Kröncke *et al.* 2011), as well as anthropogenic stressors such as fishing, dredging and eutrophication (Kröncke *et al.* 2011). The effect of temperature can be ruled out, since water masses at the BPNS are well mixed (MUMM 1996) and the studied areas experience similar water temperatures and eutrophication influence. Also, the effect of fisheries and dredging are trivial, since these activities are not permitted in the OWF. The studied banks are, however, influenced by different water masses: the Thornton Bank is situated on the edge between the clear water of the English Channel and the more turbid coastal water (Lacroix *et al.* 2004). The Bligh Bank, situated 40 km offshore, is influenced exclusively by English Channel water masses, which is reflected in a difference in organic matter content of the overlying waters. However, study of the water column was not included in this study. Therefore, we can only relate the observed differences in effects of distance to the turbine to natural spatial variability (Ysebaert & Herman 2003) or to the effect of foundation type.

The natural spatial variability in macrobenthic communities on the Thornton Bank and the Bligh Bank did not allow us to specifically test for the effect of foundation type across both sandbanks (jacket vs. monopile foundations), but the difference in effect of foundation presence on the sediment characteristics and on macrobenthic communities (BB: higher TOM levels far from foundation, but no community differences; TB: no TOM differences but different communities at the two distances) hints that there might be an effect of foundation type.

4.2. Effect of foundation type

Both the current study and the one of Reubens *et al.* (2016) contradict the observations of finer sediment and concomitantly of different communities closer to the turbines in Coates *et al.* (2014). The underlying reason is primarily the difference in scale (distance to the turbine: < 50 m in Coates *et al.* 2014 vs. < 250 m in this study and Reubens *et al.* 2016). However, another reason to consider might be the difference in turbine foundation type. We therefore investigated the sediment characteristics and macrobenthic community structure around two types of foundations at the Thornton Bank: gravity based and jacket foundation. We specifically focused on the Thornton Bank, to exclude the bank (location) effect (the fact that the original communities from the TB and BB are different). Twelve samples were taken at jacket foundations and only three samples at a gravity based turbine. The new sampling design (since 2015) was focused on a stratified random sampling in order to take samples close and far from the turbines, without taking into account the different turbine types, so only three of the six gravity based foundations were sampled. Because of this, no hard conclusions can be made, but our results do give an indication of the effect of foundation type. Gravity based foundations are concrete cylindrical/conical structures. They are support structures held in place by their own gravity (www.C-Power.be).

The large concrete base profoundly affects local current flow (Leonhard & Pedersen 2005). Decreased current flow in the wake of the turbine prevents the resuspension of finer sands and enriched TOM close to the turbines (Reubens *et al.* 2016). On the other hand, jacket foundations are steel structures with four legs connected to each other with braces (www.C-Power.be). They are open structures allowing the main current flow to pass through the construction (Lancelot *et al.* 1987).

At locations with reduced currents (such as in the wake of gravity based turbines), the organic material can accumulate (Reubens *et al.* 2016). This is not seen in our results as the TOM values at the gravity based turbines (0.43%) were no different from those at the jacket foundations (0.48%). Clear results may not be apparent due to the fact that only three samples were taken at gravity based foundations; still, also at the monopiles of the Bligh Bank (which are similar to the gravity based foundation), the TOM values were even lower (0.37%) than at the jacket foundations of the TB.

However, we did not observe differences in sediment characteristics, nor in macrobenthic communities between gravity based and jacket foundations. Again, if any, the effect of turbine foundation type on benthic communities may be manifested in the immediate vicinity (< 50 m) of the turbine. This is confirmed by Coates *et al.* (2014a) where an increase in total abundance, species richness and biomass was observed in samples at 50 m and even more so, closer to a gravity based turbine (on the South West side). As such, the data from the samples taken at 50 m do not provide any conclusive result.

5. Conclusion and recommendations

It can be concluded that the installation of offshore wind turbines can induce changes in the macrobenthos. This is mainly seen at the Thornton Bank, where communities of

the far sites differed significantly from the close sites, with a higher diversity at the far sites. These community changes occurred independently from the measured environmental variables (GS, TOM and 2 mm fraction), which remained unchanged with respect to turbine presence. However, on the Bligh Bank, a higher organic matter content was found further from the turbines, but this did not result in differences between the communities of the two distances to the turbine. No differences were observed for both the abiotic and the biotic variables between jacket and gravity based foundations. This may be due to a small sample size at the gravity based foundations. Alternatively, the effect of turbine presence and foundation type might manifest itself only within close vicinity of the turbines (< 50 m) and as such remain unconcealed by the current sampling design.

To enable long term studies, it is recommended to continue monitoring macrobenthic communities and their environment following the current sampling design (but with a higher number of samples at the gravity based foundations [6]). In addition, it would be highly interesting to perform a targeted monitoring study to investigate potential changes in sedimentology and organic enrichment in the close vicinity (7-100 m) of the three turbine types present in the BPNS, as different physical and biotic interactions can occur depending on turbine type, and since discrepancies between our results and those of other studies (Leonhard & Pedersen 2005; Coates *et al.* 2014) may well relate to differences in proximity of sample collection to turbines.

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CHAPTER 5

EFFECTS OF BELGIAN OFFSHORE WINDFARMS ON SOFT SEDIMENT EPIBENTHOS AND FISH: AN UPDATED TIME SERIES

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Abstract

Since 2005, ILVO has been performing beam trawl monitoring aimed at evaluating the potential effects of wind farms on the soft sediment epibenthos and fish in between turbines. The study effort is concentrated on the Thornton and Bligh Bank offshore wind farms (OWFs). In this chapter, an update on the time series collected between 2005 and 2016 is presented.

The main conclusions are: 1) soft sediment epibenthos and fish assemblages in between the turbines (at distance > 200 m) have not really changed 6 years after the construction of the wind turbines; 2) species assemblages within the OWFs seem to be mainly structured by temporal variability at larger spatial scales; 3) the post-construction "overshoot" of epibenthos density and biomass caused by an increase in opportunistic, scavenging species, was a temporary phenomenon lasting only 2 years post-construction; 4) no effect of fisheries exclusion is yet observed in soft sediment epibenthos and fish between the turbines; 5) monitoring effort should be increased with a higher number of replicate samples per survey to increase the statistical power of the analyses.

1. Introduction

Construction of offshore wind farms (OWFs) introduces artificial hard substrates into the typical soft bottom sandy environment in the Belgian part of the North Sea (BPNS). These hard substrates generate a new "rocky" habitat which attracts hard substrate species (Lindeboom *et al.* 2011; Kerkhof *et al.* 2012; De Mesel *et al.* 2015), and creates a reef effect for epibenthic fauna and demersal and benthopelagic fish (Reubens *et al.* 2011; 2013; Stenberg *et al.* 2015). This reef effect, in combination with fisheries exclusion in the wind farm area, may affect the original soft bottom epibenthos and fish assemblages between the wind turbines.

Currently, three OWFs are operational in the BPNS. In 2008, C-Power installed the first six gravity-based wind turbines (30 MW) at the Thornton Bank, followed by the construction of 48 more jacket foundation turbines (295 MW) in 2011, becoming operational in 2013. In 2009-2010, Belwind constructed 55 monopile turbines (165 MW) at the Bligh Bank, and additionally 50 monopile turbines (165 MW) in 2016-2017 in the adjacent Nobelwind concession zone. In between these two OWFs, Northwind NV built 72 more monopile turbines in 2013 at the Lodewijckbank.

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2. Material and methods

2.1. Sampling

Since the previous report of Derweduwen *et al.* (2016a), one extra sampling campaign was performed in autumn 2016. Due to recurring technical problems with RV Belgica (which gave us less ship time) in combination with construction works at Nobelwind, adjacent to the Belwind concession area, we could only focus on the wind farm effects and not on the fringe effects. In autumn 2016, trawl samples were taken in between the wind farms (4 within C-Power and 2 within Belwind) and at several reference locations away of the concessions (fig. 1). On these track locations, fish fauna and epibenthos were sampled with an 8-meter shrimp beam trawl (22 mm mesh in the cod end) equipped with a bolder-chain. The net was towed for 15 minutes at an average speed of 4 knots. 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 tracks 2 and 3 within the C-power concession which are perpendicular to the coastline due to the positioning of the in-field electricity cables. Epibenthos and fish were identified, counted, measured (all fish, crabs and shrimps) and wet weighted (all epibenthos) onboard. The samples that could not be fully processed onboard were frozen and further processed in the lab.

2.2. Data used and statistical analyses

The time series of trawl samples in both C-Power and Belwind dates back to respectively 2005 and 2008. However, within the sampling period 2005-2016, the sampling design had to be adapted based on previous monitoring results, wind farm accessibility, weather conditions, and research vessel availability. An overview on sampled stations in autumn during the entire time period is given in table 1 and 2. For an overview map of all track locations, the reader is referred to Vandendriessche *et al.* (2015).

For this chapter, we tested wind farm effects for two ecosystem components (epibenthos and demersal-benthopelagic fish) in a two-factorial PERMANOVA design with factors "year" and "impact" for univariate parameters (species number, density, biomass), and with factors "phase" and "impact" for community structure (with "phase" we mean the state of the concession, being either T0, construction, right after construction, and operational phase). This was done for both the C-Power and Belwind concession separately. The primary aim was to analyse interaction effects between "year" and "impact" or "phase" and "impact", since these would reveal whether the changes that occurred could be attributed to the construction of the OWF. When a significant effect for the "impact x year" or "impact x phase" interaction term was found, pairwise tests were conducted to test for differences between impact and reference samples within each year or each phase. P values for pairwise test were, due to the restricted number of possible permutations, drawn from Monte Carlo (MC) permutations (Anderson & Robinson 2003).

SIMPER analyses were done to find the species responsible for the observed changes. All samples used for the tests in this chapter are highlighted in table 1 and 2. Tests were done on density, biomass (the latter only for epibenthos), species richness and

Figure 1. Overview map showing the 2016 trawl locations at the C-Power and Belwind concession area and the respective reference locations.

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Table 1. Overview table of autumn sampled locations for effects of the C-Power wind farm within the time period 2005-2016 with indication of different phases within the wind farm's construction

Thornton Bank design			T0 Baseline			Construction phase 1		Operational phase 1	Construction phase 2 & 3	Right after construction	1 year after construction	Operational phase		
Location	imp/ref/fri	top/gully	2005	2006	2007	2008	2009*	2010	2011	2012	2013	2014	2015	2016
ft330	ref	gully	x		x	x	x (l)	x	x	x	x	x		
ftWG2	ref	top	x	x	x	x		x		x	x	x		x
ftWT1bis	ref	gully	x			x		x		x	x	x		x
ftWT2bis	ref	top	x			x	x (l & s)	x	x	x	x	x		x
ftWT3	ref	gully	x			x	x (l & s)	x		x	x	x		x
ftWT4	impact	top	x			x		x						
ftWT5	impact	top	x			x	x (l & s)	x						
ftWT6	impact	gully	x			x		x						
ftWT7	fringe	gully	x			x	x (l)	x		x	x	x		
ftWT8	impact	top	x	x	x	x	x (l)	x						
ftWT9	fringe	gully	x			x	x (l & s)	x		x	x	x		
ftWT10	fringe	gully								x	x	x		
ftWT11	fringe	gully								x	x	x		
ftTrack1	impact	top								x				
ftTrack 2	impact	top								x	x			x
ftTrack 3	impact	top								x	x			
ftTrack4	impact	top								x				
ftTrack 5	impact	top								x	x	x		x
ftTrack 6	impact	top								x	x	x		x

* Before 2009 all fish tracks were 2 NM – 30' trawls. In 2009, there was a switch from long (l) (2 NM – 30' trawl) to short (s) (1 NM – 15' trawl) fish tracks and afterwards all tracks were short trawls (see Derweduwen et al. 2010). Highlighted samples were used for the current analyse.

Table 2. Overview table of autumn sampled locations for effects of the Belwind wind farm within the time period 2005-2016 with indication of different phases within the wind farm's construction

Bligh Bank design			T0 Baseline	Construction Belwind		Right after construction	1 year after construction	Operational phase			
Location	imp/ref/fri	top/gully	2008	2009	2010*	2011	2012	2013	2014	2015	2016
ftWBB01	ref	gully	x	x	x (l & s)	x	x	x	x		x
ftWBB02	ref	top	x	x	x	x	x	x			x
ftWBB03	ref	gully	x	x	x	x	x	x	x		x
ftWBB04	fringe	gully	x	x	x	x	x	x	x		x
ftWBB05	impact	gully	x	x		x	x	x	x		
ftWBB06a	impact	top	x	x		x	x	x			x
ftWBB06b	impact	top				x	x	x			x
ftWBB07	impact	gully	x	x		x	x	x	x		
ftWBB08	fringe	gully	x	x	x	x	x	x	x		
ftWOH01	ref	gully	x		x	x	x	x			
ftWOH02	ref	top	x	x	x	x	x	x			
ftWOH03	ref	gully	x	x	x	x	x	x			

* Before 2010 all fish tracks were long (l) 2 NM 30' trawls, from 2010 onwards all fish tracks became short (s) 1 NM – 15' trawls (see Derweduwen et al. 2010). Highlighted samples were used for the current analyses.

community structure per ecosystem component and per OWF.

Pelagic species (based on www.fishbase.org) such as *Sprattus sprattus*, *Trachurus trachurus*, *Scomber scombrus*, next to jellyfish, bivalves (such as *Abra alba)* and polychaetes were excluded from the analyses, since these are not quantitatively sampled with a beam trawl. For analyses on community structure, rare species (frequency < 6% in the dataset) were excluded, data were square root transformed and similarity among samples was quantified using Bray-Curtis similarity index. PERMANOVA analyses on univariate data (species richness, density and biomass) were performed on Euclidean distance resemblance matrices with unrestricted permutation of raw data.

All analyses were executed using Primer v6 with PERMANOVA add-on software (Clarke & Gorley 2006; Anderson *et al.* 2008).

3. Results

3.1. Epibenthos

3.1.1. Species number, density and biomass

For both wind farms, the number of epibenthic species (S) remained similar over the years between impact and reference samples, and was not affected by the construction of the wind farm (interaction year x impact, Permanova p > 0.05) (fig. 2). In the Belwind area, we did observe a decrease in number of species both in impact and reference samples, which may be linked to the switch from long to short fish tracks. Density and biomass showed a similar trend in both wind farms, with an increase in the first two years after construction (*i.e.*, 2011 for C-Power phase 2 and 2010 for Belwind), and levelling off after three years post-construction (fig. 2). This trend was only significant in C-Power (interaction term "year x impact", Permanova p = 0.04 for both density and biomass), not in Belwind.

Variances between samples were higher in the first post-construction years (presented by the wider/longer bars in fig. 2), indicating a higher degree of heterogeneity within the wind farms immediately after the construction phase. Increases in density and biomass in C-Power in the two years post-construction were mainly due to the common star fish *Asterias rubens* and the flying crab *Liocarcinus holsatus* (the latter only in 2013). In Belwind, increases in both biomass and density two years post-construction were owing to *A. rubens* and serpents' table brittle star *Ophiura albida*.

3.1.2. Species composition

For C-Power, a significant wind farm effect (impact x phase, p = 0.007) was found. Pairwise tests showed that impact and reference samples differed significantly in community structure in the phase "right after construction" (p = 0.04) and "one year after construction" (p[mc] = 0.03). SIMPER analysis showed that this was not due to the occurrence of other species, but related to differences in species' densities. Much higher average densities of *A. rubens* (80 in impact vs 14 ind./1000 m² in reference), and higher densities of the hermit crab *Pagurus bernhardus* (12 vs 5 ind./1000 m²) occurred in impact samples "right after construction" (*i.e.*, 2012) compared to reference samples in the same phase (fig. 3). One year after construction, differences were again related to higher densities of *A. rubens* (avg. 21 in impact vs 7 ind./1000 m² in reference), and this time also higher densities of *L. holsatus* (avg. 18 in impact vs 4 ind./1000 m² in reference). During the first three phases (baseline, construction phase 1 and operational phase 1), the dominant species were the brown shrimp *Crangon crangon* and *O. albida* both in impact and reference samples, but after construction phase 2 & 3, densities of both species dropped (fig. 3). This is not a wind farm effect, but rather natural variation since this trend was observed in both impact and reference locations.
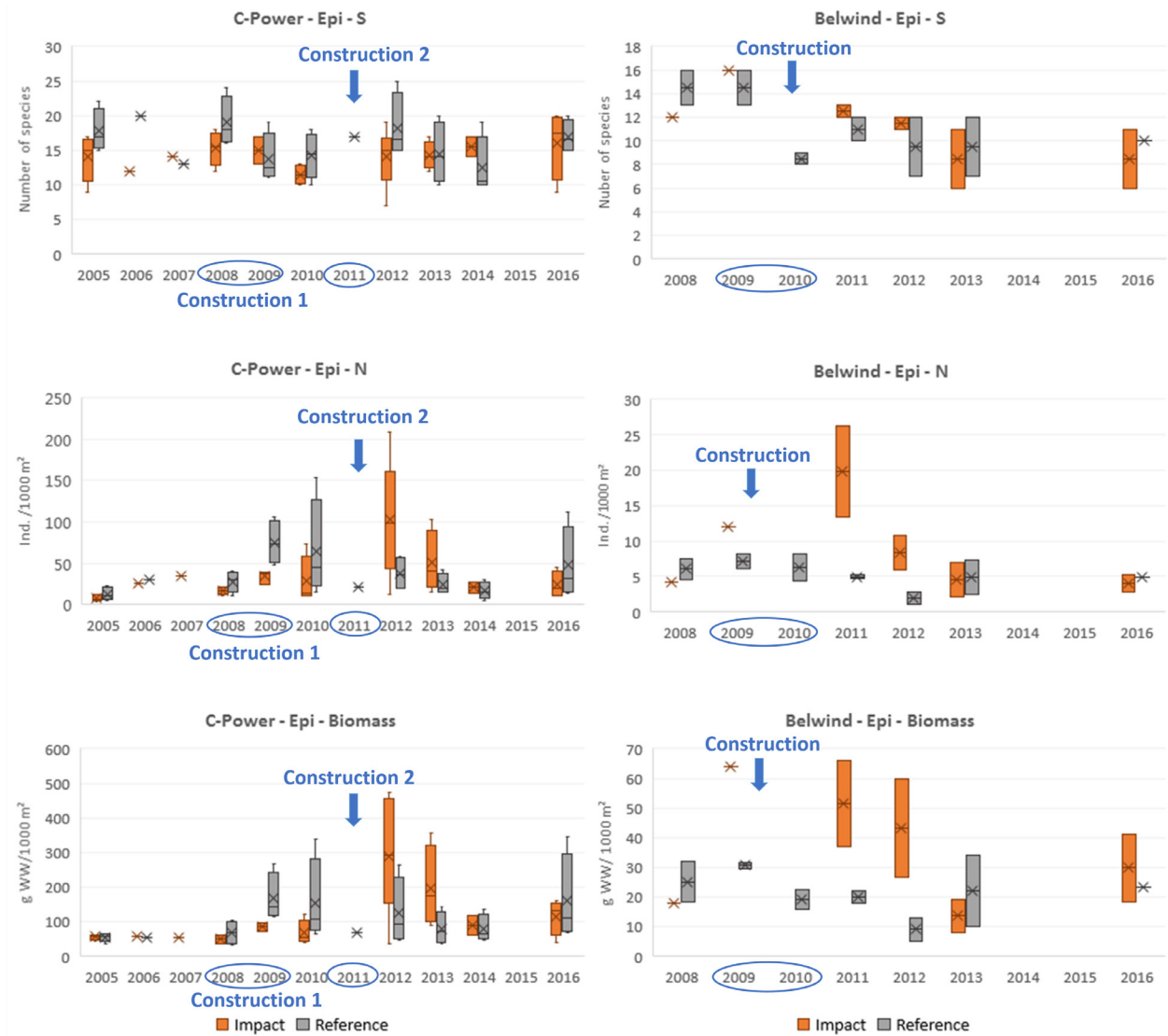
Figure 2. Boxplots showing species richness (S), density (N) and biomass for the ecosystem component epibenthos in both impact and reference samples at C-Power and Belwind wind farm. Median is shown as a horizontal line, average is indicated by x, the bars represent the 0.25 and 0.75 percentiles and the vertical lines are the minima and maximal values recorded.

For Belwind, no significant wind farm effect was observed (impact x phase, p = 0.2) for community structure. The lack of significance can be attributed to the limited number of samples, since differences in species densities were seen in the phase "right after" and "one year after" construction, although again no differences in species composition were found. Higher densities in impact samples were mainly observed for *O. albida* (3 in impact vs 0.2 ind./1000 m² in reference right after construction; 2 vs 0.1 one year after construction) and *P. bernhardus*

(8 vs 3 ind./1000 m² shortly after; 3 vs 1 one year after). This is also shown by the vector overlay based on multiple species correlation (r > 0.4) (fig. 4). *Asterias rubens* also showed much higher densities in impact samples (4 vs 0.3 ind./1000 m² "shortly after"; 2 vs 0.3 ind./1000 m² "one year after"), but since this species is highly correlated to *O. albida* (r = 0.8), it is not shown in the multiple vector overlay.
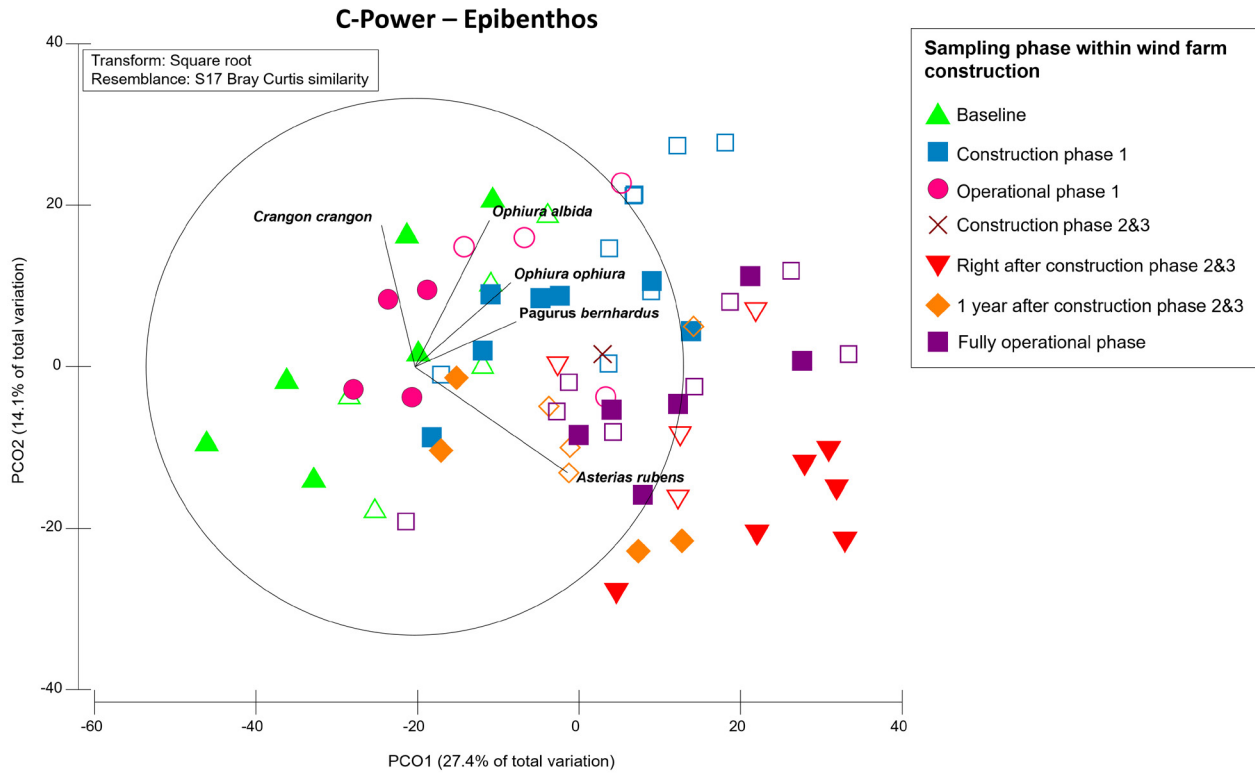
Figure 3. PCO plot of the epibenthos community at C-Power wind farm with indication of the situation phase of the wind farm. Open symbols refer to reference samples, filled symbols are impact samples. Vector overlay shows the species that are best correlated (multiple correlation r > 0.4) with the observed multivariate pattern.
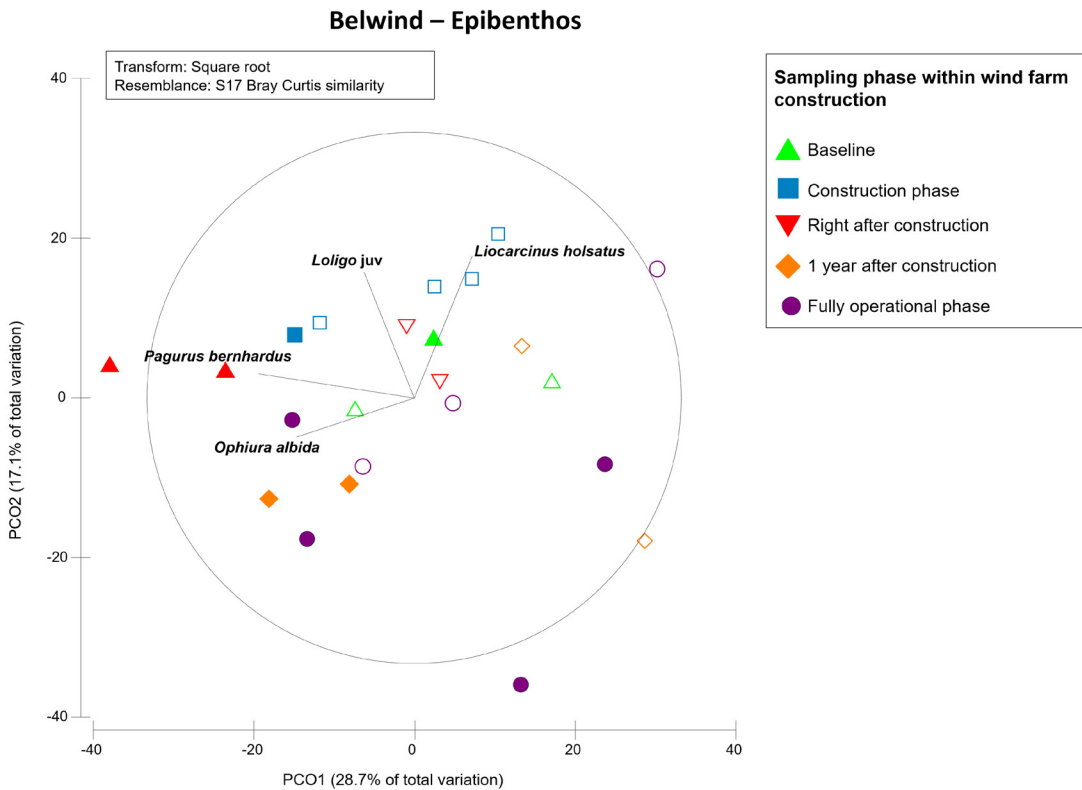

Figure 4. PCO plot of the epibenthos community at Belwind wind farm with indication of the situation phase of the wind farm. Open symbols refer to reference samples, filled symbols are impact samples. Vector overlay shows the species that are best correlated (multiple correlation r > 0.4) with the observed multivariate pattern.

3.2. Demersal and bentho-pelagic fish

3.2.1. Species number and density

For the number of species (S), no wind farm effect (interaction year x impact) was observed for the ecosystem component fish, not for C-Power (p = 0.5), nor for Belwind (p = 0.4). In Belwind, a significant year effect (p = 0.03) was observed, with a higher number of species in 2009 compared to the years 2011, 2013 and 2016, but this was the case for both impact and reference samples (fig. 5).

Fish density (N) did not show a significant wind farm effect for both C-Power (p = 0.1) and Belwind (p = 0.8). A decrease in density was observed in Belwind after 2011 in the impact samples, but this was also the case (to a lesser extent) in the reference samples, so this is probably related to natural variation (fig. 5). Especially lower densities of the dominant species lesser

weever *Echiichthys vipera* were noted from 2012 onwards, but also decreases in other species like solenette *Buglossidium luteum,* gobies *Pomatoschistus* sp. and reticulated dragonet *Callionymus reticulatus* were observed, again both in reference and impact samples. As for epibenthos, there seemed to be some more variance in fish densities directly after construction.

3.2.2. Community structure

For both wind farms, fish community structure was not affected by the construction of the OWF (impact x phase, p > 0.05). The sampling period or phase, on the other hand, was significant for both C-Power (p = 0.0001) and Belwind (p = 0.006), indicating that similar changes occurred over time both in reference and impact samples. These changes were not on the level of
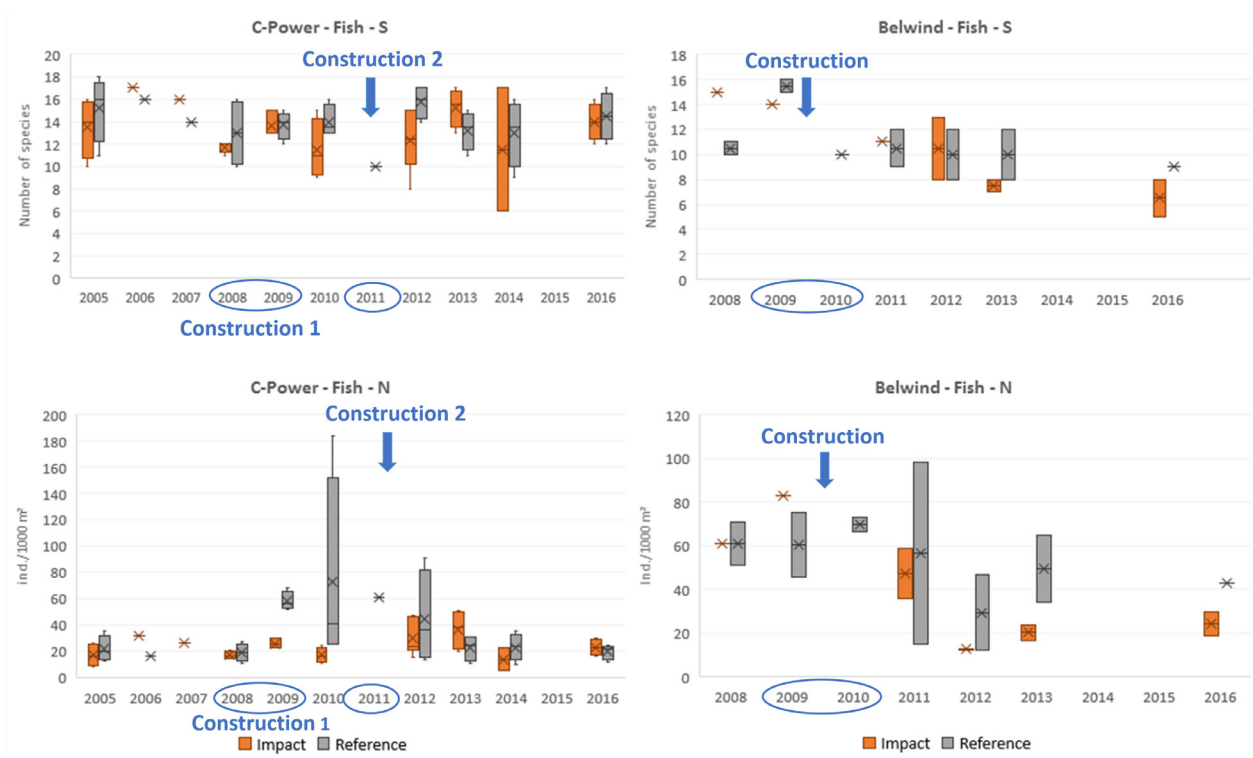

Figure 5. Boxplots showing species richness (S) and density (N) for the ecosystem component fish in both impact and reference samples at C-Power and Belwind wind farms. Median is shown with a line, average is indicated by x, the bars represent the 0.25 and 0.75 percentiles and the vertical lines are the minima and maximal values recorded.
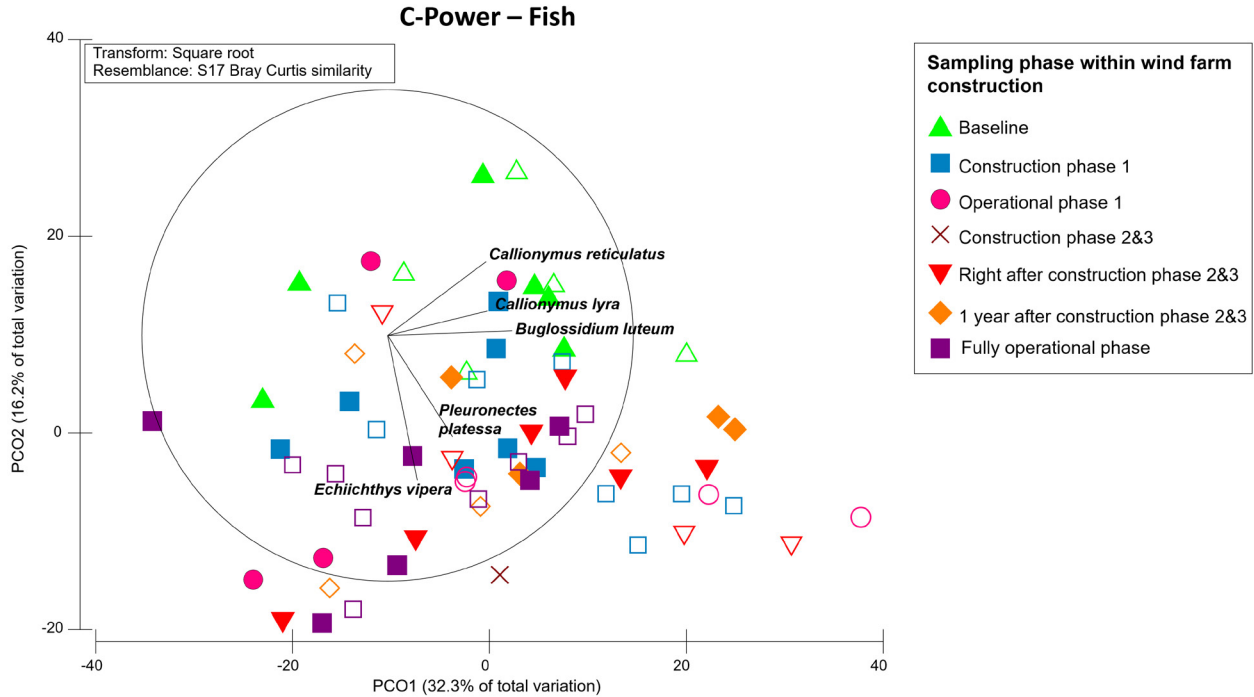
Figure 6. PCO plot of the fish community at C-Power wind farm with indication of the situation phase of the wind farm. Open symbols refer to reference samples, filled symbols are impact samples. Vector overlay shows the species that are best correlated (multiple correlation r > 0.4) with the observed multivariate pattern.

species composition, but were due to density changes at the species level for certain fish species.

Within the Thornton Bank area (C-Power), the multivariate pattern was best explained by the species *Pleuronectes platessa* (plaice), *E. vipera*, *B. luteum*, *Callionymus lyra* (common dragonet) and

C. reticulatus (fig. 6). An overall increase over time was observed for *E. vipera*, while *C. reticulatus* showed a decreasing trend, but this was noted in both impact and reference samples. Only for *P. platessa*, a wind farm effect was observed (interaction p = 0.04), a continuously increasing trend over time was seen mainly in impact samples,
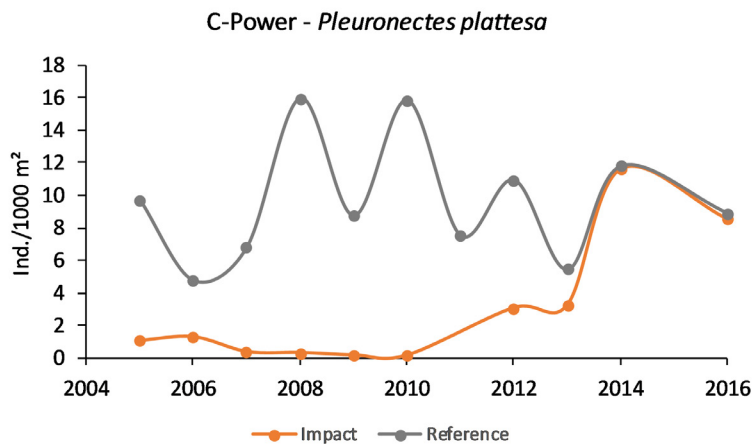

Figure 7. Timeline of average plaice densities in impact and reference samples at C-Power.
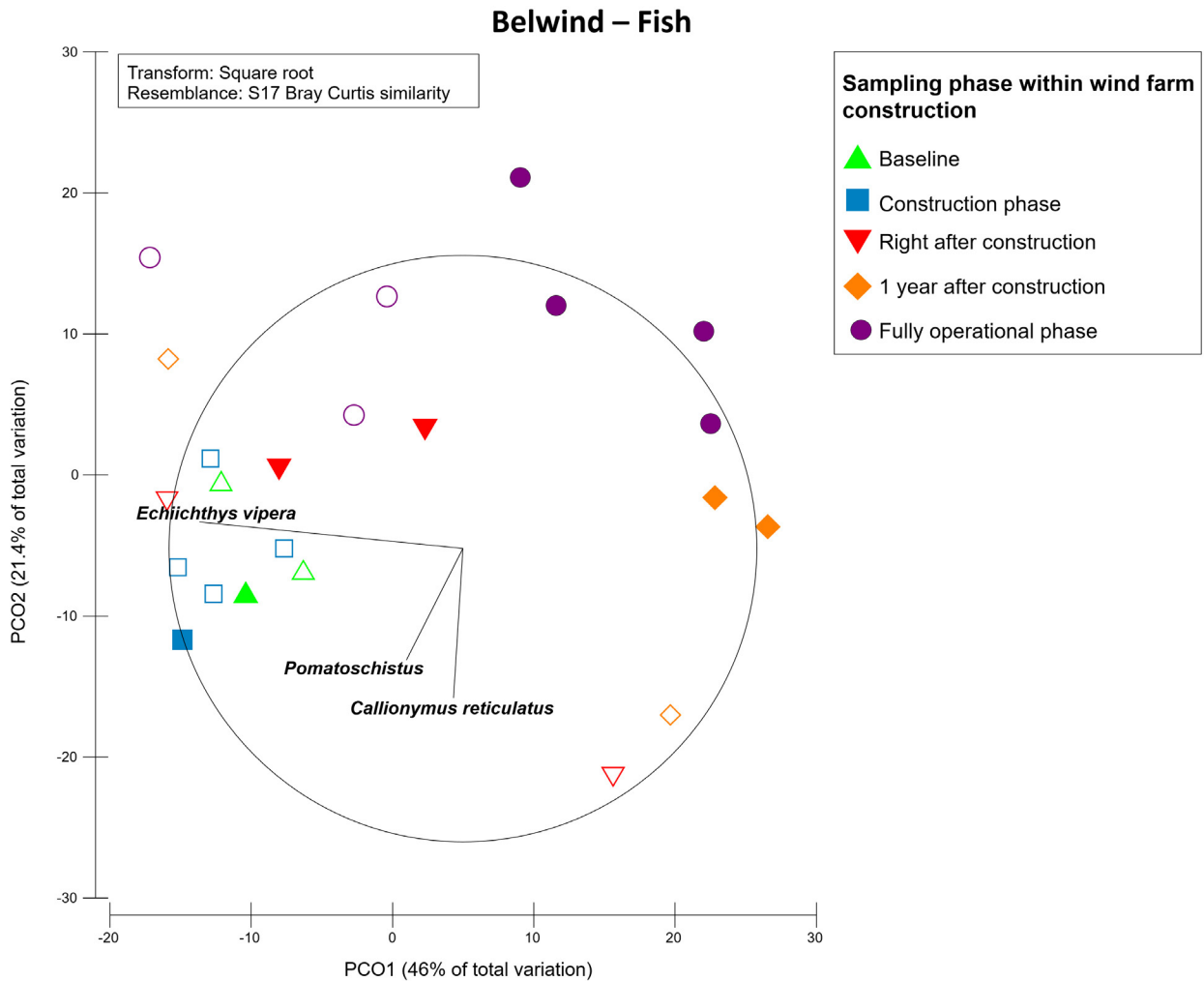
Figure 8. PCO plot of the fish community at Belwind wind farm with indication of the situation phase of the wind farm. Open symbols refer to reference samples, filled symbols are impact samples. Vector overlay shows the species that are best correlated (multiple correlation r > 0.4) with the observed multivariate pattern.

reaching similar densities as reference samples in 2013 (start of the fully operational phase) (fig. 7).

The Bligh Bank (Belwind) multivariate pattern was best explained by trends in densities of *E. vipera*, *Pomatoschistus* sp. and *C. reticulatus* (fig. 8). A decrease in *E. vipera* was observed over time, and is best related with the first PCO axis (fig. 8). This decreasing trend was most obvious in impact samples, but also in reference samples a slight decrease was also seen. *Pomatoschistus* sp. and *C. reticulatus* showed a decreasing trend over time in both impact and reference

samples. This trend is best related to the second PCO axis (fig. 8).

4. Discussion and conclusions

This chapter presents an update of the monitoring time series for epibenthos and fish sampled between 2005 and 2016 in the C-Power and Belwind concession area. Since the previous report (Derweduwen *et al.* 2016a), the time series was expanded with one sampling survey in autumn 2016. Hence, the analyses focused on the effect of a wind farm (combined influence of introduction of hard substrate and fisheries

exclusion) on the soft sediment epibenthos and fish communities. No fringe effects were investigated. Seasonality was excluded by only including autumn samples.

The main conclusions are:

• The OWFs did not directly affect the number of species or the species composition of the soft sediment epibenthos and fish assemblages. This indicates that the soft sediment ecosystem in between the turbines (at distance > 200 m) has not really changed (yet), some 5 to 6 years post-construction. The species originally inhabiting the sandy bottom are still in place and remain dominant in the species assemblages. This is in line with other studies, *e.g.*, Bergström *et al.* (2013) and Stenberg *et al.* (2015). However, there is one species, *Pleuronectes platessa* (plaice) that seems to be positively affected by the OWF, since densities increased after construction, indicating an attraction effect due to increased food availability and/or fisheries exclusion. Furthermore, although overall fish assemblages did not change, the feeding behavior of some fish species within the assemblage has changed (Derweduwen *et al.* 2016b): instead of limiting their diet to characteristic sandy bottom prey species, the investigated fish species (*i.e.*, lesser weever and dab) started preying upon species typically associated with hard substrates, so in that respect the presence of OWFs surely has an impact on the soft bottom ecosystem.

• The species assemblages within the OWFs seem to be mainly structured by temporal variability at larger spatial scales such as yearly temperature fluctuations, hydrodynamic changes, or plankton blooms. These processes influence species populations in a wider area, and that signal is picked up in our samples both in the OWFs and reference areas. For instance, the brown shrimp *Crangon crangon* was a dominant species in autumn samples at C-Power before 2012 in both impact and reference locations, but it almost disappeared afterwards. This can be linked to a change in the migration pattern or

reproduction cycle in relation to temperature differences (Boddeke 1975; Beukema 1992). For future analyses, it would be worthwhile to include environmental variables to gain a better insight in the observed patterns.

• For epibenthos, a post-construction "overshoot" of density and biomass was discerned directly after the construction phase of the windfarms. The pattern was identical for both OWFs; a density and biomass peak up to two years post-construction, but decreasing towards comparable levels as reference samples after three years post-construction. This shows that the previously observed wind farm effect (Vandendriessche *et al.* 2013; Derweduwen *et al.* 2016) was probably only a temporary phenomenon. Density and biomass peaks could be attributed to four species: *Asterias rubens* and *Pagurus bernhardus* (in both OWFs), *Liocarcinus holsatus* (C-Power) and *Ophiura albida* (Belwind). Increased densities in the common starfish (*A. rubens*) were also noted in macrobenthos samples from the area (Coates *et al.* 2014) and on the turbine foundations (Kerkhof *et al.* 2012). These four species are all typical opportunistic, scavenging species, that were probably attracted to the increased food availability due to fouling communities on the turbines. It also seems that these species had a patchy distribution, since variability between replicate samples was large in the two post-construction years compared to other years. High variability is characteristic for disturbances in biological communities, which can explain why several common species did aggregate, instead of being equally distributed. This density pattern was also noted in the multivariate analyses, especially in C-Power since *A. rubens* and *L. holsatus* were highly dominant in the assemblage during the two years post-construction.

• An effect of fisheries exclusion is not observed yet in the area in between the turbines. Near the turbines, "refugium" effects have been observed for fish (Reubens *et al.* 2013; Stenberg *et al.* 2015), but in

the current phase of the wind farms, such a refugium effect could not be observed at further distances from the turbines, except maybe for plaice that showed a clear increased density after construction. Besides the direct positive effects of fisheries exclusion on fish populations and bycatch species, that could be expected in OWFs serving as marine protected areas, a change in benthic community is also expected as an indirect effect due to the absence of trawling (Duineveld *et al.* 2007). Handley *et al.* (2014) showed that when fisheries are excluded, a change in the macrobenthic assemblages is expected with a bigger share of large long-lived species, sessile epifaunal species and species sensitive to trawling. Functionally, this can mean a shift from mobile scavengers, motile burrowing deposit feeders and predators to more suspension feeders and grazers, as observed in a soft bottom area closed to fisheries for 28 years in New Zealand (Handley *et al.* 2014). This change in benthic community triggered a functional change in the fish assemblage as well. Up till now, no changes in macrobenthos related to fisheries exclusion have been observed in the investigated wind farms in the Belgian part of the North Sea (Reubens *et al.* 2016). As such, a related

change in epibenthos and fish assemblage is also not to be expected yet. At this moment, time after construction is probably still too short, and the whole wind farm concession area is not yet large enough to signal effects of fisheries exclusion beyond the immediate vicinity of the turbine.

• Due to several reasons, such as wind farm accessibility, weather conditions and research vessel availability, the number of samples remains limited, which hampers the statistical power for thorough analyses on soft sediment epibenthos and demersal fish assemblages. Striking changes will surely be detected with this design, but more subtle changes may be difficult to be picked up. For the following years we will try to increase the number of replicate samples per survey to increase the power of the analyses.

Acknowledgements

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CHAPTER 6

ON THE REPLICABILITY OF NATURAL GRAVEL BEDS BY ARTIFICIAL HARD SUBSTRATA IN BELGIAN WATERS

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Abstract

In this contribution, we were particularly interested in qualifying the differences of natural (*e.g.*, gravel beds) versus artificial (*e.g.*, turbine foundations and scour protection) hard substrates. Therefore, we explored the epifauna data based on biological trait composition rather than the species composition of the epifouling communities. Both habitats harbour a rich species diversity and share a number of species. The initial results show that natural hard substrata harbour a much higher number of species and also more unique species than the artificial ones and that there are also some differences in life traits. Therefore, it seems that artificial hard substrata cannot act as alternatives to the loss of natural hard substrata.

1. Introduction

Gravel areas occur scattered in the soft sediment dominated southern North Sea and the Eastern Channel (Veenstra 1969; Cameron & Askew 2011). Several studies demonstrate that gravel areas, such as the Westhinder sandbank area (Belgian waters – Houziaux *et al.* 2008; Haelters *et al.* 2007), the Klaverbank (Dutch waters – Van Moorsel 2003) and the Dover Strait (French waters – Foveau *et al.* 2008), accommodate a unique community of species. These gravel beds are composed of boulders of variable sizes, shell fragments and sand. This high habitat heterogeneity leads to a huge biodiversity.

In addition to natural hard substrata, many artificial hard substrata occur in the North Sea (Zintzen *et al.* 2008; Coolen 2017) such as wrecks and wind farms. The hardening of the coastal zone, due to the increasing construction of harbours, groynes and other structures, is rapidly changing the coastal environment and also further offshore with a proliferation of wind farms and other marine infrastructure in response to the increasing demand of renewable energy, the number of man-made structures increases (*e.g.*, Mineur *et al.* 2012). In the wind farms, both the foundations of the turbines and the erosion protection around the foundations form hard substrata where species can settle. On the other hand, unspoiled natural hard substrata are decreasing due to fisheries pressure and aggregate extraction (Lindeboom *et al.* 1998).

The creation of new habitats increases the habitat diversity, which in turn increases species diversity. Hence, artificial hard substrata too are often considered hot spots of biodiversity (Wolff 1999). However, in coastal regions they often harbour introduced species that occur all over the world (Kerckhof *et al.* 2016; Reise *et al.* 1999).

From the onset of the hardening of the coast, which started in the 16th century (Wolff 1999), many hard substrata species successfully colonised this newly created habitat (*e.g.*, Mineur *et al.* 2012). Through history, shipwrecks further augmented the extent of suitable habitat for many of these hard substrata species (Zintzen & Massin 2010; Lengkeek *et al.* 2013). With the construction of offshore wind farms finally, a new habitat of artificial hard substratum was introduced in a region mostly characterized by sandy sediments, enhancing the habitat heterogeneity and biodiversity of the region (Kerckhof *et al.* 2009; 2010). The effect of the introduction of these hard substrata – the so-called reef effect – is regarded as one of the most important changes of the marine environment caused (Petersen & Malm 2006). These artificial substrata are in general rapidly colonised by fouling organisms (Horn 1974; Connell & Slatyer 1977; Kerckhof *et al.* 2010).

Because fishing activities, including bottom trawling, are prohibited in the Belgian wind farms and due to the intrinsic architectural characters of the wind turbines, the fouling communities on the artificial hard substrata of the scour protection and the piles are not disturbed by human activities. In contrast, natural hard substrata are threatened under the influence of various human activities such as bottom trawling and sand and gravel extractions (Lindeboom *et al.* 1998). The impact of bottom disturbing fisheries has increased significantly over the last 100 years due to technological advances and consequently, long-lived fragile species and erect species have declined or disappeared

while scavengers and opportunistic species are favoured (Lindeboom *et al.* 1998). Some parts of the gravel areas in the Westhinder sandbank area are practically more difficult for fishing because they are situated in a trough of sickle-shaped barchan hills (Houziaux *et al.* 2008). Here, in this somewhat sheltered zone, the biodiversity proved to be greater than in the surrounding, more fished zones. These areas can be regarded as relicts and hence important for the possible recovery of the surrounding Habitat Directive Area. Due to their higher biodiversity, they can act as a source from which disturbed areas could be recolonized.

In the royal decree establishing a maritime spatial plan for the Belgian waters (Anonymous 2014; Vandevelde *et al.* 2014), the gravel areas of the Westhinder sandbank area are part of the proposed special area for conservation "Vlaamse Banken - Flemish banks" (Anonymous 2012) (kaart MSP) that, in turn, is part of the ecological network Natura 2000 (Anonymous 1992). In the gravel bed area of the Westhinder sandbank area, two subzones have been designated for improving the seafloor integrity by reducing fisheries with bottom contacting gears, called zone 3 and zone 4 (MSP) In zone 3, no bottom disturbing fisheries are allowed at all while zone 4 experimental fishing techniques are still allowed. Zone 3 is also known as a relict (formerly called "refugium") zone (Houziaux *et al.* 2008) because of its rich epifauna since this zone is less fished than the surroundings due to the presence of barchan dune structures that hamper beam trawling.

Another anthropogenic activity with a potential impact is sand and gravel extraction (Vanaverbeke *et al.* 2007). Since 2011 in an area located 2.5 km from zone 4 of the "Vlaamse Banken", sand and gravel is extracted (Anonymous 2011). The silt that is suspended by the extraction is carried by the flow and can cause clogging of the gills of filter feeders (Vanaverbeke *et al.* 2007), many

species of which occur on hard substrate. Also, organisms on the stones can catch the sludge, so that typical hard substrate species can no longer settle (De Mesel *et al.* 2013).

Recently, artificial hard substrata are sometimes put forward as a possible alternative for the loss of natural hard substrata habitat and are even proposed to strengthen biodiversity *e.g.*, plan Zeehond (Van de Lanotte *et al.* 2012). When evaluating whether artificial hard substrata habitat may indeed strengthen and/or even replace natural hard substrata habitat, we first have to investigate the (dis)similarities in species and community composition between both types of hard substrata habitat. In this preliminary exercise, we therefore explored to what extent artificial hard substrata (*i.e.*, turbine foundations and scour protection) are comparable to natural ones (*i.e.*, gravel beds) and if, for example, they in a way contribute to and/or strengthen species and functional (*i.e.*, biological traits) diversity of naturally occurring hard substrata.

2. Material and methods

We selected three data sets on hard substrate fauna available from the same water mass, situated in clear offshore (Channel) waters (M'harzi *et al.* 1998; Lacroix *et al.* 2004). We focused on hard substrata waters influenced by Channel waters because most of the natural hard substrata in Belgian waters are situated in this type of water.

2.1. Artificial hard substrata sample selection

As an example of the artificial substrata, we selected the wind farm located on the Bligh Bank at about 50 km off the Belgian coast (see Brabant *et al.* 2011). This wind farm is furthest situated from the Belgian coast and is entirely located in the English Channel's water flow. The construction started at the end of 2009. The turbines are steel monopile foundations surrounded by a scour protection

consisting of natural stones of various sizes (Van Oord Dredging & Marine Contractors 2009). As such, the combined scour protection sites could be regarded as an artificial reef consisting of 107 separate locations, each composed of a single steel foundation pile surrounded by approximately 500 m² (~ 840 m³) of scour protection.

All subtidal samples were collected by scuba divers in 2011. The samples included 9 scrape samples taken on the turbine foundations and 9 stones gathered from the scour protection (Kerckhof *et al.* 2011). On the piles samples, were taken at a depth of 15 m by scraping 3 replicates of the fouling organisms from a sampling surface area of 6.3 dm² of turbines BBC2 (May) and BB8 (May and November). Divers collected 3 stones of the scour protection of turbines BBC2 (May) and BB8 (May and November). The scraped material and the stones were collected in plastic bags that were sealed and transported to the laboratory for further processing: sieving over a 1 mm sieve and sorting. The samples were preserved on buffered formalin 10% and further processed in the laboratory.

2.2. Natural hard substrata sample selection

To represent natural hard substrata, we selected two data sets taken in the Westhinder sandbank area, part of the special area for conservation, the "Vlaamse Banken". The first set consisted of 5 samples taken on 3 July 2013 at a depth of around 30 m in zone 3 (the so-called relict zone, *sensu* Houziaux *et al.* 2006). The second was taken in zone 4 in December 2016 and consisted of 13 samples. The samples of the natural hard substrata were taken at a depth of approximately 25 m with a 0.1 m² Hamon grab. This device is, contrary to other commonly used grabs such as the Van Veen and Box Corer, especially suitable for use in gravelly sediment although it does not collect large boulders or stones. Once on board, the samples were sieved over a 1 mm sieve and

sorted. The largest boulders (with or without growth) were separated from the coarse material, which consisted of coarse sand, shell fragments and gravel. Then the samples were preserved on buffered formalin 10% and further processed in the laboratory. We pooled both data sets of the Westhinder sandbank area because both originated from the same area.

After preservation of the samples, individual organisms were sorted and identified to the lowest taxonomic group possible – mostly species level (further called "species") – using a stereoscopic binocular microscope.

Additionally, to assess the quality of the habitats and as part of the criteria put forward in the Determination of the Good Environmental Status and Establishment of Environmental Targets for the Belgian Marine Waters (Belgische Staat 2012) as required in the framework of the Marine Strategy Framework Directive, we also looked at the presence of large erect species, in particular certain Bryozoans such as *Flustra foliacea*, *Alcyonidium* spp. and sponges such as *Haliclona oculata* and Hydrozoans.

2.3. Data selection

In this study, we took into account both countable macrofaunal (retained by a 1 mm mesh-sized sieve) organisms and uncountable crust forming and erect (bushy) epifaunal species such as Cnidaria, Bryozoa, sponges, etc. To be able to combine both types, we transformed the data to presence/absence.

The samples taken in the Westhinder sandbank area with the Hamon dredge contained both epifaunal species and infaunal species – contrary to the samples of the scour protection that only consisted of stones. Since we were only interested in species associated with hard substrata, we scored them in relation to their affinity with hard surface. Obligate hard substrate species such as forms

cemented on a surface (*e.g.*, acorn barnacles, forms attached by threads or knobby structures *e.g.*, mussels) and mobile forms (*e.g.*, snails, sea urchins) or species living in burrows under or in the vicinity of hard substrata such as certain worms received score 3. Infaunal species that do not depend on a solid surface were scored 0. A certain number of free living species is nevertheless associated with hard substrata, as they are dependent on either the substrate itself as shelter or because they are associated with species that are obligate hard substrate species as, for example, Stenothoe or nudibranchs feeding on Tubularia. Moreover, some species, such *as Lanice conchilega* or *Crepidula fornicata*, are capable to live both on hard and on soft substrata. Such species were scored 1 or 2 according to their dominant occurrence. Some species such as *Venerupis corrugata* and *Aequipecten opercularis* start their live attached to hard substrata before moving to an adult free living (Aequipecten) or infaunal (Venerupis) stage. Since we only encountered juvenile stages of these species in our samples we treated them as obligate hard substrate species, score 3.

We extracted a species list for the three habitats and only retained hard-surface species: out of the species pool of all species identified, we eliminated those species – infaunal species – that were not associated to hard surfaces. This yielded a list with genuine hard substratum species and species that are associated with hard substrata during at least part of their life cycle. The dataset contained in total 208 unique species of which 136 were considered hard substrate or hard substrate associated species.

2.4. Biological traits selection

In this exercise, we gathered information for two biological traits: the feeding method and the mobility of the adults. Information was gathered from a variety of published sources. Additionally, information was obtained from online databases *e.g.*, WORMS,
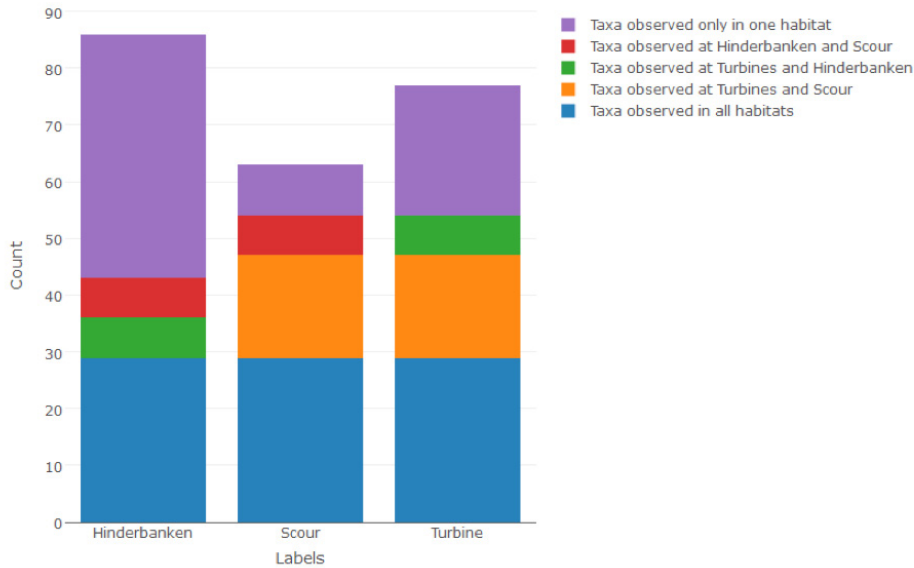
Figure 1. Number of hard substrate species observed per habitat.

BIOTIC or specialized literature. In case the information was not available for a particular species, we looked at other species of the same genus, or family.

We recognized five feeding traits: suspension feeder, deposit feeder, parasite, predator/scavenger and grazer. A "suspension feeder" is an organism that feeds on organic particles filtered out of the water column, a "deposit feeder" feeds on fragmented organic particles deposited onto the bottom. We combined predator and scavenger into one category, as most predators are also to some extent feeding on carrion.

For the mobility trait, we recognized five categories: sessile, hemi sessile, permanently attached, crawler and digger. We used the category hemi sessile for organisms that usually stay on the same spot, but can move to a limited extent in a limited area, such as the amphipod *Jassa herdmani*, the plumose anemone *Metridium senile*, or the mussel *Mytilus edulis*. Sessile was used for organisms living in rigid tubes firmly attached to the substrate or organisms firmly attached to the substrate (*i.e.*, permanently attached), *e.g.*, Cnidaria, sponges. Species belonging to the digger category live and move into the soil. *Venerupis corrugata* and

Aequipecten opercularis start their lives attached to hard substrata and are treated here as hemi sessile.

2.5. Criteria for the classification of traits

Because some species can belong to more than one category we used the "fuzzy coding" method (Chevenet *et al.* 1994). For each type, a score of zero to three was assigned to each category of traits. Zero means that the species does not belong to that category, three means that the type belongs to that category and one or two means that the species exhibits that property but with lower or higher affinity for that category. If the species belongs to two categories, one or two will be assigned to those categories. For each species, the sum of all categories of one feature is equal to three.

3. Results

3.1. Species richness

The three habitats shared an equal species number (29) (fig. 1). The scour and the turbine of the wind farm habitats shared 17 species, and 7 species were present on both Scour and Hinderbanken and the Westhinder sandbank area shared 7 species with the turbines.
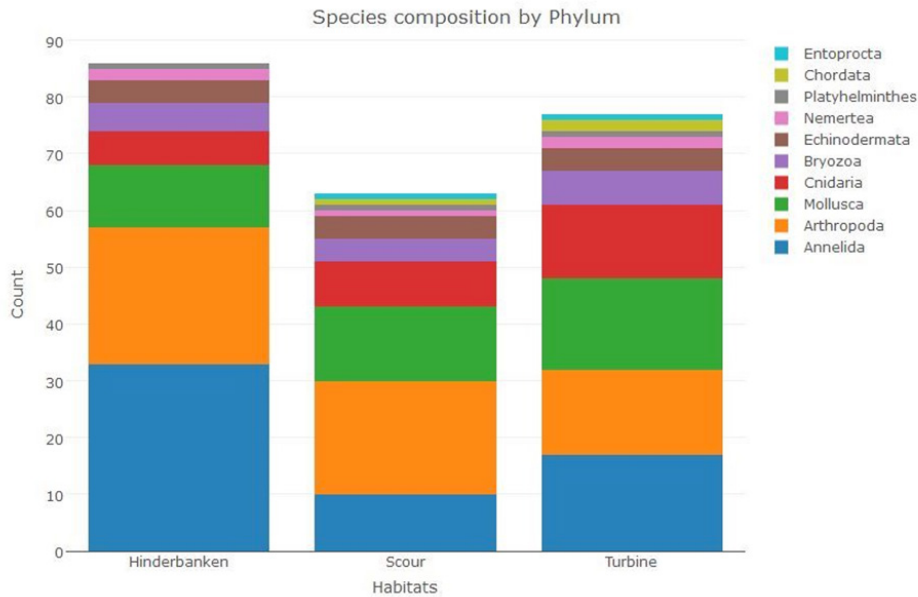
Figure 2. Taxonomic species composition per phylum in the different habitats is presented in.

The Westhinder sandbank area harboured the highest number of unique species. The number of unique species was much lower in the wind farm habitats with 9 on the scour and 23 on the turbines.

3.2. Taxonomic composition

In all three habitats Annelida, Arthropoda and Mollusca were the three most species-rich phyla, followed by Cnidaria and Bryozoa (fig. 2). Cnidaria were relatively more species rich in the Westhinder sandbank area.

The distribution of the feeding classes is similar over the three habitats (fig. 3) with the exception of sessile species of which the number is clearly higher on the turbines than on the other habitats. Crawlers are in the majority in all three habitats while swimmers and diggers are virtually absent.
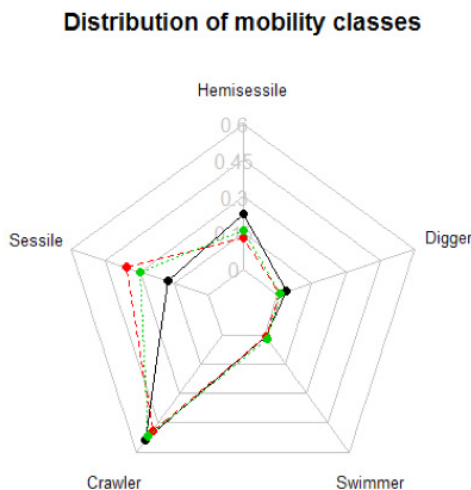

Figure 3. Radar chart of mobility classes indicating the relative composition of mobility classes of the species encountered in the three habitats (red: turbines; black: Westhinder sandbank area; green: scour protection).
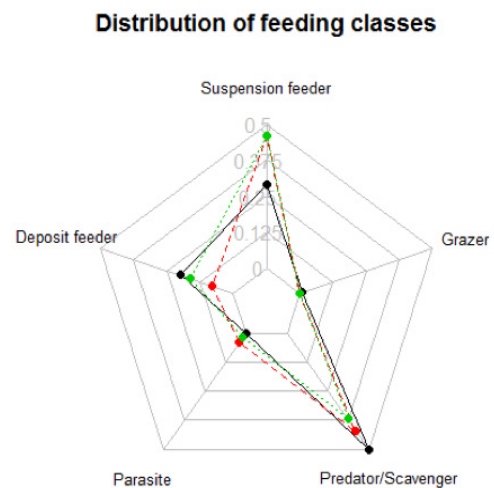

Figure 4. Radar chart of feeding classes indicating the relative composition of feeding classes of the species encountered in the three habitats (red turbines, black Westhinder sandbank area, green scour protection).

Suspension feeders are nearly twice as common on the artificial hard substrata of the piles and scour than on the natural hard substrata while deposit feeders are slightly more numerous on the natural hard substrata of the Westhinder sandbank area than on the wind turbine habitats. The Westhinder sandbank area contained the highest number of predator/scavengers their number was slightly lower the two artificial habitats. Grazers and parasites are virtually absent in all three habitats.

3.3. Number of long-lived and erect species

The number of long-lived and erect species found was very low in all three habitats including the natural gravel beds. Such erect species as the bryozoans *Flustra foliacea*, *Alcyonidium* spp. or the sponge *Haliclona oculata* were not found at all and from *Alcyonium digitatum* only small colonies were present.

4. Discussion

Despite this limitation mentioned above, the natural hard substrata of the Westhinder sandbank area harbour more and unique species than the other two habitats, including the scour protection. However, a rich community can only develop if the habitat is not strongly subjected to natural and/or anthropogenic disturbance. Bulleri *et al.* (2000) suggested that certain man-made structures in the marine environment could act as surrogate rocky shores but further research showed that artificial reefs could not be considered as substitutes for natural habitats in terms of relevant ecological processes (*e.g.*, Bulleri & Chapman 2010).

The species assemblages in the three habitats are characteristic for a hard substrate community and the contribution of dominant phyla to the species list is similar. There is a slight difference in species composition, species numbers, phylum composition and difference in biodiversity between the Westhinder sandbank area and the

erosion protection. For example, we observed a higher species richness with many unique spies in the natural hard substrata of the Westhinder sandbank. In reality, the number could be much higher because the preliminary data we used for these exercise clearly have limitations as for example no large stones were collected in the Westhinder sandbank area. However, the data allows to form an idea of a possible difference in species, species, dominant phyla, biodiversity and functional groups. In the future, data of larger stones will be taken into account as we will add data collected with the Gilson dredge, a device aimed to target larger stones and boulders. So it is likely that the species richness will increase even more if more data will be used. The possible underestimation of species richness potential, also holds true for the erosion protection that was in place for only two years. It should be noted however that the scour protection is expected to remain in place only for a period of 20-30 years in accordance with the conditions stipulated in the environmental permit for the construction and exploitation of the wind park (BMM 2007). The number of long-lived and erect species found was zero in all three habitats including the natural gravel beds. This was not expected as in a healthy natural hard substrate community their number should be higher. This could be a consequence of the sampling technique and/or degradation of the environment by *e.g.*, fishing/aggregate extraction. The relict zone is probably still touched by fishing activities despite the zone being less accessible. Both natural hard substrata and the scour protection are situated in a very dynamic environment, influenced by the movements of strong sand waves that sometimes cover the stones completely. It is unclear why deposit feeders and predators/scavengers are more numerous on the natural hard substrata than on the artificial hard substrates.

5. Conclusion

Our results – although preliminary – suggest that artificial hard substrata cannot readily be put forward as an alternative for declining natural substrata. We noted a difference in species numbers, functional groups and a difference in biodiversity between natural and hard substrata. This illustrates the importance of maintaining the Westhinder sandbank area, and thus the Vlaamse Banken, as part of the Natura 2000 network.

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CHAPTER 7

SEABIRD MONITORING AT THE THORNTON BANK OFFSHORE WIND FARM

UPDATED SEABIRD DISPLACEMENT RESULTS AS AN EXPLORATIVE ASSESSMENT OF LARGE GULL BEHAVIOUR INSIDE THE WIND FARM AREA

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Abstract

Since 2005, the Research Institute for Nature and Forest (INBO) has been performing monthly BACI-designed surveys to study seabird displacement following the construction of offshore wind farms (OWFs) in the Belgian part of the North Sea. Here we report our findings for the C-Power wind farm at the Thornton Bank after four years of post-construction monitoring. Following the concern on potentially high levels of collision mortality among large gull species, we also report the first results of our behavioural study, making use of our transect count data, GPS tracking data and observations with a fixed camera installed on turbine I5 in Thornton Bank OWF.

As expected, considering the rather small amount of data added during the monitoring year 2016, our displacement study results are highly similar to those reported in the previous monitoring report (Vanermen *et al.* 2016). The impact area appeared to be avoided by four species, being northern gannet, little gull, black-legged kittiwake and common guillemot, these having dropped in numbers by no less than 97%, 89%, 75% and 69% respectively. The Thornton Bank OWF attracted great black-backed gulls, numbers of which increased by a factor 6.6 compared to the control area and the period before impact. Sandwich tern too was attracted to the OWF at the Thornton Bank, the effect being significant for the buffer zone only, where we observed a factor 5.7 increase in numbers. Only for herring gull was there a shift in the estimated wind farm effect since the latest report. While the OWF coefficient for herring gull was estimated to be close to zero after three years of monitoring, it now showed a (borderline) significant increase in numbers (factor 2.9). The buffer zone, however, saw a significant decrease in numbers of herring gull.

Though it is still too soon to draw any definite conclusions out of our behavioural monitoring, there were already some indicative results. Great black-backed gulls for example clearly favor outer turbines

for roosting, suggesting a partial barrier effect. Based on our tracking data, lesser black-backed gulls seemed to spend half of their time inside the OWF area roosting on the jacket foundations, and spent less time flying inside compared to outside the wind farm. While mostly observed roosting, with the fixed camera we assessed that 9% of the large gulls observed on the jacket foundations were actually foraging. Sustaining the current effort throughout 2017 will allow us to analyse tidal and diurnal patterns in the presence and behaviour of large gulls inside the Thornton Bank OWF. Importantly, the results of this behavioural study might shed new light on the currently expected collision risk of large gulls at OWFs, and may highlight the need for proper post-construction monitoring. Because next to a possible post-construction change in numbers, any behavioural shift (*i.e.*, a decrease in time flying) will have a strong effect on the anticipated collision mortality among large gulls.

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 steadily and at the end of 2016, 3589 offshore wind turbines were fully grid-connected in European waters, totalling 12.6 GW (EWEA 2017). Currently, three OWFs 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 Thornton Bank, 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, Northwind NV built 72 turbines at the Lodewijckbank, 37 km offshore, in the course of 2013.

Since 2005, the Research Institute for Nature and Forest (INBO) performs seabird counts specifically aimed at studying seabird displacement caused by OWFs. In this report we present the results of our seabird displacement study at the Thornton Bank OWF after 4 years of operation ("baseline monitoring").

Earlier results from the Bligh Bank OWF showed attraction of large gull species and therefore increased levels of collision risk, which could lead to population level effects in a (realistic) scenario of 10,000 wind turbines across the North Sea (Brabant *et al.* 2015). The behaviour and presence of large gulls inside OWF areas should therefore be subject of a "targeted monitoring" scheme. The design of such a monitoring scheme, however, is hampered by ongoing budgetary and logistic constraints. Nonetheless, the GPS tracking of large gulls breeding along the Belgian and Dutch coast does open possibilities to study their behaviour inside OWFs more closely. A fixed camera located at one of the jacket foundations on the edge of the Thornton Bank OWF further allows for behavioural observations of gulls on and around the turbines. Here we report the results of a first and explorative analysis of presently available behavioural data, mainly focusing on the gulls' association with the turbine foundations.

2. Material and methods

2.1. Thornton Bank offshore wind farm

The Thornton Bank wind farm is located 27 km off the coast of Zeebrugge, and consists of 2 subareas of 24 and 30 wind turbines, measuring 10.7 and 9.2 km² respectively (see fig. 2). The water depth of the turbine-built area ranges between 12 and 28 m (C-Power 2016). Distances between the turbines range 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.

2.2. Displacement study

2.2.1. 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 afterwards ("distance analysis"). The transect is therefore divided in four distance categories (A = 0-50 m, B = 50-100 m, C = 100-200 m and D = 200-300 m). Counting all flying birds crossing this transect, however, would cause an overestimation and would be a measure of bird flux rather than bird density (Tasker *et al.* 1984). Flying birds are therefore counted through one minute interval counts of a quadrant of 300 by 300 m inside the transect ("snapshot counts"). 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 was linked to the corresponding GPS coordinates registered 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 of birds observed per km², *i.e.*, a seabird density at a specific location. Up to 2012, observations were aggregated in ten minute bouts, which were cut off to the nearest minute at waypoints. Since 2013, resolution was 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 recorded inside 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.

2.2.2. Distance analysis

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 0-0.5 m / 0.5-1.0 m / 1.0-2.0 m / 2.0-3.0 m, …), both variables 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. As such, we 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.

2.2.3. Monitoring set-up

Monitoring was performed according to a Before-After Control-Impact (BACI) set-up. The OWF footprint area was 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 could 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 (Vanermen *et al.* 2005). Meanwhile, the distance between the control and impact area 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 Thornton Bank study area was counted on a highly regular basis from 2005 until present (figs 1-2). During this dedicated monitoring program, the study area 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 higher than 2 m and/or poor visibility). Before this dedicated monitoring program, the study area was 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.

For our displacement analysis, only data falling within the "reference period" and "impact period" (phase I, II and II) were used (table 1). Note that phase III was not yet operational before September 2013, while the impact period defined in table 1 starts in October 2012 (when phase II became operational). This is justified by the fact that access for monitoring was not allowed where active construction activities of phase III were going on, so data collected during that period account for the operational part of the OWF only.

Table 1. Definition of the reference, construction and impact periods at the Thornton Bank study area as applied in the impact analyses

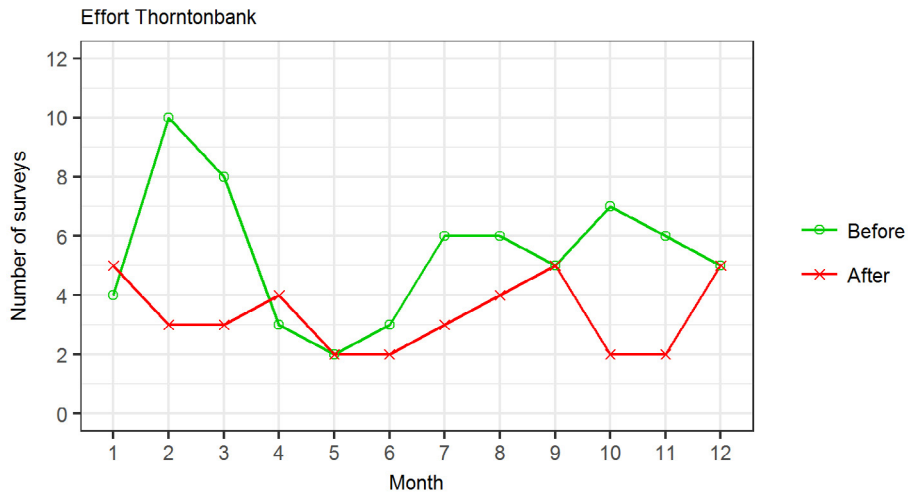| OWF	Phase	Period
Thornton Bank	Reference period	< 04/2008
	1st construction period	04/2008 => 05/2009 (highly restricted access)
	Impact period (phase I)	06/2009 => 04/2011 (6 turbines)
	2nd construction period	05/2011 => 09/2012 (variable access)
	Impact period (phase I, II & III)	10/2012 => present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

Figure 1. Count effort in the Thornton Bank study area indicated by the number of surveys performed before the construction of the phase I turbines (< 04/2008) and after the construction of the phase II turbines (> 09/2012).
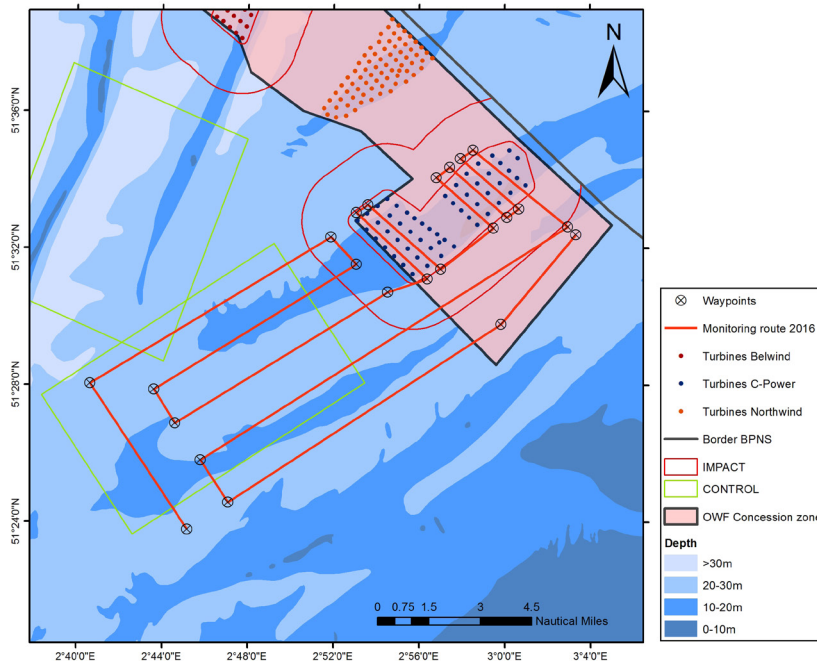

Figure 2. Monitoring route through the Thornton Bank OWF study area in 2016.

Compared to the previous monitoring report (Vanermen *et al.* 2016), data from eight monitoring days could be added to the dataset. During only four of these, however, we visited the OWF footprint area itself. The four other trips were sailed for reference monitoring of the future Norther OWF, during which monitoring inside the study area was confined to the two most south eastern tracks as shown in figure 2, only partly crossing the Thornton Bank OWF buffer zone.

2.2.4. BACI analysis

Introduction

For the BACI modelling, we aggregated our count data per area (control/impact) and per monitoring day, resulting in day totals for both zones. As such, we avoided spatio-temporal correlation between counts. We further selected only those days on which both the control and impact area were visited, minimizing day-to-day variation in seabird abundance.

Modelling was performed for twelve seabird species occurring regularly in the OWF area, *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 guillemot (*Uria aalge*) and razorbill (*Alca torda*). For each of these species, we modelled three different impact datasets (OWF footprint + 0.5 km, OWF footprint + 3 km, buffer 0.5-3 km; see fig. 3).
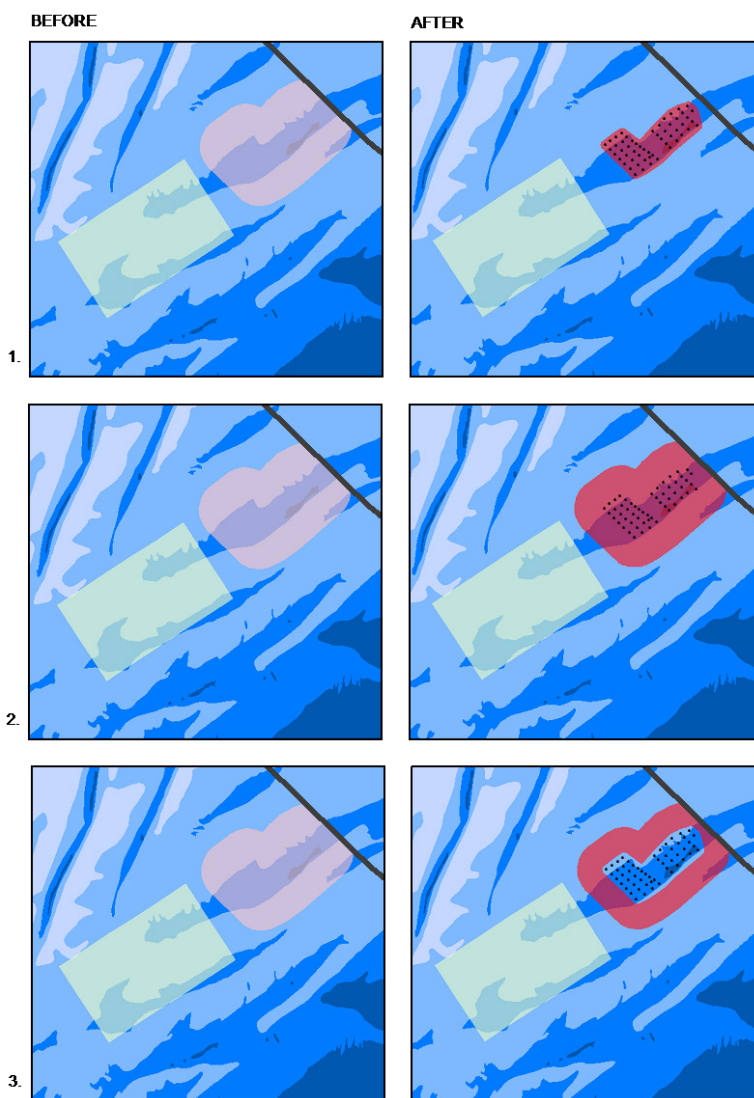

Figure 3. Overview of the BACI polygons used for data selection to study OWF induced seabird displacement at the Thornton Bank (green = control area / red = impact area; 1 = "OWF footprint + 0.5 km"; 2 = "OWF footprint + 3 km"; 3 = "buffer 0.5-3 km")

Response variable

The response variable (Y) of our displacement models equaled the number of birds observed inside the transect and during snapshot counts, aggregated per area and per monitoring day. For the large gull species herring, lesser black-backed and great black-backed gull we also modelled an "adjusted response variable". Because (i) the corridors between the C-Power turbines used during seabird monitoring (fig. 2) vary in width between 650 and 850 m, and (ii) the research vessels aimed to sail right in the middle of these corridors for security reasons, birds associated with the turbines were always right outside our 300 m wide transect. Our adjusted response variable is therefore calculated by adding (i) the number of birds that would have been counted inside the transect if the turbine-associated birds would have occurred homogenously spread across the area to (ii) the number of birds counted inside the transect and during snapshot counts (*i.e.*, the original response variable). This is best illustrated with an example: at 28 August 2015 we counted no less than 161 great black-backed gulls resting on the jacket foundations, as opposed to only 1 bird observed inside our transect (the original response) 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 Thornton Bank 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. If these birds would have occurred homogenously spread across the area, and knowing we counted 7.4 km², the number of birds inside the transect would be about 42 ($\approx (5.6*7.4) + 1$), which is our adjusted response. The original and adjusted response variable were always analysed both, and the difference is clearly indicated in the graphs and tables.

Explanatory variables

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) a time factor BA (before/after construction), (ii) an area factor CI (control/impact area), (iii) an offshore wind farm factor OWF (wind farm present/absent) and (iv) a fishery factor F (fishing vessels present/absent in the area). For the latter we only considered fishing vessels observed within a distance of 3 km from the monitoring track, and was considered only for species known to aggregate around fishing vessels (and therefore not used for little gull, Sandwich tern, common guillemot and razorbill). Finally, the continuous variable month (m) was used to model seasonal fluctuations by fitting a cyclic smoother or alternatively a cyclic sine curve, the latter described through 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 and/or an asymmetric seasonal pattern. Eventually, we considered five different "full" models:

- no seasonal variation:
$Y \sim BA + CI + OWF + F$

- 12 month period sine curve:
$Y \sim BA + CI + OWF + F + \sin(2\pi*m/12) + \cos(2\pi*m/12)$

- 12 + 6 month period sine curve:
$Y \sim BA + CI + OWF + F + \sin(2\pi*m/12) + \cos(2\pi*m/12) + \sin(2\pi*m/6) + \cos(2\pi*m/6)$

- 12 + 4 month period sine curve:
$Y \sim BA + CI + OWF + F + \sin(2\pi*m/12) + \cos(2\pi*m/12) + \sin(2\pi*m/4) + \cos(2\pi*m/4)$

- cyclic smoother:
$Y \sim BA + CI + OWF + F + s(m)$

Model selection

For the distribution and model selection we first considered the "OWF footprint + 3 km" dataset (fig. 3). 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 on the other hand mostly occur strongly aggregated in (multi-species) flocks, resulting in "over-dispersed" count data which can often be analyzed with a negative binomial (NB) distribution (Ver Hoef & Boveng 2007; Zuur *et al.* 2009). On the other hand, when the data exhibit (much) more zeros than can be predicted through a Poisson or NB distribution, it may be necessary to apply a zero-inflated (ZI) distribution (Potts & Elith 2006; Zeileis *et al.* 2008), which consists of two parts: (i) a "count component" modelling the data according to a Poisson or NB distribution and (ii) a "zero component" modelling the excess in zero counts.

As such, the five different full models were fitted applying these four 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 five full models were fitted applying the selected distribution. Again based on the resulting AIC matrix, the most likely covariate combination was chosen. When the best-fitting model did not contain the OWF factor, it was added to the model afterwards in order to estimate its effect. Next, the selected model was also applied to the "OWF footprint + 0.5 km" and "buffer 0.5-3 km" datasets.

In the results section, we often refer to (i) the OWF coefficient, being the model coefficient of 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 factor combination, with the offset variable set to 1 km². Note that the OWF coefficient is always reported in its untransformed form, and

that it is actually a factorial term. A coefficient of 0 for example is transformed by taking the exponential function *e* to the power 0, which equals 1, meaning no effect. On the other hand, a coefficient of 1 is transformed by doing *e* to the power 1, equalling 2.718, implying that numbers inside the OWF area are almost three times higher compared to the control area.

2.3. Behavioural study of large gulls inside the offshore wind farm

2.3.1. Observations of turbine-associated birds during transect counts

During the seabird monitoring tracks through the OWF at the Thornton Bank (fig. 2) we carefully checked each adjacent turbine foundation on the presence of birds. Ever since September 2014, we also registered the turbine number of all counted turbines, resulting in turbine-specific information on the presence of birds on 13 monitoring days, totaling 487 records. When the full monitoring route was sailed, 43 turbines could be counted reliably. Due the circumstantial situations – mostly adverse weather conditions –, the monitoring route as displayed in figure 2 sometimes needed to be cut off, explaining the lower number of counted turbines on 6 out of 13 occasions (table 2).

After selecting the best-fitting distribution based on an information theoretic criterion (AIC), we applied a mixed modelling strategy (including random effects *date* and *turbine*) to test the effect of *distance to edge* (fixed effect) on the numbers of birds associated with the turbines (response variable).

2.3.2. Tracking data of lesser black-backed gull

Between 2013 and 2016, 112 lesser black-backed gulls breeding at Zeebrugge (Belgium) and Vlissingen (the Netherlands) have been equipped with a UvA-BiTS tracker (Bouten *et al.* 2013). Some of these birds visited the Thornton Bank OWF, allowing

Table 2. Count effort regarding turbine-specific information on the presence of birds

Date	Number of turbines
09/09/2014	43
29/10/2014	36
18/11/2014	43
16/12/2014	16
27/01/2015	34
22/04/2015	43
25/09/2015	39
21/01/2016	43
16/02/2016	43
17/03/2016	43
30/09/2016	39
14/12/2016	43
24/03/2017	22
Total	487

a characterization of their behaviour in and around this specific OWF. In a first and explorative analysis, we focused on their association with the turbine foundations, the proportion between flying versus resting in and around the OWF and diurnal patterns in their presence and behaviour. As the resolution of the recorded tracks varied strongly from 10 to 3600 seconds, we selected one data point per hour in all calculations except when assessing the actual time spent in a certain area. This way we avoided a higher weight of birds tracked at higher resolutions and also avoided temporal correlation between records (Ross-Smith *et al.* 2016).

2.3.3. Fixed camera

A fixed camera (AXIS Q6044-S) located at one of the jacket foundations in the Thornton Bank OWF (turbine I5) allowed to count and observe gulls associated with the turbine foundations within the viewing and/or zooming range of the camera. The view is limited to one side of the jacket foundation

of turbine I5, but in good weather conditions it was also possible to assess the presence of gulls on turbines I4 and J2. As such, we have performed 349 counts since January 2017, allowing to look for tidal and diurnal patterns in the gulls' presence and behaviour. Current efforts will be sustained at least throughout 2017, and the first data analysis results will be reported in the 2018 monitoring report. Below, however, we do already report on the numbers and species observed up until now, and we further show some tentative graphs of tidal and diurnal patterns.

2.4. Statistics

All data handling and modelling was performed in R.3.3.3 (R Core Team 2017), making use of the following packages:

- RODBC (Ripley & Lapsley 2016);
- foreign (R Core Team 2016);
- date (Therneau *et al.* 2017);
- ggplot2 (Wickham 2009);
- compare (Murrell 2015);
- reshape (Wickham 2007);
- plyr (Wickham 2011);
- MASS (Venables & Ripley 2002);
- mgcv (Wood 2011);
- pscl (Jackman 2015);
- glmmADMB (Skaug *et al.* 2016);
- distance (Miller 2016);
- mrds (Laake *et al.* 2016);
- rgdal (Bivand *et al.* 2016);
- data.table (Dowle & Srinivasan 2017);
- rgeos (Bivand & Rundel 2017);
- sp (Pebesma & Bivand 2005);
- spatialEco (Evans 2016).

3. Results

3.1. General observations

Since the Thornton Bank OWF became operational, most of the birds observed inside the OWF footprint area were gulls (92% of all non-passerine birds – see table 3). Most of these belong to one of the three "large gull" species, *i.e.*, herring, lesser black-backed and great black-backed gull. With over 1000 individuals observed, great black-backed gull was by far the most numerous species of all. Great black-backed gull also showed a much higher preference to the turbine foundations compared to the other two large gull species (79% versus 21% and 36% for lesser black-backed and herring gull, respectively). Cormorants too showed a clear preference to the turbines, as 89% of the great cormorants and 79% of the European shags were observed roosting on the jacket foundations.

Table 3. Number of birds and sea mammals observed inside the Thornton Bank (626 km of surveying)

		Total	Number present on turbines	Percentage present on turbines
BIRDS				
Northern fulmar	*Fulmarus glacialis*	1	0	
Northern gannet	*Morus bassanus*	42	0	
Great cormorant	*Phalacrocorax carbo*	53	47	89%
European shag	*Phalacrocorax aristotelis*	14	11	79%
Unidentified cormorant	*Phalacrocorax sp.*	3	1	33%
Eurasian sparrowhawk	*Accipiter nisus*	1	0	
Bar-tailed godwit	*Limosa lapponica*	1	0	
Arctic skua	*Stercorarius parasiticus*	1	0	
Little gull	*Hydrocoloeus minutus*	10	0	
Black-headed gull	*Chroicocephalus ridibundus*	16	0	
Common gull	*Larus canus*	122	3	2%
Lesser black-backed gull	*Larus fuscus*	622	131	21%
Herring gull	*Larus argentatus*	109	39	36%
Great black-backed gull	*Larus marinus*	1033	817	79%
Unidentified large gull		551	418	76%
Black-legged kittiwake	*Rissa tridactyla*	255	1	0%
Sandwich tern	*Sterna sandvicensis*	17	0	
Common tern	*Sterna hirundo*	1	0	
Common guillemot	*Uria aalge*	69	0	
Unidentified auk	*Alca torda* or *Uria aalge*	14	0	
Razorbill	*Alca torda*	32	0	
Domestic pigeon	*Columba livia "domestica"*	1	0	
Common starling	*Sturnus vulgaris*	122	3	2%
other passerines		31	4	13%
SEA MAMMALS				
Harbour porpoise	*Phocoena phocoena*	4	0	
Grey seal	*Halichoerus grypus*	1	0	

Table 4. Results of the multi-covariate distance analysis

Species	Detection function	Covariates	Detection probability
Northern fulmar	Hazard-rate	log(group size) + wave height	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	0.65
Common gull	Hazard-rate	log(group size) + wind force	0.52
Lesser black-backed gull	Hazard-rate	log(group size) + wind force	0.68
Herring gull	Hazard-rate	log(group size) + wind force	0.66
Great black-backed gull	Hazard-rate	log(group size) + wind force	0.73
Black-legged kittiwake	Hazard-rate	log(group size) + wave height	0.57
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	0.57
Razorbill	Hazard-rate	log(group size) + wind force	0.64

Despite the reported avoidance of OWFs by gannets and auks, these birds did regularly enter the OWF footprint area. As such, we observed 42 northern gannets, 69 common guillemots and 32 razorbills.

3.2. Distance analysis

For all species except for great skua, hazard-rate detection models fitted our data better than half-normal detection functions (table 4). In general, either wave height or wind force proved to affect the detectability of seabirds significantly, except for great skua and both terns. The natural logarithm of group size was retained for all species except for northern gannet and great skua, while for common guillemot group size was preferred over the logarithm of 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.

3.3. BACI modelling results

3.3.1. Northern fulmar

During the operational phase of the Thornton Bank OWF, numbers of northern fulmar were low both in the control area and impact area, in line with an overall decrease in densities as observed in the BPNS. Within the "OWF footprint + 0.5 km" area no birds were observed at all, explaining the empty space in figure 4 and the extreme values in table 5 (a strongly negative OWF coefficient of -23.08 opposed to a high p-value of 0.999). In both the "OWF footprint + 3 km" and "buffer 0.5-3 km" areas, the OWF coefficients were strongly negative (-2.13 and -1.52), yet neither one was proved significantly different from zero. In conclusion, despite indications of avoidance, no significant effect of the Thornton Bank OWF on the numbers of northern fulmar could be found.

3.3.2. Northern gannet

Northern gannets showed clear avoidance of the OWF at the Thornton Bank, and compared to the control area and the period before impact, numbers dropped by 97% in the "OWF footprint + 0.5 km" area, by 70% in the "OWF footprint + 3 km" area and by 53% in the "buffer 0.5-3 km" area. All three OWF coefficients proved statistically significant (P < 0.05, see table 5). These results confirm earlier results from the Thornton Bank and the strong decrease in densities of 82% found at the Bligh BankOWF (Vanermen *et al.* 2016).

3.3.3. Great skua

As for northern fulmar, no great skuas were observed inside the "OWF footprint + 0.5 km" area after impact, hampering meaningful statistics and explaining the empty space in the left panel of figure 6. For the "OWF footprint + 3 km area", the OWF coefficient was close to zero (illustrated by the highly parallel BACI graph in the right panel of figure 6), while it was slightly positive (0.62) yet not significantly different from zero for the "buffer 0.5-3 km" area (P = 0.525). In conclusion, there was no apparent effect of the Thornton Bank OWF on great skua numbers.

3.3.4. Little gull

As already reported in Vanermen *et al.* (2016), little gull showed a distinct pattern of avoidance of the OWF footprint area as opposed to increased numbers in the surrounding buffer zone. Compared to the control area and the period before impact, little gulls significantly decreased in numbers by 89% in the "OWF footprint + 0.5 km" area (OWF coefficient = -2.22, P = 0.006), and showed a (non-significant) increase in numbers in the "buffer 0.5-3 km" area (OWF coefficient = 1.02; P = 0.088).

3.3.5. Common gull

Between the reference and impact period, numbers of common gull strongly increased in the study area as a whole. This increase, however, is less prominent in the wind farm area and its immediate surroundings resulting in quite strongly negative OWF coefficients (ranging between -0.81 and -1.30) for all three data selections. As none of these significantly differed from zero, we conclude that there was no apparent effect of the Thornton Bank OWF on the presence of common gull.

3.3.6. Lesser black-backed gull

The OWF coefficients found for lesser black-backed gull were all close to zero, also when taking in account birds roosting on the turbine foundations (*i.e.*, model results based on the adjusted response variable). As opposed to the strong attraction effect reported at the Bligh Bank OWF (Vanermen *et al.* 2015; 2016), there were no signs of attraction of lesser black-backed gulls to the Thornton Bank OWF area.

3.3.7. Herring gull

The updated results for herring gull differ from the results in the previous monitoring report (Vanermen *et al.* 2016). While earlier no post-construction change in numbers was observed in the OWF, we now found 2.9 times higher numbers in the "OWF footprint + 0.5 km" area compared to the control area and the period before impact. This estimated increase applies to data including birds roosting on the turbines and the corresponding coefficient was found borderline significant (OWF coefficient = 1.06; P = 0.050). The model results for the data in- and excluding turbine-associated birds, however, were highly comparable. In contrast, but meanwhile similar to the result reported by Vanermen *et al.* (2016), we observed significantly lower numbers in the buffer zone (OWF coefficient = -1.88; P = 0.008).

3.3.8. Great black-backed gull

We found significant attraction of great black-backed gull towards the Thornton Bank OWF, provided we include birds roosting on the turbines. This was not unexpected considering the high numbers observed in the area and the high percentage associated with the turbines (table 3). For the "OWF footprint + 0.5 km" area the OWF coefficient equaled 1.88, implying a significant increase in numbers with a factor 6.6 compared to the control area and the period before impact (P < 0.001). In the "buffer 0.5-3 km" area, the OWF coefficient approached zero while the result for the "OWF footprint + 3 km" area was intermediate between the footprint and buffer area results.

3.3.9. Black-legged kittiwake

Post-construction numbers of black-legged kittiwake in the impact area appeared to be significantly lower compared to the period before impact, as opposed to a stable trend in the control area. In the "OWF footprint + 0.5 km" area numbers

significantly decreased by no less than 75% (OWF coefficient = -1.39; P = 0.009), and decreased by 51% in the "buffer 0.5-3 km" area, the latter coefficient no longer being significantly different from zero (OWF coefficient = -0.72; P = 0.123).

3.3.10. Sandwich tern

Generally, we used year-round data for modelling, but due to fitting problems, we only used Sandwich tern data collected from March till September, while no longer considering seasonal variation. In doing so, Sandwich terns showed a less marked decrease in numbers in the impact area compared to the control area, resulting in positive OWF coefficients for all three data selections. For the buffer zone only, the effect was significant (OWF coefficient = 1.74; P = 0.018). Despite this statistical significance, results should be interpreted with care considering the low number of positive observations after impact. On the other hand, this result is in line with the attraction of Sandwich terns to the 3 km buffer zone around the phase I Thornton Bank OWF (Vanermen *et al.* 2013), when only six turbines were present (OWF coefficient = 2.46; P = 0.001).

3.3.11. Common guillemot

With a negative OWF coefficient of -1.16 (P = 0.001), common guillemots significantly avoided the "OWF footprint + 0.5 km" area. In the buffer zone too numbers decreased, but the latter change was no longer significant (OWF coefficient = -0.33; P = 0.252). Back-transforming the coefficient of -1.16, the corresponding decrease of 69% as found for the Thornton Bank is highly comparable to the 75% decrease reported for the Bligh Bank (Vanermen *et al.* 2016).

3.3.12. Razorbill

The models for razorbill estimated a negative OWF coefficient for the "OWF footprint + 0.5 km" area, a positive coefficient for the buffer area and an intermediate result
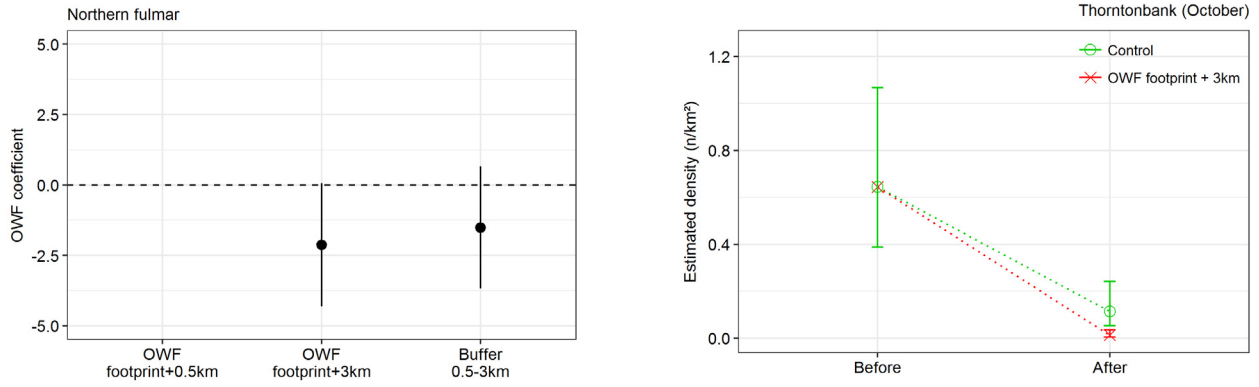
Figure 4. Modelling results for northern fulmar in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right.
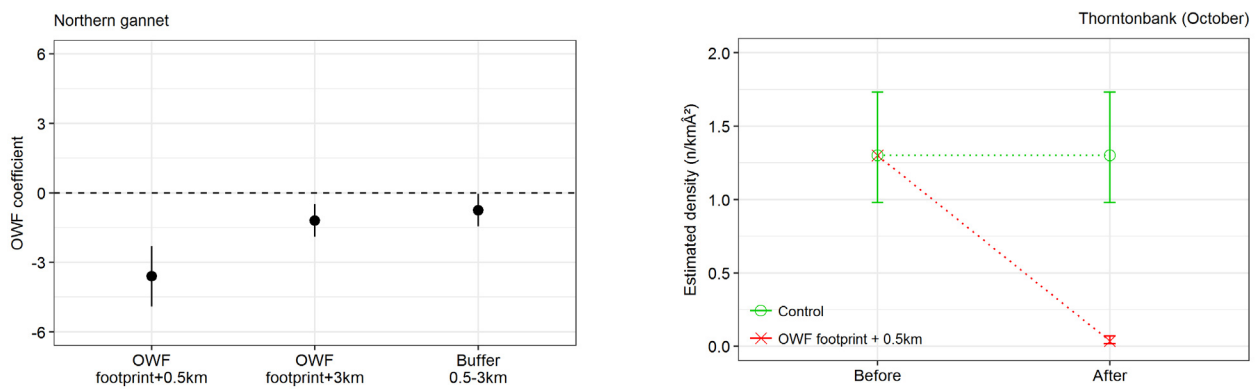

Figure 5. Modelling results for northern gannet in the Thornton Bank study area with OWF coefficients and their 95% CI's on the left and BACI density estimates for the month with maximum numbers on the right.
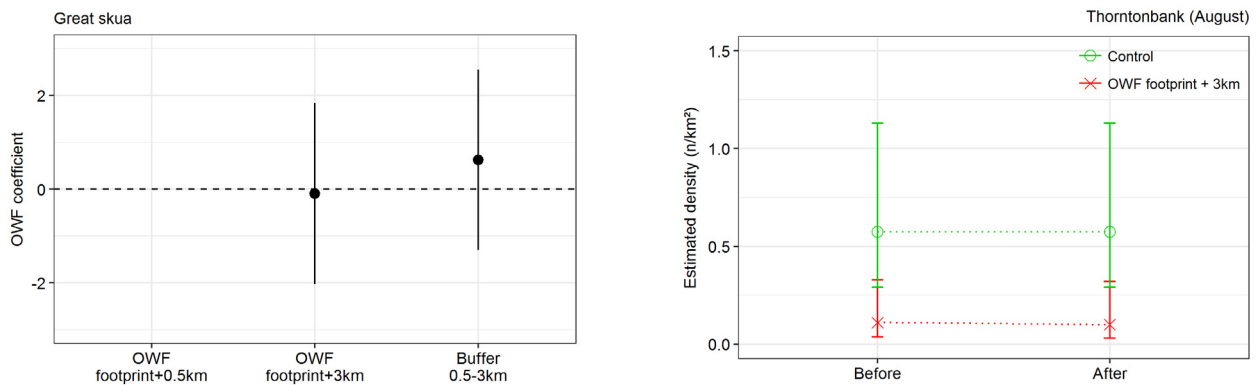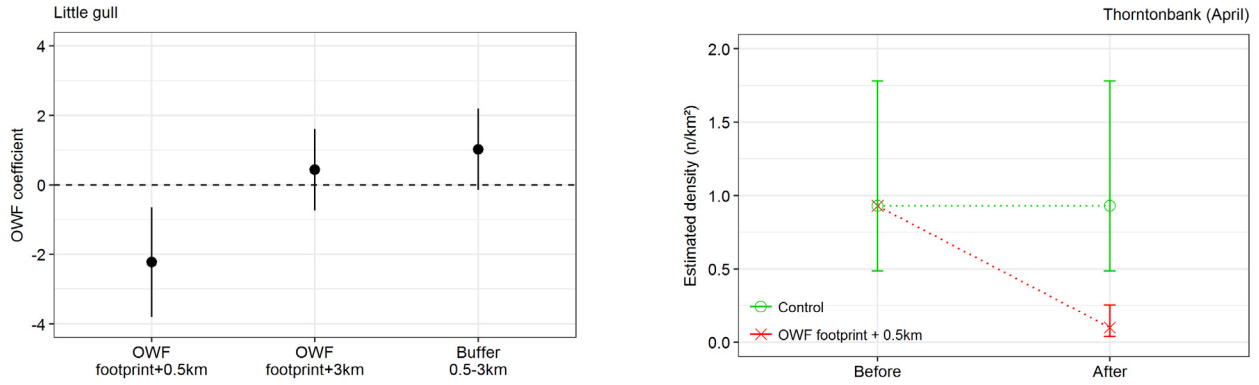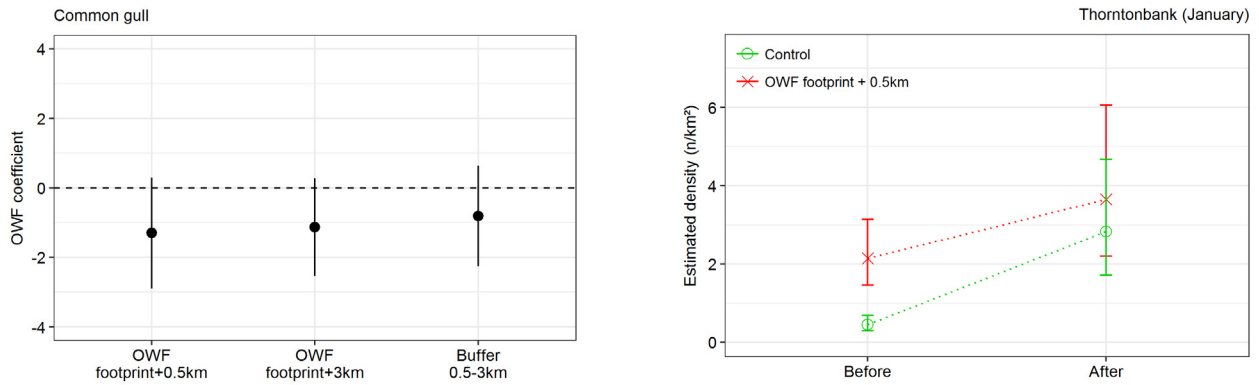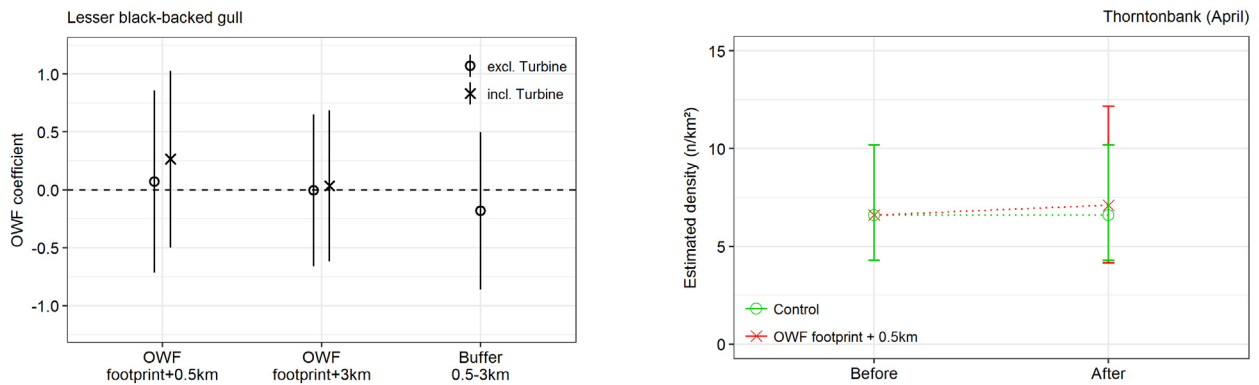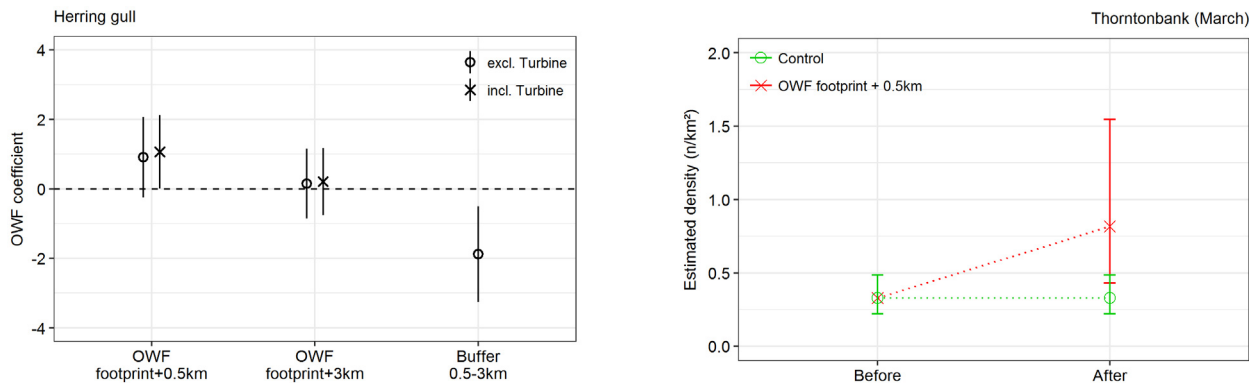

Figure 6. Modelling results for great skua in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the monthwith maximum numbers on the right (but note a zero-inflation of 72%).

Figure 7. Modelling results for little gull in the Thornton 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.

Figure 8. Modelling results for common gull in the Thornton 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.

Figure 9. Modelling results for lesser black-backed gull in the Thornton Bank 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.
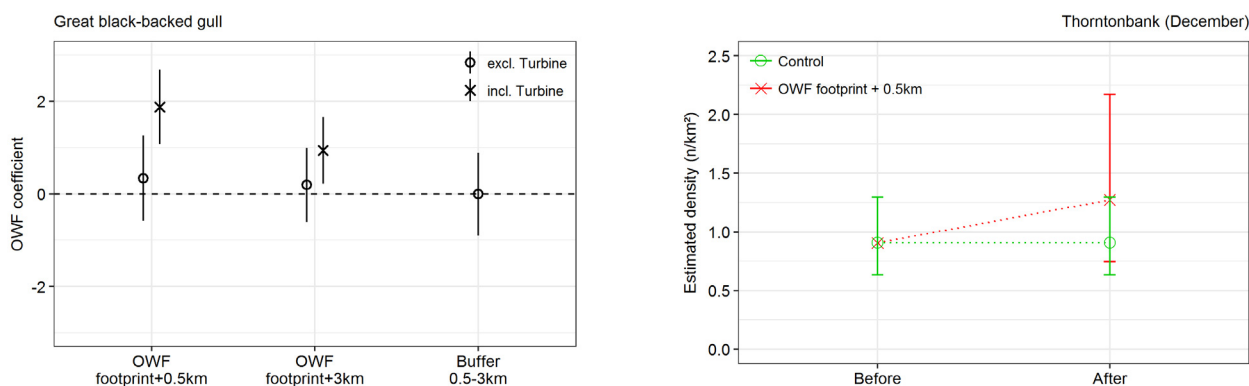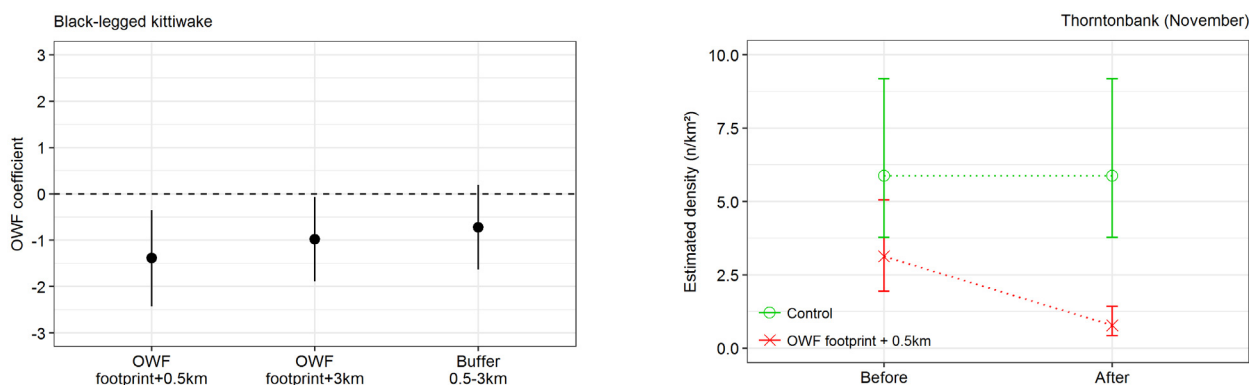
Figure 10. Modelling results for herring gull in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers (exclusive turbine-associated birds) on the right.

Figure 11. Modelling results for great black-backed gull in the Thornton Bank 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 12. Modelling results for black-legged kittiwake in the Thornton 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.
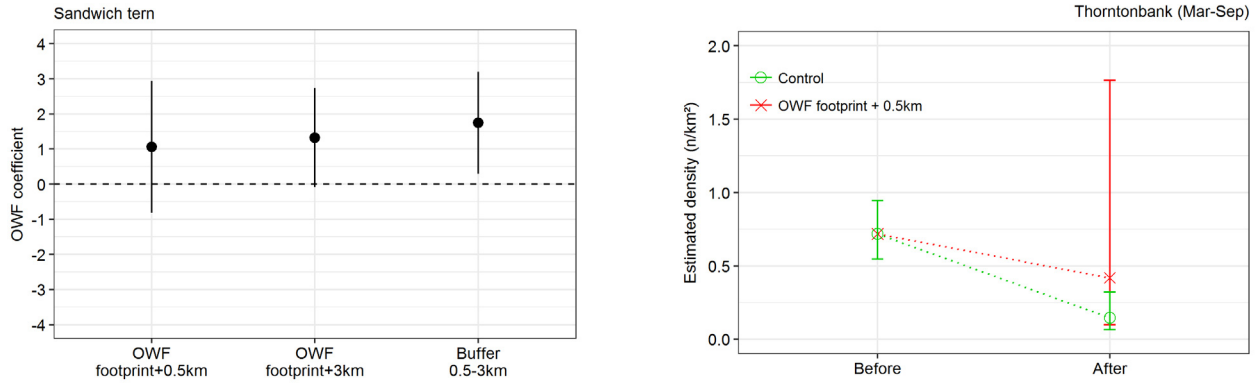
Figure 13. Modelling results for Sandwich tern in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the period March to September on the right (but note that zero-inflation equals 75%).
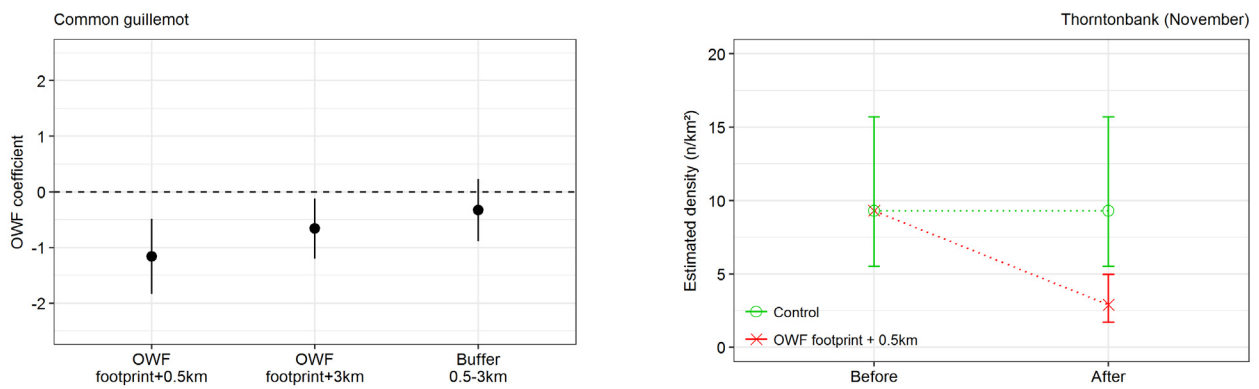

Figure 14. Modelling results for common guillemot in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right (but note that zero-inflation equals 10%).
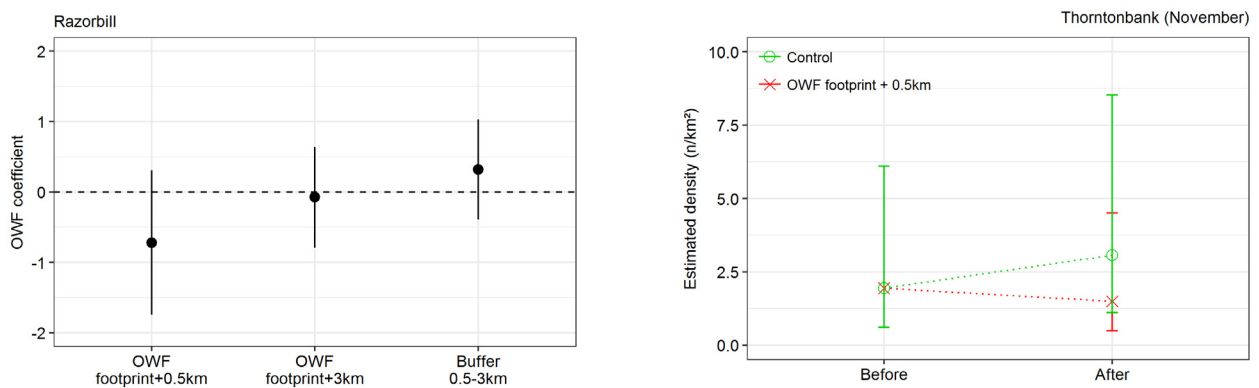

Figure 15. Modelling results for razorbill in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates for the month with maximum numbers on the right (but note that zero-inflation equals 18%).

of almost zero when both areas are analyzed together ("OWF footprint + 3 km"). None of these coefficient values, however, significantly differed from zero (P > 0.05), and therefore no apparent effect of the Thornton Bank OWF on the numbers of razorbill was observed.

3.3.13. Summarizing tables

Our BACI monitoring results are summarized in table 5, which lists all OWF coefficients and corresponding P values as estimated through the modelling process. All impact model coefficients are displayed in table 7 in the Appendix.

   After four years of post-impact monitoring at the Thornton Bank OWF, the impact area appeared to be avoided by four species, *i.e.*, northern gannet, little gull,

black-legged kittiwake and common guillemot. In the "OWF footprint + 0.5 km" area, these species dropped in numbers by no less than 97%, 89%, 75% and 69% respectively. The Thornton Bank OWF further attracted great black-backed gulls, this species having increased in numbers by a factor 6.6. Sandwich tern too appeared to be attracted to the OWF at the Thornton Bank, the effect being significant for the buffer zone only. All of these results are highly similar to the results reported last year. Only for herring gull we observed a shift in the estimated wind farm effect. While the OWF coefficient for herring gull was estimated to be close to zero after three years of monitoring, it now showed a borderline significant increase in numbers by a factor 2.9. In contrast, a significant decrease in numbers of herring gull was observed in the buffer zone.

Table 5. BACI monitoring results for the C-Power wind farm at the Thornton Bank after 4 years of operation, with indication of the displacement-related OWF model coefficients and their respective P values; model results of the adjusted response variable are indicated by "(T)" in the species column (P < 0.05*; P < 0.01**; P < 0.001***; red cells indicate significant avoidance, green cells indicate significant attraction)

	OWF footprint + 0.5 km		OWF footprint + 3 km		Buffer 0.5-3 km	
	OWF Coefficient	P-Value	OWF Coefficient	P-Value	OWF Coefficient	P-Value
Northern fulmar	-23.08	0.999	-2.13	0.057.	-1.52	0.171
Northern gannet	-3.60	0.000***	-1.19	0.001***	-0.75	0.036*
Great skua	-18.56	0.998	-0.10	0.922	0.62	0.525
Little gull	-2.22	0.006**	0.43	0.468	1.02	0.088
Common gull	-1.30	0.110	-1.13	0.117	-0.81	0.271
Lesser black-backed gull	0.07	0.857	0.00	0.989	-0.18	0.600
Lesser black-backed gull (T)	0.27	0.495	0.03	0.917		
Herring gull	0.91	0.125	0.15	0.767	-1.88	0.008**
Herring gull (T)	1.06	0.050	0.21	0.670		
Great black-backed gull	0.34	0.473	0.19	0.636	0.00	0.992
Great black-backed gull (T)	1.88	0.000***	0.94	0.011*		
Black-legged kittiwake	-1.39	0.009**	-0.98	0.035*	-0.72	0.123
Sandwich tern	1.06	0.269	1.32	0.066	1.74	0.018*
Common guillemot	-1.16	0.001***	-0.66	0.017*	-0.33	0.252
Razorbill	-0.72	0.169	-0.08	0.836	0.32	0.376

3.4. Association with turbines

3.4.1. Transect counts

We used data of 13 monitoring days during which we crossed the Thornton Bank OWF and checked the adjacent turbine foundations (n = 487) on the presence of birds. This resulted in a total number of 3 European shags, 33 great cormorants, 9 lesser black-backed gulls, 29 herring gulls, 510 great black-backed gulls and 30 unidentified large gulls. Figure 16 shows the distribution of the mean numbers per turbine of great cormorant and great black-backed gull, illustrating both species' preference to the outer turbines.

We tested the hypothesis that the number of great cormorants and great black-backed gulls associated with the turbines decreases towards the center of the OW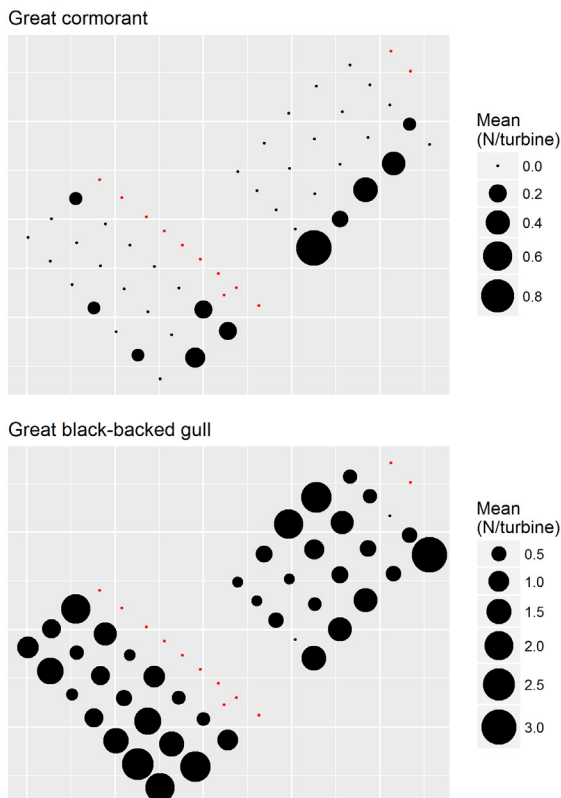F through a mixed model with *distance to edge* as a fixed effect, and *date* and *turbine* as random effects. For great cormorant a negative binomial distribution model was selected, and *distance to edge* did negatively affect the number of birds present on the turbine foundations (P = 0.012). For great black-backed gull too we selected a negative binomial distribution and again *distance to edge* proved significant (P < 0.001). Model predictions are illustrated in figure 17.
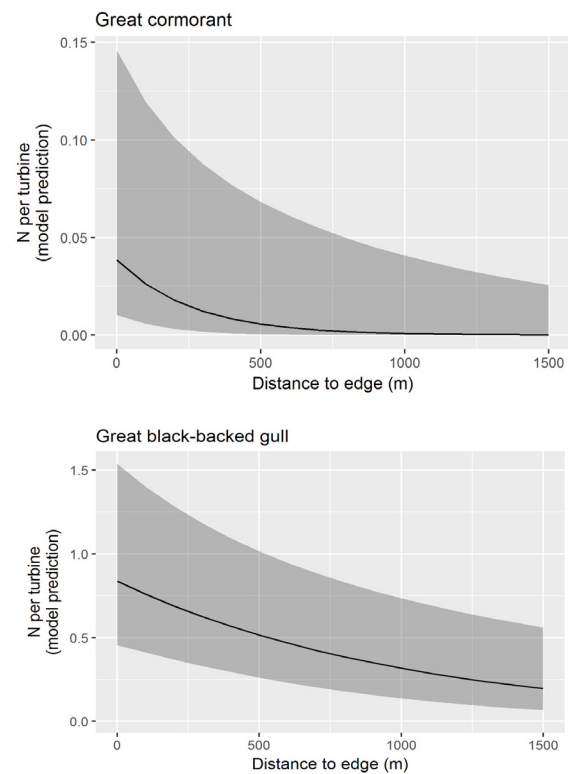

Figure 17. Model predictions of the numbers of great cormorant and great black-backed gull present on the turbine foundations in relation to *distance to edge* at the Thornton Bank OWF.

3.4.2. Tracking data

In order to assess potential attraction of lesser black-backed gulls towards the jacket foundations in the Thornton Bank OWF, track log positions were overlaid with 100 m buffer areas around the turbines. Out of a total of 41 individual birds logged inside the Thornton Bank OWF boundaries, 20 individuals were recorded at least once inside these 100 m buffer areas.

Figure 16. Mean number of great cormorant and great black-backed gull present per turbine during 13 seabird monitoring days through the Thornton Bank OWF (turbines coloured red were not counted).

Exploring the characteristics of the selected logs, most (96%) referred to *non-flying* birds (*i.e.*, logs with a speed below 4 m/s) located at a mean height of 17 m above sea level, and were therefore considered to be resting on the jacket foundations. The fact that tracked lesser black-backed gulls were often resting on the turbine foundations is also nicely illustrated when comparing the histograms of the logged altitudes of *non-flying* birds in the Thornton Bank control versus footprint area (see fig. 18). While the histogram centres around zero for *non-flying* birds logged in the control area (*i.e.*, swimming birds), there are two peaks of logged altitudes in the "OWF footprint + 0.5 km" area: one around zero, and one at about 20 m above sea level.

Next, we calculated the total time spent in (i) the OWF as a whole and (ii) the turbine buffer areas by summing the time intervals between the first and last log of each visit to the respective areas. This implies that single "isolated" logs were not taken into calculation, but also that we assume that birds stay within the area boundaries between two subsequent logs inside these boundaries. As such, lesser black-backed gulls appeared to spend 51% of their time inside the Thornton Bank OWF resting on the jacket foundations. When using the selection of one log per hour (see methods section) and calculating the proportion of the number of logs within the turbine buffer areas versus the total number of logs inside the OWF, we obtained a very similar result of 49%. Considering the huge difference in surface between the OWF footprint area and the turbine buffer areas, we can safely conclude that the tracked lesser black-backed gulls showed a high preference towards the turbine foundations.

Figure 19 illustrates the total time spent per turbine. As in the previous paragraph, we tested the hypothesis that birds prefer the outer turbines. Based on a negative binomial model, however, *distance to edge* did not significantly affect the time spent on the turbines ($P = 0.249$).
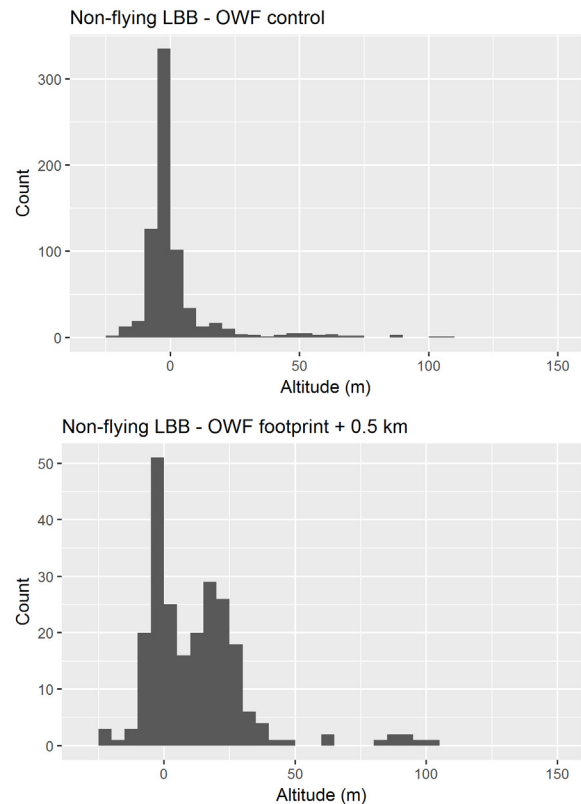

Figure 18. Distribution of logged altitudes of tracked lesser black-backed gulls in the Thornton Bank control versus footprint area (see also fig. 3).
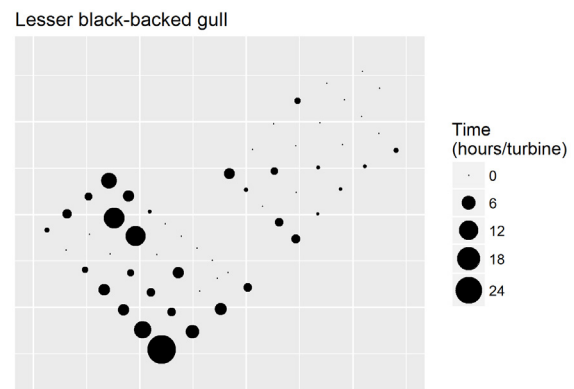

Figure 19. Time spent per turbine by lesser black-backed gulls tracked inside the Thornton Bank OWF.

3.5. Activity patterns in- versus outside the Thornton Bank OWF (tracking data)

In total, 41 tracked individuals were logged inside the Thornton Bank OWF boundaries, with the number of logs varying from only 1 for gulls *Annelies & Imme* to 440 for gull

Romelo. Apart from the actual time spent inside the OWF, the number of logs strongly depended on the logging resolution, the latter varying from 10 to 3600 seconds. As already mentioned in the methods section, we therefore selected one log per hour for all calculations in the paragraph below.

Birds were classified as *flying* when having a calculated speed of over 4 m/s. Resulting, 44% of the logs in the BPNS were identified as *flying*, opposed to a much lower 19% in the Thornton Bank study area. Within the study area itself there was less difference in the proportion of birds flying, with 20% and 15% flying in the control and impact area respectively (fig. 20). Hence, despite the rather small difference, lesser black-backed gulls appeared to spend more time resting (*non-flying*) inside compared to outside the Thornton Bank OWF.

Regarding the diurnal rhythm in flying activity, the study area (including both the wind farm and control area) was also found to be markedly different from the BPNS as a whole.

At the BPNS, the presence of the tracked birds was lowest during night hours (from 9 pm to 2 am), while peaking in the early morning (4 am) and the evening (7 pm). More than 70% of the birds staying out at sea between 9 pm and 2 am were classified as *non-flying*. This percentage was about 50% during the rest of the day with a slight secondary peak in the *non-flying* proportion around noon (11 am) (fig. 21). Strikingly, this pattern of increased presence and activity in the morning and afternoon was highly consistent throughout the year (not illustrated).

In contrast, presence in the study area was highest before midday from 6 am to 12 am, showing only one peak instead of two, while the proportion of *non-flying* birds kept a much higher level during the full diurnal cycle (mostly above 70%). As in figure 21, the *non-flying* proportion did show (much less obvious) peaks during the night
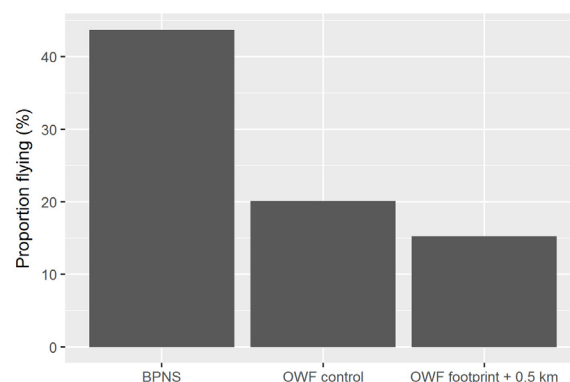

Figure 20. The proportion of GPS-logged birds flying in the BPNS as a whole on the one hand, and in the Thornton Bank OWF control and impact area on the other hand (see also fig. 3).
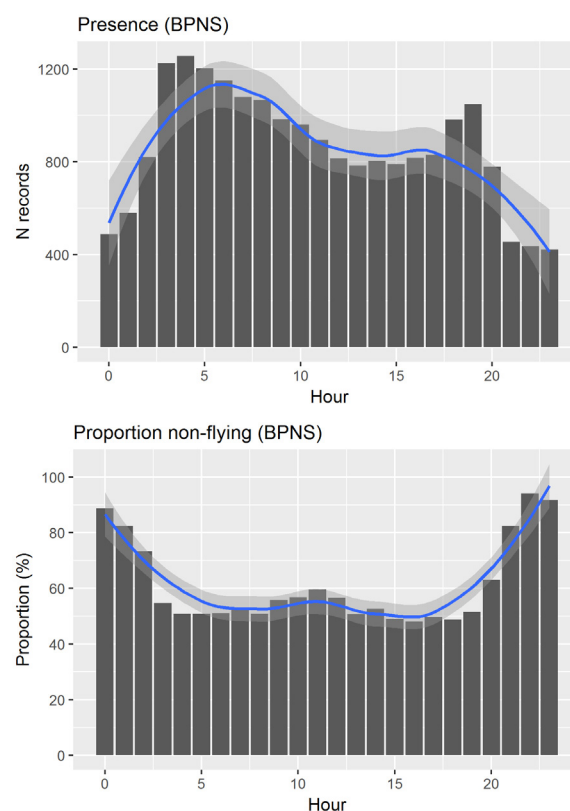

Figure 21. Diurnal pattern of the presence and *non-flying* behaviour of tracked lesser black-backed gulls in the BPNS.

and around midday. Patterns in the control and impact area appeared very much alike (figs 22-23).

While the Thornton Bank study area is on the boundary of the species' offshore distribution, it appears that the diurnal pat
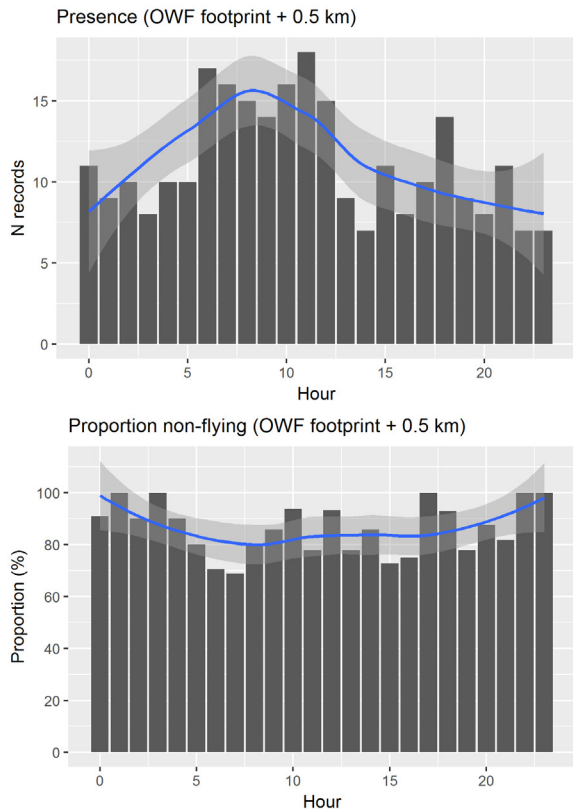
105

Figure 22. Diurnal pattern of the presence and *non-flying* behaviour of tracked lesser black-backed gulls in the Thornton Bank "OWF footprint + 0.5 km" area.
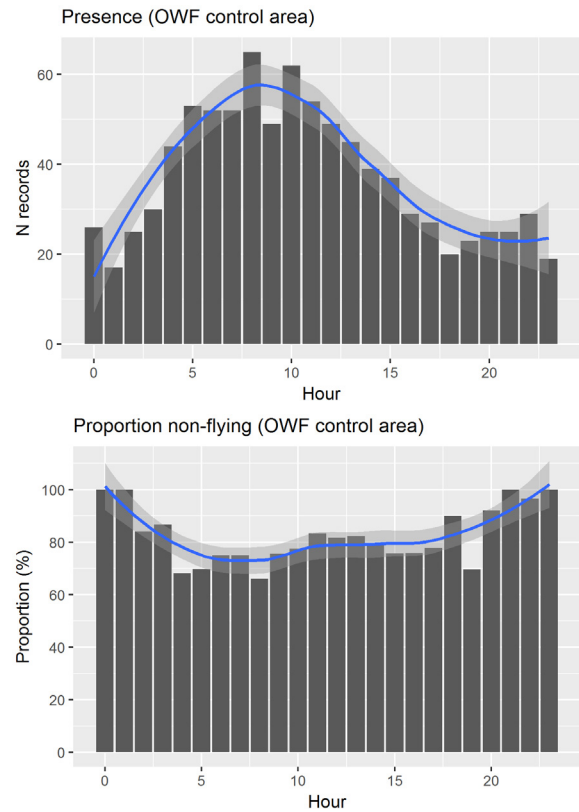

Figure 23. Diurnal pattern of the presence and *non-flying* behaviour of tracked lesser black-backed gulls in the Thornton Bank OWF control area.

tern and high level of flying activity at the BPNS as a whole is partly determined by commuting flights between land and offshore foraging areas. The early morning peak in flying activity at the BPNS (fig. 21, right panel) for example is followed by increased presence before noon in the Thornton Bank study area. The evening peak in flying activity on the other hand is not followed by increased presence in the study area, suggesting that the evening activity of lesser black-backed gulls reaches less far out at sea.

As calculated in §3.4.2, about 50% of the birds inside the OWF at the Thornton Bank concentrate around the turbines. But while we expected this proportion to be higher during the night, the opposite seems true. During midnight, less than 30% of their time
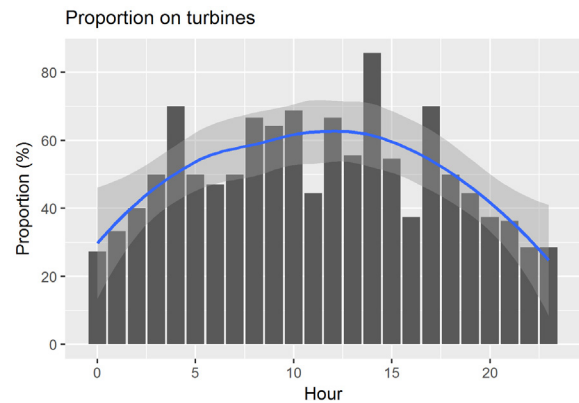

Figure 24. Diurnal pattern of the proportion of birds present on the turbines in the Thornton Bank OWF.

is spent on the turbines, while this proportion was about 60% during the day. Apparently, during the night, lesser black-backed gulls feel safer on the water than on the turbines.

3.6. Fixed camera

From January until the beginning of May 2017 we performed 349 counts of birds associated with turbine I5, on the side of which the fixed camera is installed. Neighboring turbines I4 and J2 were counted 235 and 212 times respectively. Count results are shown in table 6. Note that turbine I5 is only partly visible, and so numbers are not representative for the turbine as a whole.

Based on the counts of I4 and J2, the mean number of large gulls per turbine was 0.98. This is comparable with the mean number of 1.21 gulls per turbine as assessed during the transect counts. The proportion between species on the other hand is strikingly different from the proportion observed during transect counts. While on I5, herring gull made up for 34% of all large gulls, this proportion was only 5% during transect counts. We should note that the transect count results account for the OWF as a whole and were performed on a relatively limited number of (year-round) occasions. In contrast, counts with the fixed camera were performed during the period January to April of this year only and had only very limited spatial coverage.

Out of the 180 large gulls observed on turbine I5, 20 birds were actively foraging on the lower reaches of the jacket foundations (11.1%) (see fig. 25). These were mostly herring gulls (15 birds), as opposed

Table 6. Number of species counted per turbine as observed with the fixed camera

	I5	I4	J2
Great cormorant	0	1	0
European shag	1	0	0
Unidentified cormorant	0	1	5
Common gull	1	0	0
Lesser black-backed gull	3	0	0
Herring gull	62	0	0
Great black-backed gull	96	3	3
Unidentified large gull	19	161	272

to only 3 great black-backed gulls and 2 unidentified large gulls. Birds always seemed to feed on mussels growing on the lower intertidal zone of the jacket foundations. At turbines I4 and J2, we counted 36 birds foraging on the intertidal zone of the jacket foundations, which makes 8.2% of the total number of large gulls present.

Below we show some preliminary graphs of the mean numbers of large gulls associated with the observed turbines in relation to wind, tide and time of day.

Figure 25. Large gulls foraging on the lower intertidal reaches of the turbine I5 jacket foundation.
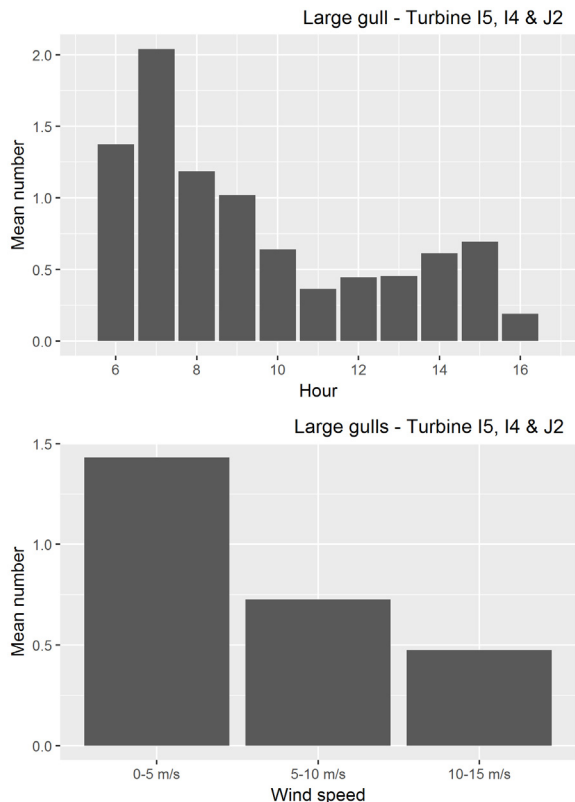
Figure 26. Mean number of large gulls present on the turbines I4, I5 and J2 in relation to time of day and to wind speed.

Figure 27. Mean number of large gulls present (panel at the top) and foraging (panel below) on the turbines I4, I5 and J2 in relation to tidal height.

In coming reports, we will do the same analyses for each large gull species separately, but not before we have collected at least one cycle of year-round data.

Numbers of gulls associated with the jacket foundations seemed to peak early morning at 7 am, with a slight secondary peak at 3 pm. As expected, gull presence was negatively correlated with mean wind speed, and by far the highest numbers were observed on calm days with wind speeds below 5 m/s (fig. 26).

In relation to tidal height, numbers clearly peaked during the lowest tidal height category (< 0 cm above TAW) (fig. 27). Doing the same for foraging gulls only, we see highly increased numbers below 100 cm above TAW, and numbers dropping to zero for tidal heights higher than 300 cm above TAW (fig. 28).
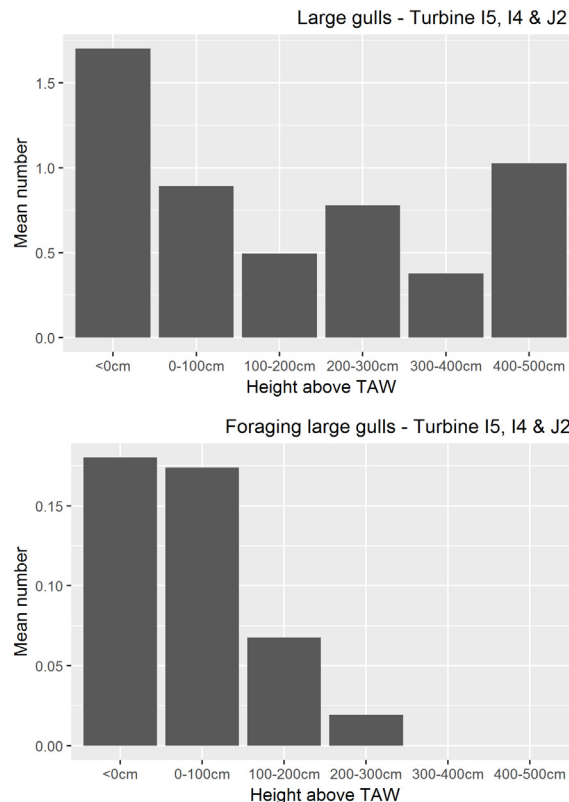
4. Conclusions

After four years of post-impact monitoring at the Thornton Bank OWF, the impact area appeared to be avoided by four species, being northern gannet, little gull, black-legged kittiwake and common guillemot. In the OWF footprint area, these species dropped in numbers by no less than 97%, 89%, 75% and 69% respectively. Not unexpectedly, considering the rather small amount of data added in the course of the monitoring year 2016, these results are highly similar to those reported in the latest monitoring report (Vanermen *et al.* 2016). At the Bligh Bank, we also observed a significant decrease in numbers of northern gannet and common guillemot, while for the latter site, results for little gull and black-legged kittiwake remained inconclusive.

The Thornton Bank OWF attracted great black-backed gulls, this species having increased in numbers by a factor 6.6. Sandwich tern too appeared to be attracted to the OWF at the Thornton Bank, this effect being significant for the buffer zone only. Again, these results are highly similar to the results reported last year, but for herring gull there was in fact a shift in the estimated wind farm effect. While the OWF coefficient for herring gull was estimated to be close to zero after three years of monitoring, it now showed a borderline significant increase in numbers by a factor 2.9. On the other hand, a significant decrease in numbers of herring gull was observed in the buffer zone.

The reported attraction of large gulls to OWFs has raised concern on the number of expected collision victims, and considering the upcoming large scale exploitation of offshore wind in the North Sea, collision mortality might even affect these species on a population level (Brabant *et al.* 2015). Up until now, however, there is little information on the behaviour of large gulls inside OWF areas, and it remains unclear whether these birds visit the wind farms because of enhanced foraging conditions or simply for roosting. Gaining more insight in this matter, however, is considered crucial for a reliable collision risk assessment. At the Thornton Bank OWF, roosting possibilities are particularly numerous as 48 out of 54 turbines are built on jacket foundations which offer easy access to the intertidal fouling communities during low tide. In order to unravel part of the remaining knowledge gaps, we started studying the occurrence and behaviour of large gull species in the Thornton Bank wind farm area using (i) the results of our dedicated ship-based seabird counts, (ii) GPS tracking data and (iii) observational data through a fixed camera installed on one of the turbines.

While the limited number of data collected up until now does not allow to draw any definite conclusions, first results showed that the time spent resting was higher inside compared to outside the wind farm. Based on our transect count data, almost 80% of the great black-backed gulls observed inside the OWF were associated with the turbine foundations. Tracking data of lesser black-backed gulls showed that birds entering the OWF spend about 50% of their time roosting on the jacket foundations. Great black-backed gulls further seemed to prefer the outer turbines, suggesting a partial barrier effect. Turbine foundations were mainly used for roosting, but during a short time period around low tide, small numbers of birds were observed foraging on mussels growing on the lower reaches of the foundations. In total, 9% of the large gulls observed on the jacket foundations within viewing range of the fixed camera were actually foraging. Herring gull in particular seemed to favour this temporary but daily available food source.

The results of our behavioural study might shed new light on the currently expected collision risk to large gulls at OWFs, and may highlight the need for proper post-construction monitoring. Pre-construction studies for example tend to extrapolate past and/or current numbers and behaviour to feed collision risk models. But next to a possible post-construction change in numbers, any behavioural shift (i.e., a decrease in time flying) too will have a strong effect on the anticipated collision mortality among large gulls.

Acknowledgements

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Appendix

Table 7. Impact model coefficients for all species studied at the Thornton 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	OWF footprint + 0.5 km	-1.40	-0.84	0.50					-1.79			-23.08	0.08	
	OWF footprint + 3 km	-1.37	-1.00	0.14					-1.73			-2.13	0.08	
	Buffer 0.5-3 km	-1.37	-1.00	0.14					-1.72			-1.52	0.08	
Northern gannet	OWF footprint + 0.5 km	-0.48			s(month)							-3.60	0.29	
	OWF footprint + 3 km	-0.55			s(month)							-1.19	0.30	
	Buffer 0.5-3 km	-0.55			s(month)							-0.75	0.30	
Great skua	OWF footprint + 0.5 km	-2.94	-2.03	-0.06			0.38	0.88		-1.91		-18.56		0.68
	OWF footprint + 3 km	-2.77	-1.76	0.00			0.54	0.70		-1.65		-0.10		0.72
	Buffer 0.5-3 km	-2.78	-1.78	0.00			0.56	0.69		-1.64		0.62		0.72
Little gull	OWF footprint + 0.5 km	-2.22			s(month)							-2.22	0.12	
	OWF footprint + 3 km	-2.44			s(month)							0.43	0.12	
	Buffer 0.5-3 km	-2.45			s(month)							1.02	0.12	
Common gull	OWF footprint + 0.5 km	-3.94	2.19	2.36					1.84	1.56		-1.30	0.24	
	OWF footprint + 3 km	-3.87	2.14	2.29					1.63	1.55		-1.13	0.27	
	Buffer 0.5-3 km	-3.86	2.09	2.32					1.61	1.51		-0.81	0.26	
Lesser black-backed gull	OWF footprint + 0.5 km	-0.37			s(month)						0.74	0.07	0.29	
	OWF footprint + 0.5 km (T)	-0.32			s(month)						0.59	0.00	0.32	
	OWF footprint + 3 km	-0.33			s(month)						0.48	-0.18	0.31	
	OWF footprint + 3 km (T)	-0.37			s(month)						0.73	0.27	0.30	
	Buffer 0.5-3 km	-0.33			s(month)						0.60	0.03	0.32	

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	OWF footprint + 0.5 km	-2.32	1.21	0.06							0.75	0.91	0.13	
	OWF footprint + 0.5 km (T)	-2.35	1.14	0.14							0.77	0.15	0.15	
	OWF footprint + 3 km	-2.55	1.48	0.19							1.37	-1.88	0.16	
	OWF footprint + 3 km (T)	-2.33	1.22	0.05							0.79	1.06	0.16	
	Buffer 0.5-3 km	-2.35	1.15	0.11							0.82	0.21	0.16	
Great black-backed gull	OWF footprint + 0.5 km	-1.73			s(month)						1.58	0.34	0.22	
	OWF footprint + 0.5 km (T)	-1.92			s(month)						1.65	0.19	0.25	
	OWF footprint + 3 km	-1.92			s(month)						1.65	0.00	0.21	
	OWF footprint + 3 km (T)	-1.62			s(month)						1.65	1.88	0.27	
	Buffer 0.5-3 km	-1.71			s(month)						1.64	0.94	0.28	
Black-legged kittiwake	OWF footprint + 0.5 km	-0.40			s(month)					-0.63	1.07	-1.39	0.25	
	OWF footprint + 3 km	-0.60			s(month)					-0.67	1.36	-0.98	0.27	
	Buffer 0.5-3 km	-0.62			s(month)					-0.69	1.50	-0.72	0.27	
Sandwich tern	OWF footprint + 0.5 km	-0.33							-1.60			1.06	1.52	0.75
	OWF footprint + 3 km	-0.40							-1.76			1.32	1.15	0.71
	Buffer 0.5-3 km	-0.36							-1.74			1.74	1.24	0.73
Common guillemot	OWF footprint + 0.5 km	-2.71	1.43	6.43	-1.14	-1.80			0.46			-1.16	0.96	0.10
	OWF footprint + 3 km	-2.95	1.62	6.72	-1.32	-2.00						-0.66	0.93	0.11
	Buffer 0.5-3 km	-3.04	1.74	6.84	-1.41	-2.03						-0.33	0.90	0.10
Razorbill	OWF footprint + 0.5 km	-6.17	1.03	9.49	-1.08	-3.61			0.44			-0.72	0.77	0.18
	OWF footprint + 3 km	-6.30	1.28	9.76	-1.42	-3.72						-0.08	0.95	0.21
	Buffer 0.5-3 km	-6.16	1.24	9.59	-1.39	-3.69			0.42			0.32	1.02	0.23

CHAPTER 8

AUTUMN BIRD MIGRATION REGISTERED WITH VERTICAL RADAR AT THE THORNTON BANK IN THE BELGIAN PART OF THE NORTH SEA

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Abstract

The Belgian part of the North Sea (BPNS) is part of a very important seabird migration route through the Southern North Sea. Also, large numbers of non-seabirds (mainly songbirds) are known to migrate at sea. The development of offshore wind farms in the North Sea might impact these migrating birds as they can collide with the turbines, which results in an increased mortality rate.

Radar observations greatly contribute to the understanding of the spatial and temporal patterns of bird migration because of the ability to register birds continuously at a large spatial scale and at high altitudes. Using a bird radar, installed in an offshore wind farm, the objectives of this study are to determine (1) the seasonal phenology of migrating birds across the North Sea; (2) the diurnal patterns of migrating birds at sea; (3) the vertical distribution (altitude) of migrating birds and (4) the link between bird migration and meteorological conditions.

Elaborate tests have shown that the radar antenna used in this study is performing suboptimally at detecting birds at low altitude (0-150 m above sea level). This has led to the decision to replace this antenna with a conventional magnetron X-band antenna. However, some conclusions could still be drawn from our data.

The migration traffic rates (MTR, birds. km^{-1}.hr^{-1}) values show that migration at sea, as registered by the radar, was most intense during the nights of October and early November.

The observed diurnal pattern of these months is similar to the pattern measured in the Dutch part of the North Sea. Especially in October a clear peak in MTR values occurs at dusk. A second smaller peak is noticeable at dawn.

For this study period, no clear pattern with weather conditions could be revealed, although it seems that MTR values are higher if the wind was coming from the N, NE, E and SE and when wind speed was lower than 13 m/s.

The altitude profile suggests that migration at night is happening at higher altitudes compared to daytime movements. While passerines (*i.e.*, non-seabird species) tend to dominate nighttime migration, daytime

migration tends to be a mixture of seabird and non-seabird species.

1. Introduction

Twice a year, during autumn and spring, hundreds of millions of birds fly over Europe during their migration towards and from their wintering grounds. The Belgian part of the North Sea (BPNS) is part of a very important seabird migration route through the Southern North Sea. Because of its shape, this part of the North Sea acts as a migration bottleneck, concentrating birds during migration (Stienen *et al.* 2007). Also, large numbers of non-seabirds (mainly songbirds) are known to migrate at sea (Bourne 1980; Buurma 1987; Alerstam 1990; Lensink 2002). Estimates of the number of birds seasonally travelling through the Southern North Sea vary from 85 million (Lensink *et al.* 2002) up to several hundreds of millions (estimates of Helgoland mentioned in Hüppop *et al.* 2006). This songbird migrations mainly occurs along two routes: (1) between breeding grounds on the mainland of northern Europe / Scandinavia and the UK; (2) between northern Europe / Scandinavia and wintering grounds in southern Europe and Africa (Lack 1959-1963; Lensink *et al.* 2002; Krijgsveld *et al.* 2015).

Migrating birds fly at all altitudes from sea-level up to 10 km and a general phenomenon is that birds fly high with tailwind and that they fly at a lower altitude with headwind (Bruderer 1971; Buurma 1987; Lensink *et al.* 2002).

Migrating birds suffer from ever increasing human pressures (*e.g.*, increased mortality due to desertification, loss of suited stop-over places or collision with man-made structures; Erickson *et al.* 2005; Strandberg *et al.* 2009). The development of offshore wind farms in the North Sea might impact these migrating birds as they can collide with the turbines, which results in an increased mortality rate.

Both from a purely scientific and a conservation point of view, it is crucial to understand and monitor bird migration. Radar observations greatly contribute to the understanding of the spatial and temporal patterns of bird migration because of the ability to register birds continuously at a large spatial scale and at high altitudes (Eastwood 1967; Bruderer 1997; Gauthreaux *et al.* 2003). Radars offer several advantages compared to visual observations as they are not limited to lower altitudes, daylight and good visibility. They also do not suffer from observer bias. However, there are also several restrictions to this technique: the recorded radar data have low taxonomic resolution and radars record objects other than birds (*e.g.*, sea surface, ships, rain). The latter unwanted detections are referred to as clutter.

The objectives of this study are to determine:

- the seasonal phenology of migrating birds across the North Sea;

- the diurnal patterns of migrating birds at sea;

- the vertical distribution (altitude) of migrating birds;

- the link between bird migration and meteorological conditions.

2. Material and methods

2.1. Radar hardware

In this study, we make use of a Merlin bird radar (DeTect-inc., Florida, USA) which is installed on the offshore platform inside the C-Power wind farm on the Thornton Bank in the BPNS (fig. 1). The radar antenna (Kelvin-Hughes Sharpeye solid state S-band) is rotating in the vertical pane, creating a vertical "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
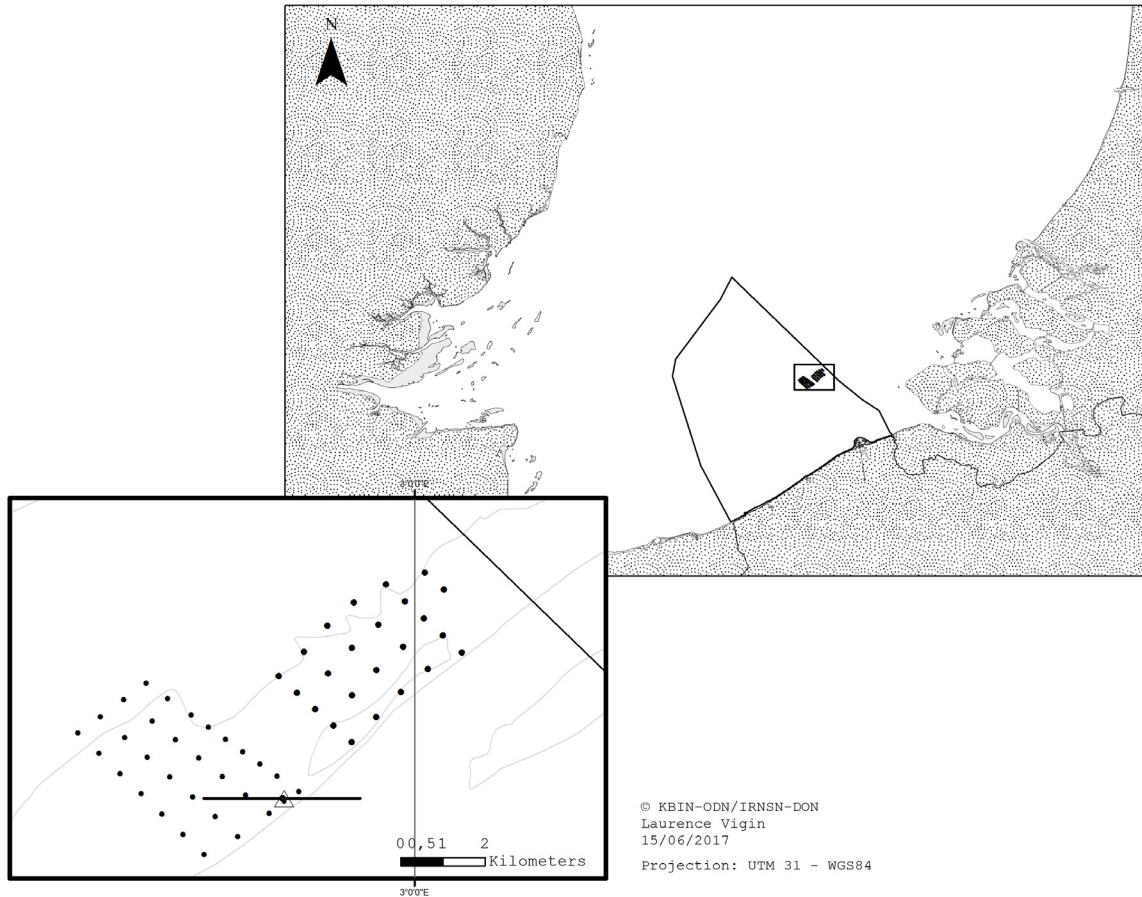
Figure 1. Map of the Belgian part of the North Sea (black polygon) with indication of the C-Power wind farm on the Thornton Bank (black marks). The location of the individual turbines (dots) and the radar location on the transformer platform (triangle) are shown in detail. The black line indicates the orientation of the vertical radar from east to west.

traffic rate (MTR), *i.e.*, number of birds that pass across a one kilometre line during an hour (birds.km^{-1}.hr^{-1}; Schmaljohann 2008). The orientation of the radar is east to west (fig. 1, zoom), which was the only possible practical set-up due to restrictions on the top deck of the platform. Ideally, the radar antenna should be positioned perpendicular to the main migration direction (*i.e.*, mainly northeast-southwest, which is perpendicular to the coastline).

2.2. Radar software and data post-processing

The detection range of the radar antenna can be specified in the system's settings and is set at one nautical mile. The radar operates continuously year-round and the system is remotely controlled. The system is

operated by the Merlin software which is specifically designed to track individual birds (DeTect Inc. 2010; Brabant *et al.* 2012). The Merlin software links consecutive registrations of a target, and thus registers the flight path of a moving target.

However, these processed data still contain a large amount of clutter coming from different sources (*e.g.*, rain, waves, ships, wind turbines, side lobes). As we use the radar data to determine the flux of birds in the area, it is very important to remove clutter as accurate as possible. To do so, we have developed a data-filter. The reader is referred to Brabant *et al.* (2016) for more details on the data filtering.

After the data filtering, two columns of 500 m wide were selected from the entire measurement volume. We only retained data
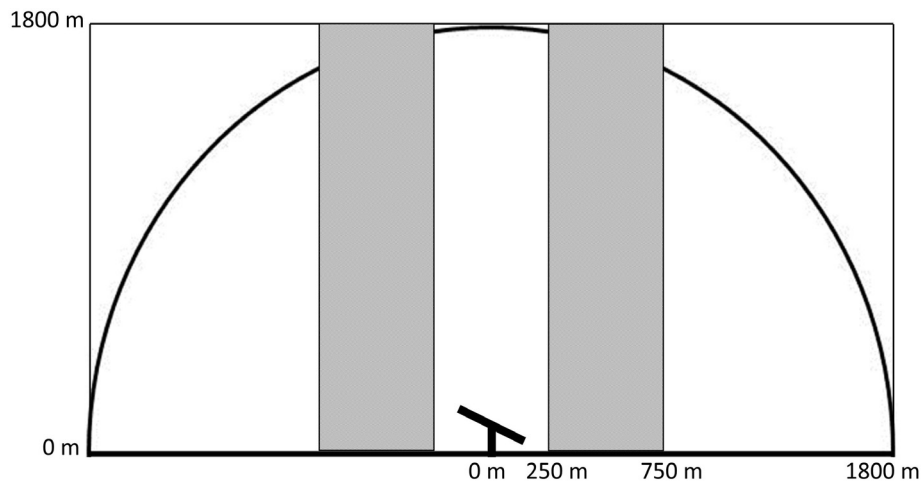
Figure 2. Vertical radar data used to determine the MTR.

from 250 until 750 m distance from the radar, both to the east and west (fig. 2). Doing so, we avoid using the data close to the radar location, which is saturated with reflections of the radar platform, and further than 750 m from the radar to avoid detection loss at further distance from the radar (Fijn *et al.* 2015). The number of bird tracks within those two columns in one hour equals the MTR. As the radar is not able to differentiate single birds from a small group of birds, the MTR for this type of radar is actually the number of groups of birds.km^{-1}.hour^{-1} or a minimum estimate of the number of birds.km^{-1}.hour^{-1}.

2.3. Data analyses

Four different analyses were done with the radar registrations between the 23 August 2016 and the 16 November 2016, here representing the autumn migration season.

> 1. Mean MTR values were calculated for every day and night. We used the daily time of sunrise and sunset to determine the length of day and night.

> 2. The diurnal pattern for every month was calculated by averaging the MTR and associated standard error for every hour of day (HoD) for the different months (*e.g.*, the mean of all MTR values from 0:00 to 1:00 am, for all days in October).

> 3. The total number of counts per night and per day, within altitude layers of 50 m were calculated. We considered data up to an altitude of 1800 m ASL, although it is known that there is detection loss at higher altitudes. Fijn *et al.* (2015) describe that for a magnetron radar (25 kW Furuno FR1525 MK3 X-band), this detection loss starts at 900 m for smaller birds. In this case, a solid state antenna is being used which has three different pulses (short, medium and long). For this type of radar, detection loss will occur within every pulse. However, at this point, it is not possible to quantify this and is hence here considered more or less random throughout the altitudinal range.

> 4. We investigated how migration intensity was influenced by the wind direction and speed. Wind direction and speed were taken from the C-Power meteorological observations from a wind turbine near the offshore platform (temporal resolution: 10 minutes).

It is important to note that we know the radar antenna is performing suboptimal at detecting birds at low altitude (0-150 m above sea level [ASL]). This was shown during elaborated tests in collaboration with the radar supplier and could not be resolved at this point. Krijgsveld *et al.* (2011; 2015)

and Fijn *et al.* (2015) however showed that in similar circumstances a large part of the migration at sea is occurring in those lower altitude layers. This has led to the decision to replace this antenna with a conventional magnetron X-band antenna. Replacement is foreseen in summer 2017.

Calculations and graphs were made in R version 3.2.2. (R Core Team 2015), making use of the packages ggplot2 (Wickham 2009), cowplot (Wilke 2016), reshape2 (Wickham 2007) and plyr (Wickham 2011).

3. Results

3.1. Autumn migration phenology

In general, the nighttime mean MTR values (fig. 4, lower panel) are higher than during daytime (fig. 4, upper panel). Highest numbers are recorded in October, especially during the first few days of that month. In August, MTR values are very low, both during day and night.

A scatterplot of the log-transformed mean daytime versus the log-transformed

mean nighttime MTR shows there is a significant relation between both (p-value: 3.531e^{-09}, R-squared = 0.37; fig. 3).

3.2. Diurnal pattern

All months showed a diurnal pattern with a peak at sunset (fig. 5). This is especially the case for the month of October where MTR values peak at sunset and decrease during the night. A smaller second peak at sunrise is also noticeable in October and November.

3.3. Flying altitudes

As was already shown in figure 4, absolute numbers are much higher during the night compared to daytime (fig. 6). During day, the highest number of counts was recorded from 100 to 150 m ASL. At night, this was the case in the layers from 200 to 300 m.

Given the radar's poor performance in the lower altitudes (up to 100-150 m, *i.e.,* the two to three lowest bars in figure 6), the number of birds counted in these altitude layers is therefore considered not reliable (see materials and methods).
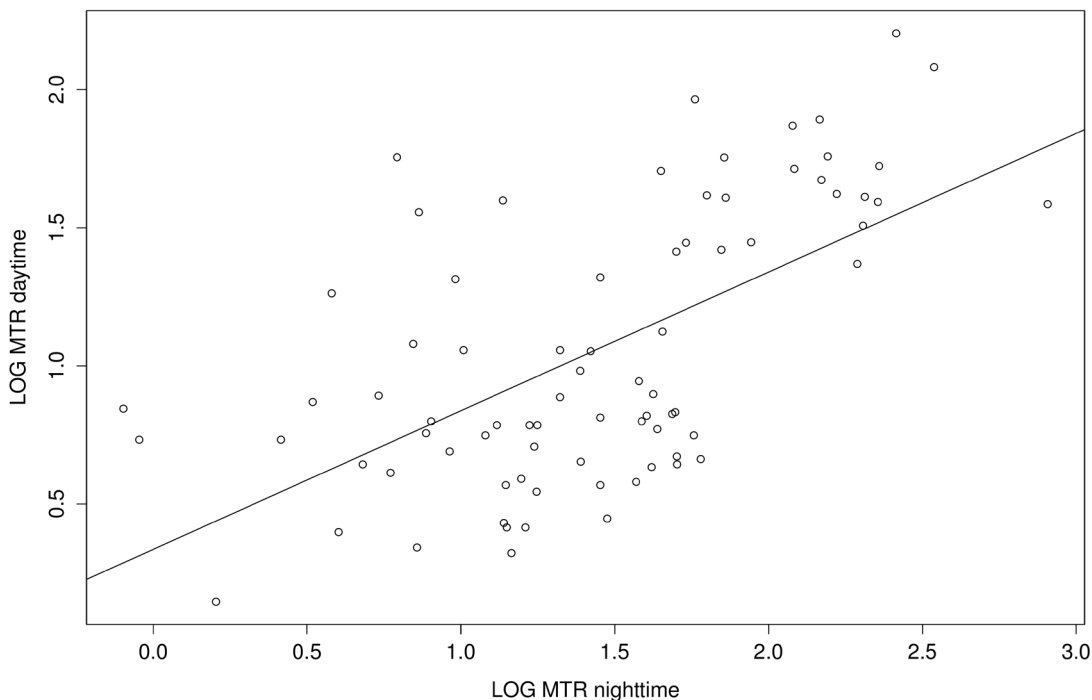

Figure 4. Scatterplot of the mean MTR at nighttime versus at daytime (log transformed).
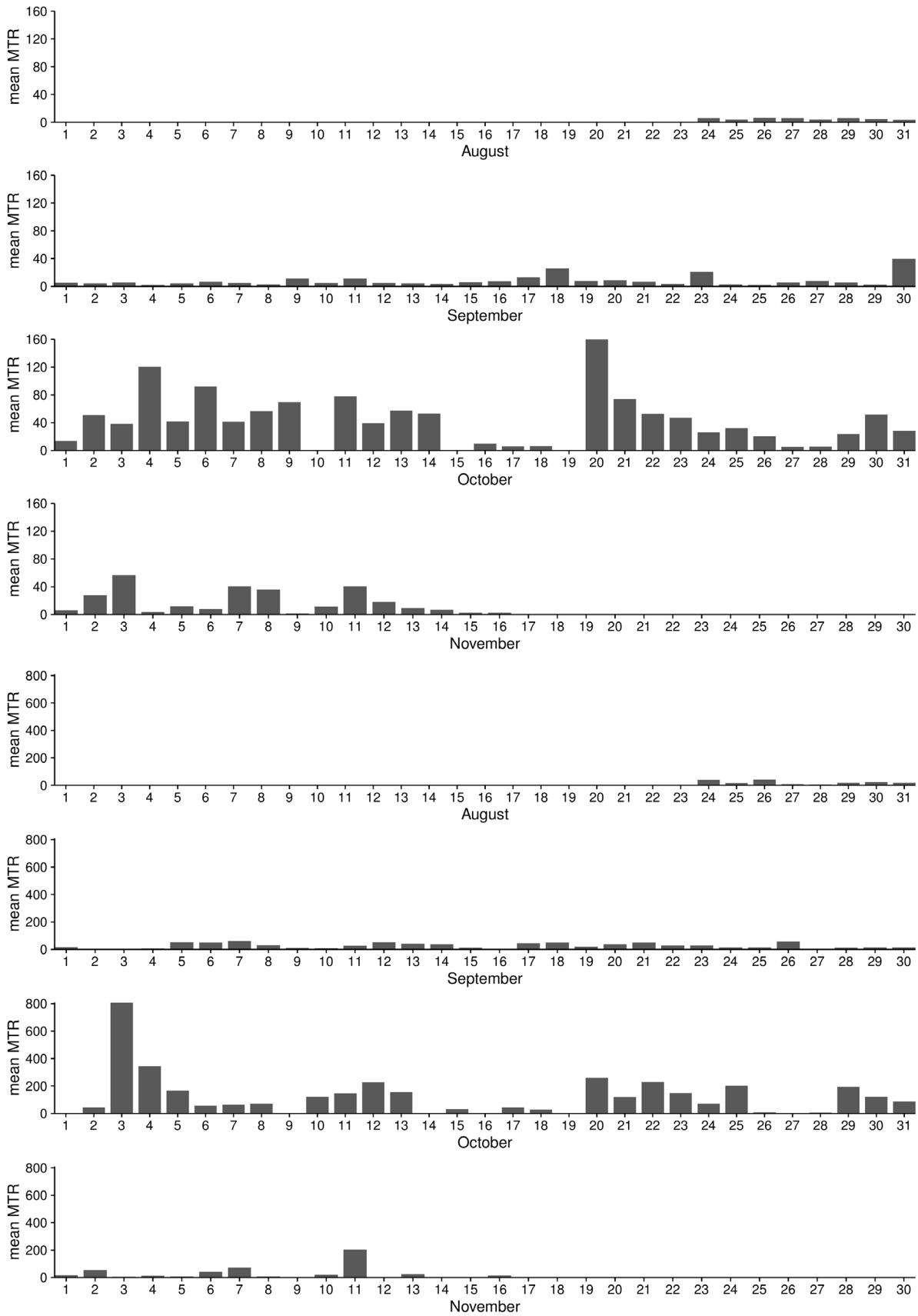
Figure 3. Average MTR ([groups of] birds.km^{-1}.hour^{-1}) per day (upper panel) and night (lower panel) for the autumn of 2016. Note that the Y-axis scale is different for the two plots.

Diurnal pattern August

Diurnal pattern September

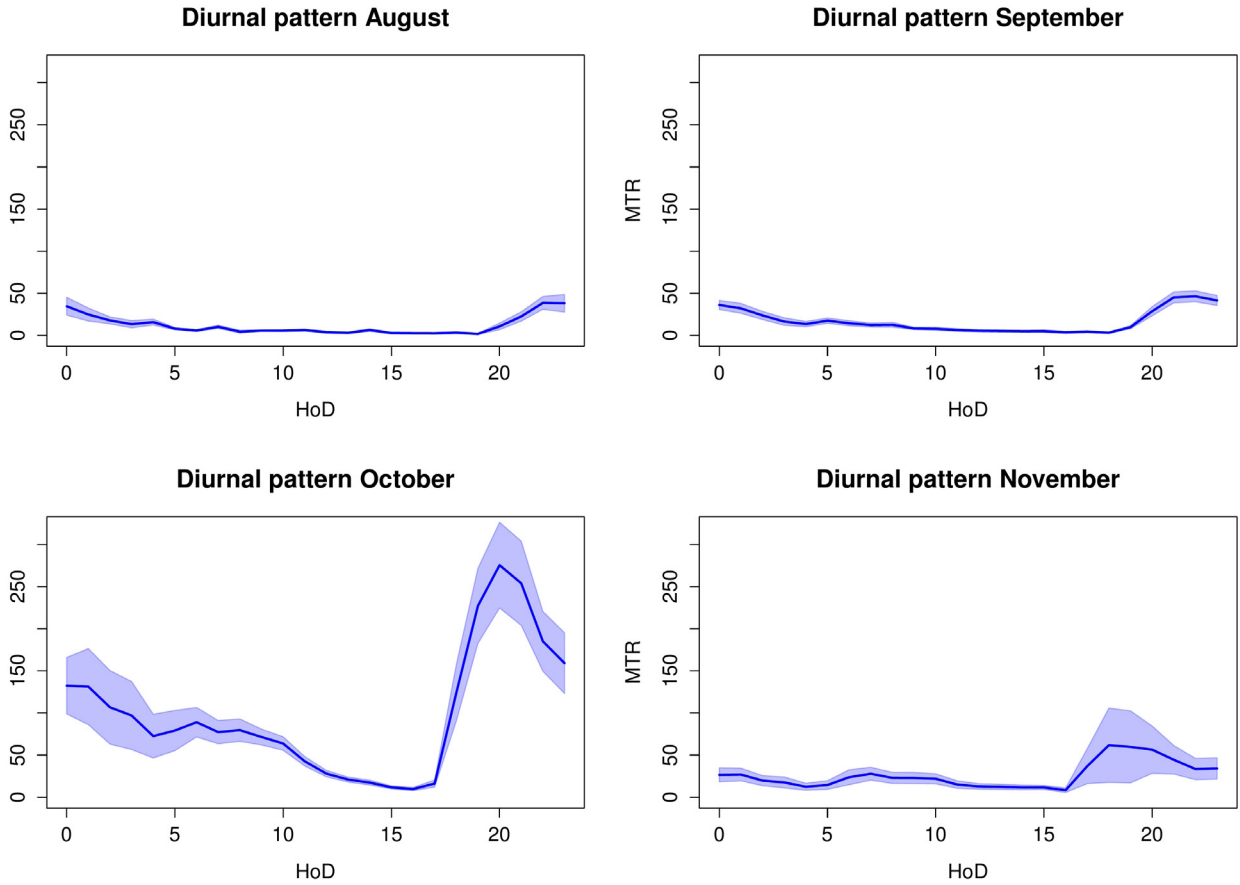## Diurnal pattern October

Diurnal pattern November

Figure 5. Average MTR value per hour of day (HoD) in UTC (blue line) ± standard error (light blue polygon).
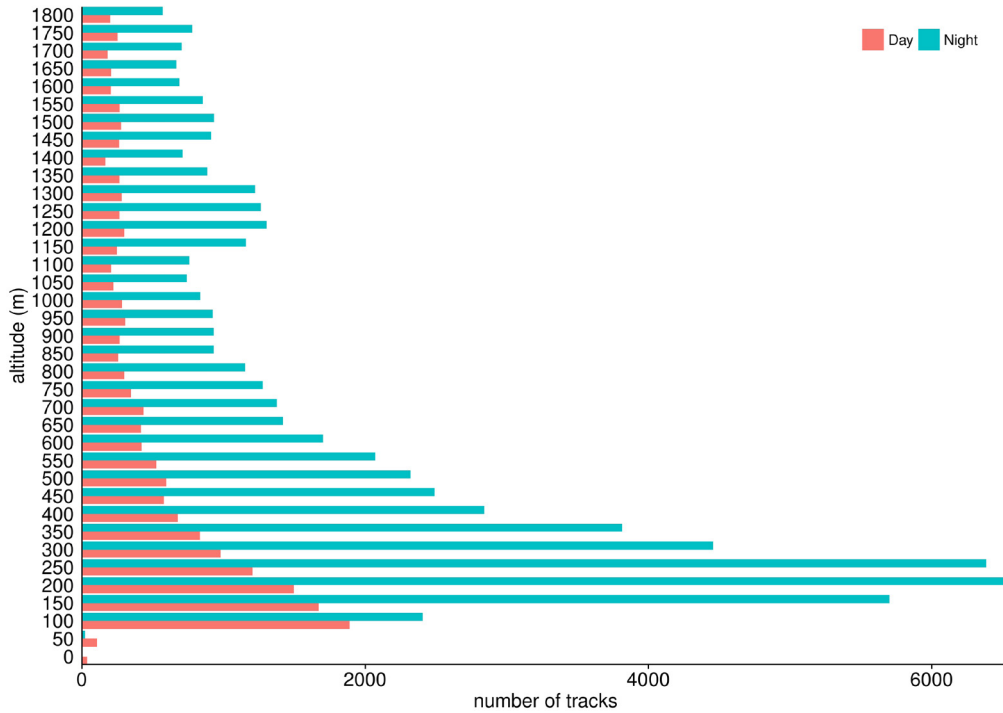
Figure 6. Absolute number of counts during day and night per 50 m altitude layer.
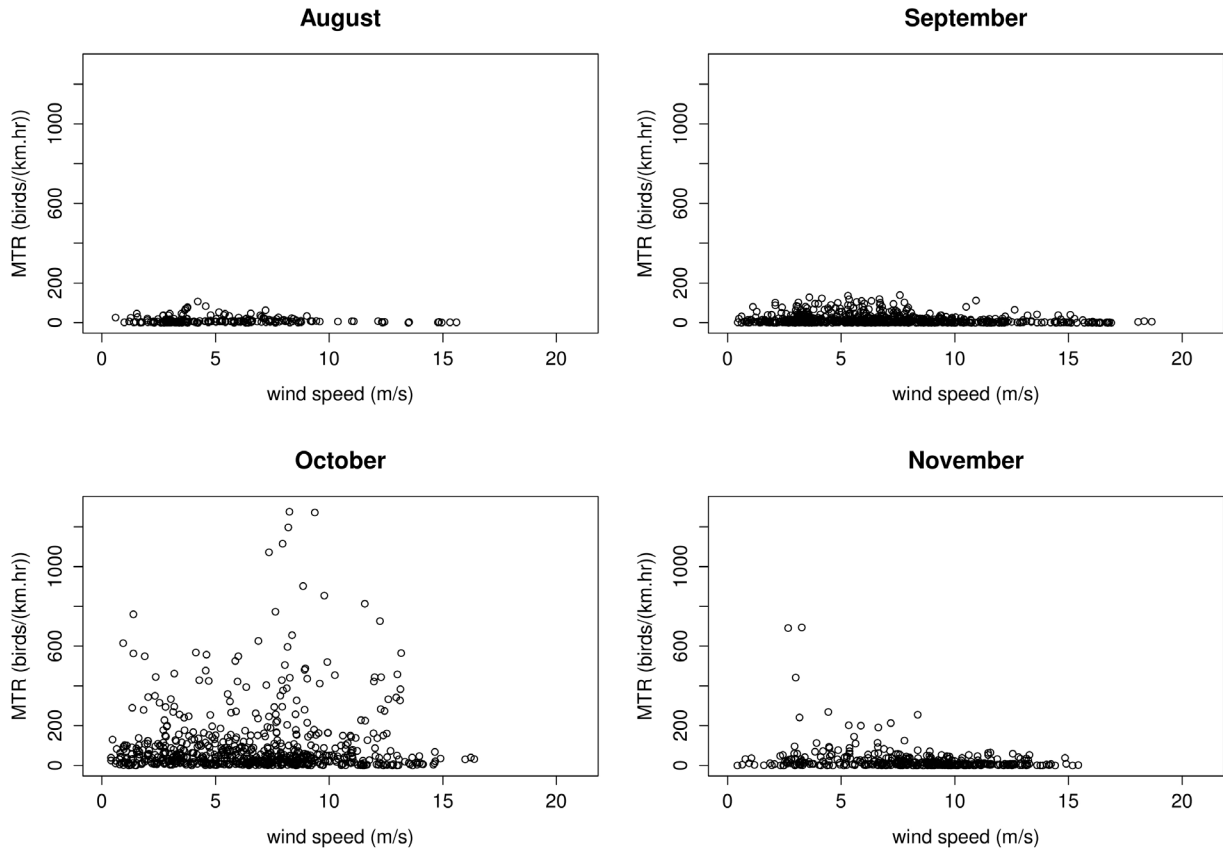
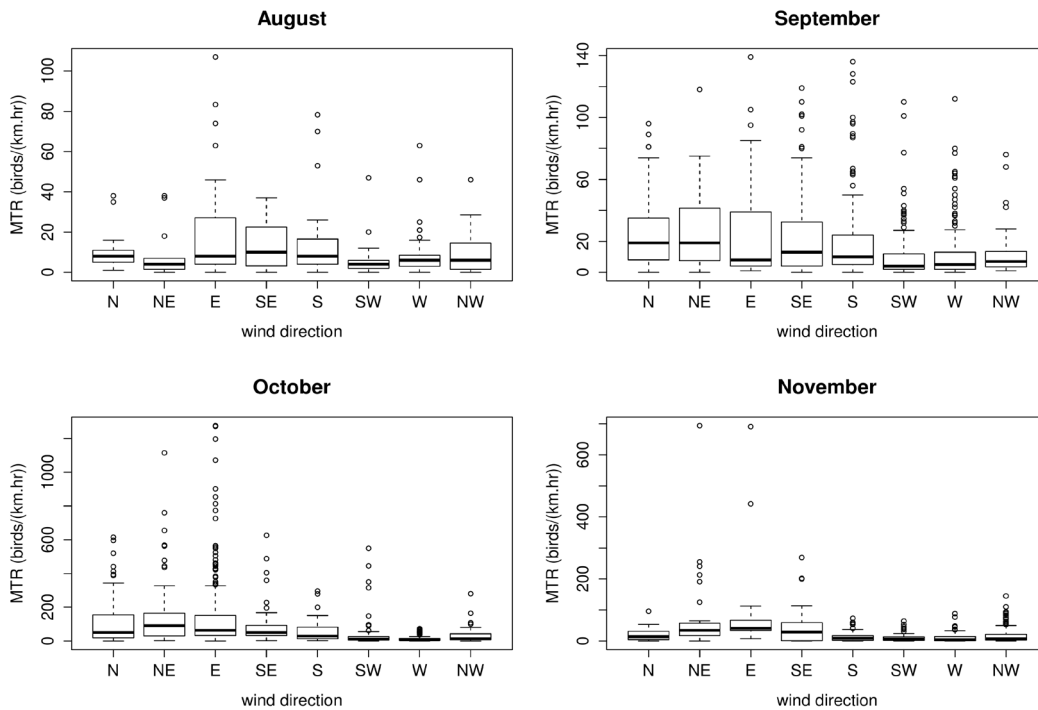**Figure 7.** Scatterplot of the hourly MTR (birds.km⁻¹.hr⁻¹) and wind speed (m/s).

Figure 8. Boxplot of the hourly MTR (birds.km-1.hr-1) and the wind direction. Line in the box is the median value. Lower and upper limits of the box represent 25th and 75th percentile of the data, respectively. The upper whisker is defined as 75th percentile + (1.5 x spread). The lower whisker is 25th percentile - (1.5 * spread), the spread being 75th - 25th percentile.

Figure 9. Boxplot of the wind direction and wind speed during the entire measurement period. Line in the box is the median value. Lower and upper limits of the box represent 25th and 75th percentile of the data, respectively. The upper whisker is defined as 75th percentile + (1.5 x spread). The lower whisker is 25th percentile - (1.5 * spread), the spread being 75th - 25th percentile.
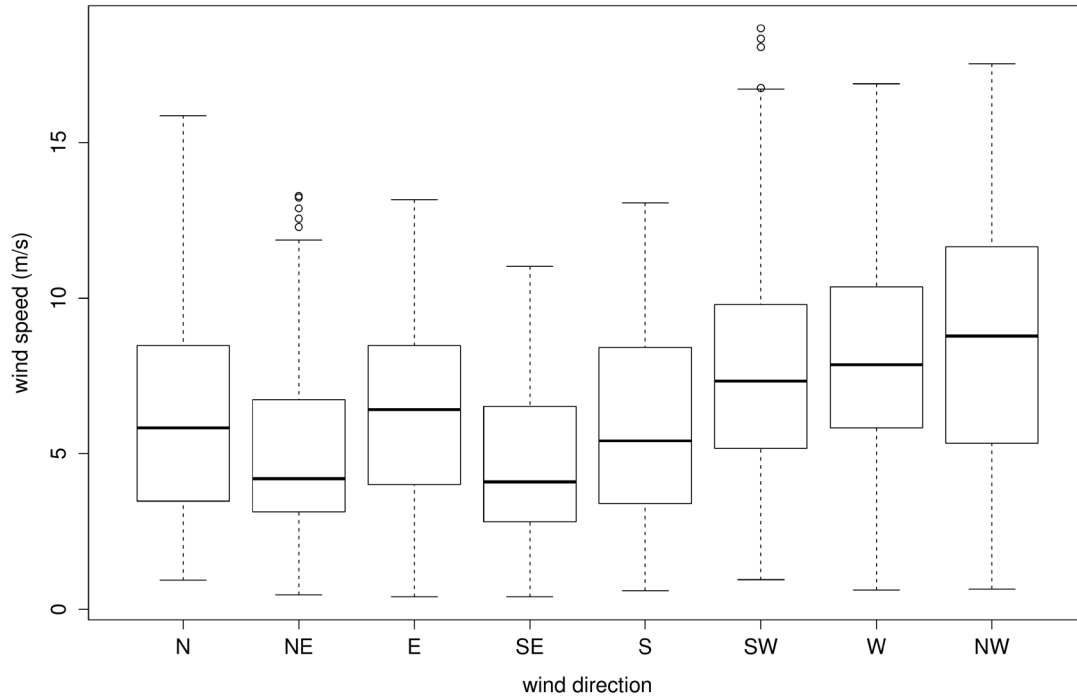
3.4. MTR in relation to wind speed and wind direction

There is no clear observed pattern between MTR and wind speed (fig. 7). The highest MTR were recorded in October and November. High MTR values (> 200 birds. km^{-1}.hr^{-1}) were however never recorded when the wind speed was higher than 13 m/s. The maximum wind speed during the study period was 20.9 m/s.

Figure 8 suggests that the October and November MTR values are higher when the wind is coming from the N, NE, E and SE. In August and September, this is not the case. During the measurement period, the wind speed was highest coming from the SW, W and NW (fig. 9). Wind from the NE and SE had the lowest speed.

4. Discussion

Overall, the measured MTR values are lower than expected. This has three reasons. (1) As mentioned in the methodology section, the radar is performing suboptimally in the lowest 150 m ASL. Looking at the results of Krijgsveld *et al.* (2015) and Fijn *et al.* (2015), in similar circumstances, it was shown that 50% of the total flux occurred below 115 m. The pattern we see in this study is caused by the limits of the solid state S-band radar antenna which is currently being used. (2) The current antenna has a wavelength in the S-band spectrum (7.5-15 cm), which is less suited to register smaller birds. So, presumably, the number of songbirds is being underestimated. (3) Lastly, the orientation (E-W) of the radar antenna is not ideal. An orientation perpendicular to the main migration direction is preferred to correctly measure the flux of birds (van Gasteren *et al.* 2002). This was logistically not possible in this case. If the flight direction is other than perpendicular to the radar orientation, the bird numbers is inevitably underestimated. Van Gasteren *et al.* (2002) describe a

formula to compensate for this by correcting the surface area of the sampled air of the radar. If the flight direction is 45°, relative to the radar orientation, the correction factor is 1.41. If it is 22.5°, then the correction is 1.08 (Fijn *et al.* 2015). Fijn *et al.* (2015) also made the argument that the vertical radar has a specific beam width and thus records flux in a volume rather than along a line, the underestimation is therefore at least smaller and in many cases close to the measured flux. Because no confirmed flight direction data is available in this study, no corrections could be made.

Compared to the total flux measured in an entire autumn season (September-November) by Fijn *et al.* (2015), the total flux in this study is about a factor 10 smaller. Not taking the lowest 150 m into account, this is still a factor 5. This has led to the decision to replace the currently deployed antenna with a conventional magnetron X-band antenna, similar to the one successfully being used in the Dutch part of the North Sea (Krijgsveld *et al.* 2011; Fijn *et al.* 2015). By then, the horizontal radar will be operational again, providing flight direction data. This will then be used to correct the measured flux, if necessary.

Although this antenna was not ideal to register bird migration, some useful information is gained from the data. The MTR values show that migration at sea, as registered by the radar, was most intense during the nights of October and early November. Field observations (auditory recordings of vocal calls) carried out at night by Krijgsveld *et al.* (2011) at the OWEZ wind farm in the Dutch part of the North Sea, indicate that these high nocturnal fluxes reflected mostly migrating passerines. Especially Blackbird *Turdus merula*, Song Thrush *Turdus philomelos*, Redwing *Turdus iliacus* and Robin *Erithacus rubecula* were recorded. This is supported by earlier studies by Bourne (1980), Buurma (1987), Alerstam (1990) and Lensink (2002).

High daytime fluxes measured in October and November, correspond with coastal observations of high numbers of migrating meadow pipits *Anthus pratensis*, European starling *Sturnus vulgaris* and Chaffinches *Fringilla coelebs* (www.trektellen.nl). Also large numbers of Brant *Branta bernicla* were counted, a species regularly seen in the BPNS (Vanermen *et al.* 2006). As this latter species tends to fly at a lower altitude, it is unlikely that it was detected by the radar.

The observed diurnal pattern of these months is similar to the pattern measured by Fijn *et al.* (2015). Especially in October a clear peak in MTR values occurs at dusk. A second smaller peak is noticeable at dawn.

Wind direction is the main driver of autumn migration (Alerstam 1990). For this period, no clear pattern with weather conditions could be revealed. It seems that MTR values are higher if the wind was coming from the N, NE, E and SE and when wind speed was lower than 13 m/s. In autumn, easterly winds are known to give rise to concentrated migration near the coast and at sea (Lensink *et al.* 2002). This was also the case in the beginning of October, when the highest fluxes of this study were measured. At that time, a storm front covered Germany and Poland, forcing birds to a more westerly migration route, which led to high numbers of birds in Belgium and the Netherlands.

Birds are registered up to 1800 m (highest altitude bin taken into account in this study). The altitude profile, although not complete, suggests that migration at night is happening at higher altitudes compared to daytime movements. This is also what Fijn *et al.* (2015) observed during autumn. While passerines (*i.e.*, non-seabird species) tend to dominate nighttime migration, daytime migration tends to be a mixture of seabird and non-seabird species. Seabird migration

(divers, terns, seaducks) migrate at an altitude lower than 25 m. This can sometimes go up to 50 m, but rarely higher (Krüger & Garthe 2001). The current radar configuration hence is not suited to monitor seabird migration.

Starting autumn 2017, the recorded bird flux data will be analysed with an explanatory model approach, to identify the variables driving the observed migration at sea (*e.g.*, wind direction and speed, hour of day, Julian day, bird flux at the previous day). This can then lead to prediction models which can be used to apply mitigation measures for offshore wind farms. For instance, a requirement for the neighbouring Dutch Borssele wind farms is to shut down the turbines when the bird flux at rotor height exceeds 500 birds.km^{-1}.hr^{-1}. To practically apply such a measure will require a lot of cooperation of all involved parties and model results can assist in this process.

Acknowledgements

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CHAPTER 9

DETERMINING THE SPATIAL AND TEMPORAL EXTENT OF THE INFLUENCE OF PILE DRIVING SOUND ON HARBOUR PORPOISES

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Abstract

Piling driving sound is known to impact harbour porpoise (*Phocoena phocoena*) distribution, but to date detailed knowledge on the combined spatial and temporal components of this impact over longer time periods remains lacking. From May to September 2016, pile driving was taking place at the Nobelwind wind farm located on the Bligh Bank in Belgium. In this period, porpoise activity was recorded using passive acoustic monitoring (PAM) using Continuous Porpoise Detectors (C-PoDs), at various distances from the construction site ($1 - > 55$ km). In this study, we compared porpoise detections before, during and after pile driving. During piling, porpoise detections decreased at stations located up to 20 km from the location of the piling event. At larger distances (20-55 km), porpoise detections either remained the same or increased slightly during piling events, which may be due to displaced porpoises entering the area.

Underwater sound levels were extrapolated for the different locations. Pile driving sound levels at the furthest distance where reductions in porpoise detections were observed were ~159 dB re 1μPa (L_{z-p}), which is close to the threshold level for major disturbance for harbour porpoise proposed in literature.

1. Introduction

The harbour porpoise (*Phocoena phocoena*) is the most common marine mammal in the Belgian part of the North Sea (BPNS) and is protected by both national and EU law. In the North Sea, the harbour porpoise is considered vulnerable because of high by-catch levels and increasing sound pollution. Impulsive pile driving sound originating from the construction of offshore wind farms (OWF) has been shown to affect porpoises up to distances of 20 km from the sound

source (Haelters *et al.* 2013; Brandt *et al.* 2016). On the basis of seasonally high porpoise densities in Belgian waters, a pile driving ban is in force from the start of January up to the end of April (Rumes *et al.* 2011; 2012; 2014). However, The Netherlands have the Borssele offshore wind farm at only one kilometer away from the Belgian offshore wind farm zone, and do not enforce a seasonal pile driving ban. Instead, seasonally fluctuating underwater sound limits are set for construction sound (Ministerie van Economische Zaken 2015). There is a need for improved insights into the spatial and temporal extent of the impact of pile driving sound on porpoises in order to determine the consequences of pile-driving at the (local) population scale using demography-based modelling. This can then serve as a basis for a more objective assessment of the effects

and stimulate better regional alignment of mitigation measures.

In this study we use continuous passive acoustic monitoring (PAM) to study the spatial and temporal extent of the influence of pile driving sound on harbour porpoises.

2. Material and methods

2.1. Data collection

Passive acoustic monitoring (PAM) of porpoises was conducted using 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 for each click
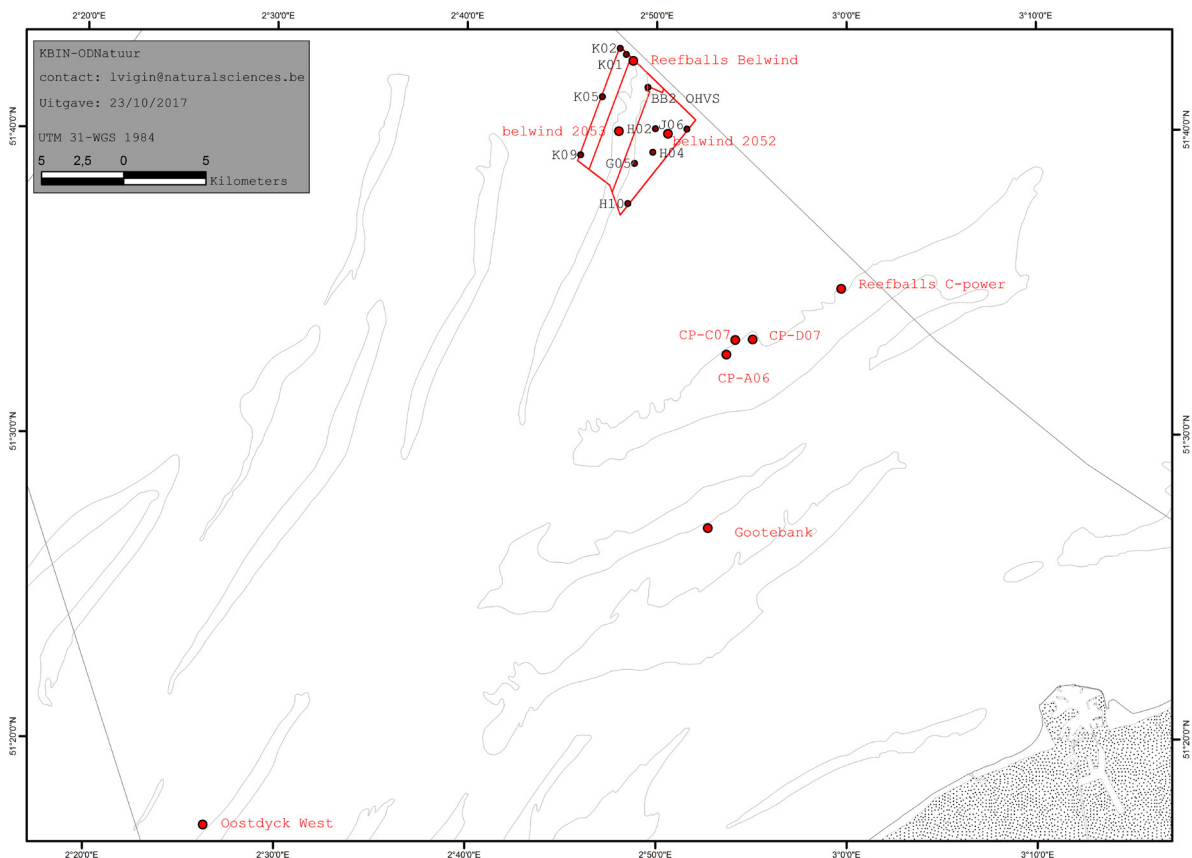

Figure 1. Location of the PoD deployments used in this study with indication of the Nobelwind wind farm (red outline) and the locations of the piling events used in this study.

Table 1. Overview of PoDs used in this study with indication of their location, deployment period, mooring type and range of distance from the piling events. Two additional PoDs deployed inside the Belwind windfarm on 4 April 2016 remain lost at sea.

Station	Latitude	Longitude	Start record	End record	Mooring type	Range
Belwind 2052	51°39.875'	02° 50.590'	11 August 2016	22 February 2017	Bottom	Inside
Belwind 2053	51°39.956'	02°47.999'	4 April 2016	1 July 2016	Bottom	Inside
Reefballs Belwind	51°42.265'	02°48.756'	14 July 2016	9 September 2016	Bottom	2-10 km
Reefballs C-Power	51°34.800'	02°59.729'	30 August 2016	4 October 2016	Bottom	10–20 km
CP-A06	51°32.639'	02°53.687'	24 April 2016	18 August 2016	Bottom	10–20 km
CP-C07	51°33.116'	02°54.150'	24 April 2016	27 May 2016	Bottom	10–20 km
CP-D07	51°33.137'	02°55.067'	24 April 2016	17 August 2016	Bottom	10–20 km
Gootebank	51°26.950'	02°52.720'	28 June 2016	24 October 2016	Surface Buoy	20-30 km
Oostdyck West	51°17.150'	02°26.320'	27 June 2016	13 October 2016	Surface Buoy	45-55 km

characteristics such as 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 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 m. PoDs have autonomy of up to 200 days (www.chelonia.co.uk).

For this study, data were used from PoDs deployed at nine locations in the BPNS, five of which were specifically deployed for this study with the other four forming part of the VLIZ EU Lifewatch observatory (Flanders Marine Institute 2015; fig. 1). Mooring locations were divided into five range classes: inside the piling area, 2 to 10 km from the piling events, 10-20 km distance, 20 to 30 km distance and 45 to 55 km.

As PoDs were anchored in different ways and at different depths, which influence detection rates (Sostres, Alonso & Nuuttila 2015), comparisons of detections between those PoDs are not justified. Therefore, we limited ourselves to comparing the relative differences in detection rates through time at the different stations. Data from different locations was only aggregated when the same type of anchoring was used.

2.2. Data selection and analysis

Pile driving for the Nobelwind wind farm comprised 51 piling events from 16 May 2016 up to 22 September 2016. Pile diameter ranged from 4.5 to 6.8 m, penetration depth lay between 29 to 39 m and total piling time varied between 1.27 h and 4.31 h. The contractor was legally obliged to turn on an acoustic deterrent device (ADD; in this case a Lofitech Seal Scarer was used) 30 min before the start of piling and to use a soft start procedure.

As in Brandt *et al.* (2016), we selected only those piling events where at least 96 h had passed since the end of the previous pi-

Table 2. Overview of the Nobelwind piling events included in this study

Location	Pile order	Date	Time started	Time stopped	Total blows	Latitude	Longitude
K01	1	16 May 2016	14:11	18:15	3539	51° 42.477' N	2° 48.381' E
K02	2	21 May 2016	05:02	09:29	3510	51° 42.676' N	2° 48.064' E
K05	5	4 June 2016	01:08	04:00	3211	51° 41.088' N	2° 47.118' E
K09	9	15 June 2016	11:10	14:36	3921	51° 39.177' N	2° 45.981' E
J06	13	28 June 2016	01:15	03:30	2894	51° 40.032' N	2° 51.590' E
BB2 OHVS	25	25 July 2016	18:46	23:19	5157	51° 41.400' N	2° 49.531' E
G05	26	31 July 2016	20:56	23:08	3215	51° 38.906'	2° 48.830' E
H04	32	15 August 2016	10:35	13:58	3955	51° 39.271' N	2° 49.792' E
H02	35	23 August 2016	03:08	05:20	3098	51° 40.039' N	2° 49.926' E
H10	40	5 September 2016	08:45	10:10	3603	51° 37.582' N	2° 48.478' E

ling event. Our analyses are limited to the time period starting 48 h prior to the start of the activation of the ADD up to a maximum of 48 h after the end of a piling event. The time period was shorter when there was a consecutive piling within 48 h of the previous event. Hours during which the ADD was turned on or during which piling took place were counted as 0. All this was done in order to minimize the effect of consecutive piling events, *i.e.*, we assumed that harbour porpoise densities had returned to the original level 48 h after piling ended. Details on the piling events included in our analysis are listed in table 2.

PoD data were automatically processed with the proprietary software C-POD.exe version 2.044 (Tregenza 2011) using the KERNO classifier and the settings for "porpoise-like" click sequences in the classes "Hi" and "Mod" . For the analysis we used the following measures for porpoise presence:

• detection positive minutes per day (DPM/d), or the number of minutes in a day, in which porpoise click trains were detected; also detection positive 10 min per hour (DPM10/h) was used;

• click intensity per hour represents the number of porpoise clicks recorded during that hour;

• waiting time (WT) is the interval length of periods of more than 10 min without detections and thus a measure for the amount of time between different porpoise encounters (Dähne *et al.* 2013).

We compared click intensity per hour, DPM/d, DPM10/h and WT between encounters and at various distances from the piling event.

All analyses were executed using R (version 3.4.0, The R Foundation for Statistical Computing) and Rstudio (2009-2016 Rstudio, Inc.).

2.3. Underwater sound

Impulsive underwater sound was measured during piling operations in the framework of the RBINS wind farm monitoring programme using a calibrated moored hydrophone (B&K 8104 hydrophone with a
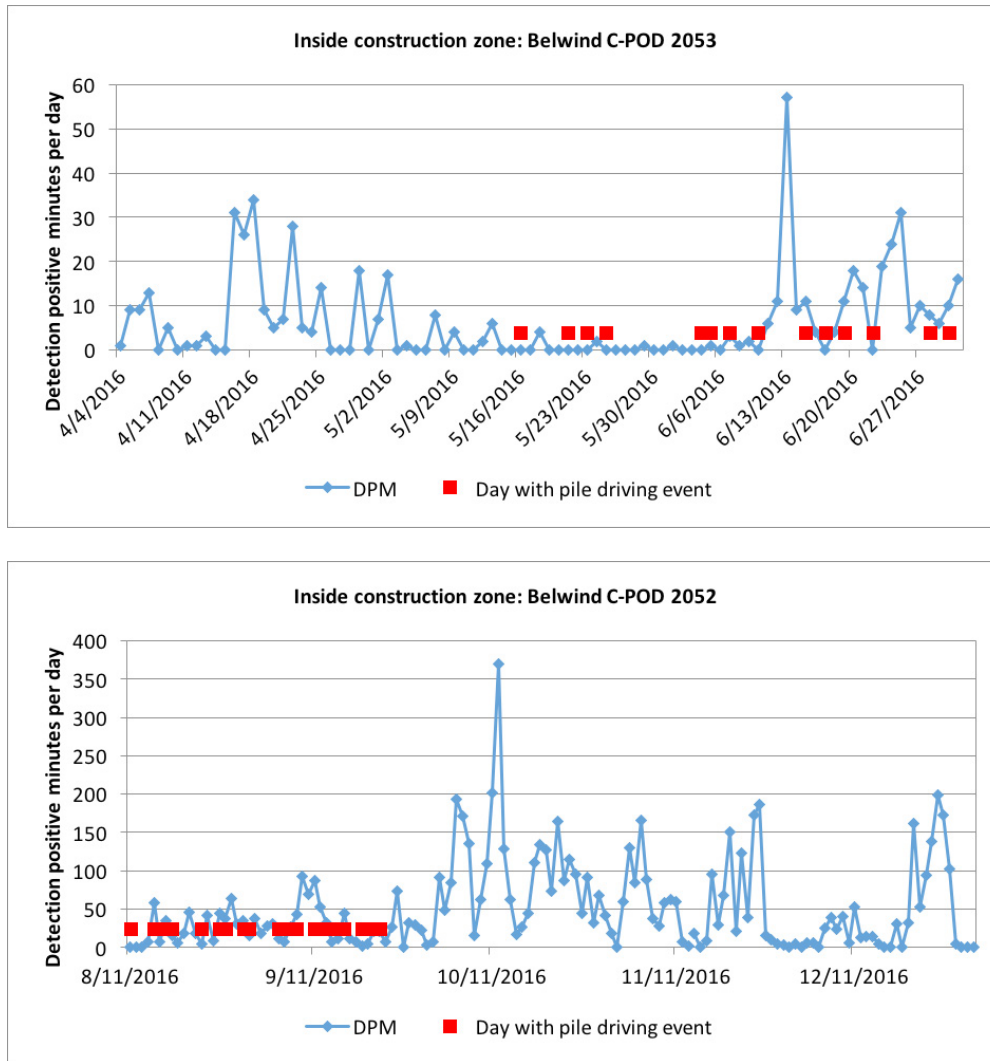
Figure 2 (part one). Detection positive minutes per day for the two PoDs moored inside the pile driving area.

RTsys EA-SDA14 recorder) located within the Nobelwind area (at the same location as the Belwind 2052 PoD). The recorder was operated at a sampling rate of 78,125 Hz. Using reference signals, the sound level and frequency distribution (spectral analysis) of selected sections of the recordings were analysed. Zero to peak level (L_{z-p}) as well as Sound exposure level for siggle strike SEL_{ss} and cumulative sound exposure level SEL_{cum} were calculated using MATLAB. A propagation model (Norro *et al.* 2013) was used to extrapolate the sound levels at various distances from the source.

3. Results

3.1. Passive acoustic monitoring

3.1.1. Detection positive minutes per day

The interannual variability and seasonal patterns in harbour porpoise densities in the Southern North Sea make it difficult to interpret changes in porpoise detections throughout the piling period. A visual inspection of the DPM/d vs piling does illustrate the range in variability present in the dataset. Inside the work area, we observed on average lower detection rate (DPM/d) during piling days (fig. 2, part 1). There was no clear
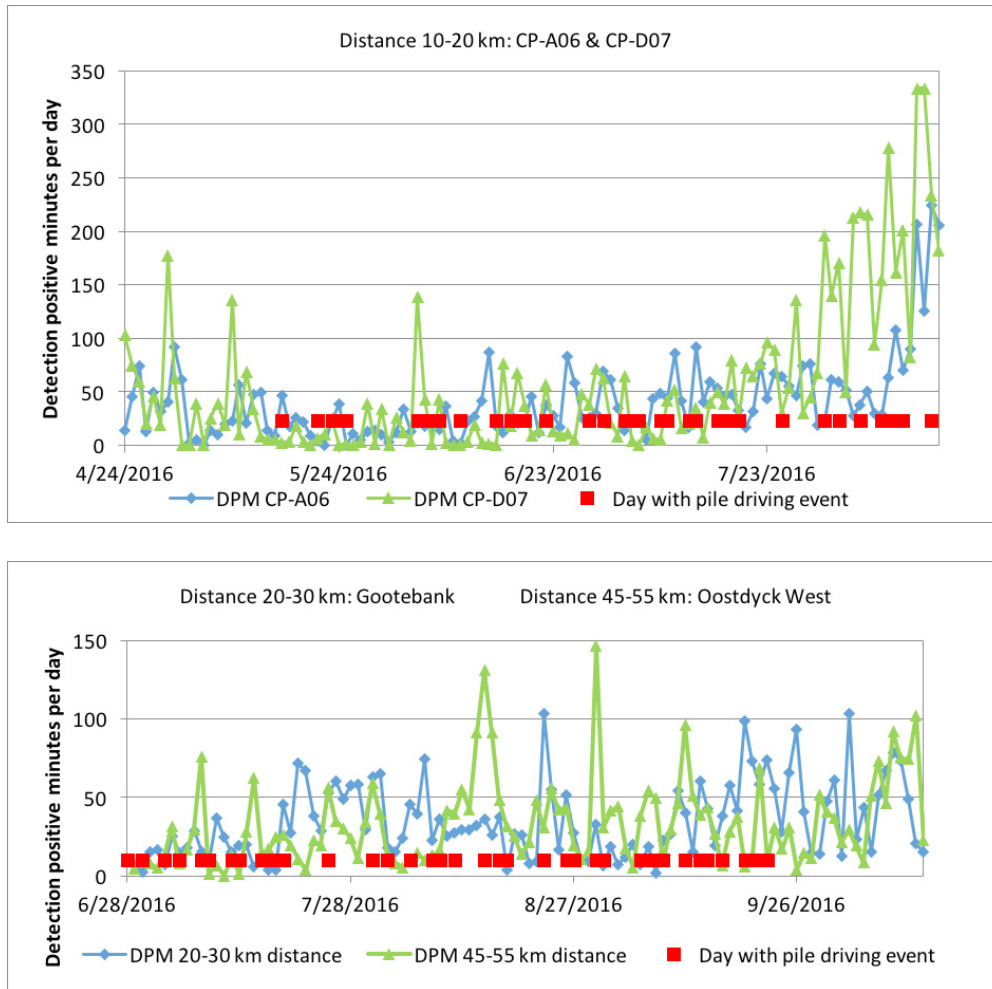
Figure 2 (part two). Detection positive minutes per day for the PoDs moored at increasing distances from the pile driving area.2.1.2 Click intensity per hour

relation between detection rate and piling at the other stations, with natural variability in density probably playing a major role (fig. 2, part 2).

In case of a low density of porpoises, such as usually in early summer months in Belgian waters (Haelters *et al.* 2016), the number of porpoise clicks per hour is often zero. This notwithstanding, we could still observe a reduction in the number of porpoise detections (to virtually 0) inside the work area during the hours of acoustic deterrence and piling. At locations further away detections were higher during these time intervals (fig. 3).

In the run up to and during the piling event, click intensity decreased strongly

inside of the piling area only to recover less than 48 h later (fig. 4). A smaller reduction in click intensity was observed in the vicinity (2-10 km distance) of the piling area. At larger distances, click intensity either remained largely the same or it temporarily increased (fig. 4).

3.1.2. Detection Positive 10 minutes per hour (DPM10/h)

In the run up to and during the piling events, porpoise detections (DPM10/h) decreased both inside and at distances up to 20 km from the piling area, with the decrease starting later further away from the work area. In contrast, at larger distances (> 20 km distance) DPM10/h increased during the piling events (fig. 5).
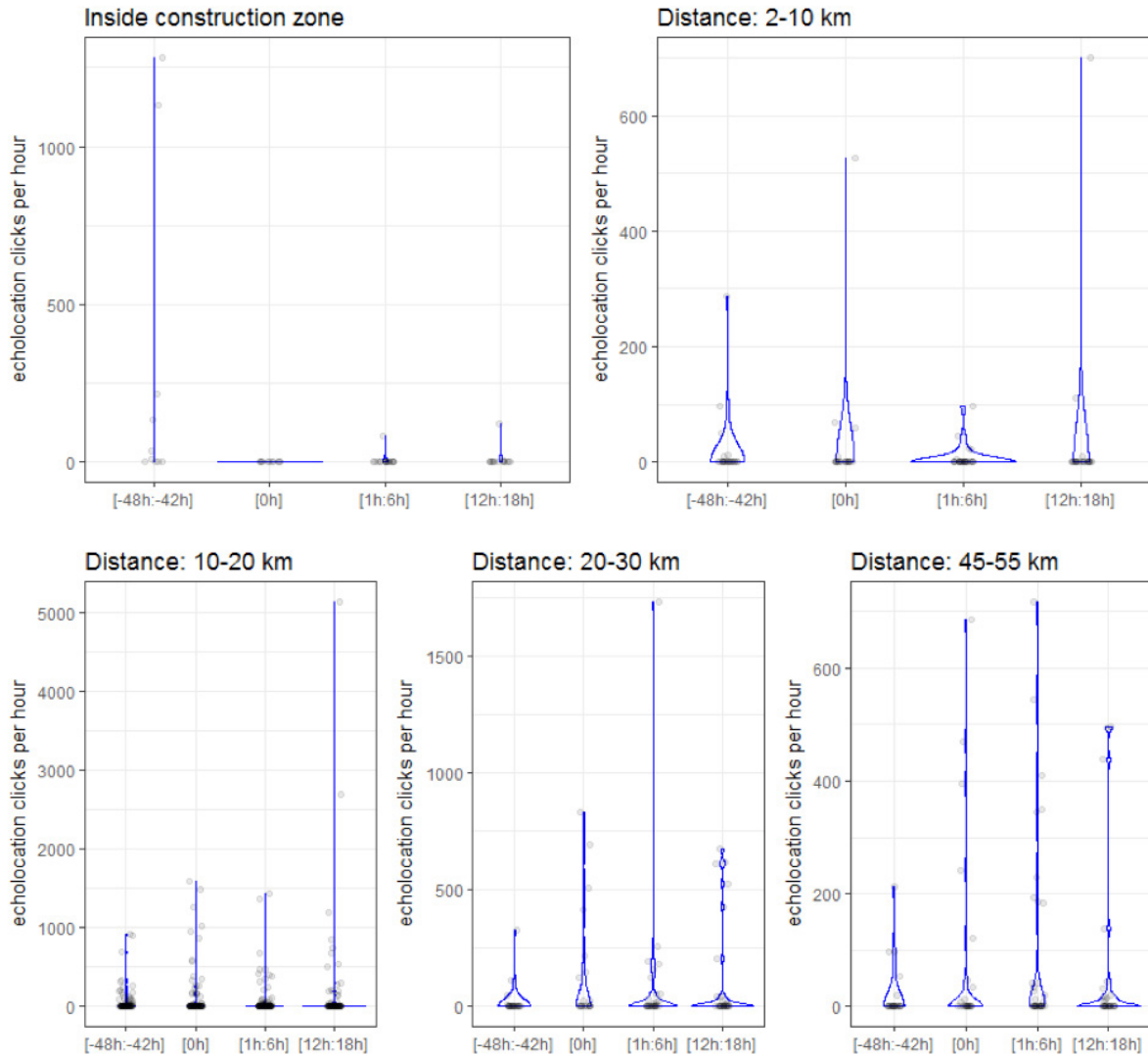
Figure 3. Violin plot of the number of Harbour porpoise (*Phocoena phocoena*) echolocation clicks per hour for four time intervals (48-42 h before the start of deterrence – used here as the baseline, during acoustic deterrence or piling, 1-6 h after piling ended and 12-18 h after piling ended) at five distance ranges from the piling event.

3.1.3. Waiting time (WT)

Waiting time, a measure for the amount of time between different porpoise encounters, temporarily increased both inside and at distances up to 20 km from the piling area (fig. 5). At larger distances WT (temporarily) decreased in this time period.

3.2. Underwater sound

3.2.1. Underwater sound levels inside the pile driving area

Underwater sound was recorded during the construction period using a moored hydrophone (B&K 8104 hydrophone with a RTsys EA-SDA14 recorder). During the piling events extremely high sound levels were recorded (up to L_{z-p} 198 dB re 1µPa and SEL_{ss} 174 in dB re 1µPa^2 s both @ 750 m, Norro, this volume). For the BPNS, ambient underwater sound levels were documented
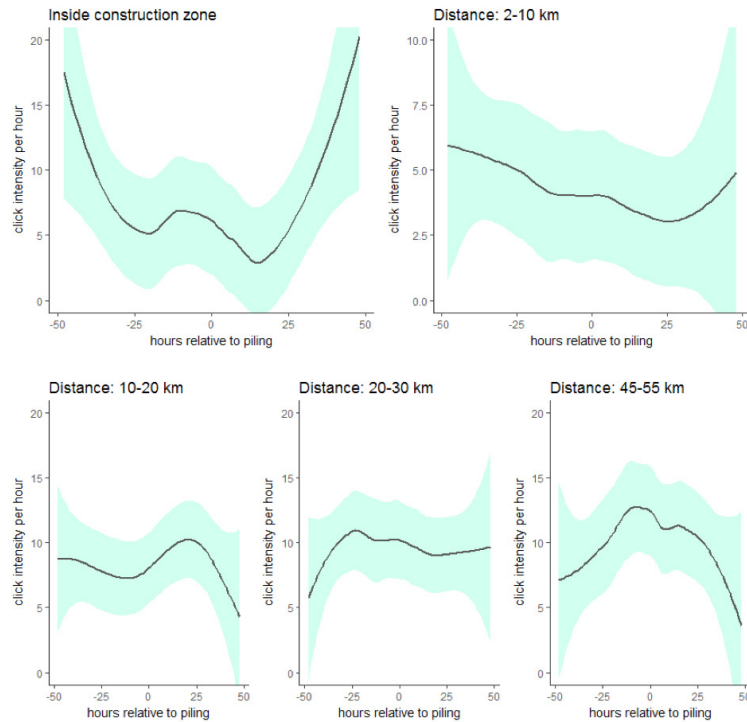
Figure 4. Click intensity per hour for a time period starting 48 h before acoustic deterrence started and ending 48 h after piling was terminated at five distance ranges from the piling event. All data from the time period starting with the start of deterrence and ending with the end of piling is included in the 0 h data point. Error bars (shaded area) represent the 95% confidence interval. For the distance interval 10-20 km data from CP-A06 was used.
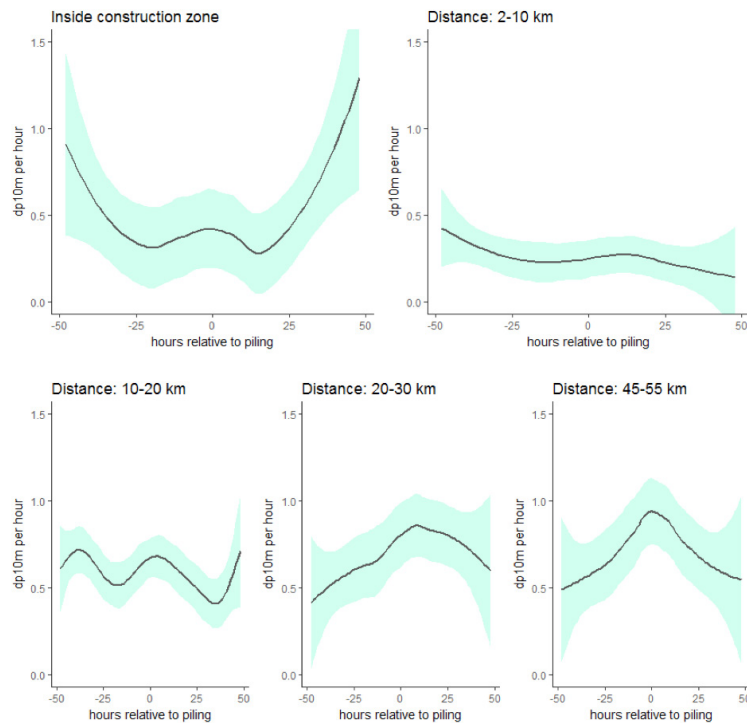

Figure 5 (part 1). Detection Positive 10 minutes per hour (0-6) for a time period starting 48 h before acoustic deterrence started and ending 48h after piling was terminated at five distance ranges from the piling event. All data from the time period starting with the start of deterrence and ending with the end of piling is included in the 0 h data point. Error bars (shaded area) represent the 95% confidence interval.

Figure 5 (part two). Waiting time for a time period starting 48 h before acoustic deterrence started and ending 48 h after piling was terminated at five distance ranges from the piling event. All data from the time period starting with the start of deterrence and ending with the end of piling is included in the 0 h data point. Error bars (shade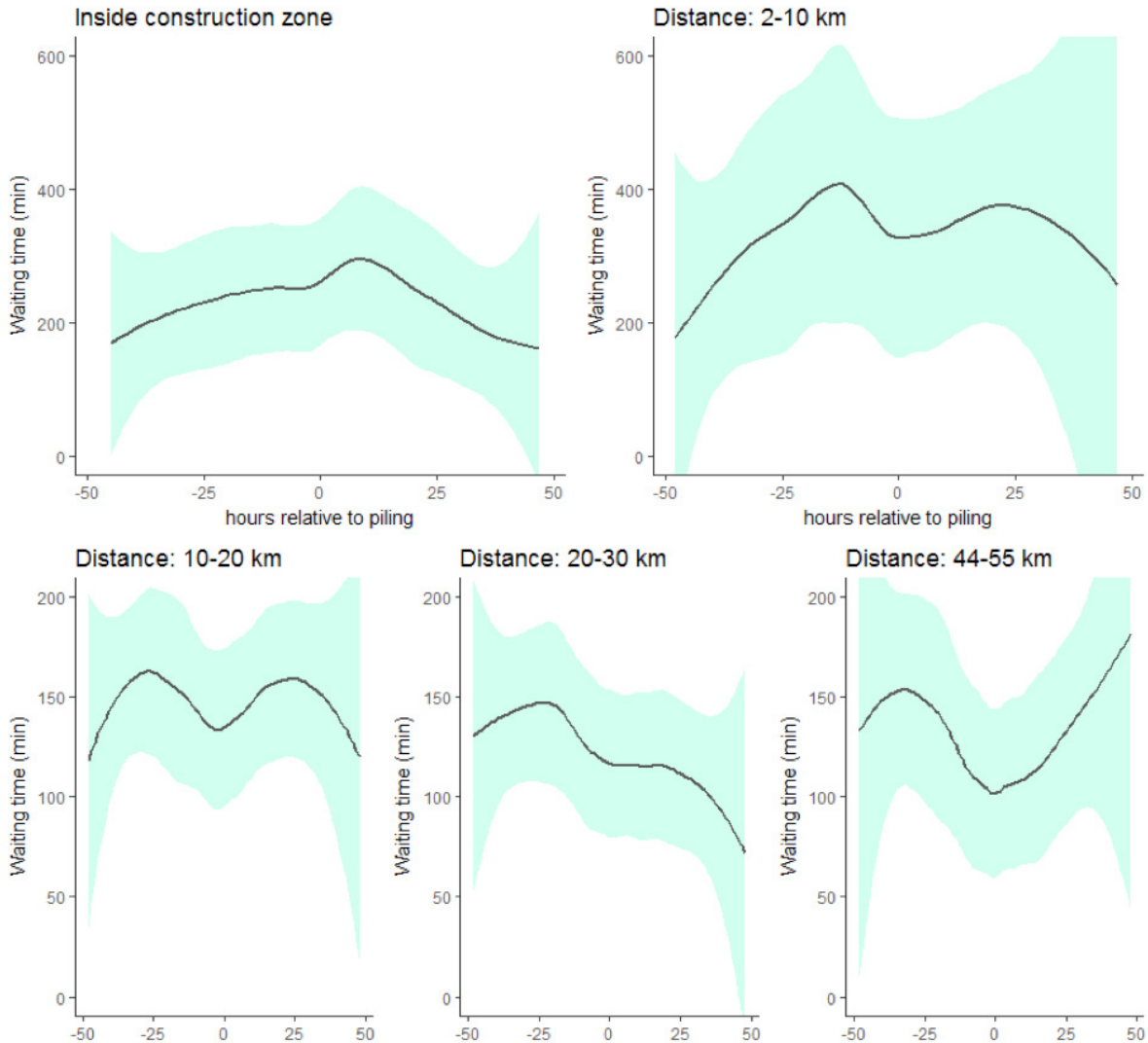d area) represent the 95% confidence interval. Note the difference in scale between the top two and bottom three locations.

prior to wind farm construction with reference to sound pressure levels (SPL) of about 100 dB re 1 µPa at the Thornton Bank and Bligh Bank (Henriet *et al.* 2006; Haelters *et al.* 2009). Underwater sound levels in the construction area also temporarily increased outside of piling events with extended periods of near continuous sonar when the pile driving vessel was on site.

3.2.2. Pile driving sound levels at various distances

Using the updated propagation model of Norro *et al.* (2013) and recorded pile driving sound levels we extrapolated pile driving sound levels to the different spatial ranges (table 3). Bailey *et al.* (2010) propose 149 dB SPL L_{z-p} re 1 µPa as the threshold level for major disturbance for harbour porpoise.

Table 3. Extrapolated pile-driving sound level from the Nobelwind pile driving operations (propagation model of Norro *et al.* 2013)

Distance to source (in km)	Pile driving sound level (L_{z-p} in dB re 1µPa)	Pile driving sound exposure level (SEL$_{ss}$ in dB re 1µPa^2 s)
1	196	172
2	187	163
10	168	144
20	159	136
30	155	131
45	150	126
55	147	124

4. Discussion

4.1. Spatial extent of deterrence

Elevated levels of underwater sound can affect harbour porpoise in several ways ranging from injury and death to discomfort and the masking of communication (Kastelein & Jennings 2012). While the thresholds for these impacts are as of yet unknown, it is well-established that porpoises will temporarily vacate too noisy areas even if these are otherwise suitable (Culik *et al.* 2000). In this study, we observed a reduction in detections of porpoises at stations up to 10-20 km from the location of the piling event. We extrapolated that pile driving sound levels at this distance were ~159 dB re 1µPa (L_{z-p}) (SEL$_{ss}$ = 136 dB re 1 µPa^2 s) which is close to the threshold level for major disturbance for harbour porpoise proposed by Bailey *et al.* (2010). Previously, Haelters *et al.* (2013) using aerial survey data, found decreased porpoise densities up to 20 km from the piling event. The observed spatial extent of deterrence is consistent with the results of similar research in other parts of the North Sea (Brandt *et al.* 2011; 2016; Tougaard *et al.* 2006; 2009)

At larger distances, porpoise detections either remained the same or increased slightly, which may be due to displaced porpoises entering the area. In the German waters, Dähne *et al.* (2013) showed a negative impact of pile-driving on relative porpoise detection rates at distances less than 10.8 km and increased detection rates were at 25 and 50 km distance, suggesting that porpoises were displaced towards these positions.

4.2. Temporal extent of deterrence

Inside the work area detections decreased well before the start of piling works. This is in line with results from the German Bight (Brandt *et al.* 2016) and suggests that porpoises leave prior to the start of pile-driving possibly due to increased work vessel traffic sound and sonar which act as a deterrent. In fact, overall detections inside the construction area decreased throughout the entire construction period whether there was pile driving ongoing or not. This may be due to the effect of consecutive pile driving events which prevent the stabilisation of porpoise densities. However, this may also be due to seasonal fluctuations in porpoise densities with decreasing numbers in function of

time at the start of the construction period (Haelters *et al.* 2016).

With increasing distance from the pile driving event we would expect changes in porpoise detections to be less pronounced, start later, and last shorter (as in Diederichs *et al.* 2010; Brandt *et al.* 2011; 2016). However, while this appears to be correct for the stations at 15-20 km distance further stations (25-55 km distance) do not follow this trend. As argued in Tougaard *et al.* (2009), this may be due to limited data availability.

4.3. Future work

In order to more accurately assess the spatial and temporal extent of pile-driving induced deterrence of harbour porpoise we need to understand the consequences of repeated piling events. Although Thompson *et al.* (2010) suggested that the distance over which cetaceans are disturbed becomes larger with each successive piling event, no such effect was observed in the German Bight (Brandt *et al.* 2016). In our current study, we avoided this issue by selecting only those piling events where there was an interval of at least 96 h between the end of the previous piling event and the start of acoustic deterrence. However, this meant we limited ourselves to only 10 out of 51 piling events. Our next step is to use generalized additive modelling to also take into account the effects of successive piling events, seasonality, and diurnal patterns on porpoise detections.

As we gain insight into both the seasonally fluctuating porpoise densities in the BPNS (Haelters *et al.* 2016) as well as the spatial and temporal extent of pile-driving induced deterrence we can start to more accurately determine the number of porpoises affected by wind farm construction. This is part of the information we need to determine the consequences of pile-driving at (local) population scale using demography-based modelling, such as the interim *Population Consequences of Disturbances* (PCoD; Harwoord *et al.* 2014) and the *Disturbance Effects on the Harbour Porpoise Population in the North Sea* (DEPONS; Van Beest *et al.* 2015). Both models will be applied to estimate the cumulative effects of the planned piling in the BPNS and are expected to contribute to the choice of appropriate sound mitigation measures.

Acknowledgements

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