Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Assessing and Managing Effect Spheres of Influence

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

**ASSESSING AND MANAGING
EFFECT SPHERES OF INFLUENCE**

Edited by
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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 onto 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 the targeted or process monitoring, focusing on the cause-effect relationships of *a priori* selected impacts. As such, the 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 and 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 up to and including 2017.

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EXECUTIVE SUMMARY

ASSESSING AND MANAGING
THE EFFECT SPHERE OF INFLUENCE

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Today, environmental monitoring of offshore wind farms is ever more targeting an impact assessment at the larger spatial scales at which ecosystems are functioning. Observing increased densities of cod *Gadus morhua* at the erosion protection layer of wind turbines for example needs to be put into the wider perspective of a rapidly increasing number of offshore wind farms within the geographic distribution of the species to assess the impacts onto the population dynamics of the species. Similarly, the threat of invasive non-indigenous species present in the intertidal zone of wind turbines or the possible impact on the status of harbour porpoise *Phocoena phocoena* populations can only be assessed in the same cumulative perspective. With 238 km² reserved for offshore wind farms in Belgium (*i.e.*, by the end of 2018: 274 offshore wind turbines representing an installed capacity of 1152 MW, Chapter 1), 344 km² in the adjacent Dutch Borssele zone and 122 km² in the French Dunkerque zone, cumulative ecological impacts are likely to form a major concern in the coming years. While the importance of an upscaling of locally observed effects hence is widely recognised, the key to such upscaling still is based on an understanding in depth of what happens at the wind turbine or wind farm scale. Local scale effects indeed are at the basis of possible knock-on impacts onto the wider ecosystem. Furthermore, an eventual mitigation of unwanted impacts through management measures will also most likely take place at this local scale. At-source mitigation generally is considered a better option than *e.g.* compensation of eventual ecological damage.

When considering the local scale effects, knowledge about the extent of the sphere of influence is indispensable. The sphere of influence comprises four dimensions, *i.e.*, the two horizontal dimensions (distance from source of disturbance), the vertical spatial dimension (throughout the water column and in the air) and time (seasonal and yearly variation). In this report, new findings with regards to the extent of sphere of influence and its possible management are presented. These findings can roughly be allocated to (1) assessing the effect sphere of the wind turbines, (2) assessing the effect sphere of the wind farm and (3) managing the sphere of influence of offshore wind turbine construction.

Assessing the sphere of influence of offshore wind turbines

At the smallest spatial scale, the sphere of influence of an individual wind turbine is expected to comprise the biofouling and direct hard substrate-associated communities, and the surrounding (often soft sediment and less mobile) communities that are affected by the presence of the structure and its biofouling communities. This sphere of influence could actually be called the first-order artificial reef effect.

Artificial hard substrates are known to be attractive to many hard substrate-associated species, among which several fish species. For the latter, these offshore structures provide shelter, suitable habitat and a source of food. In Belgium, a total of 25 fish species were observed in the immediate vicinity of the wind turbines, 15 of which are also known to dwell around wrecks (Chapter 6). In contrast to that of the surrounding sandbanks, the fish community around wind turbines may hence be considered relatively unique. Four species, the tadpole fish *Raniceps raninus*, the tompot blenny *Parablennius gattorugine* and the longspined bullhead *Taurulus bulbalis* were previously rarely or, in the case of the ballan wrasse *Labrys bergylta*, only once reported from Belgian waters. This, however, does not necessarily mean that they are rare. Most of the obligate hard substrate fish species that were observed are frequently recorded in the oyster beds and boulder fields of the nearby Eastern Scheldt estuary. Sampling efforts, designs and techniques all co-determine the perceived rarity of a species. We show that, in order to obtain a good insight into the fish fauna dwelling hard substrates, the use of a suite of varied sampling techniques is necessary. We expect that hard substrate-frequenting fish species will increasingly benefit from the continued expansion of offshore wind farms in the Southern North Sea. Because these communities primarily consist of hard substrate-associated species, the immediate sphere of influence is limited to the spatial extent of artificial hard substrate.

The sphere of the first-order artificial reef effect also comprises the surrounding (often soft sediment) communities that are impacted by *e.g.* the deposition of faecal pellets from the biofouling communities, altered hydro- and hence morphodynamics, and/or increased predation pressure by attracted fish. The extent of this sphere of influence still is under investigation by the scientific community and depends on *e.g.* the communities under consideration and, the size and age of the artificial structure. We compared the soft sediment macrobenthos (*i.e.*, the fauna retained on a 1 mm mesh-sized sieve and inhabiting the soft sediments) at 350-500 m away from the artificial structures with that close by (37.5 m) (Chapter 5). Turbine-related effects were detected at close distances from jacket-based foundations at the Thornton Bank with fining (median grain size: 343 ± 22 μm *vs* 378 ± 49 μm) and organic enrichment (total organic matter content: $0.72 \pm 0.39\%$ *vs* $0.53 \pm 0.17\%$) of the sediment together with higher macrofaunal densities (934 ± 1112 *vs* 343 ± 329 ind. m^{-2}), diversity (number of species: 18 ± 9 *vs* 8 ± 4, diversity: 1.92 ± 0.46 *vs* 1.57 ± 0.44) and shifts in communities at close distance and not further off. The sphere of influence hence stretches out to at least some 40 m, but less than 350 m, from the jacket-based turbines (*i.e.*, ~5000 m²). In contrast, effects around monopile foundations at the Bligh Bank were significantly different between close by and further off sampling locations for community composition only. These contrasting results might be due to a combination of structural differences (*in casu* jacket *vs* monopile foundations) but also site-specific (*in casu* transitional *vs* offshore waters), justifying further research into the subject of the sphere of influence onto soft sediment macrobenthos. Site specific differences are exemplified by a clear north-south gradient within the wider offshore wind farm area for both soft sediment epibenthos and

demersal-benthopelagic fish assemblages (*i.e.*, larger fauna living on the soft sediments) (Chapter 4). The concession area closest to shore (ca. 23 km) exhibited much higher densities (1200 *vs* ca. 80 ind. 1000 m^{-2} for epibenthos and 120 *vs* 25 ind. 1000 m^{-2} for fish) and biomass (3900 *vs* 180 g WW 1000 m^{-2} for epibenthos) and also community structure differed from the more offshore concession areas. The area close to the shore is inhabited by an assemblage most related to a typical coastal community, while further offshore a typical offshore assemblage prevails.

When further considering the epibenthos and demersal-benthopelagic fish trends, remarkable was that two epifaunal animals, *i.e.*, blue mussels *Mytilus edulis* and anemones Anthozoa spp. known to be fouling on the foundations, were quite abundant in soft sediment samples collected in one of the investigated wind farms (resp. 5 and 3 ind. 1000 m^{-2}). Both were totally absent or present in much lower densities (resp. 0.04 and 0.3 ind. 1000 m^{-2}) in the reference locations outside the offshore wind farms (Chapter 3). This could indicate that the 'reef' effect is starting to expand beyond the direct vicinity of the turbines, as such expanding the sphere of influence with time. However, a detailed follow-up would be needed to validate whether this is a one-off observation or a persistent wind farm effect reflecting the effect of time after construction. Overall, no direct wind farm effect, nor indirect fisheries exclusion effect was yet observed for the soft-bottom epibenthos and demersal-benthopelagic fish assemblage in 2017. Aside from the difference for blue mussels and anemones, species composition, species number, density and biomass (for epibenthos only) of the soft-bottom assemblage inside the offshore wind farms remained very similar compared to the assemblage in reference locations. The epibenthic and demersal-benthopelagic fish species originally inhabiting the soft sediments of both offshore wind farm areas remain dominant.

Another example of the sphere of influence of individual turbines is given by bats exploring and migrating across the marine environment. Several bat species known to migrate long distances between summer and winter roosts also cross the North Sea and may hence encounter offshore wind farms. The developments of offshore wind farms in the North Sea therefore represents a potential risk for migrating bats. To investigate the altitude-specific activity of bats at sea and as such the risk of collision, we installed eight acoustic bat detectors at four turbines in the wind farm on the Thornton Bank (Chapter 9). Four were installed on the platform of the transition piece (17 m above mean sea level, amsl) and four were installed on the nacelle of the turbines in the centre of the rotor swept area (94 m amsl). A total of 98 recordings of bats were made by all eight Batcorders during 19 different nights during the entire study period (from the end of August 2017 until the end of November 2017). The detections at nacelle height were only ~10% of the detections made at low altitude. The observations made by the detectors at nacelle height give a first indication of the activity of bats at that altitude. Given the limited detection range of the detectors, this does not yet allow to make sound conclusions about the collision risk for bats, especially not in the lower part of the rotor swept zone. Therefore, there is a need for studies assessing bat activity at the entire rotor swept zone.

Assessing the sphere of influence of offshore wind farms

The sphere of influence for other, often more mobile species is less likely to be concentrated at the scale of a single wind turbine, as was observed for macrobenthos, epibenthos and demersal-benthopelagic fish, but rather at the spatial scale of a wind farm or a multitude of wind farms. This second-order artificial reef effect particularly holds true for marine mammals but also seabirds that may be attracted to the offshore wind farms

because of *e.g.* improved foraging conditions and availability of roosts.

With regards to seabirds, we analysed GPS data of lesser black-backed gulls *Larus fuscus* caught and tagged in the colonies at Ostend and Zeebrugge (Chapter 7). Three modelling exercises were performed to study the response of lesser black-backed gulls to a Belgian offshore wind farm at a fine spatial scale. These exercises confirmed that much more time was spent roosting on outer than on inner turbines located 500 m from the wind farm edge (2.5 *vs* 0.5 h per turbine). Next, we found a significant and gradual increase in the number of logs of flying birds going from the centre of the wind farm (~0.6 logs) up to 2000 m from the wind farm edge (~1.4 logs), beyond which the response seemed to stabilise. For non-flying birds too, the model showed a minimum number of logs (~0.5) in the centre of the wind farm and a flattening of the smoother at about 2000 m (~3.5 logs), yet with a spike of increased presence right at the wind farm's edge, representing birds roosting on the outer turbine foundations. The last model aiming to assess temporal variation in the presence of lesser black-backed gulls in and around the Thornton Bank offshore wind farm showed that the birds were increasingly wary entering the wind farm during times of strong winds (> 14 m/s) with fast moving rotor blades. The results of this study illustrate that the sphere of influence of offshore wind farms to lesser black-backed gulls is subject to both temporal and (within-offshore wind farm) spatial variation, which can be used to further refine collision risk models.

Managing the sphere of influence of offshore wind turbine construction

With a proper understanding of the (negative) effects, mitigation measures to directly manage the sphere of influence can be designed. For offshore wind farms, the production of high levels of impulsive underwater sound, when large steel turbine foundations are hammered into the seabed, is one of the most pertinent stressors for *e.g.* marine mammals. Sound mitigation measures recently became mandatory for such pile driving activities in Belgian waters.

In 2017, during construction of the Rentel wind farm, a single big bubble curtain (BBC) was used as sound mitigation measure. With BBC deployed, the zero to peak sound level (L_{z-p}) normalized to 750 m distance from the source and ranged from 185 to 194 dB re 1µ Pa (for 7.8 m diameter steel monopiles, 4000 kJ max. hydraulic hammer) (Chapter 2). L_{z-p} was estimated to have been reduced with maximum 11-13 dB re 1 µPa by the BBC compared to the extrapolated values of L_{z-p} that would have been produced in case of absence of sound mitigation. Therefore, the efficiency of the BBC was assessed to be in the lower range of the values that can be found in literature. More than one mitigation measure will thus be needed for future projects to comply with the Belgian Marine Strategy Framework Directive requirements (L_{z-p}: max. 185 dB re 1µ Pa) and hence reduce the effects of underwater impulsive sound to ecologically acceptable levels.

Although not enough to comply with the Belgian standards, current sound mitigation measures will have reduced the extent of the effect sphere of influence. Ecological damage can further be limited by a careful timing and preparation (*e.g.*, acoustic deterring device, ADD) of piling activities. We therefore tested seventeen 'mitigation' scenarios for the effects of the likely construction schedules for three future Belgian wind farms onto the harbour porpoise; this with and without various mitigating measures (Chapter 8). The interim Population Consequences of Disturbance (iPCOD) model was used to quantify how differences in regulatory regimes with regards to offshore wind farm construction impact a simulated harbour porpoise population. The impact of pile driving on the harbour porpoise

population proved to be strongly influenced by the timing of the activities, because of the seasonal changes in spatial distribution of the species. Regardless of timing however, the impulsive sound effect sphere of influence is reduced (by up to 90%) when noise mitigation measures such as BBC and/or a noise mitigation screen are in place. The combination of a seasonal pile driving restriction and an ADD alone was not enough to lower the additional risk of a 5% decline of the porpoise population to less than 10%. Our results further suggest that building a wind farm every year would negatively affect the harbour porpoise population more than constructing two wind farms at the same time.

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. By the end of 2018, an installed capacity of 1152 Megawatt (MW), consisting of 274 offshore wind turbines, will be operational in the Belgian part of the North Sea (BPNS). Four other projects are scheduled for the next few years after that. With 238 km² reserved for offshore wind farms in Belgium, 344 km² in the adjacent Dutch Borssele zone, and 122 km² in the French Dunkerque zone, cumulative ecological impacts are likely to form a major concern in the coming years. These anticipated impacts both positive and negative, triggered an environmental monitoring programme focusing on various aspects of the marine ecosystem components, but also on the human appreciation of offshore wind farms. This report provides an overview of the offshore renewable energy development in the BPNS.

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 BPNS are expected to make an important contribution to achieve that goal.

With the Royal Decree of 17 May 2004, a 264 km² area within the BPNS is reserved for the production of electricity from water, currents or wind. It is located between two major shipping routes: the north and south traffic separation schemes. In 2011, the zone was adjusted on its Northern and Southern side in order to ensure safe shipping traffic in the vicinity of the wind farms. After this adjustment the total surface of the area amounted to 238 km² (fig. 1).

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
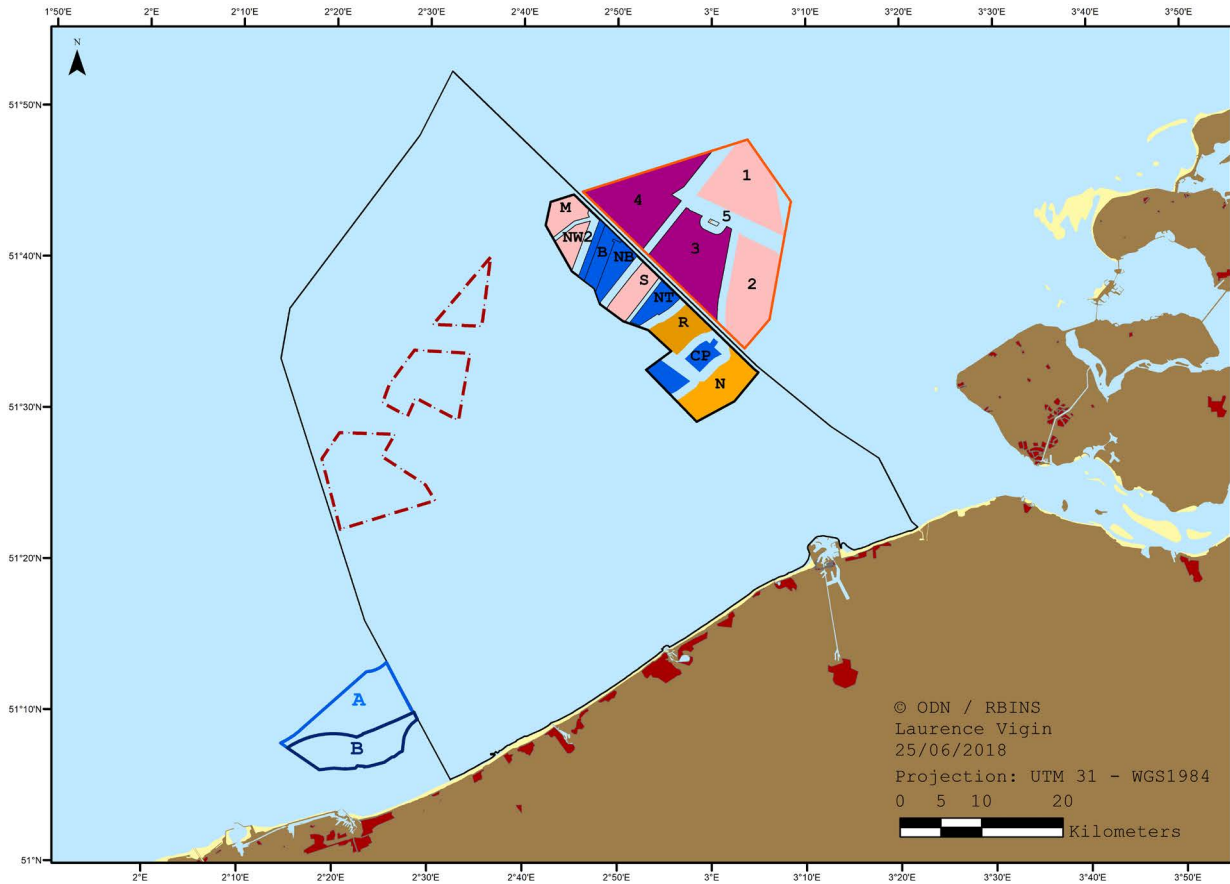
Figure 1. Current and planned zones for renewable energy in and around the Belgian Part of the North Sea with indications of wind farms that are operational (blue), currently under construction (orange) or set to start construction in 2019 (pink) or 2020 (purple). The proposed sites for the Dunkerque offshore wind farm are indicated by A & B. Locations of the new renewable energy zone, as proposed in the draft of the marine spatial plan 2020-2026, are shown by the dashed lines.
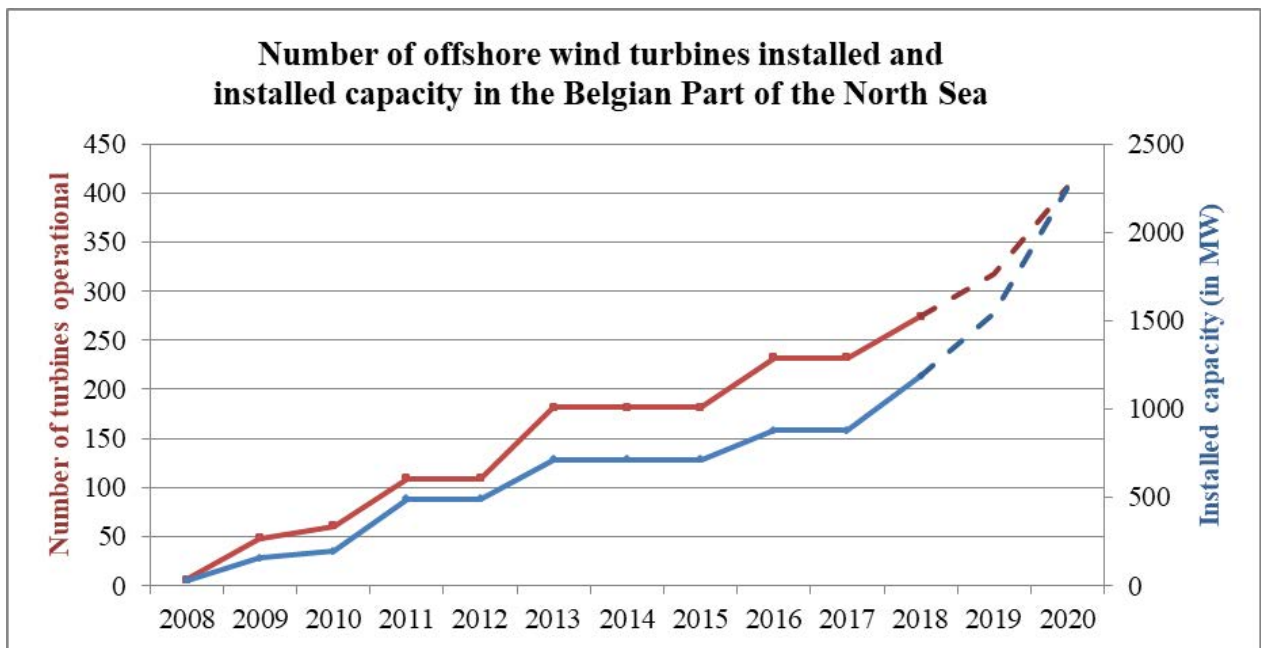
Figure 2. Number of offshore wind turbines installed and installed capacity in the Belgian Part of the North Sea since 2008.

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.

At present, nine projects were granted a domain concession and an environmental permit (from South to North: Norther, C-Power, Rentel, Northwind, Seastar, Nobelwind, Belwind, Northwester II & Mermaid (table 1). When all Belgian wind farms are built, there will be a little over 400 wind turbines in the Belgian part of the North Sea (fig. 2). The entire area with its 9 parks will have a capacity of 2250 MW and cover up to 10% of the total electricity needs of Belgium or nearly 50% of the electricity needs of all Belgian households.

The environmental permit includes a number of terms and conditions intended to minimize 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. Based on the results of the monitoring programme, and recent scientific insights or technical developments, permit conditions can be adjusted. As a result, in 2017, the Rentel project became the first project in Belgium to use noise mitigation (a big bubble curtain) during construction (hydraulic pile driving) to reduce the impact of underwater sound on marine mammals (fig. 3).

Figure 3. Big Bubble Curtain active during the construction of Rentel.

On 13 May 2017, the NEMOS project received an environmental permit for the construction and exploitation of a temporary research structure for wave energy conversion, at a distance of about 500 m 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 was scheduled to be dismantled and removed. However, on 19 January 2018, the POM West-Vlaanderen introduced a request for an environmental permit to continue the exploitation of this maritime innovation and development platform until 2033.

Table 1. Overview of wind farms in the Belgian part of the North Sea (situation on 20 May 2017)

Project		Number of turbines	Capacity (MW)	Total capacity (MW)	Concession obtained	Environmental permit obtained	Status
C-Power	phase 1	6	5	325	YES	YES	Phase 1 operational since 2009
	phase 2 & 3	48	6.15		YES	YES	Phase 2 and 3 operational since 2013
Belwind	phase 1	55	3	171	YES	YES	Phase 1 operational since 2011
	Alstom Demo project	1	6		YES	YES	Demo turbine operational 2013
Nobelwind		50	3.3	165	YES	YES	Operational since 2017
Northwind		72	3	216	YES	YES	operational since 2014
Rentel		42	7.35	275	YES	YES	Construction started July 2017
Norther		44	8	320	YES	YES	Construction started July 2018
Seastar		30	8.4	252	YES	YES	Construction foreseen to start in 2019
Mermaid		28	8.4	235.2 + 5*	YES	YES	Construction foreseen to start in 2019
Northwester 2		23	9.5	218.5	YES	YES	Construction foreseen to start in 2019

* including 5 MW of wave energy

2. Beyond 2020: the marine spatial plan 2020–2026

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. The current marine spatial plan is valid for a period of six years and thus in 2020 a new plan will come into effect. This new plan will include a proposal for a new large area for renewable energy (fig. 1). Storage of energy and grid reinforcement continue to be major hindrances to the further integration of renewables into the electricity grid and locations are foreseen for a possible energy atoll and reinforcing the offshore electricity grid.

3. Grid reinforcement and the Modular Offshore Grid (MOG)

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. This project was completed in November 2017.

The currently 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 with the MOG expected to be operational by September 2019.
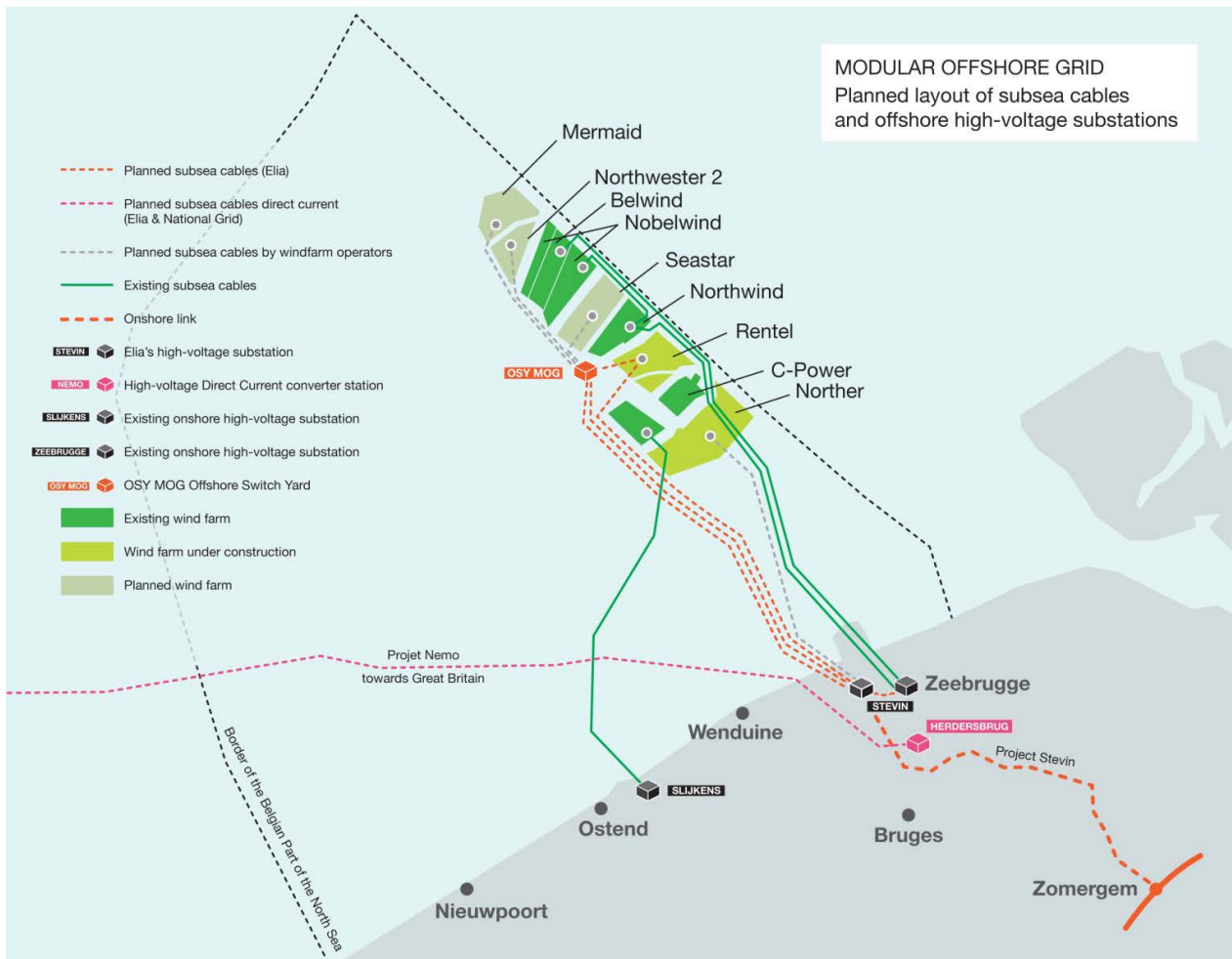
Figure 4. Design for the Modular Offshore Grid (MOG) (source: http://www.elia.be).

CHAPTER 2

ON THE EFFECTIVENESS OF A SINGLE BIG BUBBLE CURTAIN AS MITIGATION MEASURE FOR OFFSHORE WIND FARM PILING SOUND IN BELGIAN WATERS

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Abstract

The construction works of the Rentel wind farm off the Belgian coast was monitored for the emission of energy into the sea by means of underwater sound (pressure). Thirteen complete piling events were monitored, covering the driving to full depth of 13 steel monopiles of 7,8 m diameter using a hydraulic hammer with a maximum power of 4000 kJ. Sound mitigation in the form of a single big bubble curtain (BBC) was used. Measured zero to peak level (L_{z-p}) normalized to 750 m distance from the source showed values ranging from 185 to 194 dB re 1µ Pa at the end of the piling event when maximal hammer energy is used (2100-4000 kJ). The efficiency of the BBC is in the lower range proposed by the literature with a reduction of a maximum of 11-13 dB re 1 µPa (L_{z-p}). More than one mitigation measure should be used simultaneously in order to comply with the Belgian Marine Strategy Framework Directive requirements for such project.

1. Introduction

The size of commercially available wind turbines has increased in the last decades. Whereas in 1991, the first offshore wind farm used 450 kW turbines currently projects typically use 8 MW or larger turbines. The offshore wind energy sector has had to adapt turbine foundation design in order to keep up with this increase in size. Taking into account the cost and construction time, the followed option has been by increasing the size of the monopile foundations. However, more powerful hammers are required to drive such XL or XXL steel monopiles into the seafloor. As a result, higher levels of impulsive sound are introduced into the marine environment raising concerns about possible negative impacts on marine life (*i.e*, Popper & Hawkins 2012; 2016). In absence of mitigation measures, pile driving of an 8 m diameter monopile would emit impulsive underwater sound zero to peak levels (L_{z-p}) of about 204 dB re 1µ Pa at 750 m distance from the source (ITAP model, see below). The reduction of the generated sound by sound mitigation measures hence no longer is an option but compulsory; this given

the need to respect maximum admissible levels of sound defined at national level for the Marine Strategy Framework Directive (MSFD). For Belgium, that limit is set at a maximum L_{z-p} of 185 dB re 1μ Pa at 750 m from the source. In Germany, maximum L_{z-p} must stay below 190 dB re 1 μ Pa at 750 m and sound exposure level (SEL) must be below 160 dB re 1 μ Pa^2 s at 750 m from the sound source.

In 2017, the company Rentel built a new wind farm off the Belgian coast located between the C-Power and the Northwind wind farms. This new wind farm consists of 42 monopiles with a diameter ranging from 7.5 m (10 piles) over 7.8 m (26 monopiles) to 8 m (6 monopiles). For this project, the noise mitigation measure proposed and used was a single big bubble curtain (BBC).

The purpose of this report is (1) to quantify the emitted underwater sound during piling events and (2) to assess and evaluate the efficiency of the noise mitigation measure (insertion loss).

2. Material and methods

2.1. Research strategy

Underwater sound generated by driving an 8 m diameter XXL steel monopile into the seabed while applying a big bubble curtain as sound mitigation measure was measured *in situ* during construction. The L_{z-p}, the sound exposure levels of a single stroke (SEL_{ss}) and the cumulative sound exposure levels (SEL_{cum}) were computed. The effectiveness of the sound mitigation measure was assessed comparing the measured value on site with the theoretical figures obtained for such monopile diameter using the ITAP model.

2.2. Construction activities

The first steel monopile of the Rentel offshore wind farm was installed on 21 July 2017 (RC03) and the last one was piled on 23 September 2017 (RD05). During construction, underwater sound was recorded by means of a moored station during 13 complete pile driving events. A Hydro hammer S-4000 from IHC IQIP (4000 kJ) was deployed from the jacking-up platform Innovation.

Sound mitigation was in place in the form of a single big bubble curtain (BBC) of 700 m long (table 1). The flow of air was provided by eight oil-free compressors (AC PTS 916) of 40.3 m^3 min^{-1} each at a maximum pressure of 10 bar.

BBCs are expected to reduce the sound levels by 14 dB L_{z-p} (range 11-17 dB) or 11 dB SEL (range 9-13 dB) (OSPAR 2014). The best sound reduction is achieved with an optimal air supply and BBC design, *e.g.*, distance between holes and dimension of the holes (OSPAR 2014; Nehls *et al.* 2015).

2.3. Underwater sound measurement equipment

Underwater sound was recorded from a moored station (figs 1 & 2). The mooring was equipped with a measuring chain consisting of

Table 1. Specificities of the single big bubble curtain, nozzle hose (as provided by the concessioner)

Inner diameter hose	102 mm
FAD (Free Air Delivery)	8x40 m³/min/compressor 320 m³/min
FAD per meter	0,44 m³/m/min
Diameter holes	2 mm every 100-300 mm

an acoustic release (Benthos 866 A/P), one underwater sound recorder (RTsys EA-SDA14), one hydrophone (B&K 8104 or HTG) and a flotation device used to maintain the systems upright and tied. One additional acoustically commanded pop-up buoy (Benthos 875-PUB) was used for recovery of the mooring block afterwards. A wood block (fig. 2) was used to assure a silent behavior of the pop-up buoy in case of strong tidal currents and wave action. The manufacturer 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 mooring was deployed on 14 July 2017 from RV Belgica at the position WGS84 N 51° 35,129; E 002° 56,037. The mooring was retrieved on the 25 August 2017 when another one was placed at the position WGS84 51° 35,114 N, 002° 56,04 E. As such, the distance between the measuring equipment and the piling locations ranged from 808 to 4691 m. No

Figure 2. Underwater sound measuring chain prior to deployment from RV Belgica. A C-POD (acoustic porpoise detector) is added to the mooring (next to the acoustic release). A pop-up buoy (far end) is used for recovery of the concrete block. Photograph by A. Norro.

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 mooring.

2.4. Underwater sound measurements and post-treatment

Sound pressure was recorded continuously at a sampling rate of 78125 Hz and stored on a hard drive coded on WAV format. During the period of deployment, 13 piling events occurred (table 2).

MATLAB was used for the post treatment of the records. SEL_{ss}, SEL_{cum} as well as the normalization of the sound levels to the reference distance of 750 m were computed following the material and methods section of Norro *et al.* (2013). Because the intensity of the sound depends on the size of the sound
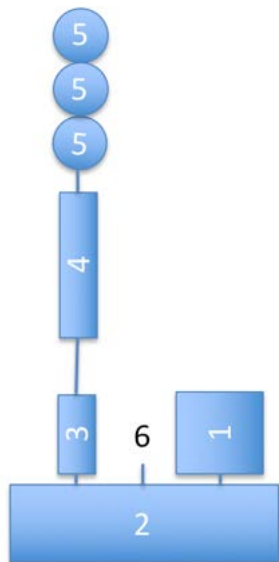

Figure 1. Mooring design of the underwater sound measurement equipment. 1 pop-up buoy (acoustic command), 2 concrete blocks 300 kg, 3 acoustic release, 4 underwater noise recorder & hydrophone, 5 rigid flotation (total 650 N buoyancy), 6 attachment for deployment, total height 3 m, all links in stainless steel cables 8 mm.

Table 2. Sound data available for the Rentel wind farm piling phase. Position of the monopile and instrument, monopile diameter, distance from the monopile to the measuring equipment (instrument), RTsys1 position for the first measuring chain deployment, RTsys2 position for the second deployment

Location	Center Point Position (Coordinates) as built (WGS84)		Distance to instrument (m)	Monopile diameter (m)
	Latitude	Longitude		
B4	51°34.934' N	2°55.412' E	808	7.8
B5	51°34.670' N	2°55.874' E	872	7.8
B6	51°34.411' N	2°56.346' E	1378	7.8
B7	51°34.164' N	2°56.813' E	1974	7.5
C4	51°34.729' N	2°56.895' E	1237	7.8
C5	51°34.485' N	2°57.305' E	1869	7.8
D2	51°35.915' N	2°56.020' E	1458	7.8
D6	51°34.819' N	2°57.882' E	2196	7.8
E2	51°35.910' N	2°57.089' E	1909	8
E4	51°35.117' N	2°58.518' E	2861	7.5
F3	51°35.857' N	2°58.411' E	3064	7.8
G7	51°35.721' N	2°59.667' E	4337	7.5
G8	51°35.485' N	3°0.058' E	4691	7.5

source and associated hammer, the intensity of the sound increases with the pile diameter.

For assessing the efficiency of the sound mitigation measure, the ITAP model was used to estimate the sound levels generated by the piling without mitigation measures.

ITAP proposes a model based on past observations (Bellmann *et al.* 2017) that allows estimating both SEL and L_{z-p} from the diameter of the monopile to be driven into the seabed.

3. Results

L_{z-p} normalized at 750 m ranged, at the start of the piling when a maximum of 500 kJ was used, from 176 to 187 dB re 1µPa, while at the end of the piling when maximum energy was used, between 186 and 193 dB re 1µPa (table 3).

SEL_{ss} ranged from 165 to 173 dB re 1 µPa² s while SEL_{cum} ranged from 200 to 208 dB re 1 µPa² s (table 4). The difference between SEL_5 and SEL_{50} (the SEL percentile 5 and 50 respectively) is less than 3 dB. The total energy used for the complete piling event and the total number of strokes

are provided as additional information often requested by bio-acousticians (Hawkins & Popper 2016) in order to better evaluate cumulative effects.

Based on the ITAP model (Bellmann *et al.* 2017), piling of a 7.8 m steel monopile produce an average L_{z-p} of 204 dB re 1µPa at 750 m distance (ranging from 199 to 209 dB re 1µPa) and a SEL_{ss} of 179 dB re 1 µPa² s (range: 174-184 dB re 1 µPa² s). With a reduction of sound of about 11-13 dB re 1 µPa (table 5), the efficiency of the BBC seems to be less than the predicted 14 dB re 1 µPa (ranging 11 to 17 dB re 1µPa) and hence would be closer to the lower limit of sound reduction cited for BBCs (OSPAR 2014). The efficiency of the BBC could probably have been enhanced by an optimal setup of the device (OSPAR 2014). Hole of 1.5 mm diameter (OSPAR 2014) in the hose instead of 2 mm (table 1) may have improved the quality of the mitigation.

Stated flow of 0.44 m³ m⁻¹ min⁻¹seems to be sufficient but one should remember that the bubble curtain is not placed at the surface but below 20 m of sea water. At that depth hydrostatic pressure is three time the

Table 3. Sound zero to peak levels normalized at 750 m distance from the piling location, measured during 13 piling events at the Rentel site. Start is at the start of the piling event within the first 10 minutes; end is for the end of the piling event when the maximum energy was used. Those results include the insertion loss from a single big bubble curtain (BBC)

Location	Distance to instrument (m)	Measured level L_{zp} START (dB re 1µ Pa)	Nomalized level L_{zp} @ 750m START (dB re 1µ Pa)	Measured level L_{zp} END (dB re 1µ Pa)	Nomalized level L_{zp} @ 750m END (dB re 1µ Pa)
B4	808	176	176	192	192
B5	872	182	183	190	191
B6	1378	181	185	189	193
B7	1974	176	182	184	190
C4	1237	180	183	188	191
C5	1869	179	186	185	191
D2	1458	181	185	188	192
D6	2196	180	187	184	191
E2	1909	179	185	185	191
E4	2861	176	184	183	191
F3	3064	175	185	182	186
G7	4337	174	184	175	186
G8	4691	174	184	176	188

Table 4. Computed SEL_{ss} and SEL cumulative SEL_{cum} normalized at 750 m distance from the 13 piling locations as well as total energy provided by the hammer and the number of strokes needed for complete penetration of the monopile. Those results include the insertion loss from a single big bubble curtain (BBC)

Location	SELss @750m (dB re 1 µ Pa^2 s)	SEL CUM (dB re 1 µ Pa^2 s)	Total E (kJ)	Strokes (n)
B5	166	201	6765005	3332
B6	168	203	6046718	3071
B4	170	205	7912667	3547
C4	169	204	4819735	3247
C5	173	208	5064200	3174
E2	167	202	4987828	3035
D6	170	205	5247363	3171
B7	167	202	5573269	2986
E4	169	204	5039575	3072
F3	166	200	4480189	2784
G8	165	199	3948502	2714
G7	166	200	4517827	2562

atmospheric pressure and therefore it a reduced pressure that is present in the SSB when immerged.

In the case presented here, we observe levels that exceed MSFD value as permitted in Germany and the Netherlands (Rumes *et al.* 2016).

Table 5. Efficiency estimate of the single big bubble curtain mitigation measure based on the difference between the theoretically produced zero to peak level of sound (L_{z-p}) at 750 m (ITAP model) of 204 dB re 1μ Pa (on average) and the observed L_{z-p} normalized to 750 m

Location	Nomalized level L_{zp} @ 750m E (dB re 1μ Pa)	Observed efficiency of the BBC (dB re 1μ Pa)	Distance to instrument (m)
B4	192	12	808
B5	191	13	872
B6	193	11	1378
B7	190	14	1974
C4	191	13	1237
C5	191	13	1869
D2	192	12	1458
D6	191	13	2196
E2	191	13	1909
E4	191	13	2861
F3	186	18	3064
G7	186	18	4337
G8	188	16	4691

4. Discussion

During the first few minutes (< 10 min) of the piling, when the energy provided by the hammer is less than 500 kJ and the BBC in place, L_{z-p} is generally below 185 dB re 1μPa which is the MSFD limit in Belgium. In the following stages of a piling event, when the energy provided by the hammer is more than 500 kJ (500-4000 kJ), L_{z-p} is well above the Belgian MSFD limit.

An optimal tuning of the BBC is necessary to obtain the full efficiency of the system and to reach a reduction of about 17 dB re 1μPa for L_{z-p} (OSPAR 2014). The numbers presented in table 5 suggest that the BBC was not optimally configured or that the flow of air inside the BBC was insufficient for optimal noise mitigation (OSPAR 2014). Moreover, even with an optimized BBC, the maximum reduction is 17 dB re 1μPa and remains insufficient to reduce L_{z-p} below 185 dB re 1μPa at 750 m distance (204 - 17 = 187). For such a project more than one mitigation measures must have been used.

Nevertheless, the apparent better efficiency of the BBC observed for F3, G7 and G8 with 18 to 16 dB reduction is due to another effect. It is an under estimation of the L_{z-p} resulting from the computation of the normalized value presented at table 5.

Norro *et al.* 2013 presented in the results section a validated propagation model better suited for the Belgian part of the North Sea and that could have been used for the normalization computation. One should remember that it is generally accepted that for such a comparison of normalized value at 750 m one uses a sound propagation law on '15 log' as introduced and used by Muller & Zerbs (2011). As a result, we surmise that, when the difference between the normalization distance (750 m) and the actual distance between sound source and measurement increases, the underestimation of L_{z-p} increases as well.

Our results demonstrate that, when it is required to install XL or XXL monopiles by pile driving, it will be necessary to use a combination of at least two sound mitigation measures in order to comply with national MSFD regulations, as had been predicted by Rumes *et al.* (2017).

5. Conclusion

With L_{z-p} in excess of 185 dB re 1μPa at 750 m even with BBC sound mitigation measure in place, the BBC proved to be less effective than predicted. For future construction activities involving such XL or XXL monopiles, it will be required to combine two or more sound mitigation measures as to comply with the Belgian MSFD thresholds for impulsive underwater sound.

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CHAPTER 3

SOFT SEDIMENT EPIBENTHOS AND FISH MONITORING AT THE BELGIAN OFFSHORE WIND FARM AREA:

SITUATION 6 AND 7 YEARS AFTER CONSTRUCTION

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Abstract

Since 2005, ILVO performs beam trawl monitoring surveys to evaluate the potential effects of offshore wind farms (OWFs) on soft sediment epibenthos and demersal-benthopelagic fish. The study effort has been concentrated on the C-Power and Belwind OWFs. The time series has been investigated in detail in a previous report, therefore this chapter focuses on the results of 2017, which is resp. 6 (C-Power) and 7 (Belwind) years after construction.

No direct wind farm ('reef') effect, nor indirect fisheries exclusion effect, was yet observed for the soft-bottom epibenthos and demersal-benthopelagic fish assemblage in 2017. Species composition, species number, density and biomass (for epibenthos only) of the soft-bottom assemblage inside the OWFs were very similar compared to the assemblage in reference locations outside the OWFs. The species, originally inhabiting the soft sediments of both OWFs, remain to be dominant.

Remarkable was that two epifaunal animals, *i.e.*, *Mytilus edulis* and Anthozoa sp., known to be fouling on the foundations, were quite abundant in the C-Power OWF soft sediment samples, and totally absent or only present in much lower densities in the reference locations outside the OWF. This could indicate that the 'reef' effect is starting to expand beyond the direct vicinity of the turbines. However, detailed follow-up is needed to validate whether this is a one-off observation or a real wind farm effect reflected with time after construction possibly because of increasing epifaunal biomass on the foundations.

1. Introduction

Since 2005, ILVO performs beam trawl monitoring surveys to evaluate the potential effects of offshore wind farms (OWFs) on soft sediment epibenthos and demersal-benthopelagic fish. Construction of OWFs introduces artificial hard substrates into the typical soft bottom sandy environment in the Belgian part of the North Sea (BPNS). Introduction of these hard substrates may affect the original soft bottom epibenthos and fish assemblages between the wind turbines. This for two reasons: (1) attraction of hard substrate species (Lindeboom *et al.* 2011; Kerckhof *et al.* 2012; De Mesel *et al.* 2015;

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Coolen 2017), and (2) creation of a reef effect for epibenthic fauna and demersal and benthopelagic fish (Reubens *et al.* 2011, 2013; Stenberg *et al.* 2015). Additionally, fisheries are excluded in the area, which is another potential effect at play to induce changes on the soft-bottom assemblages (Handley *et al.* 2014).

Our study effort has been concentrated on the C-Power (54 turbines, 325 MW) and Belwind (55 turbines, 165 MW) OWFs, the first OWFs in Belgian waters. In De Backer & Hostens (2017), an update on the time series up to 2016 (resp. 5 and 6 years after construction) was given. Results so far showed a post-construction 'overshoot' of epibenthos density and biomass caused by an increase in opportunistic, scavenging species (similar as was noted in Derweduwen *et al.* 2016a). This was, however, a temporary phenomenon lasting only two years post-construction. Overall, soft sediment epibenthos and demersal-benthopelagic fish assemblages in between the turbines (at distance > 200 m) had not really changed six years after the construction of the wind turbines, and no effect of fisheries exclusion is yet observed in soft sediment epibenthos and fish between the turbines. Nevertheless, the feeding behaviour 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. For the moment, time after construction is probably still too short, and the whole OWF operational area not yet large enough to signal effects of fisheries exclusion beyond the immediate vicinity of the turbine (De Backer & Hostens 2017).

In 2017, another survey was undertaken to extend the time series. Last year, the time series was investigated in detail

(De Backer & Hostens 2017), hence this chapter focuses on the results of 2017, 6 (C-Power) and 7 (Belwind) years after construction. We compare the results observed in 2017, with the observations described in previous years (*i.e.*, no real 'reef' and fisheries exclusion effect yet on the soft sediment assemblage between the turbines) to see whether the previous conclusions remain valid or whether effects occurred in 2017 due to increased time after construction.

2. Material and methods

2.1. Sampling

Since the previous report of De Backer and Hostens (2017), one extra sampling campaign was performed in autumn 2017 with RV Belgica. Trawl samples were taken in between the wind farms (4 within C-Power and 3 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 during 15 minutes at an average speed of 4 knots over approximately 1 nautical mile. 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 infield electricity cables. Epibenthos and fish were identified, counted, measured (all fish, crabs and shrimps) and wet weighted (all epibenthos) on board. The samples that could not be fully processed on board, were frozen and further processed in the lab.

Figure 1. Overview map showing the 2017 trawl locations at the C-Power and Belwind concession area and the respective reference locations.

2.2. Data used and statistical analyses

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 this chapter, we tested wind farm effects for sampling year 2017 for two ecosystem components (epibenthos and demersal-benthopelagic fish) for the C-Power and Belwind concession separately.

For each trawl sample, univariate variables for each ecosystem component (species number, density and biomass for epibenthos only) were calculated using the DIVERSE application in Primer v6 with PERMANOVA add-on software (Clarke & Gorley 2006; Anderson *et al.* 2008). To test for significant differences in univariate variables for 2017, one-way Permanova with factor 'impact' was done on Euclidean distance resemblance matrices with unrestricted permutations of raw data. P values were, due to the restricted number of possible permutations, drawn from Monte Carlo (MC) permutations (Anderson & Robinson 2003). However, for visualization purposes and to show the extension of the time series, 2017 results were added to time series graphs, which were produced based on average values (± standard deviation) in R 3.3.3. (R Core Team 2017) using plyr (Wickham 2011) and ggplot2 (Wickham 2009) packages. For Belwind OWF, we excluded the gully samples, both in impact (ftWBB07) and reference (ftWOH01-03 and ftWBB01-03), from the univariate analyses and only included the top samples, since univariate variables are known to be higher in gully samples compared to top samples (Vandendriessche *et al.* 2009), and this could blur effect results, which we are interested in. For the multivariate analyses looking at species composition of Belwind OWF, both top and gully samples

were included. For C-Power OWF, all samples were included in both the univariate and multivariate analyses.

Multivariate data analysis was done using a multivariate model-based approach available in the package 'mvabund' (Wang *et al.* 2012) in R 3.3.3. Square root transformed multivariate species abundance data were fitted against impact using the *manyglm* function with 'negative binomial' family. The mean-variance assumption was checked by plotting residuals versus fits. Afterwards, univariate tests for each species separately can be run as well which allows looking at individual species effects. This package allows for visualization of multivariate species data against impact by using *e.g.*, boxplots.

3. Results

3.1. Epibenthos

3.1.1. Species number, density and biomass

For 2017, no significant effects in any of the univariate variables were observed, not for C-Power, nor for Belwind. Values were very similar between impact and reference samples, and much higher for C-Power compared to Belwind (table 1).

The 2017 values were added in time series graphs (fig. 2) showing average values for species richness (S), density (N) and biomass for impact and reference samples at both C-Power and Belwind (only top samples) over time for epibenthos. The trend from 2016 to 2017 for impact and reference samples is very similar, and within the boundaries of what can be expected in natural variability (fig. 2). Overall, trends over time are very similar between impact and reference samples, with the exception of the post-construction overshoot in density and biomass in the two years following construction for both OWFs (see De Backer & Hostens 2017).

Table 1. Average epibenthos species richness (S), density (N) and biomass for 2017 of both impact and reference samples in C-Power and Belwind

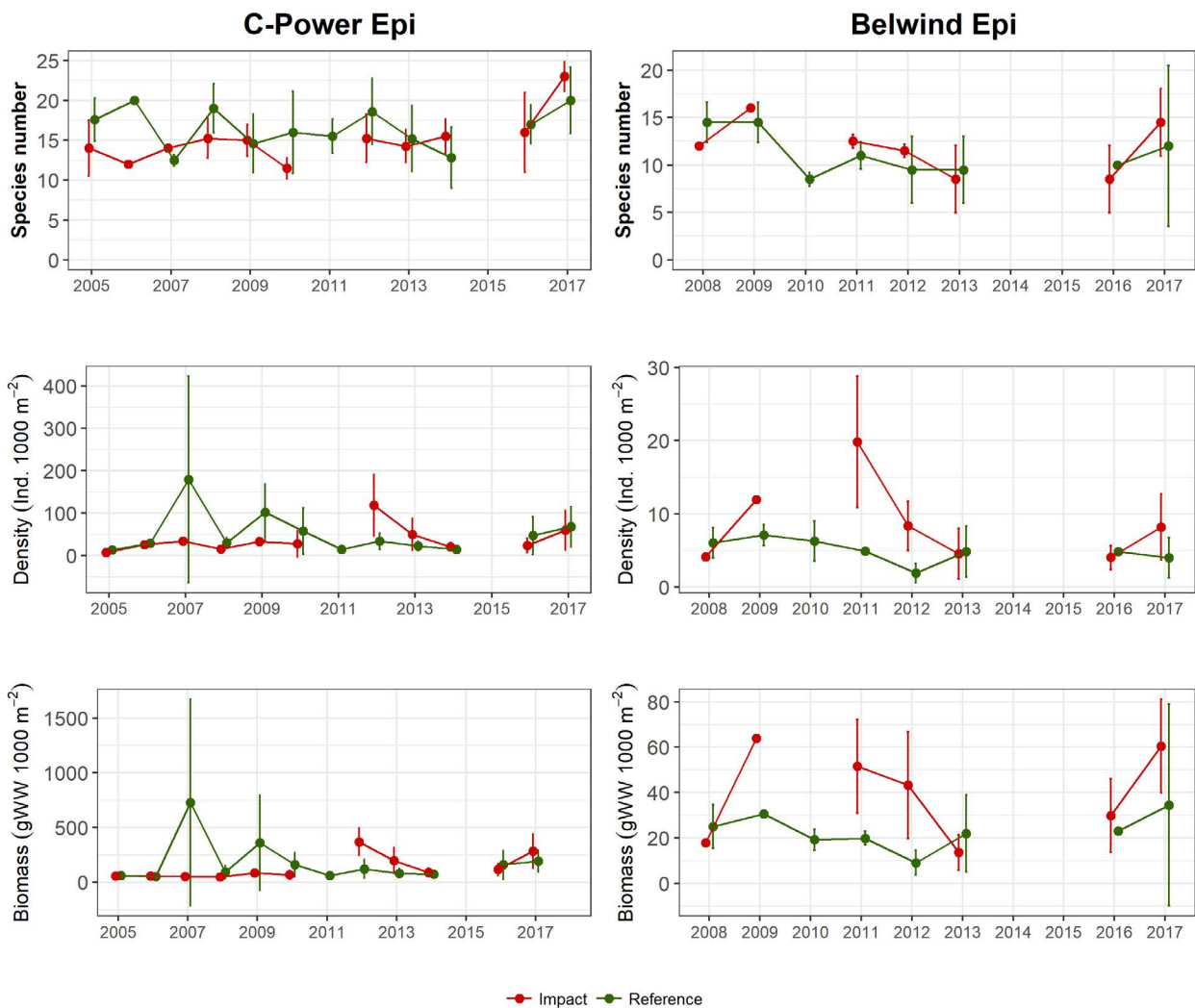| OWF	Imp/Ref	Avg. S ± SD	Avg. N ± SD (Ind. 1000 m⁻²)	Avg Biomass ± SD (g WW 1000 m⁻²)
C-Power	Imp	23 ± 2	60 ± 46	285 ± 156
	Ref	20 ± 4	68 ± 48	192 ± 100
Belwind	Imp	15 ± 4	8 ± 4	60 ± 21
	Ref	12 ± 8	4 ± 3	35 ± 44

Figure 2. Time series plots of the univariate variables species number (S), density (N) and biomass for epibenthos for both impact and reference samples at C-Power and Belwind wind farm. Average values ± SD are shown. Construction second phase C-Power in 2011, construction of Belwind in 2009-2010.

3.1.2. Species composition

The overall epibenthos species assemblage was not significantly different between impact and reference samples in 2017, not for C-Power (LRT = 39; p = 0.1), nor for Belwind (LRT = 3; p = 0.75). Looking at individual species abundances (fig. 3), occurrence and abundance was very similar between reference and impact samples. Top 3 species for C-Power were the brittle stars *Ophiura ophiura* (resp. avg. 15 and 24 Ind. 1000 m⁻²) and *Ophiura albida* (resp. avg. 13 and 22 Ind. 1000 m⁻²) and the hermit crab *Pagurus bernhardus* (resp. avg. 6 and 8 Ind. 1000 m⁻²) both in impact and reference. For Belwind, top 3 species in both impact and reference, were *Pagurus bernhardus* (resp. avg. 6 and 4 Ind. 1000 m⁻²), *Ophiura albida* (resp. avg. 2 and 1 Ind. 1000 m⁻²) and the star fish *Asterias rubens* (resp. avg. 1 and 1 Ind. 1000 m⁻²) (fig. 3).

Anemones Anthozoa and blue mussel *Mytilus edulis* were in C-Power, however, much more abundant in impact (resp. avg. 3.4 and 5 Ind. 1000 m⁻²) compared to reference samples (resp. avg. 0.3 and 0.04 Ind. 1000 m⁻²) (fig. 3). For Anthozoa, this was even significantly higher (LRT = 9; p = 0.03). For Belwind, the squid *Loligo vulgaris* was observed in higher abundances in impact (avg. 1 Ind. 1000 m⁻²) than in

reference samples (avg. 0.5 Ind. 1000 m⁻²), however, this was not at all significant.

3.2. Demersal and bentho-pelagic fish

3.2.1. Species number and density

In 2017, no significant effects for S or N were observed in neither of the two OWFs. Average values for species richness and density were very similar for both impact and reference samples (table 2).

The 2017 results were added to the time series graphs (fig. 4) for demersal and bentho-pelagic fish showing average values for species richness (S) and density (N) for impact and reference samples at both C-Power and Belwind (only top samples) over time. Both impact and reference samples show exactly the same evolution between 2016 and 2017, indicating that no wind farm effect is at play when looking at univariate variables (fig. 4). Overall, trends over the entire time series are very similar between impact and reference samples, and this for both OWFs.

3.2.2. Species composition

The overall demersal and bentho-pelagic fish species assemblage was not significantly different between impact and reference samples in 2017, not for C-Power (LRT = 3.5, p = 0.8), nor for Belwind (LRT = 9, p = 0.07).

Figure 3. Box-and-whisker-plots showing minimum, maximum, 0.25 percentile, 0.75 percentile and median sqrt densities for most abundant epibenthos species in reference (black) and impact (red) samples for C-Power and Belwind (Autumn 2017). Outliers are represented as circles. List for full species names in annex 1.

Table 2. Average fish species richness (S) and density (N) for 2017 of both impact and reference samples in C-Power and Belwind

OWF	Imp/Ref	Avg. S ± SD	Avg. N ± SD (Ind. 1000 m^{-2})
C-Power	Imp	15 ± 2	24 ± 13
	Ref	15 ± 2	19 ± 11
Belwind	Imp	11 ± 1	10 ± 2
	Ref	13 ± 0	15 ± 5

Figure 4. Time series plots of the univariate variables species number (S) and density (N) for bentho-pelagic fish for both impact and reference samples at C-Power and Belwind wind farm. Average values ± SD are shown. Construction second phase C-Power in 2011, construction of Belwind in 2009-2010.
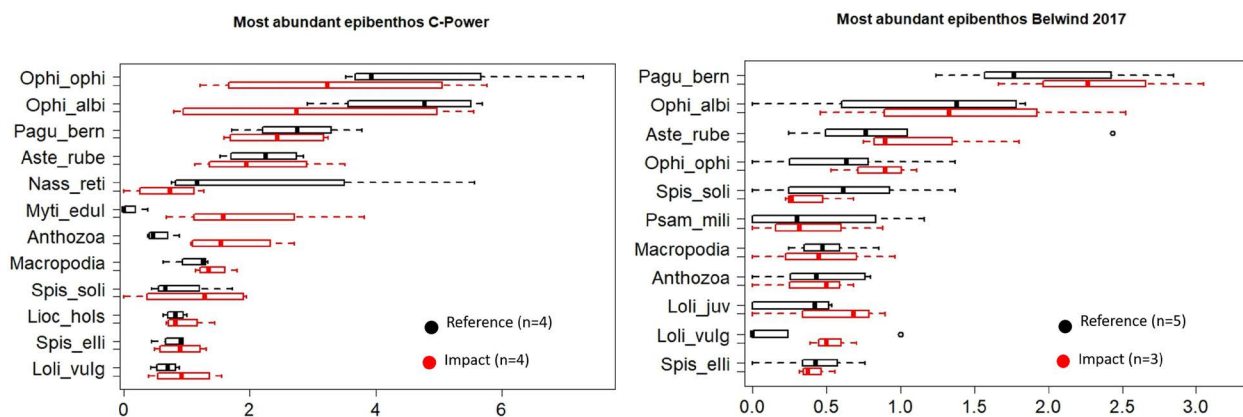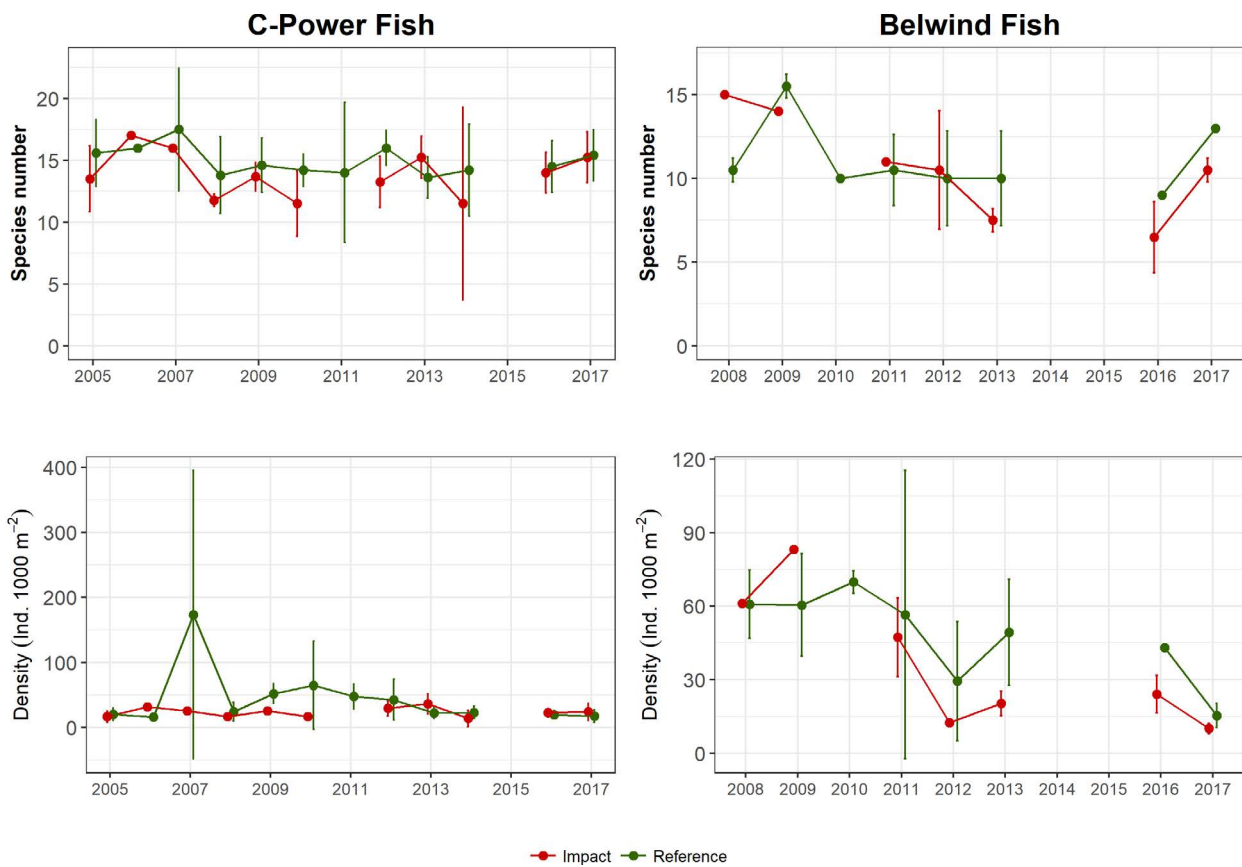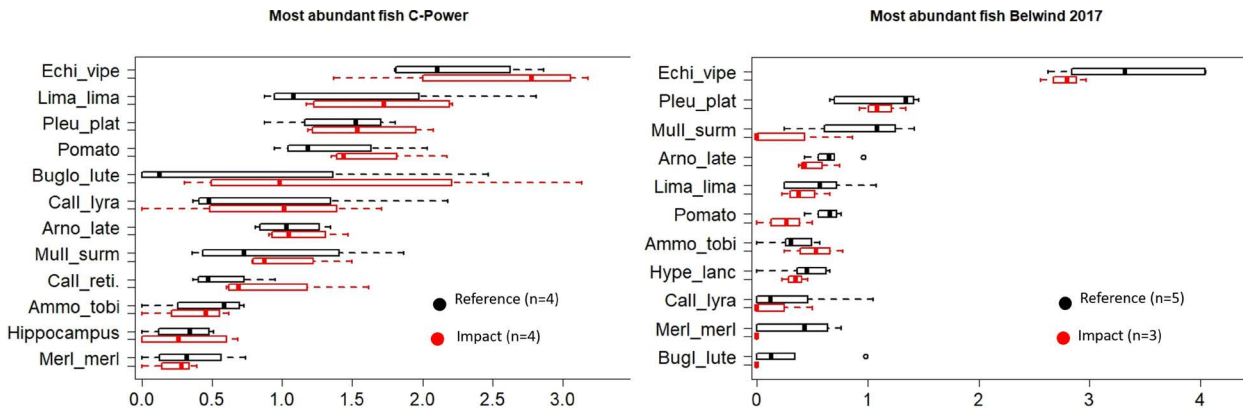
Figure 5. Box-and-whisker-plots showing minimum, maximum, 0.25 percentile, 0.75 percentile and median sqrt densities for most abundant bentho-pelagic fish species in reference (black) and impact (red) samples for C-Power and Belwind (Autumn 2017). Outliers are represented as circles. List for full species names in annex 1.

However, for Belwind, some differences in abundance could be observed, but these could mainly be attributed to the higher number of gully samples in the reference zone (3 versus 1 in impact).

For C-Power, lesser weever *Echiichtys vipera* (resp. avg. 7 and 5 Ind. 1000 m⁻²), dab *Limanda limanda* (resp. avg. 3 and 3 Ind. 1000 m⁻²) and plaice *Pleuronectes platessa* (resp. avg. 3 and 2 Ind. 1000 m⁻²) were the most dominant species in both impact and reference samples (fig. 5). Lesser weever and plaice were dominant as well in both reference (resp. avg. 12 and 1 Ind. 1000 m⁻²) and impact samples (resp. avg. 8 and 1 Ind. 1000 m⁻²) of Belwind, followed by mullet *Mullus surmuletus* in reference samples (1 Ind. 1000 m⁻²) and scaldfish *Arnoglossus laterna* in impact samples (0.5 Ind. 1000 m⁻²) (fig. 5). Other abundant species had similar density ranges for impact and reference samples in both OWFs. No significant wind farm effect was found for any of the individual species.

4. Discussion and conclusions

No direct wind farm ('reef') effect, nor indirect fisheries exclusion effect, was (yet) observed for the soft-bottom epibenthos and demersal-benthopelagic fish assemblage in 2017. Species composition, species

number, density and biomass (for epibenthos only) of the soft-bottom assemblage inside the OWFs was very similar compared to the assemblage in reference locations outside the OWFs. This is completely in line with our previous monitoring results (Derweduwen *et al.* 2016a; De Backer & Hostens 2017) and other studies *e.g.*, Stenberg *et al.* (2015), showing as well that during the operational phase of the OWF, the species originally inhabiting the soft sediments remain to be dominant.

One remarkable result in 2017 is that epifaunal animals *i.e.*, *Mytilus edulis* and Anthozoa sp. known to be fouling on the turbine foundations (Krone *et al.* 2013; De Mesel *et al.* 2015) are quite abundant in the C-Power OWF samples, and totally absent or present in much lower densities in the reference locations outside the OWF. This could be a first indication that the 'reef' effect is starting to expand beyond the direct vicinity of the turbines into the soft sediment zones between the wind turbines. Anthozoa were not identified to species level, so verification that the increase is due to the species dominant on the C-Power foundations *Metridium senile* (De Mesel *et al.* 2015) is premature but plausible. In the follow-up survey, identification to species level of Anthozoa can provide a validated answer. For *Mytilus edulis*, life mussel clumps were

observed in all samples within C-Power, most probably originating from the turbines. Survival chances of *Mytilus edulis* on mobile soft-bottoms at depths of 20 m, with high risk of burial, are probably low (Hutchison *et al.* 2016). Nevertheless, this observation is in line with the Mytilusation hypothesis (Krone *et al.* 2013), which predicted that increased mussel biomass at wind farm foundations, can produce secondary hard substrate, which may alter the soft-bottom ecosystem. Follow-up is needed to validate whether this is a one-off observation or a real wind farm effect which can increase heterogeneity in the soft-bottom sediments in between foundations. When the increased mussel occurrence between the turbines would persist, more targeted research is needed to further investigate the processes at play. This could include *e.g.*, the survival potential of these mussel clumps on the sandy bottom or the fauna which is associated with this secondary produced hard substrate.

Acknowledgements

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Annex 1

Species names with according abbreviations used in the figures in this chapter

	Species name	**Abbreviation**
Epibenthos	*Anthozoa* sp.	Anthozoa
	Asterias rubens	Aste_rube
	Liocarcinus holsatus	Liohol
	Loligo juv	Loli_juv
	Loligo vulgaris	Loli_vulg
	Macropodia sp.	Macropodia
	Mytilus edulis	Myti_edul
	Nassarius reticulatus	Nass_reti
	Ophiura albida	Ophi_albi
	Ophiura ophiura	Ophi_ophi
	Pagurus bernhardus	Pagu_bern
	Psammechinus miliaris	Psam_mili
	Spisula elliptica	Spis_elli
	Spisula solida	Spis_soli
Fish	*Ammodytes tobianus*	Ammo_tobi
	Arnoglossus laterna	Arno_late
	Buglossidium luteum	Buglut
	Callionymus lyra	Call_lyra
	Callionymus reticulatus	Call_reti
	Echiichthys vipera	Echi_vipe
	Hippocampus sp.	Hippocampus
	Hyperoplus lanceolatus	Hype_lanc
	Limanda limanda	Lima_lima
	Merlangius merlangus	Merl_merl
	Mullus surmuletus	Mull_surm
	Pleuronectes platessa	Pleu_plat
	Pomatoschistus sp.	Pomato

CHAPTER 4

DEFINING REFERENCE CONDITION (T_0) FOR THE SOFT SEDIMENT EPIBENTHOS AND DEMERSAL–BENTHOPELAGIC FISH
IN THE NORTHER AND RENTEL CONCESSION ZONES

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Abstract

In the near future, two new offshore wind farms (OWFs) will be constructed in Belgian waters, Norther and Rentel. In this chapter, we explored whether the epibenthos and demersal-benthopelagic fish assemblage in these future OWFs differed from the (reference) assemblages that are currently monitored within the WinMon program *i.e.*, for the OWFs C-Power and Belwind. Secondly, the T_0 reference conditions for both new concession zones are described. All samples were taken in autumn 2016, as such excluding both interannual variability and seasonality.

A clear north-south gradient was observed within the wider OWF area for both soft sediment epibenthos and demersal-benthopelagic fish assemblages. Norther, the concession area closest to shore (ca. 23 km), exhibited much higher density and biomass (for epibenthos), and also community structure differed from the other concession areas. Norther is inhabited by an assemblage most related to a typical coastal community, while Rentel comprises a typical offshore assemblage, comparable with Belwind and C-Power. Because of the different epibenthos and fish assemblages, a follow-up of Norther seems justified as the extrapolation of previous monitoring results from the other OWFs cannot be guaranteed. Monitoring of Rentel, on the other hand, seems redundant within the current WinMon monitoring program, as there is a high similarity with the C-power epibenthos and fish assemblages. Although, integration of Norther in the monitoring framework is recommended, it is important to consider that natural variability within this zone is very high, especially for epibenthos, which may obscure both short and long-term potential effects related to the presence of the Norther OWF, and the fisheries exclusion in this concession zone.

1. Introduction

In order to meet the targets set by the European Directive 2009/28/EC on renewable energy, the Belgian government reserved an offshore area of 238 km² for the production of electricity. Since 2017, 232 offshore wind turbines are operational in the Belgian part of the North Sea (BPNS) with an installed capacity of 877 MW. A further 309 MW is under construction in the Rentel concession area, and in 2018 construction of the Norther offshore wind farm (OWF) will start, good for another 370 MW.

Construction of OWFs introduces artificial hard substrates into the typical soft bottom sandy environment in the BPNS. These hard substrates generate a new 'rocky' habitat, attracting hard substrate species (Lindeboom *et al.* 2011; Kerkhof *et al.* 2012; De Mesel *et al.* 2015) and creating a reef effect for epibenthic fauna and demersal-benthopelagic fish (Reubens *et al.* 2011, 2013; Stenberg *et al.* 2015). This reef effect, in combination with fisheries exclusion in the wider OWF area, may affect the original soft bottom epibenthos and fish assemblages between the wind turbines.

Since 2005, ILVO performs beam trawl monitoring surveys to evaluate the potential effects of OWFs on soft sediment epibenthos and demersal-benthopelagic fish. The study effort has been concentrated on the C-Power and Belwind OWFs, the first OWFs in Belgian waters. Both OWFs are located on a sand bank, respectively on Thornton Bank and Bligh Bank, approximately 30 and 50 km offshore. Their epibenthos and fish assemblages are characterized as a typical offshore assemblage (Derweduwen *et al.* 2010). The future Rentel OWF is situated in a gully in between both OWFs, while Norther is situated closest to shore (23 km from Zeebrugge). This hints at potentially different epibenthos and demersal-benthopelagic fish assemblages. If so, results of the current impact monitoring in C-Power and Belwind may not directly be extrapolated to the future OWFs.

We sampled both future concession areas in autumn 2016:

1) to evaluate whether the soft sediment epibenthos and fish assemblages of the future OWFs (Rentel and Norther) differ from the reference zones currently monitored for the OWFs (C-Power and Belwind);

2) to determine the reference conditions (T_0) for both future OWFs (Rentel and Norther), and to evaluate the suitability of the reference locations for future impact assessments.

2. Material and methods

2.1. Sampling

Sampling for the reference condition (T_0) was performed on board RV Simon Stevin in autumn 2016. Trawl samples were taken inside the future concession areas of Rentel and Norther, and at potential reference locations outside the concession areas (fig. 1). At the location of ftNor2, a fishing vessel had been fishing right before our sampling, with a lot of dead fish and epibenthos in the sample as a consequence. This dead material was not taken into account in the analyses. Additionally, in this same monitoring survey, we also performed the impact monitoring in the Belwind and C-Power OWFs and their respective reference zones (see De Backer & Hostens 2017). On all track locations, epibenthos and fish fauna 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 during 15 minutes at an average speed of 4 knots over approximately 1 nautical mile. 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 the depth contours parallel to the coastline, thereby minimizing the depth variation within a single track, except for ftNor1 and ftNor2 within the Norther concession which are perpendicular to the coastline due to the future positioning of the infield electricity cables. On board, epibenthos and fish were identified, counted, measured (fish, crabs and shrimps) and wet weighted (only epibenthos). The samples that could not be fully processed on board were frozen and further processed in the lab.

2.2. Data used and statistical analyses

Pelagic species (based on www.fishbase.org) such as *Sprattus sprattus*, *Trachurus trachurus*, *Scomber scombrus*, next to jellyfish and polychaetes were excluded from

the analyses, as these are not quantitatively sampled with a beam trawl.

2.2.1. Community analysis of the wider Belgian offshore wind farm area

The reference locations of the Belwind and C-Power impact monitoring of 2016 were included to test for differences in the soft sediment epibenthos and fish assemblage between the future concession areas Norther and Rentel and the reference zones of the operational OWFs Belwind and C-Power (fig. 1; De Backer & Hostens 2017). This allowed for a community analysis in the wider OWF area. For this analysis, we tested the area effect for two ecosystem components (epibenthos and demersal-benthopelagic fish) in a one-way PERMANOVA design with factor 'concession area' for univariate variables (species number, density, biomass) and for community structure. Multivariate data were fourth 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 permutations of raw data. PERMDISP test was used to test for homogeneity of dispersion within groups for a correct interpretation of the PERMANOVA results. Whenever a significant 'concession' effect was found, pairwise tests were performed to determine where the differences were situated. 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 appoint the species most responsible for the observed differences.

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

2.2.2. Reference condition (T_0) for Norther and Rentel

To determine the T_0 in both future OWFs, a descriptive analysis was executed at the sample level in order to be able to observe the degree of variability between samples in one area. The number of samples (2 impact and 2 reference) was too low for a statistically sound evaluation of the suitability of the reference locations for Norther and Rentel. Therefore, univariate measures, species number, density and biomass (for epibenthos only), were calculated for each fish track, together with relative abundance of the dominant species. Univariate measures were then visualized in ArcGIS, allowing for an expert judgement on the suitability of the reference locations for future impact assessments of Norther and Rentel.
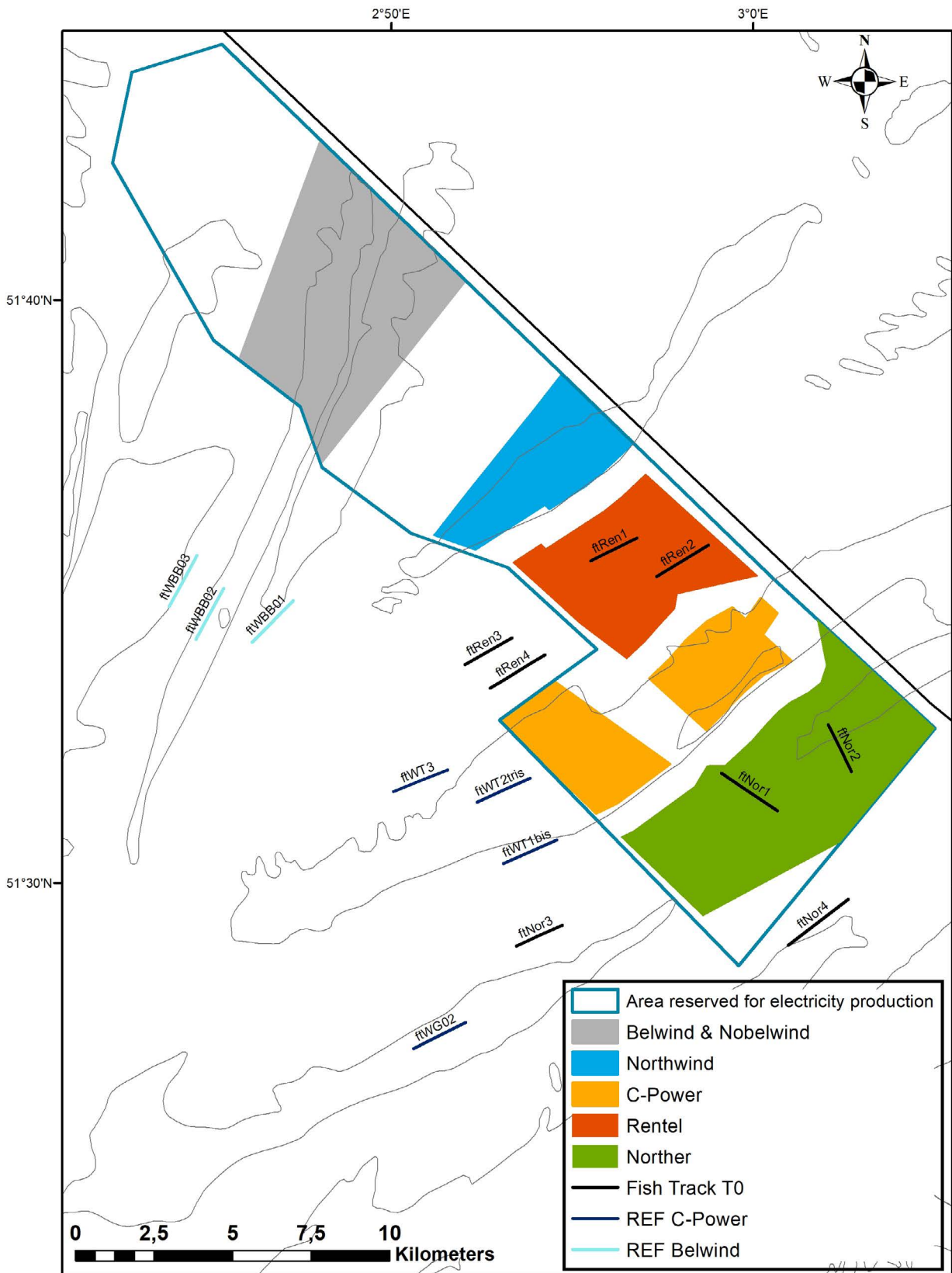
Figure 1. Overview map showing the T0 trawl locations in the Norther and Rentel concession areas and their respective reference locations (black). Dark blue are reference locations for C-Power and light blue reference locations for Belwind, which were included for the wider offshore wind farm area community analysis.

3. Results

3.1. Community analysis of the wider offshore wind farm area

3.1.1. Epibenthos

Number of species (S) per sample did not differ significantly between the four zones. Average S ranged between 15 species in Belwind and 18 species in Norther and Rentel. Density and biomass showed an overall significant effect (resp. p = 0.02 and 0.04) with a very high average density and biomass in the Norther concession zone, resp. 1212 ind. 1000 m^{-2} and 3921 g · 1000 m^{-2} (fig. 2). However, pairwise differences did not prove to be significant due to the high variation observed within the Norther samples.

Epibenthic community structure was significantly different between the different zones (p = 0.0002), and dispersion differed significantly as well (Permdisp p = 0.01).

Pairwise tests showed that Norther differed significantly from all other concession zones, both in community structure and dispersion level, indicating a high degree of heterogeneity in the Norther zone (fig. 3). Rentel and Belwind showed as well a significantly different community structure.

Norther differed significantly from the other zones (average dissimilarity with C-Power 55%, Rentel 53% and Belwind 65%) due to the high relative abundance of the sea-urchin *Psammechinus miliaris*, the star fish *Asterias rubens* and the brown shrimp *Crangon crangon*, and the lower relative abundance of the hermit crab *Pagurus bernhardus* (fig. 4). Rentel differed significantly from Belwind (avg. dissim. 41%) due to high relative abundance of the dog whelk *Tritia reticulata* and the serpent star *Ophiura ophiura*, and a lower relative abundance of Cephalopoda species (fig. 4).
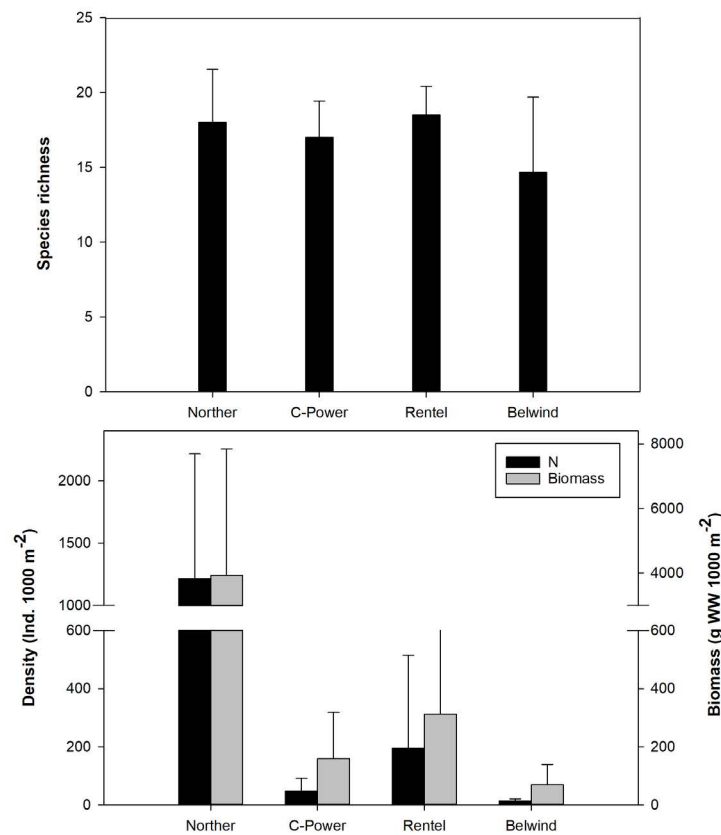
Figure 2. Bar plots showing average species richness, density and biomass (± SD) for epibenthos in the different concession/reference zones.

Figure 3. PCO plot of the epibenthos community in the wider offshore wind farm area with indication of the different concession/reference zones. Vector overlay shows the species that are best correlated (multiple correlation r > 0.35) with the observed multivariate pattern.

Figure 4. Relative abundance of the most common epibenthos species in the different concession/reference zones.

3.1.2. Demersal and bentho-pelagic fish

No significant differences in number of species (S) or density (N) were observed between the four zones. Average S ranged between 12 species in Norther and 17 species in Rentel. Average density was lowest in C-Power with 20 ind. 1000 m^{-2} and highest in Norther with 126 ind. 1000 m^{-2} (fig. 5).

Fish community structure differed significantly between the different zones (p = 0.0001), as did dispersion (Permdisp p = 0.004). Norther differed most from Belwind (average dissimilarity = 64%), but it also differed significantly from C-Power (avg. diss. = 44%) and Rentel (avg. diss. = 42%) (pairwise test p < 0.02) (fig. 6). Rentel only differed significantly in community structure from Belwind (avg. diss. = 39%).

Norther had high relative abundances of dragonet *Callionymus lyra*, whiting *Merlangius merlangus,* pouting *Trisopterus luscus* and hook nose *Agonus cataphractus* compared to the other zones. Relative abundance of lesser weever *Echiichthys vipera* was much lower (fig. 6; fig. 7). The Belwind reference zone is mainly dominated by lesser weever, while solenette *Buglossidium luteum* is characteristic for C-Power and Rentel (figs 6 & 7).
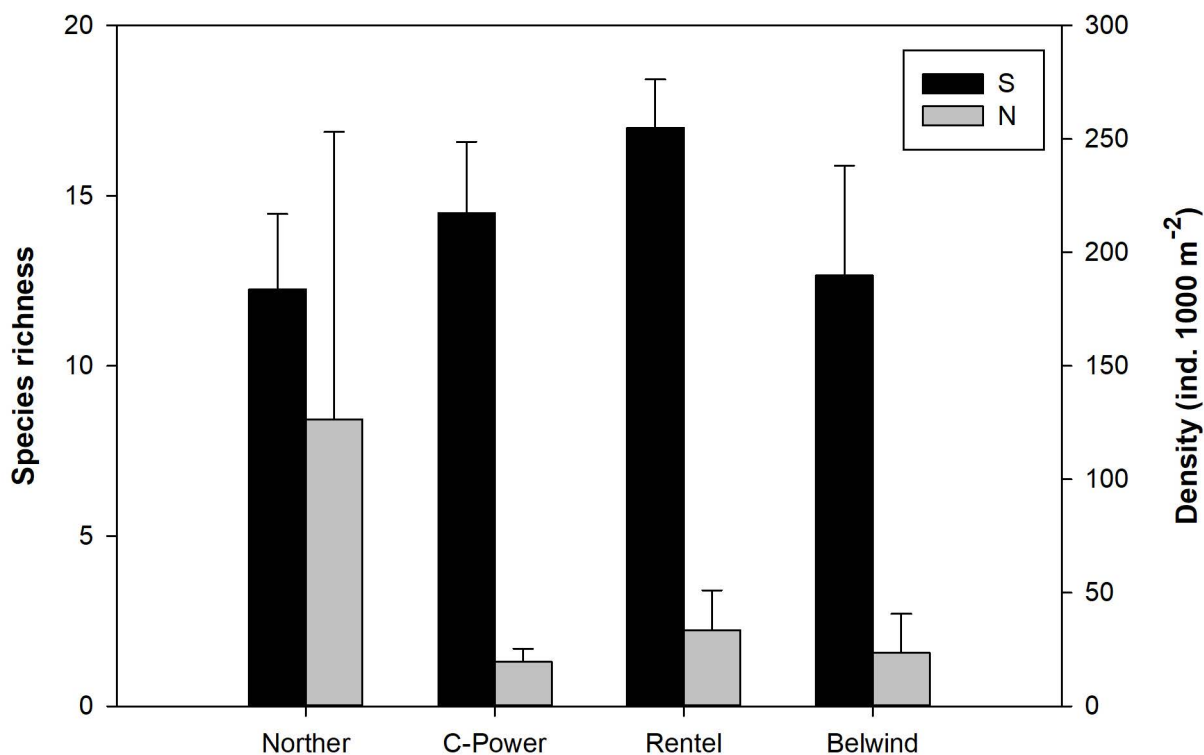
Figure 5. Bar plots showing average species richness and density for demersal-benthopelagic fish in the different concession/reference zones.

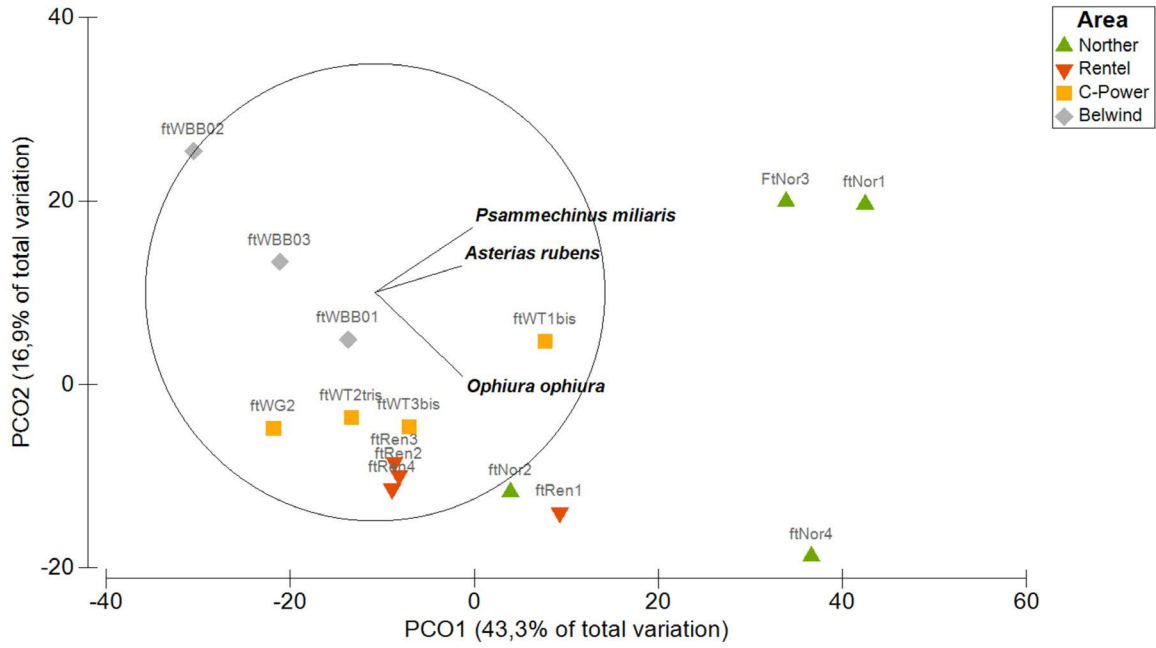**Figure 6.** PCO plot of the fish community in the wider offshore wind farm area with indication of the different concession/reference zones. Vector overlay shows the species that are best correlated (multiple correlation r > 0.4) with the observed multivariate pattern.
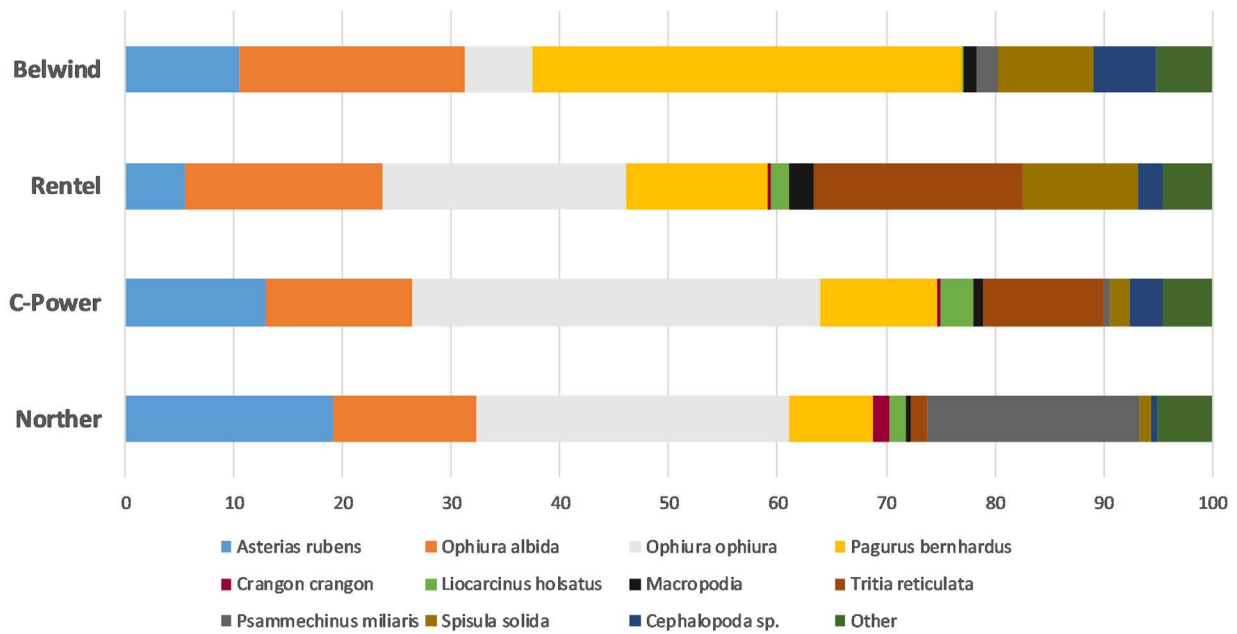

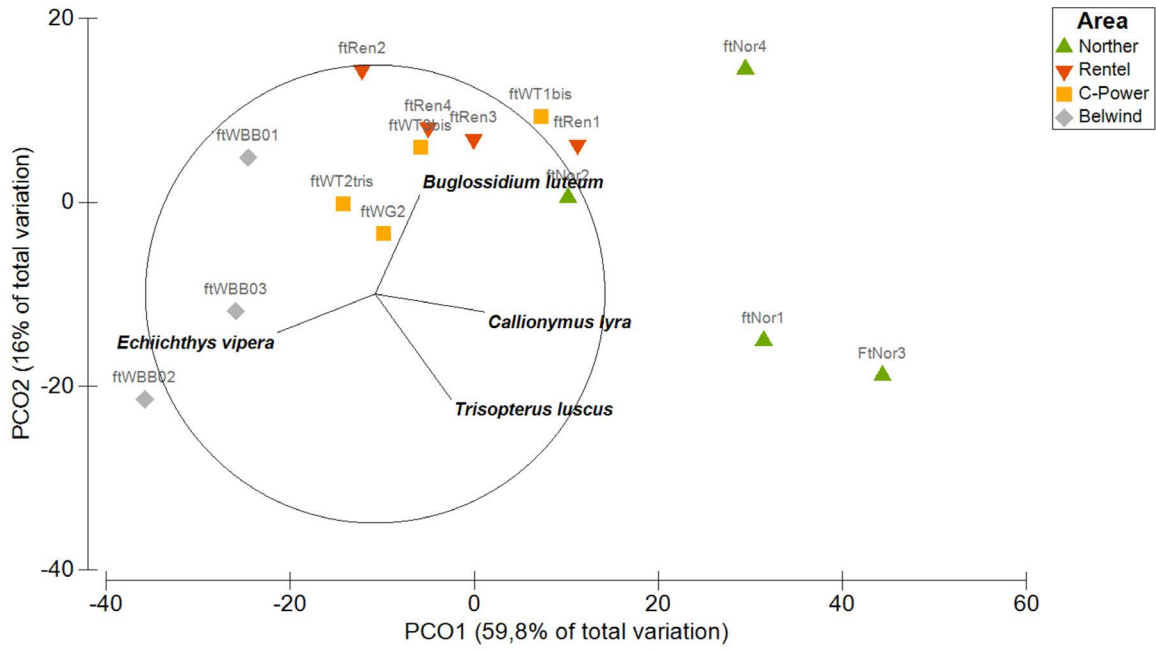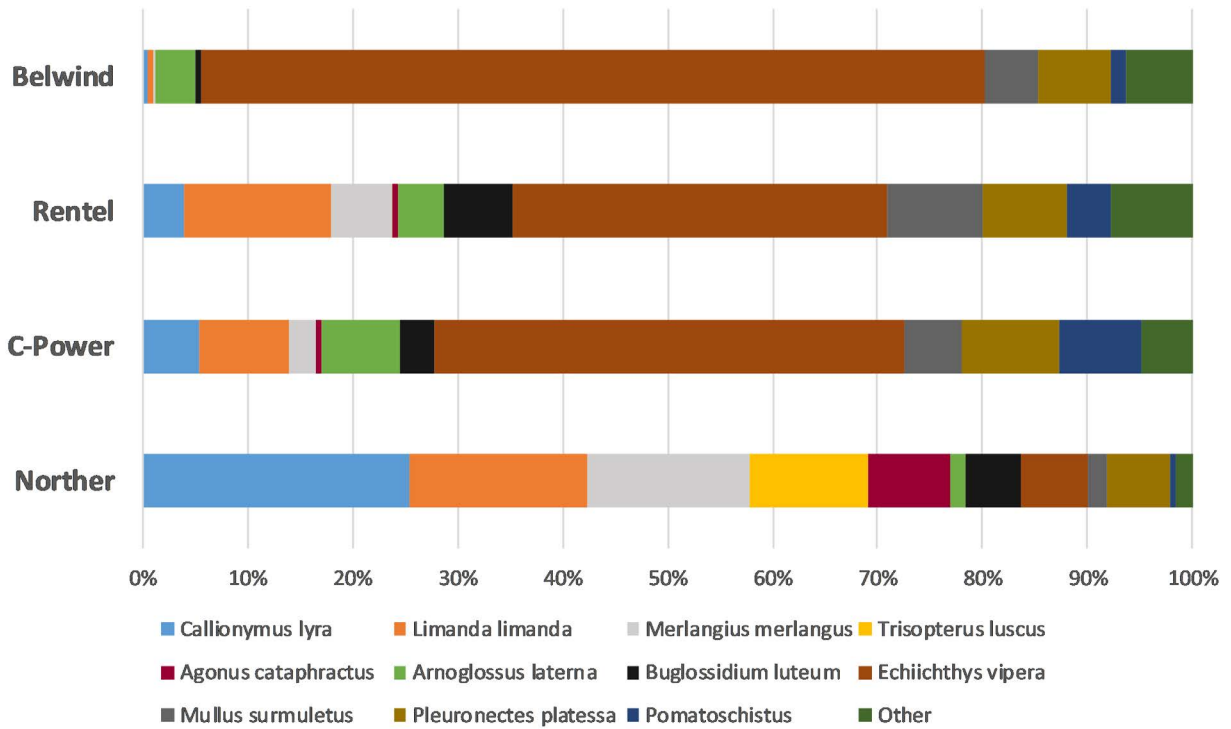**Figure 7.** Relative abundance of the most common demersal-benthopelagic fish species in the different concession/reference zones.

3.2. T$_0$ situation in Norther and Rentel

To determine the T$_0$ situation for Norther and Rentel, we zoom in on the fish tracks sampled in both concession areas and their respective reference tracks.

3.2.1. Epibenthos

For Norther, a high variability in species richness, density and biomass was noted between fish tracks for epibenthos. Number of species ranged between 15 and 22, density between 58 and 2270 ind. 1000 m^{-2}, and biomass between 238 and 10571 g WW . 1000 m^{-2}. Density and biomass were especially low in ftNor2, where fishing activity was taking place during sampling (table 1; fig. 8).

Also relative abundance of species was highly variable within Norther: similar species were present, but dominant species differed between fish tracks (fig. 9). The sea urchin *P. miliaris* formed the bulk density of ftNor1 (68%), the serpent star *O. ophiura* (34%) and the hermit crab *P. bernhardus* (23%) dominated in ftNor2, while in the reference fish tracks the star fish *A. rubens* (55%) together with *O. albida* (27%) dominated ftNor3, and *O. ophiura* (78%) dominated ftNor4 (fig. 9).

For Rentel, variability in epibenthos measures between fish tracks was relative low, only ftRen1 showed much higher density and biomass. The other fish tracks all showed highly similar values around 35 ind. 1000 m^{-2} for density and around 180 g WW · 1000 m^{-2} for biomass (table 1; fig. 8). Number of species ranged between 17 and 21.

Relative abundance of species was also very similar for the different fish tracks, except for ftRen1 which was dominated by *Tritia reticulata* (62%) (fig. 10). In the other fish tracks, species composition was more evenly distributed with most important species being *A. rubens, O. albida, O. ophiura, P. bernhardus* and *Spisula solida* (fig. 10).

Table 1. Epibenthos species richness (S), density (N) and biomass for each fish track in the Norther and Rentel concession and respective reference zones, with indication whether the track is located inside (C) or outside (R) the concession zone

Zone	Station	Conc/Ref	S	N (Ind. 1000 m^{-2})	Biomass (g WW 1000 m^{-2})
Norther	ftNor1	C	15	2270	10571
	ftNor2*	C	22	58	238
	ftNor3	R	20	723	1992
	ftNor4	R	15	1797	2883
Rentel	ftRen1	C	17	675	697
	ftRen2	C	17	31	181
	ftRen3	R	19	36	181
	ftRen4	R	21	36	189

* commercial fishing activity right before sampling
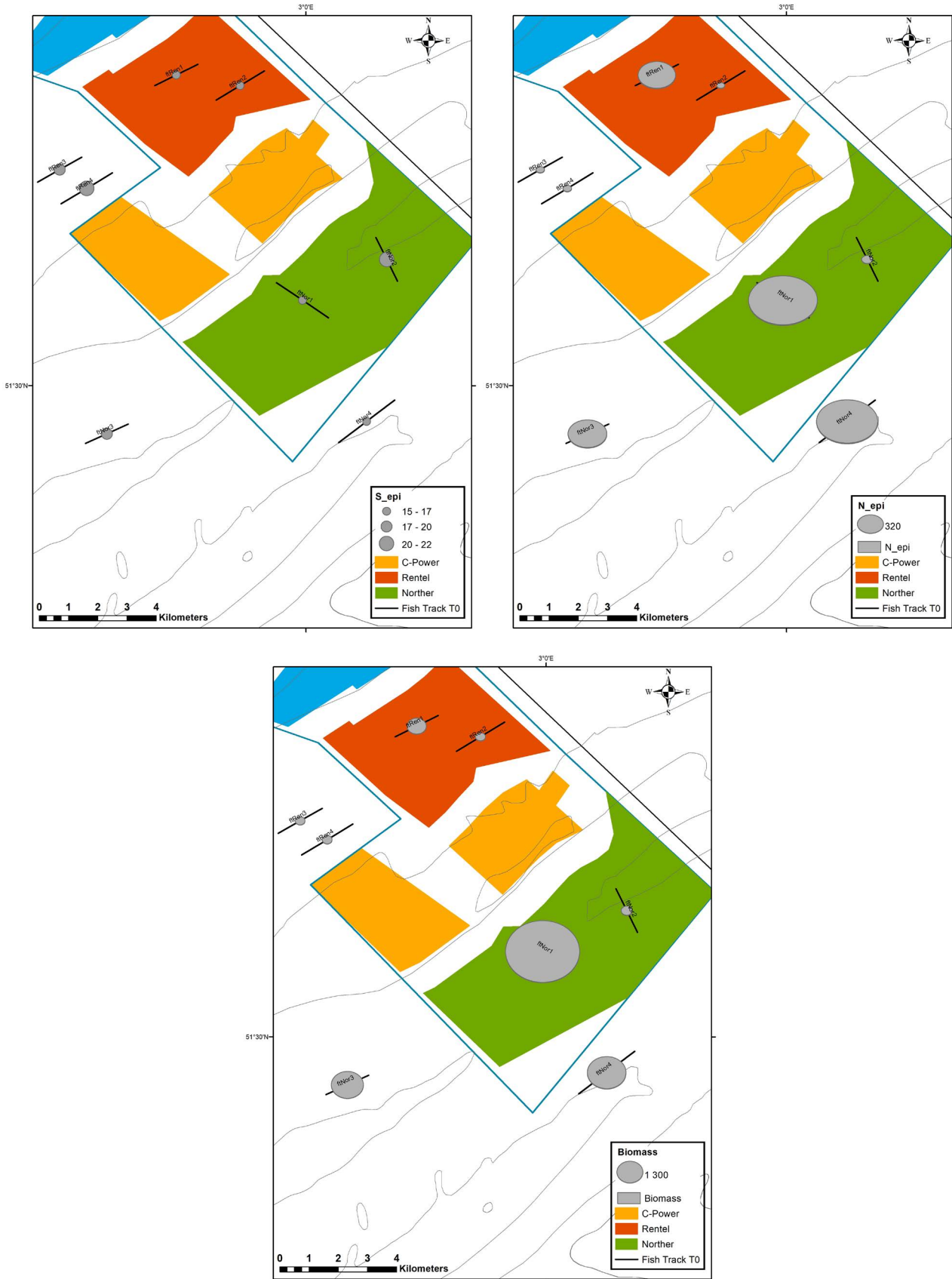
Figure 8. Map visualising number of species (left), density (middle) and biomass (right) for epibenthos of each fish track in the Norther and Rentel concession area. Size of pie charts varies with the values of each parameter.

Figure 9. Relative abundance of epibenthic species in the different fish tracks of the Norther concession and reference zone. List for full species names in annex 1.

Figure 10. Relative abundance of epibenthic species in the different fish tracks of the Rentel concession and reference zone. List for full species names in annex 1.

3.2.2. Demersal-benthopelagic fish

Within the Norther concession, a relatively high variability was noted between fish tracks for demersal-benthopelagic fish measures. Species richness ranged between 10 and 15 species. Fish tracks with the highest number of fish species had the lowest number of epibenthos species and the other way around (table 2; fig. 11). Density was highly variable with very low density in ftNor2, where there had been fishing activity just before sampling, and as for epibenthos, density was highest in ftNor1 with almost 300 ind. 1000 m⁻².

Species composition was less variable: three species were responsible for 40 to 75% of the cumulative relative abundance in all four fish tracks namely dab *Limanda limanda*, whiting *Merlangius merlangus* and common dragonet *Callionymus lyra* (fig. 12). Other species showed a higher variability in relative abundance between fish tracks: lesser weever *Echiichthys vipera* (25%) and plaice *Pleuronectes platessa* (18%) were only relatively dominant in ftNor2, pouting *Trisopterus luscus* (38%) was dominant in ftNor3, while in ftNor4 solenette *Buglossidium luteum* (18%) and hook nose *Agonus cataphractus* (15%) were relatively abundant (fig. 12).

Within the Rentel concession and reference zone, variability in fish measures was again lower between the different tracks compared to Norther. Species richness (S) ranged between 15 and 18 fish species. Density ranged between 19 and 58 ind. 1000 m⁻² (table 2; fig. 11).

For relative species composition, ftRen2 to 4 were almost identical with lesser weever as dominant species (40 to 45% relative abundance) (fig. 13). As for epibenthos, ftRen1 showed a different pattern compared to the other samples: there was not one single dominant species and species composition was more evenly distributed over dab, whiting, common dragonet, solenette, plaice and lesser weever (fig. 13).

Table 2. Demersal-benthopelagic fish species richness (S) and density (N) for each fish track in the Norther and Rentel concession and respective reference zones with indication whether the track is located inside (C) or outside (R) the concession area

Zone	Station	Conc/Ref	S	N (ind. 1000 m⁻²)
Norther	ftNor1	C	13	297
	ftNor2*	C	11	11
	ftNor3	R	10	144
	ftNor4	R	15	53
Rentel	ftRen1	C	17	58
	ftRen2	C	15	19
	ftRen3	R	18	34

* = commercial fishing activity right before sampling
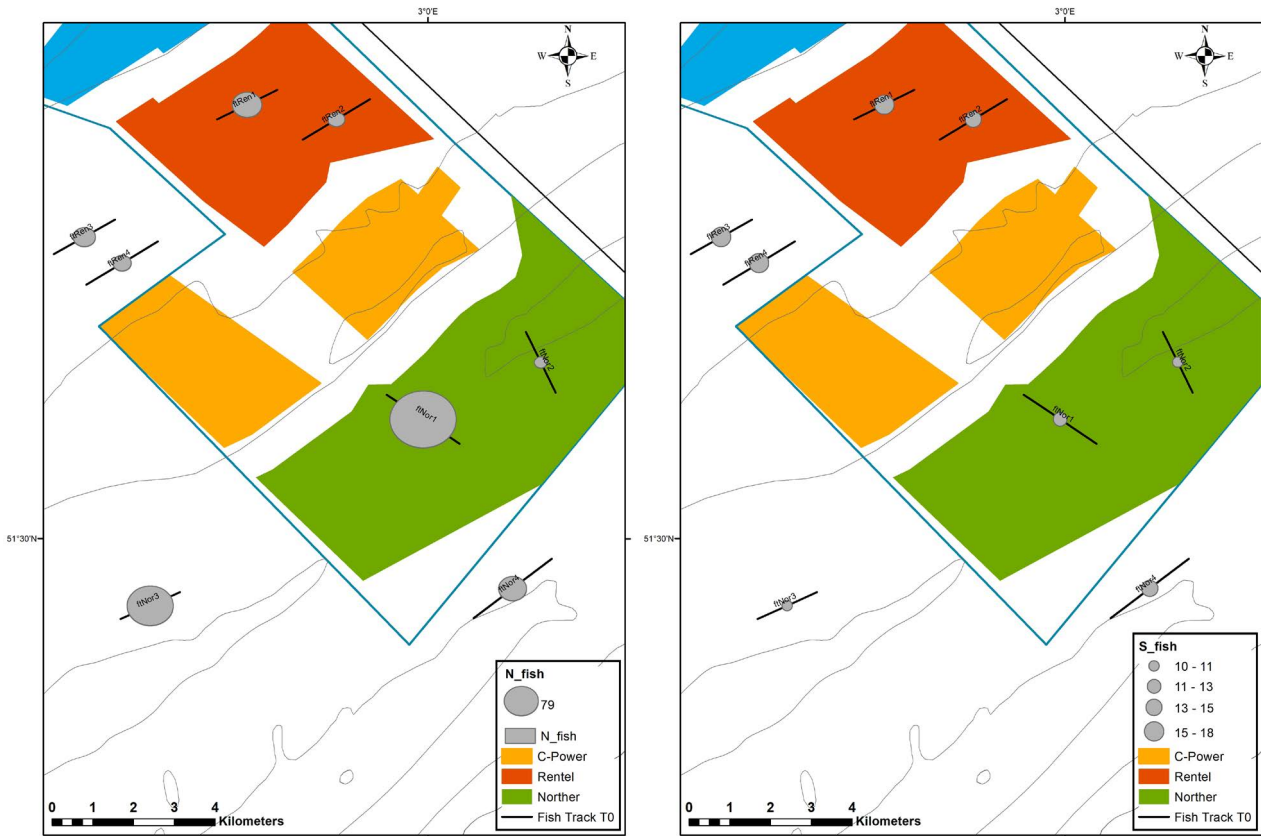
Figure 11. Map visualising number of species (left) and density (right) for demersal-benthopelagic fish of each fish track in the Norther and Rentel concession and reference zones. Size of pie charts varies with the values of each parameter.
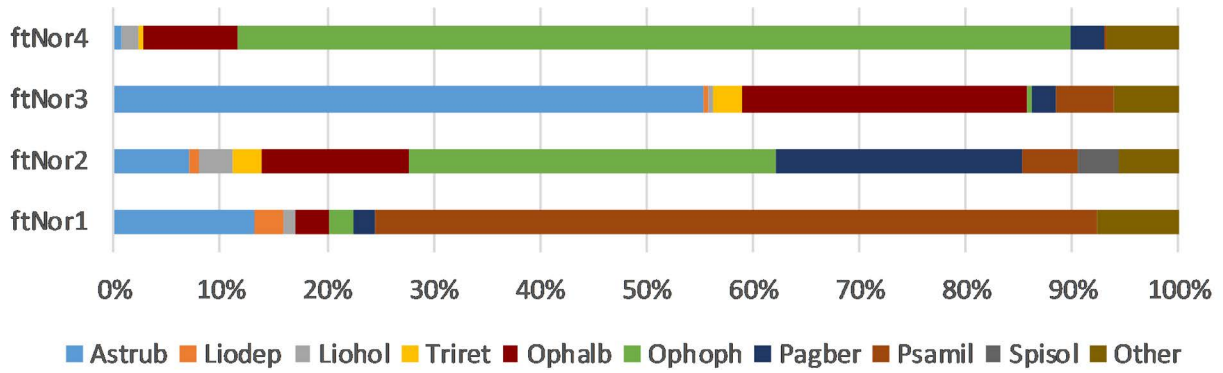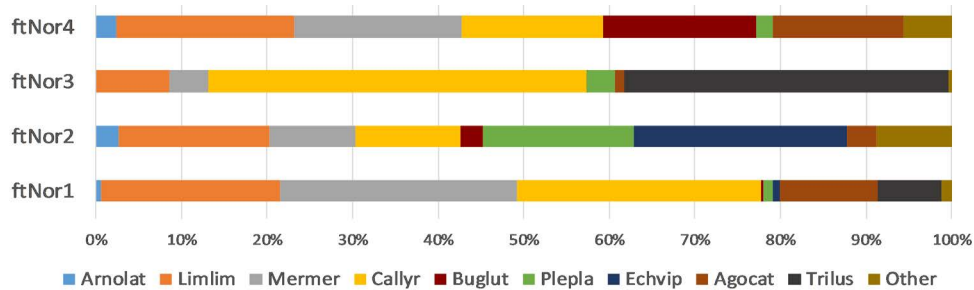

Figure 12. Relative abundance of demersal-benthopelagic fish species in the different fish tracks of the Norther concession and reference zone. List for full species names in annex 1.
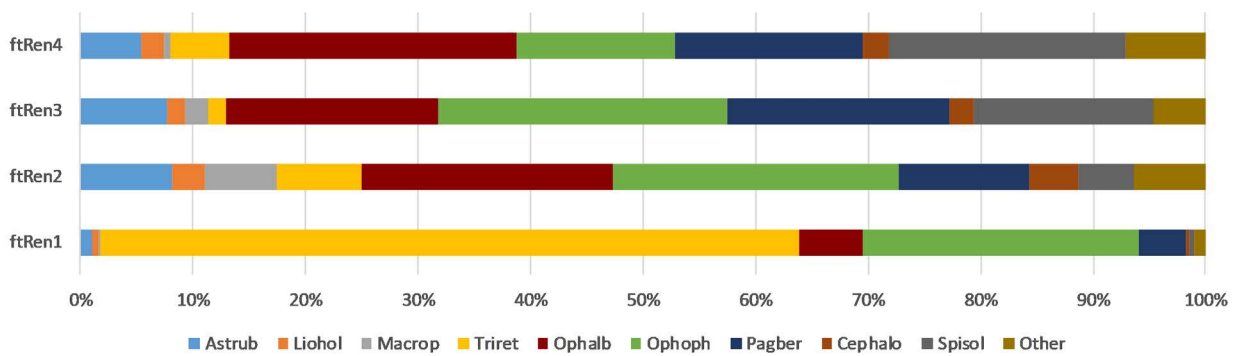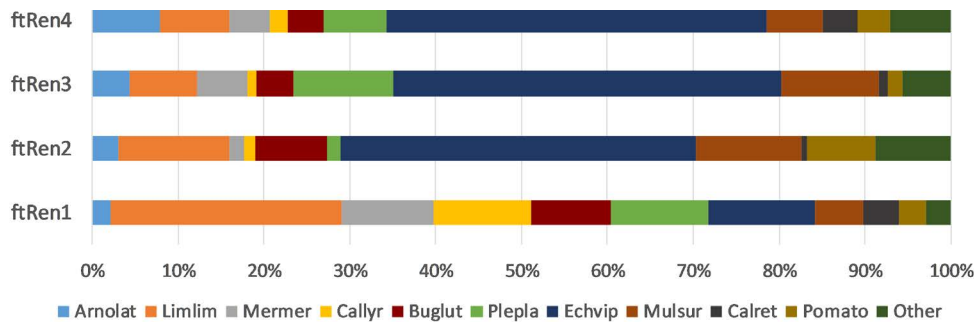

Figure 13. Relative abundance of demersal-benthopelagic fish species in the different fish tracks of the Rentel concession and reference zone. List for full species names in annex 1.

4. Discussion and conclusions

The aim of this chapter was twofold. First, it explored whether the epibenthos and demersal-benthopelagic fish assemblage in the future OWFs Norther and Rentel differed from the assemblages that are currently monitored in the reference zones of the C-Power and Belwind OWFs. If soft sediment epibenthos and fish assemblages are different, the existing results of previous monitoring surveys in C-Power and Belwind cannot directly be extrapolated to the future OWFs. Secondly, the reference condition for both concession areas was described, together with a suitability evaluation of potential new reference locations for both OWFs. As all samples were collected in autumn 2016, we excluded the influence of interannual variability and seasonality.

The main conclusions are:

- A clear onshore-offshore gradient was observed within the wider OWF area for both soft sediment epibenthos and demersal-benthopelagic fish assemblages. Norther, the concession zone closest to shore (23 km), exhibited much higher density and biomass (for epibenthos) compared to the other zones, while the Belwind reference zone, 50 km offshore, showed the lowest density and biomass. A similar gradient was observed for community structure with Norther differing from all other concession zones. For epibenthos, this was due to high densities of *Ophiura ophiura*, *Asterias rubens* and *Psammechinus miliaris* and the occurrence of brown shrimp *C. crangon*. When going further offshore towards Belwind, the hermit crab *P. bernhardus*, the bivalve *Spisula solida* and squid species Cephalopoda sp. gained relative importance. For fish, the Norther concession/reference zone was characterized by high densities of dragonet *C. lyra*, dab *L. limanda* and whiting *M. merlangus*, and the typical occurrence of pouting *T. luscus* and hook nose

A. cataphractus. When going further offshore, lesser weever *E. vipera* was the only dominant species. The Rentel concession/reference zone comprises a typical offshore assemblage, as described for Belwind and C-Power and their reference zones in Derweduwen *et al.* (2010). Rentel and C-Power are very similar, and differ from Belwind in epibenthos due to the occurrence of the netted dog whelk *T. reticulata* and high densities of *O. ophiura*. For fish, dominance of lesser weever is lower in Rentel and C-Power and solenette *B. luteum*, dab, whiting and dragonet occur in relative higher abundances compared to Belwind. The epibenthos and fish assemblage observed at the Norther concession/reference zone best related to the coastal 1 assemblage described by Vandendriessche *et al.* (2009), which was characterized by high density and diversity, although the current data rather characterize the Norther assemblage as a transition between the coastal 1 and the typical offshore assemblages as observed in the other concession/reference zones.

- Since Norther exhibits a quite different epibenthos and fish assemblage compared to the other zones, a follow-up of this future OWF seems justified, as an extrapolation of the results of the ongoing monitoring in the existing OWFs (C-Power and Belwind) cannot be considered reliable. Rentel, on the other hand, has a soft sediment epibenthos and fish assemblage which is very similar to C-Power, indicating that results of C-Power can be extrapolated to this area, assuming that effects of jacket and monopile foundations are comparable at further distances from the turbines. Preliminary results on macrobenthos and sediment characteristics near different foundation types showed no differences between foundation types (Colson *et al.* 2017). Monitoring of Rentel seems

redundant within the current WinMon monitoring program. Nevertheless, if it were to be included in the program, the sampling design is adequate: variability between the different fish tracks is low and reference locations are suitable for the concession area. Only ftRen1 inside the concession area differed due to a more even distribution of densities over the different species.

- Integration of Norther in the WinMon monitoring framework is recommended. However, the high degree of variability observed between the different fish track locations in the Norther concession and reference zone could be a bottleneck for the effect monitoring. Species occurrence is quite similar between locations both inside and outside the concession zone, but densities and relative abundance of common species differ largely from one fish track to the other, especially for epibenthos. This can hamper a sound impact assessment, since it will be difficult to measure effects of the future OWF when natural variability is so high, both inside and outside the concession zone. Furthermore, for ftNor2 located within the Norther concession zone, we measured the impact of a fishing vessel that just passed by, leaving very low densities and lots of dead fish and epibenthos, making this not the most ideal sample for a T_0 reference condition. In that respect, it would be best to exclude this outlier sample for future impact assessments. Nevertheless, including this sample in the current chapter gives an indication on what effect fisheries activity can have on the epibenthos and fish assemblage, and thus also on what might be expected when fisheries are excluded once the OWF comes in place *i.e.*, a richer and more diverse assemblage.

- The decision on whether or not to include Norther in the overall WinMon monitoring program should be made by taking into account as well the results for the other ecosystem components sampled by UGent, being hyperbenthos and macrobenthos. For epibenthos and demersal-benthopelagic fish, we showed that results of previous monitoring in C-Power and Belwind cannot be directly extrapolated to Norther, since it is a different assemblage inhabiting the concession zone. However, the natural variability in the Norther concession and reference zone is high. Consequently, effects will only be picked up when the impact is huge or after a certain amount of time when time series in the area are long enough to be able to detect a potential fisheries exclusion effect. On the other hand, the fisheries exclusion effect in this more diverse and richer epibenthos and demersal-benthopelagic fish assemblage might be even stronger than for the offshore assemblages in the other OWFs.

Acknowledgements

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Annex 1

Species names with according abbreviations used in the figures in this chapter.

	Species name	Abbreviation
Epibenthos	*Asterias rubens*	Astrub
	Cephalopoda sp.	Cephalo
	Liocarcinus depurator	Liodep
	Liocarcinus holsatus	Liohol
	Macropodia sp.	Macrop
	Ophiura albida	Ophalb
	Ophiura ophiura	Ophoph
	Pagurus bernhardus	Pagber
	Psammechinus miliaris	Psamil
	Spisula solida	Spisol
	Tritia reticulata	Triret
Fish	*Agonus cataphractus*	Agocat
	Arnoglossus laterna	Arnolat
	Buglossidium luteum	Buglut
	Callionymus lyra	Callyr
	Callionymus reticulatus	Calret
	Echiichthys vipera	Echvip
	Limanda limanda	Limlim
	Merlangius merlangus	Mermer
	Mullus surmuletus	Mulsur
	Pleuronectes platessa	Plepla
	Pomatoschistus sp.	Pomato
	Trisopterus luscus	Trilus

CHAPTER 5

EFFECTS OF WIND TURBINE FOUNDATIONS ON SURROUNDING MACROBENTHIC COMMUNITIES

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Abstract

The installation of artificial hard substrates such as wind turbines is likely to affect the surrounding environment. Fining and organic matter enrichment were observed around one gravity-based foundation on the Thornton Bank, but subsequent basic monitoring did not reveal any of these effects in the vicinity of the turbine types at Thornton- and Bligh Bank. It was suggested that effects are restricted to close distances (< 50 m) from the turbines and that impacts could differ between turbine types. Therefore, the sampling strategy within this study was adjusted by comparing far with very close locations (37.5 m). Our results confirm turbine-related effects at very close distances around jacket-based foundations at the Thornton Bank. Within very close samples, fining and enrichment of the sediment was detected together with higher macrofaunal densities, diversity and shifts in communities. In contrast, effects around monopile-based foundations at the Bligh Bank were less pronounced and a significant difference in community composition only was found between both distances. We suggest that these contrasting results might be due to a combination of site-specific dispersive capacities and structural differences between foundation types (jackets *vs.* monopiles) and their associated epifouling communities. Consequently, we recommend

performing a targeted monitoring study comparing the three different turbine foundation types (monopiles, jackets and gravity-based foundations) used in the BPNS.

1. Introduction

Currently, three offshore wind farms (OWF: C-Power, Belwind, Northwind) are operational within the concession zone for renewable energy in the eastern part of the Belgian part of the North Sea (BPNS) (Rumes & Brabant *et al.*2017). A fourth OWF will be constructed in close proximity to the coast in 2018-2019 by NV Norther.

The installation of artificial hard substrates in soft sediments could possibly affect the seafloor-inhabiting macrofauna communities. Macrobenthic communities play a crucial role in bentho-pelagic coupling and are considered an important food source for higher trophic species such as crabs and fish (Vandendriessche *et al.* 2015). Changes within these communities are therefore likely to alter overall food web energy flows (Colson *et al.* 2017; Danheim *et al.* 2014). Benthic communities are less sensitive to local-scale impacts in areas with high natural physical disturbance (Cooper *et al.* 2011). Therefore, short-term impacts through post-installation mortality are believed to be limited in the species-poor communities thriving in the
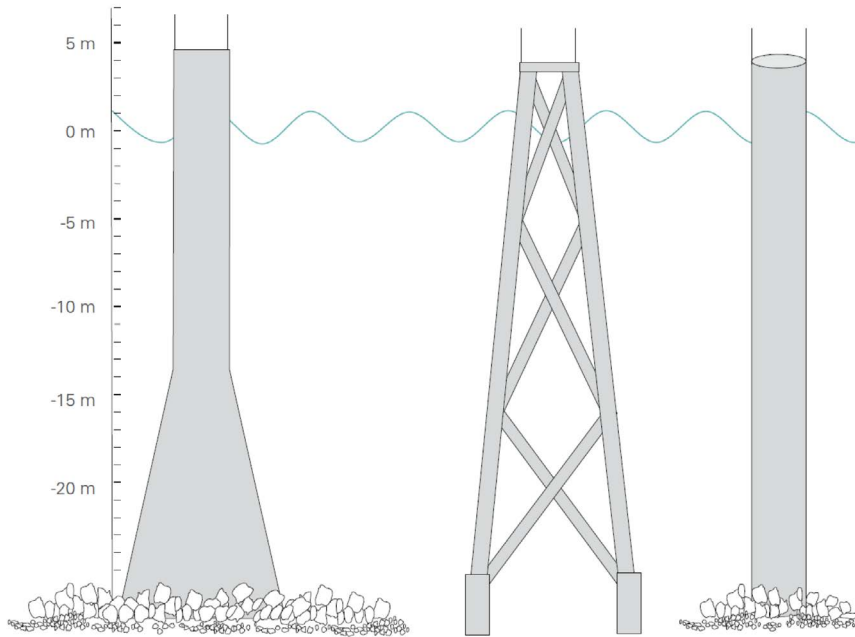
Figure 1. Three foundation types present in the Belgian part of the North Sea, from left to right: gravity based, jacket (both in C-Power) and monopile foundation (Belwind and planned in Norther) (Rumes *et al.* 2013).

highly dynamic offshore sediments of the BPNS (Van Hoey *et al.* 2004). Consecutive monitoring within two offshore wind farms (C-Power and Belwind) indeed demonstrated a relatively fast recovery (1-2 years) of the naturally occurring macrobenthic communities after wind farm construction (Reubens *et al.* 2009; Coates *et al.* 2014).

However, longer-term effects are expected. Fisheries exclusion in offshore wind farms may alter the marine environment at different levels (De Mesel *et al.* 2013; 2015; Reubens *et al.* 2013, 2014), including macrobenthic communities (Coates *et al.* 2016). In addition, the permanent presence of the wind turbines changes the physical properties of the surrounding habitat (De Backer *et al.* 2014). Vertical structures in the water column alter local hydrodynamics and sediment transport, and induce higher shear stress (Baeye *et al.* 2015; Barros *et al.* 2001). Abundant epifouling communities are known to colonize the foundations, thereby affecting the organic matter deposition to the sediment (De Mesel *et al.* 2013; Jak & Glorius 2017). Fining and organic matter enrichment of the sediment have

indeed been observed in close vicinity of one gravity-based foundation on the Thornton Bank (Coates *et al.* 2013). In the macrobenthic communities within the 50 m surrounding this specific gravity-based foundation, some typical hard substrate fauna was found, next to suspension-deposit feeding species usually observed in fine sandy and organic matter-rich sediments. The subsequent basic monitoring studies (C-Power and Belwind) thereafter did not show evidence of this fining and organic enrichment in the vicinity (50 m) of any of the turbine foundation types (Reubens *et al.* 2016; Colson *et al.* 2017). Results found by Reubens *et al.* (2016) were based on samples in C-Power, which mostly consist of jacket-based turbines. Hence, a possible reason for the contrasting results with Coates *et al.* (2013) was attributed to the differences in turbine foundation types being studied. Foundation types are mainly selected according to the environmental conditions (*e.g.*, water depth and sediment type), together with production and installation costs, and other socio-economic considerations. OWF developers have hitherto used three different foundation

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types: gravity based, jacket and monopile foundations (fig. 1), each with different (pre-)construction-related activities such as dredging and pile driving (Coates 2014).

However, the main reason for the lack of fining and enrichment effect in the last basic monitoring studies (Reubens *et al.* 2016; Colson *et al.* 2017) may equally be the distance to the foundation at which the communities were sampled. The fining and enrichment effect is hypothesized to be restricted to very close distances (< 50 m) from the turbine foundation (Coates *et al.* 2013; Colson *et al.* 2017). For these reasons, the basic monitoring scheme comparing far (350-500 m distance from turbines) to close locations (50 m) was slightly altered in 2017 to a comparison of far with very close locations (37.5 m distance from turbine center, *i.e.*, the closest distance that is still feasible and safe to sample) within this study. In this report, we test whether a fining and enrichment effect is found at such a close distance to the turbine foundation, and whether this is reflected in the macrobenthic community structure. Furthermore, we verify whether this effect is present in the vicinity of both monopile and jacket foundation types.

A second part of the report assesses the before-impact (T0) communities in the NV Norther concession area and evaluates the suitability of the possible reference area. Conditions within the future Norther OWF differ from those already being monitored in C-Power and Belwind as this OWF is not located on a sandbank and is located very close to the coast (< 25 km). Due to the differences in sedimentology and the general distribution of macrobenthic communities along the onshore-offshore gradient described by Van Hoey *et al.* (2004), we expected to find different soft-sediment communities in this area.

2. Material and methods

2.1. Study area

Within the BPNS, sampling was conducted in the concession areas of two existing offshore wind farms (C-Power and Belwind) and one planned turbine park (Norther) (fig. 2). C-Power is located on the Thornton Bank (TB), situated approximately 30 km from the Belgian coastline. This park consists of 54 turbines with 2 types of foundations: 6 gravity-based (constructed in 2008) and 48 jacket foundations, which were built between 2011 and 2013 (Rumes*et al.* & Brabant 2017). Belwind is located at the Bligh Bank (BB) and represents the north western-most turbine park within this study (46 km from the port of Zeebrugge). Belwind contains a total of 55 monopile-based turbines that were constructed between 2009 and 2010 (Rumes & Brabant *et al.*2017). The concession for the Norther wind farm was granted in 2009 and the construction of 44 monopile-based turbines is expected to start in 2018-2019. The park will be situated 23 km from the Belgian coastline (port of Zeebrugge) and lies within the south eastern-most part of the concession area. The reference site (REF) was chosen directly beyond the south eastern border of the future wind park to correspond to the sediment characteristics found within the future Norther impact area (fig. 4).

2.2. Sample design, collection and treatment

By applying systematic stratified sampling designs, this study was able to conduct two one-way spatial comparisons as described in table 1. Within the first analysis, potential effects of turbine presence on macrobenthic communities were tested in two operational wind farms (C-Power and Belwind). Samples were collected at two distances from the turbines during autumn 2017 on board the vessels RV Simon Stevin and Aquatrot (fig. 3; table 1). 'Very close' samples were taken at approximately 37.5 m from the center of the
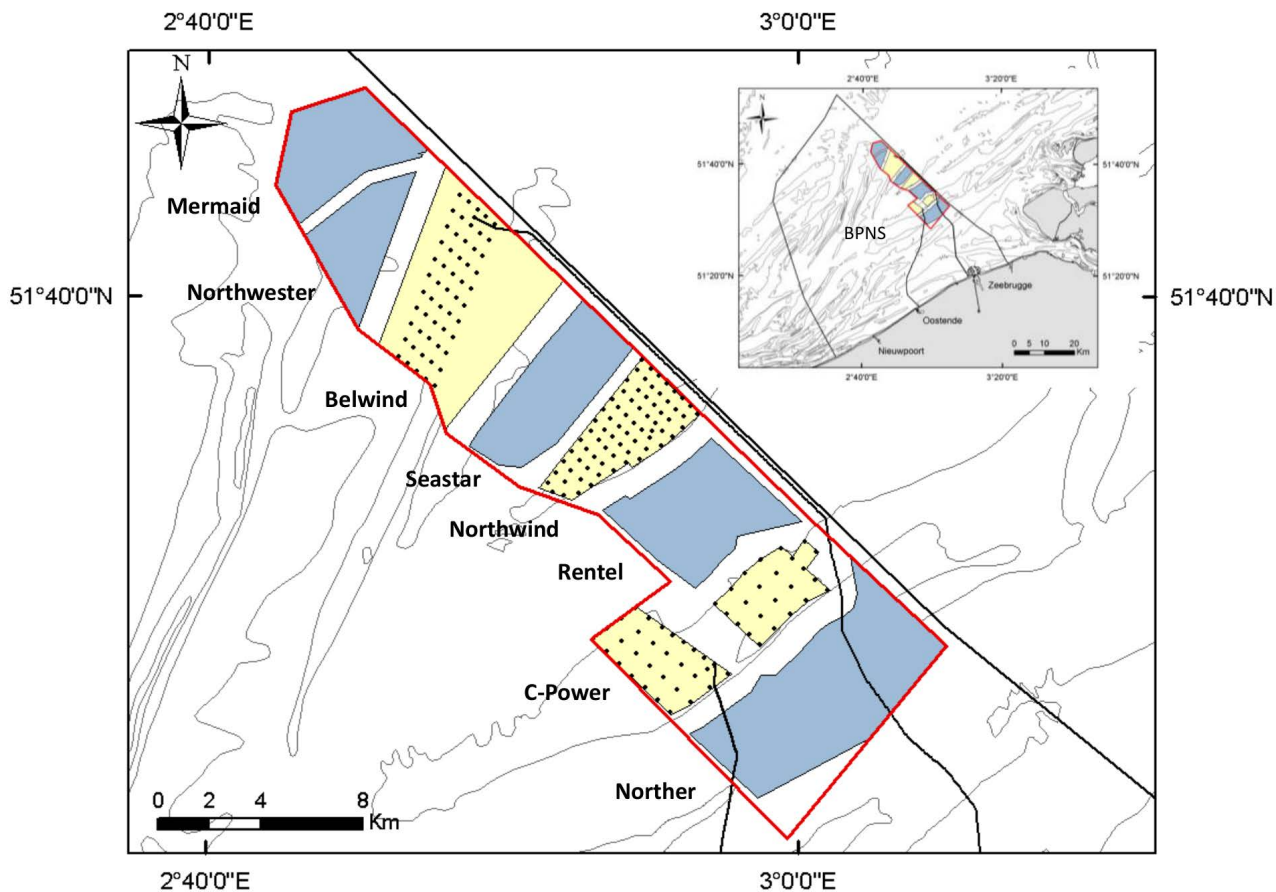
Figure 2. Wind farm concession area (red area) in the Belgian part of the North Sea. Yellow areas represent three operational offshore windfarms (C-Power, Northwind and Belwind), while blue areas are domains for which concessions have been granted (Norther, Rentel, Seastar, Northwester and Mermaid) (Coates, 2014).

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

Type of analysis	Date of sampling	Vessels	Station	# samples
Effects of turbine presence (far *vs.* very close) within C-Power and Belwind windfarms	Autumn 2017 (Oct-Dec)	RV Simon Stevin, Aquatrot	TB_FAR	31
			BB_FAR	36
			TB_VERY_CLOSE	16
			BB_VERY_CLOSE	15
Baseline (T0) analysis for future offshore windfarm Norther	Autumn 2016 (Nov-Dec)	RV Simon Stevin, Stream	NORTHER_FAR	22
			Reference site (REF)	18
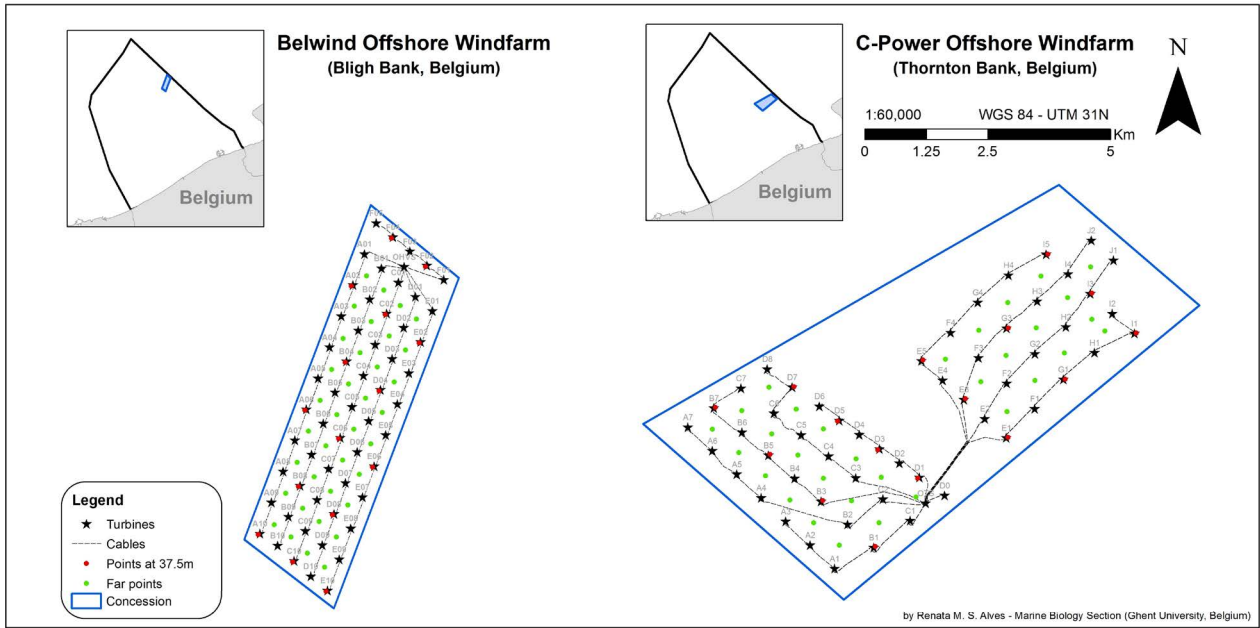
Figure 3. Overview of far and close samples at the Bligh Bank (left) and Thornton Bank (right).
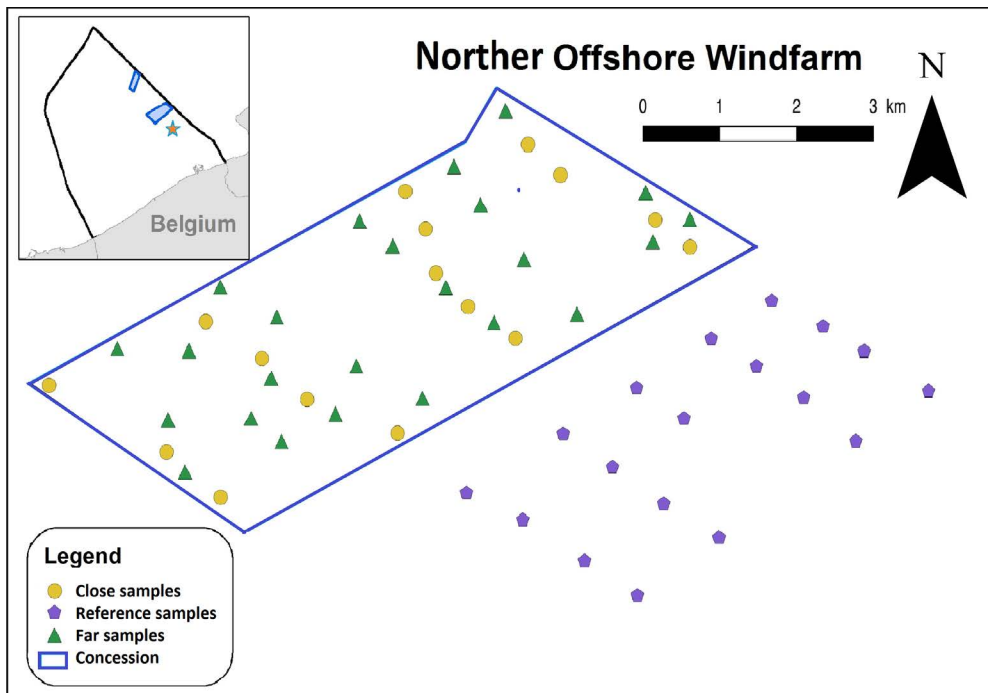
Figure 4. Overview of close and far samples (green triangles) and samples at the reference site (purple) for the future offshore wind farm Norther.

turbine, whereas far samples were collected in the middle between the four surrounding wind turbines (*i.e.*, farthest possible distance), *i.e.*, at distances between 350 and 500 m from any windmill. A second analysis was performed to establish the baseline (T0) for long-term monitoring within the future wind farm park Norther and test the validity of the proposed reference site. To this aim, samples were collected within the Norther area during autumn 2016. A similar sampling design was applied with close samples at approximately 50 m and far samples at least at 250 m from the future turbines (fig. 4; table 1). Within this study, only the far samples collected at Norther were used. In addition, 18 samples were also taken within the proposed reference area. Both the 'impact' site (Norther) and the reference site sampled in 2016 represent source samples of the area before the impact of wind turbine construction within the Before After Control Impact (BACI) design.

The samples were collected from the vessels by means of a 0.1 m² Van Veen grab. A Plexiglass core (Ø 3.6 cm) was taken from each Van Veen grab sample to collect the environmental data which include: grain size distribution (reported: median grain size [MGS]), total organic matter content (TOM) and sediment fraction larger than 2 mm (> 2 wamm). After drying at 60°C, the grain size distribution was measured using laser diffraction on a Malvern Mastersizer 2000G, hydro version 5.40. 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 dry weight (48 h at 60°C) and ash-free dry weight (2 h at 500°C).

The rest of the sample was sieved on board (1 mm mesh-sized sieve), and the macrofauna was preserved in a 4% formaldehyde-seawater solution and stained with Rose Bengal. In the laboratory, organisms were sorted, counted and identified to the lowest possible taxonomic level. Biomass was also determined for each taxon level as blotted wet weight (mg). Within this report, these taxa are further referred to as species. From the obtained dataset, hyperbenthic species were excluded, and in case of uncertain identification, some taxa were lumped (*e.g.*, genus level: *Melitta* spp.).

2.3. Data Analysis

The samples collected at gravity based foundations were removed from the analyses (3 very close and 3 far samples) to test the effect of distance from the turbine, so that only samples at jacket foundations were included for the Thornton Bank. Prior to statistical analysis, the total abundance (ind. m⁻²), biomass (mg WW m⁻²), number of species (S), Shannon-Wiener diversity index (H') and Piélou's evenness (J') were calculated from the dataset. Univariate analysis (1-way ANOVA) was performed in R (version 3.2.2) to assess differences between distances from the turbines (far *vs.* very close) and location (Thornton Bank *vs.* Bligh Bank; Norther *vs.* reference site) in terms of the above-mentioned biological parameters and the sediment parameters MGS, fraction > 2 mm and TOM. Assumptions of normality and homogeneity of variances were tested by Shapiro-Wilk – and Levene's tests –, respectively, and log transformations were performed if these assumptions were not met. If after transformation the assumptions were still not fulfilled, a PERMANOVA (Permutational Anova, based on Euclidean distance matrix) was performed, allowing us to perform univariate ANOVAs with p-values obtained by permutation (Anderson & Millar 2004), thus avoiding the assumption of normality. Additionally, multiple linear regression analysis was used to develop a model to predict the biotic variables that showed significant differences after univariate analysis from TOM, MGS and sediment fraction > 2 mm. Outliers were detected and removed from the models. Normal distribution of the residuals was tested

(Shapiro-Wilk) and potential multicollinearity was determined to use a Variance Inflation Factor (VIF).

Multivariate analysis was performed in PRIMER (version 6.1.11) with PERMANOVA add-on to investigate the potential effects of distance/location on macrobenthic community structure. These tests were based on a Bray-Curtis resemblance matrix (fourth-root transformed data) and were performed by using a fixed one-factor design (distance, levels: far *vs.* very close and location, levels: Norther *vs.* reference site). Homogeneity of multivariate dispersions was tested using the PERMDISP routine (distances among centroids). Principal coordinate analysis (PCO) was used to visualize the data, while similarity percentages (SIMPER) analysis was performed to determine the contribution of species to the distinction between groups and/or to the similarity of samples within a group (Anderson *et al.* 2008; Clarke & Gorley 2006). Finally, a distance-based linear model (DistLM, adjusted R^2 with stepwise criterion) was run to investigate the potential relationship between biological and environmental variables (Anderson *et al.* 2008). Due to the unbalanced sampling design (table 1), type 'III' sums of squares were used for every statistical test, and a significance level of $p < 0.05$ was applied. Quantitative results are expressed as mean values and corresponding standard deviation (mean ± SD). Permdisp results were only reported when significant.

3. Results

3.1. Effects of turbine presence

Thornton Bank (TB) and Bligh Bank (BB) displayed similar values in terms of TOM and sediment fraction > 2 mm. However, MGS was significantly higher at BB compared to TB (1 way ANOVA, $p < 0.01$). Higher macrobenthic densities and biomass were found at TB (1 way ANOVA, $p < 0.01$). In addition, multivariate analysis revealed that macrobenthic communities differed significantly between sandbanks (PERMANOVA, $p = 0.001$). Based on these results and to enable the comparison with the two previous reports (Colson *et al.* 2017; Reubens *et al.* 2016), it was decided to conduct further analyses testing potential effects of turbine presence for each sandbank separately.

3.1.1. Thornton Bank (C-Power)

Sediments within TB consisted of medium sands (250-500 μm), except for the sample TB28_FAR with an exceptionally high MGS (509 μm). MGS was significantly affected by distance from the turbines (1 way ANOVA, $p < 0.05$), with finer sands within the very close samples (342 ± 22 μm) compared to far samples (378 ± 49 μm). This refinement of the sediment with decreasing distance to the turbines was particularly found in the 125-250 μm fraction with the average percentage of fine sand being 20 ± 5% in very close samples, while this was only 13 ± 6% in the far samples (1 way ANOVA, $p < 0.01$). TOM content varied from 0.31%-1.86%, with significantly higher average values within the very close samples (0.72 ± 0.39%) compared to the far samples (0.53 ± 0.17%) (1 way ANOVA, $p < 0.05$). The sediment fraction > 2 mm within TB was variable and ranged from 0.04%-15.51% with higher average values within the very close samples, but no significant difference was found.

Samples closer to the turbines displayed significantly higher macrobenthic densities, species richness and Shannon-wiener diversity, whereas evenness was significantly lower (1 way ANOVA $p < 0.05$; fig. 5; table 2). No significant difference was found between both distances in terms of biomass.

Multivariate analysis on macrobenthic community structure revealed that within TB, different communities are found for both distances (PERMANOVA, $p < 0.001$; fig. 6). SIMPER results showed that very close samples had an average similarity of
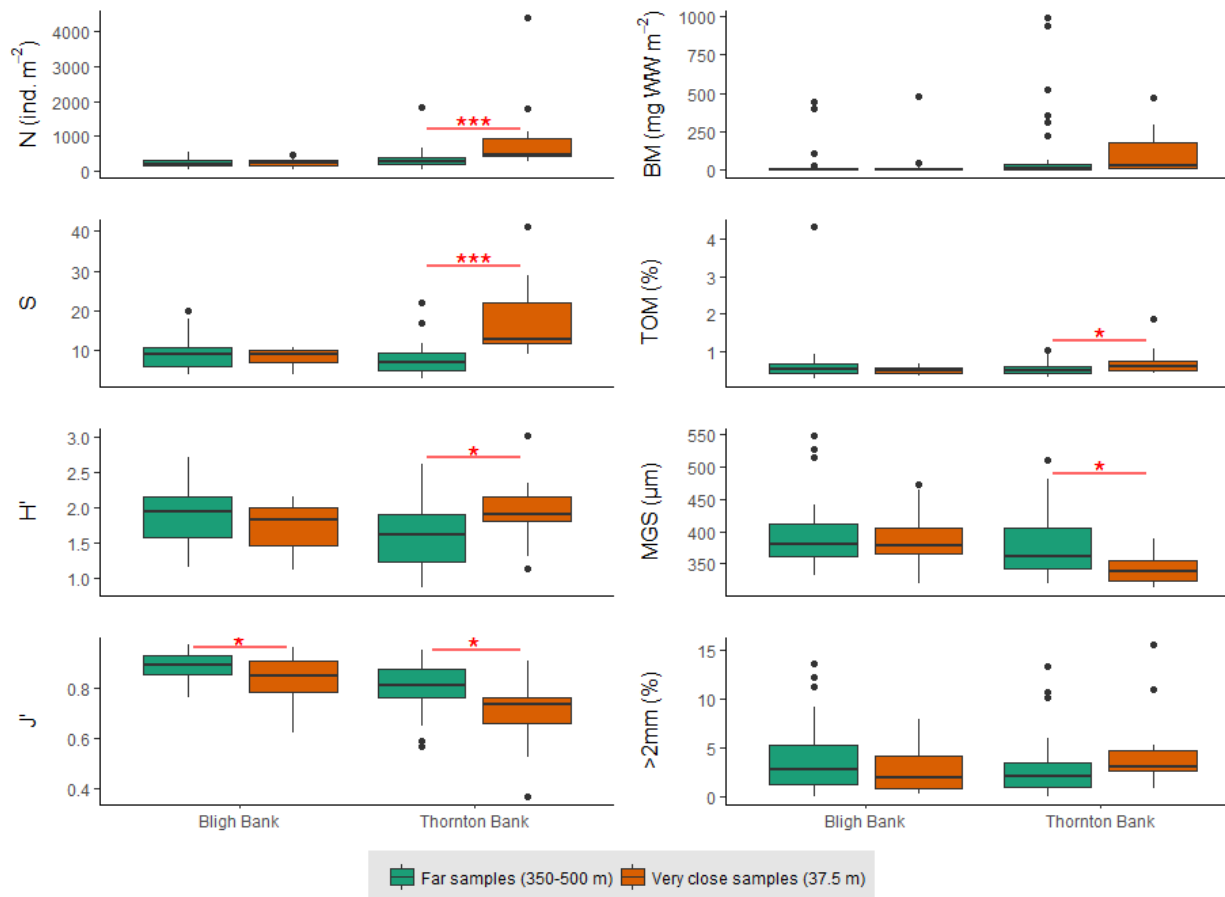
Figure 5. Overview boxplots of the biotic variables: abundance (N), biomass (BM), species richness (S), Shannon-wiener diversity (H'), evenness (J') and abiotic variables: total organic matter (TOM), median grain size (MGS), sediment fraction above 2 mm (> 2 mm) per sampling site for the very close and far samples. Black dots represent outliers.

37.25% with *Urothoe brevicornis* (22.68%), *Nephtys cirrosa* (17.45%), *Nephtys* juveniles (10.17%) and *Nemertea* sp. contributing about 60% to the total abundances. Far samples showed a higher average similarity (39.23%), but these communities were dominated by *Nephtys cirrosa* and *Nephtys* juveniles, which contributed 60% to the total abundances, while *Urothoe brevicornis* contributed another 17.45%. The average dissimilarity between communities at the two distances (far *vs*. very close) amounted to 67.33%. *Nemertea* sp. (5.58%), *Urothoe brevicornis* (5.27%), *Spiophanes bombyx* (4.17%), *Bathyporeia elegans* (4.05%), *Nephtys* juveniles (3.39%) and *Echinocardium cordatum* (3.13%) together explained about 25% of this dissimilarity and all six species showed higher average

abundances in the very close samples. Many other species contributed to a lesser extent (contribution < 3%; table 3) indicating that differences between communities cannot be attributed to a few dominant species.

Multiple regression revealed that MGS and TOM were significant predictors of macrobenthic densities (N), species richness (S') and Shannon-wiener diversity (H'). This model best explained species richness (R^2_{adj} = 0.60) , followed by macrobenthic densities (R^2_{adj} = 0.41) and Shannon-Wiener diversity (R^2_{adj} = 0.17). TOM proved to be the only significant predictor (R^2_{adj} = 0.07) of Piélou's evenness (J'). All three abiotic variables (MGS, TOM and > 2 mm) had a significant relationship with the multivariate data and explained 22.83% of the total variation (DistLM analysis).

Table 2. Overview of calculated community descriptors (mean ± SD) for spatial comparisons: between both distances from a turbine in two operational wind farms at Thornton BankBank (TB) and Bligh Bank (BB), baseline analysis within a future wind farm (Norther – Reference site). Numbers that differ significantly are indicated in bold

Spatial analysis	Effects turbine presence				Baseline (T0) study	
	TB Very Close	TB Far	BB Very Close	BB Far	Norther	REF
Total abundance (N, ind. m^{-2})	**934 ± 1112** ***	343 ± 329	255 ± 118	239 ± 120	8855 ± 20612	2588 ± 2442
Biomass (BM, mg WW m^{-2})	110 ± 145	132 ± 274	39 ± 122	31 ± 99	164 ± 279	228 ± 263
Number of species S	**18 ± 9** ***	8 ± 4	8 ± 2	9 ± 4	30 ± 14	27 ± 11
Evenness J'	**0.71 ± 0.15** *	0.80 ± 0.10	**0.84 ± 0.09** *	0.89 ± 0.06	0.74 ± 0.11	0.69 ± 0.10
Shannon-Wiener H'	**1.92 ± 0.46** *	1.57 ± 0.44	1.74 ± 0.31	1.87 ± 0.42	2.40 ± 0.48	2.22 ± 0.35
Median grain size (MGS, µm)	**342 ± 22** *	378 ± 49	391 ± 42	392 ± 51	355 ± 89	334 ± 94
Total organic matter (TOM, %)	**0.72 ± 0.39** *	0.53 ± 0.17	0.50 ± 0.10	0.65 ± 0.64	**1.09 ± 0.49** ***	1.60 ± 0.50
Sed. fraction > 2 mm (> 2 mm, %)	4.68 ± 4.11	3.10 ± 3.38	2.93 ± 2.57	4.04 ± 3.76	11.45 ± 10.67	7.45 ± 9.21

Signif. codes: '***' 0.001, '**' 0.01, '*' 0.05

3.1.2. Bligh Bank (Belwind)

In contrast to the results found within TB, all environmental variables were comparable between distances within BB. Sediments in BB were mainly composed of medium sands (250-500 µm), except for the samples: BB22_FAR (MGS: 547 µm), BB33_FAR (MGS: 514 µm) and BB36_FAR (MGS: 526 µm). The sediment fraction > 2 mm ranged from 0.07%-13.68% and TOM contents from 0.28%-4.31% with most values between 0.40% and 0.80%.

Average macrobenthic densities and biomass were slightly higher in samples closest to the turbines, while an opposite trend was found for all the diversity indices but none of these differences proved to be statistically significant. Only evenness was significantly lower (1 way ANOVA, p < 0.05) in the very close samples (0.84 ± 0.09) compared to far samples (0.89 ± 0.06).

Macrobenthic community structure did, however, differ between far and very close samples within BB (PERMANOVA, p < 0.01; fig. 6). The average similarity for the very close samples was 39.04% and communities were mainly composed (cumulative contribution of 57.26%) of the polychaetes *Nemertea* sp. (21.43%), *Nephtys cirrosa* (18.67%) and *Nephtys* juveniles (17.15%). Far samples had a higher average similarity (43.70%) with the species *Nephtys cirrosa* (24.42%), *Nephtys* juveniles (20.15%) and *Bathyporeia elegans* (13.81%) contributing approximately 60% to the difference in total abundances. Communities of far and very close samples had an average dissimilarity
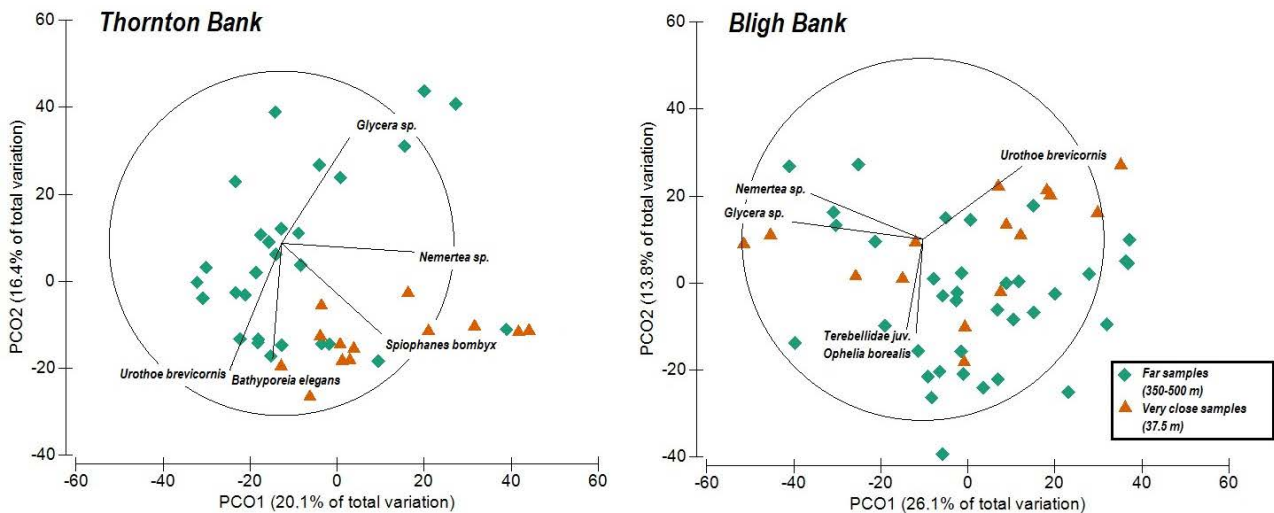
Figure 6. PCO (Principal coordinates analysis) plots based on Bray-Curtis resemblance matrix of fourth root transformed macrobenthic density data at two sandbanks (Thornton Bank and Bligh Bank) at two distances from the turbines (very close – far). Vector overlay was based on Pearson correlations (> 0.5).

of 61.19%. *Urothoe brevicornis* (7.25%), *Bathyporeia elegans* (5.93%), *Ophelia borealis* (5.47%), *Nemertea* sp. (5.46%) and *Glycera* sp. (5.24%) contributed almost 30% to this dissimilarity. Higher abundances of *Urothoe brevicornis* and *Nemertea* sp. were observed in the very close samples while the other three species were more abundant in the far samples. Comparable to the SIMPER results found at TB, many other species contributed to a lesser extent to the observed dissimilarity between distances (table 3).

Multiple regression revealed that only the sediment fraction > 2 mm was a significant predictor for Piélou's evenness, but the model showed a low fit (R^2_{adj} = 0.08). MGS and sediment fraction > 2 mm together explained 12.68% of the total variation in the macrobenthic community structure of BB.

3.2. Baseline analysis at Norther

Sediments found within the future impact area (Norther) and proposed reference area (REF) ranged from very fine sand to coarser sand (MGS: 96 μm-517 μm), but average values were comparable between both locations. The sediment fraction > 2 mm varied from 0.24%-39.46% with higher average values found within the Norther samples compared

to the REF samples. Univariate analysis, however, revealed no significant differences in MGS and sediment fraction > 2 mm between locations. Organic matter content values were significantly higher (all > 1.00%) within REF (1.60 ± 0.50%) compared to samples of the future wind farm area (1.09 ± 0.49%; 1 way ANOVA, p < 0.001).

Relatively high macrofauna densities were found within both locations, and higher average densities were reported within the Norther samples compared to the REF samples (table 2). This tendency, albeit less pronounced, was also found for all the diversity indices (S, J', H'). In contrast, macrobenthos biomass showed a higher average value for the REF samples compared to the Norther samples. However, none of these differences proved to be significant (1 way ANOVA, p > 0.05; table 2).

Multivariate analysis of the macrobenthic community structure revealed significant differences between locations (Permanova, p < 0.05; fig. 8). Similarities within groups were higher for the REF sites (40.20%) compared to Norther sites (37.19%), but *Spiophanes bombyx* was the most dominant species within both locations (16.30% and 10.60% respectively, SIMPER). For the REF

Table 3. SIMPER results with species that contributed to the difference in community composition between the very close and far samples up to a cumulative value of approximately 50% for both sandbanks

Thornton Bank	Group Far	Group Very close	Average dissimilarity between groups 66.17 %	
Species	Avg. abundance	Avg. abundance	Contribution (%)	Cumulative contr. (%)
Nemertea sp.	1.08	2.63	5.58	5.58
Urothoe brevicornis	2.06	3.26	5.27	10.85
Spiophanes bombyx	0.51	1.74	4.17	15.2
Bathyporeia elegans	0.88	1.37	4.05	19.07
Nephtys juv.	2.04	1.70	3.39	22.46
Echinocardium cordatum	0.65	0.93	3.13	25.59
Nototropis swammerdamei	0.25	0.96	2.93	28.52
Terebellidae juv.	0.06	1.10	2.87	31.39
Spio sp.	0.35	0.86	2.46	33.85
Chaetognatha sp.	0.43	0.71	2.45	36.30
Gastrosaccus spinifer	0.43	0.59	2.45	38.75
Ophelia borealis	0.62	0.27	2.14	40.89
Urothoe poseidonis	0.27	0.62	2.00	42.89
Nephtys cirrosa	2.63	2.67	1.97	44.86
Glycera sp.	0.44	0.30	1.88	46.74
Thia scutellata	0.33	0.41	1.68	**48.42**

Bligh Bank	Group Far	Group Very close	Average dissimilarity between groups 61.19 %	
Species	Avg. abundance	Avg. abundance	Contribution (%)	Cumulative contr. (%)
Urothoe brevicornis	0.65	1.48	7.25	7.25
Bathyporeia elegans	1.86	1.79	5.93	13.18
Ophelia borealis	1.24	0.75	5.47	18.65
Nemertea sp.	1.53	2.06	5.46	24.11
Glycera sp.	1.09	0.80	5.24	29.35
Bathyporeia guilliamsoniana	0.00	1.17	5.04	34.39
Nephtys juv.	2.13	1.78	4.97	39.36
Nephtys cirrosa	2.27	0.30	4.59	43.95
Spio sp.	0.95	0.30	4.53	**48.48**
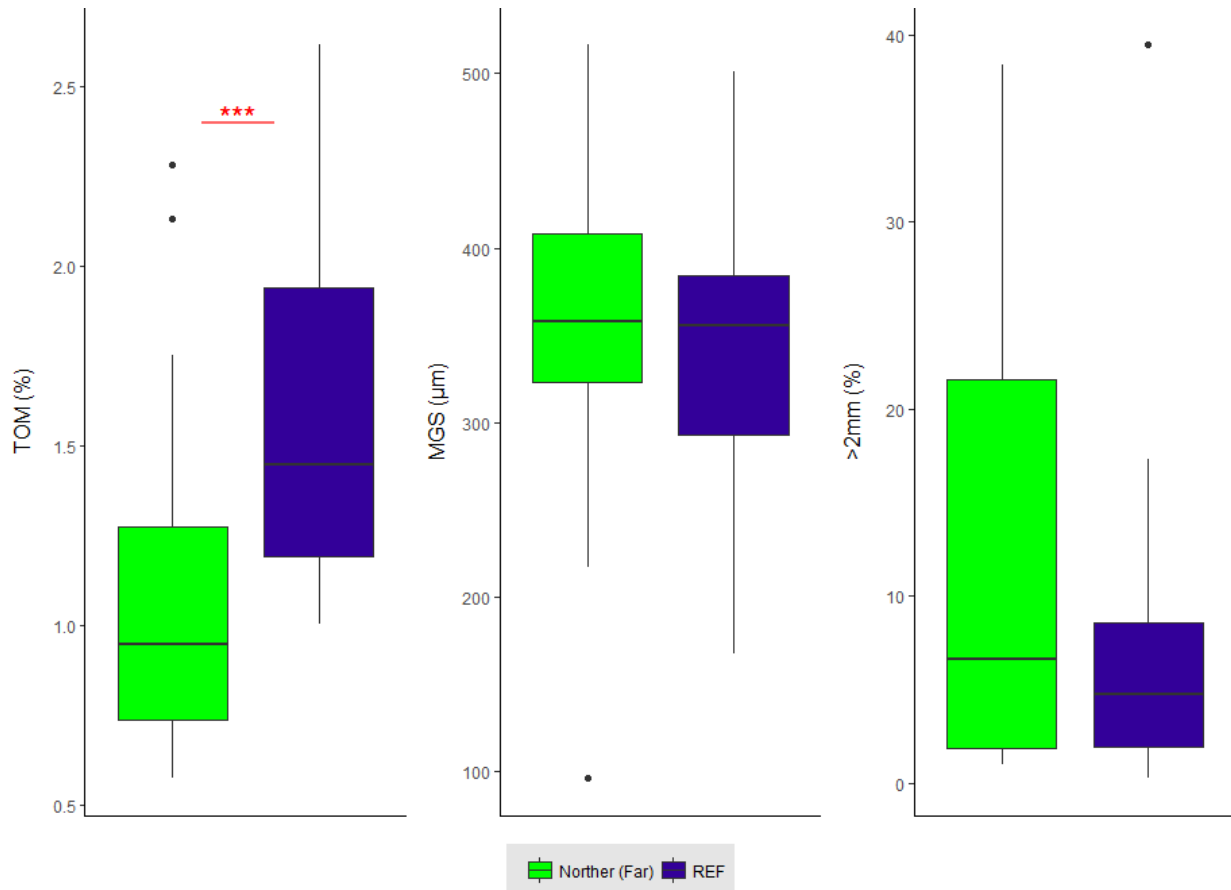
Figure 7. Overview boxplots of abiotic variables: total organic matter (TOM), median grain size (MGS), sediment fraction above 2 mm (> 2 mm) within each location (Norther vs. Reference site). Black dots represent outliers.

sites, the other two most abundant species were *Nemertea* sp. (14.30%) and *Nephtys cirrosa* (5.43%), while for the Norther samples these included *Urothoe breviconis* (9.49%) and *Nemertea* sp. (7.99%). The average dissimilarity between Norther and REF sites was 64.08%. The five most important species contributing over 10% of this differentiation included: *Urothoe brevicornis* (2.47%), *Spiophanes bombyx* (2.16%), *Edwardsia* sp. (2.14%), *Eumida sanguinea* (2.10%) and *Echinocyamus pusillus* (2.08%). All of these species showed higher average abundances in the Norther samples, except for the polychaete *Spiophanes bombyx* which was more abundant in the REF samples. It must be stated, however, that overall relative contributions were low and that many other species contributed to a lesser extent (relative contribution < 2%). A comparable

analysis for multivariate biomass data revealed similar results (Permanova, p < 0.05). Similarities were again higher within the REF samples where biomass was dominated by *Spiophanes bombyx* (13.90%), *Nephtys cirrosa* (9.29%) and *Nemertea* sp. (8.46%). Within the Norther samples, *Nephtys cirrosa* (13.5%) contributed most to overall biomass followed by *Spiophanes bombyx* (9.22%) and *Urothoe breviconis* (8.90%). The average dissimilarity between locations was 68.54% and was mostly due to the species *Echinocardium cordatum* (5.22%), *Ophiura ophiura* (3.96%), *Spisula* sp. (3.46%), *Lanice conchilega* (2.84%) and *Ophiura albida* (2.73%), explaining approximately 20% of the dissimilarity. The first three species had higher average abundances in the REF samples, while the opposite was found for *Lanice conchilega* and *Ophiura albida*.
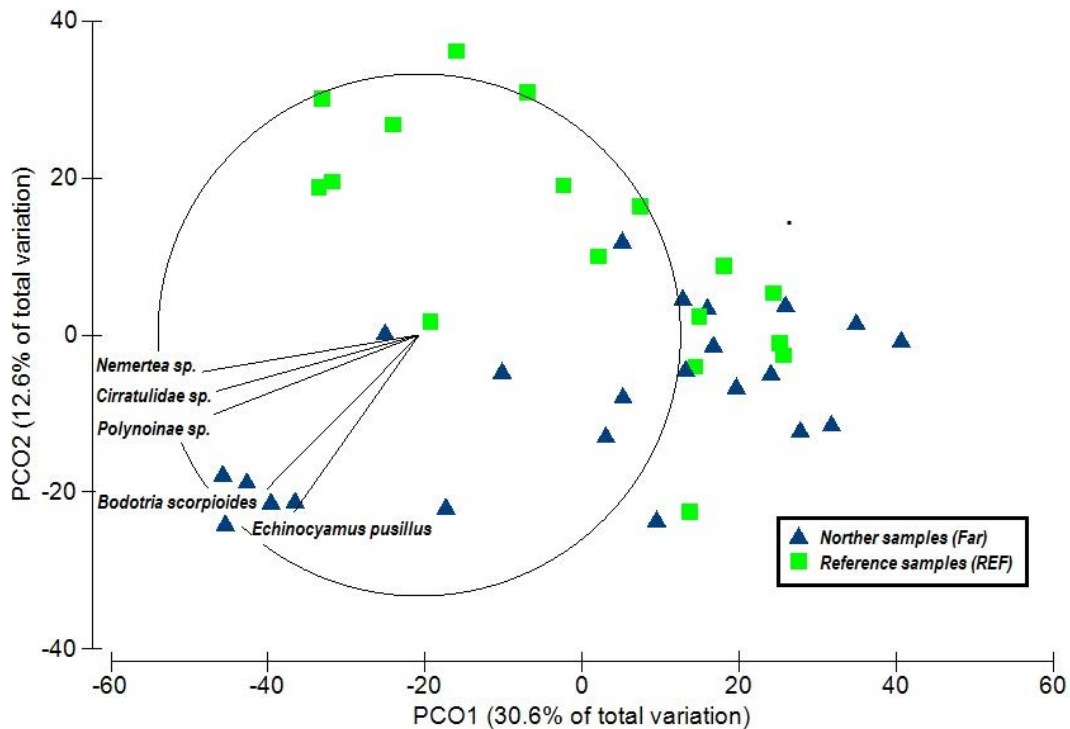
Figure 8. PCO (Principal coordinates analysis) plots based on Bray-Curtis resemblance matrix of fourth root transformed macrobenthic density data at two locations (Norther and reference site). Vector overlay was based on Pearson correlations (> 0.5).

The DistLM analyses showed that all environmental variables had a significant relationship with the multivariate abundance and biomass data, which explained 25.79% and 24.67% of the total variation, respectively.

4. Discussion

4.1. Effects turbine presence on soft sediment macrobenthic communities

The patterns observed on the Thornton Bank correspond to predictions and findings of earlier work describing the 'positive effects' of turbine presence and associated fouling communities on local macrobenthic communities (very) close to the structures (De Backer *et al.* 2014; Coates *et al.* 2014; Martin *et al.* 2005; Maar *et al.* 2009). Turbine foundations are known to change sediment characteristics by modifying local current flows and through the creation of sheltered areas (Leonard & Pedersen 2005). In this study, refinement of the sediment closer to the

turbines is reflected both in terms of a smaller median grain size and an increased fine sand fraction (125-250 µm). The combined effects of these changes to the natural sediment and the local increase of biodiversity due to colonizing epifouling communities seem to have resulted in higher total organic matter concentrations in sediments closer to the turbines. The positive correlation between organic content and fine sediment fraction is a well-known phenomenon (Coates *et al.* 2014; Snelgrove & Butman 1994). Finer sediments have a lower permeability, which in turn facilitates the retention of deposited organic matter (De Backer *et al.* 2014; Janssen *et al.* 2005). Additionally, epifouling communities are known to increase local organic matter input through the deposition of faecal pellets and detritus (De Backer *et al.* 2014; Maar *et al.* 2009; Coates *et al.* 2014).

These changes in sedimentology (grain size and organic matter) also affected the surrounding soft-substrate macrobenthos as predicted by De Backer *et al.* (2014). The

local increase in densities close to the turbines was accompanied by a higher diversity (S, H') and lower evenness (J'). Within the BPNS, abundance and species richness are highly correlated (Van Hoey *et al.* 2004) and rich communities such as the *Abra alba – Kurtiella bidentata* community are generally found in fine to medium sandy sediments (< 300 μm) with significant mud contents. A typical species for this community is the habitat structuring tube polychaete *Lanice conchilega*, which has positive effects on local faunal abundance and richness through its bioengineering capacities (Rabaut *et al.* 2007). Within TB, this species seems to be rare and was only found in one very close sample (TB_VC_16). Despite its cosmopolitan distribution and occurrence in sediments ranging from mud to coarse sands, highest *Lanice conchilega* densities are usually found in shallow muddy and fine sands in coastal areas (Van Hoey *et al.* 2008; Degraer *et al.* 2006). Additionally, Van Veen grabs have a low sampling efficiency for this species due to its rapid retracting ability (up to 20 cm), leading to a potential underestimation of actual densities (Van Hoey *et al.* 2006). SIMPER analysis, however, revealed that the opportunistic polychaete *Spiophanes bombyx* was almost completely absent from the samples far from the turbines. The occurrence of *Spiophanes bombyx* appears to be positively associated with *Lanice conchilega* (Rabaut *et al.* 2007; De Backer *et al.* 2014) and contributes a significant share of the described *Abra alba – Kurtiella bidentata* community along the Northern French and Belgian coast (Van Hoey *et al.* 2004; Van Hoey *et al.* 2005; Desroy *et al.* 2002). In addition, Coates *et al.* (2014) related the enrichment of macrofaunal abundances to the occurrence of *Asterias rubens*, *Lanice conchilega* and *Spiophanes bombyx* close to the studied GBF. Therefore, the higher relative abundances of this species together with other species (*Urothoe brevicornis*, B*athyporeia elegans* and *Nemertea* sp.) indicate a shift towards communities with higher density and diversity.

In contrast to the findings within the TB, no strong effects of turbine presence were found on the Bligh Bank as none of the studied univariate variables differed between distances, except for a lower evenness in very close samples. This is in accordance with a review paper by Jak & Glorius (2017) summarizing current research on macrobenthos in offshore wind farms within the North Sea. It was concluded that effects of turbine presence on soft sediment benthos are unclear and that if effects were found, they were either subtle, temporary or even opposite to expectations. It must be considered, however, that most of the studies were performed relatively shortly after constructions and that minimum distances from windmills were further (> 100 m) compared to our study. Nevertheless, Colson *et al.* (2017) also did not find any effects of turbine presence within BB at a distance of 50 m. In the present study, macrobenthic communities did differ between distances. However, compared to TB, communities from different distances showed lower dissimilarities and less pronounced differences. SIMPER results, however, did show some similarities with results at TB as very close samples showed higher abundances of *Nemertea* sp. and *Urothoe brevicornis*. In addition, very close to the turbines, we observed lower relative abundances of *Glycera* sp. and *Ophelia borealis*, both indicator species for the very low density and diversity *O. borealis-Glycera lapidum* community (Type I SA6; Van Hoey *et al.* 2004). This, together with a weak tendency of higher densities and lower evenness within very close samples, indicates a potential effect of turbine presence in very close vicinity of the structures.

The fact that somewhat different patterns are observed among both banks confirms the lack of consistent responses in current literature: impacts of artificial structures appear to be site-specific and can vary over different spatial scales (Martin *et al.* 2005). It also confirms that distant enrichment effects can be rather subtle and difficult to detect

(Keeley 2013; Jak & Glorius 2017). Changes in sediment type and food supply explained a substantial amount of the turbine-related increases in densities and diversity (S, H') at TB, but not at BB. Nevertheless, DistLM analyses revealed that the environmental variables used in this study did not seem to clarify macrobenthic community structure, especially within BB. This indicates that other local-scale factors may play an important role as well.

Difference in timing of construction between both OWFs might be a temporal factor driving the contrasting results. C-Power has been fully operational since 2013 (4 years), while Belwind since 2011 (6 years). This time-lag can have an impact on the state of the fouling communities on the actual structures, as species richness increases with age since installation (Van der Stap et al. 2016). Therefore, epifauna on the turbines might be in a different phase of succession (Colson et al. 2017; Leonard & Pedersen 2005). Studies on other artificial reefs (platforms, shipwrecks) do show a significant impact of time (Coolen et al. 2015; Van der Stap et al. 2016) and indicate that actual colonization with stable communities is attained after 5-6 years (Leeuwis et al. 2000; Hiscock 2010). As Belwind foundations have been installed for a longer time period, we would expect to find 'stable' epifouling communities with potentially stronger impacts on the environment surrounding the monopiles, but an opposite trend was found. However, offshore wind farm development is a 'young' industry in the North Sea, so long-term data on epifauna communities and studies on their dynamics within these OWFs are scarce. In addition, trends of fouling communities on artificial structures are less predictable than natural reefs and probably depend on several other abiotic factors such as depth, distance from the coast and water currents (Van der Stap et al. 2015).

An alternative explanation may be found in the difference in turbine foundation structure and its associated epifauna: Belwind consists of monopiles, while C-Power has constructed both gravity-based foundations and jackets (Colson et al. 2017; Reubens et al. 2016). Current literature demonstrates a clear vertical zonation on the turbines which appears to be consistently quite different between foundation types (Jak & Glorius 2017 and references therein; De Mesel et al. 2013). De Mesel et al. investigated subtidal fouling communities within TB and BB. Results showed that the Mytilus-zone was well developed (1 m width) on the concrete gravity based foundations on TB, while this zone was much narrower (50 cm) on the steel monopiles at BB. It was also found that communities in the subtidal zone are mostly the same, but that some species were only found on the GBFs and that overall, higher relative abundances were reached at TB. While epifaunal communities on jacket structures have not been studied in detail within TB, these turbines are fully covered with mussels (Reubens, pers. comm.). Krone et al. (2013) studied epifouling dynamics at an offshore platform (FINO-1) comparable to the jacket-based foundations at TB. While species composition on this structure was comparable to findings by De Mesel et al. (2013) and others, it was considered a 'biomass hotspot' with very high densities and biomass of the blue mussel Mytilus edulis ('Mytilisation'). Additionally, offshore oil rigs within the central and northern North Sea were dominated by Mytilus edulis up to depths of 20 m (Whomersby & Picken 2003). It appears that these jacket-like structures are extremely favourable for Mytilus edulis colonization. These bivalves are believed to have a strong impact on the surrounding environment (Krone et al. 2013; Maar et al. 2009). They affect biological activity by influencing particle and sediment fluxes and enrich surrounding sediments through their faeces/pseudo-faeces (Maar et al. 2009). Moreover, their shells provide secondary hard substrate enhancing spatial heterogeneity and associated local diversity (Maar et al. 2009; Krone

et al. 2012; Svane *et al.* 2001). In addition, the amount of newly available substrate differs per foundation type (Rumes *et al.* 2013; Krone *et al.* 2013). Introduced surface area for epifouling colonization in the subtidal zone is highest for jackets (1280 m²) followed by gravity based foundations (671 m²) and monopiles (518 m²) (Rumes *et al.* 2013). These combined effects of lower surface area and 'poorer' epifouling communities in terms of densities and richness may partially explain the contrasting results found in this study.

Finally, it can be expected that the spatial extent of enrichment effects will be dependent on local resuspension processes, transporting organic particles from the 'footprint' area to the adjacent sediments (Keeley 2013). Dispersive capacities of a site are determined by its physical properties such as sediment type, water masses, depth and current speed. Especially the latter two will determine the 'flushing' potential of a site, which affects the accumulation of TOM and nutrient mineralisation (Keeley 2013; Coates *et al.* 2004). With medium to coarse sediments, TB and BB can be considered as highly permeable areas. However, some differences could result in other dispersive properties between both sites. Firstly, both sandbanks are influenced by dissimilar water masses and differ in their relative position and distance from the coastline (Van Hoey *et al.* 2004; Lacroix *et al.* 2004). Secondly, the larger MGS suggests that stronger current velocities are present at BB compared to TB. Finally, regression analysis showed that TOM was not a significant predictor of abundance, diversity or community composition within BB. Therefore, an additional explanation for the lack of a significant enrichment effect at BB could be that BB represents a higher energy/flow system with intense resuspension and ephemeral organic enrichment, leading to no or at most very subtle effects at very close distances.

Previous studies have shown that monopiles are being colonized by epifauna (De Mesel *et al.* 2005) and that these structures alter local hydrodynamics (Leonard & Pederson 2005). The spatial extent of turbine-related effects, however, probably depends on interrelated factors such as a site's dispersive capacity and turbine-specific epifouling potential. As a result, impacts on local soft-sediment communities may only be detectable at distances even closer (< 37.5 m) from the monopile turbines at BB.

4.2. Baseline analysis

Whereas most wind farms are being constructed in more offshore areas, Norther will be situated in the coastal zone. A reference area was chosen directly below the future wind park (southeast border) and is thereby located even closer to the Belgian coastline. The median grain size in both reference and future impact area of Norther was variable, ranging from very fine to coarser sands. However, average values fell within the range of medium sands (250-500 μm), which are widely found within the BPNS (Van Hoey *et al.* 2004; Degraer *et al.* 1999). The higher total organic matter content in the more onshore reference area can be attributed to the onshore-offshore gradient that is established in terms of nutrient availability within the Southern North Sea (Brockemann *et al.* 1990). The higher TOM values within the reference area did, however, not result in notable differences between both areas in terms of macrobenthic diversity, biomass and densities.

While multivariate statistics revealed differences in community structure between the Norther site and the reference area, the PCO (fig. 8) also suggests strong variability within both locations and especially for samples in the future impact area. In addition, SIMPER results indicate that dissimilarities between locations are mostly due to subtle differences in less abundant species (low relative contribution, < 2%) and that true

discriminating species are difficult to confirm. These findings corroborate the results of Van Hoey *et al.* (2004), who observed a high heterogeneity in granulometry and macrobenthic communities within the nearshore area of the BPNS.

Within the reference area, certain samples (REF_13, REF_17, REF_18) showed most similarities with the subtidal *Abra alba – Kurtiella bidentata* community (Type I, SA1). This community is found in nearshore areas with finer sands and high total organic matter content. This community is characterized by high densities (> 2000 ind. m^{-2}) and diversity ($\geq$ 30 spp. sample^{-1}) and by the occurrence of species such as the amphipod *Parambius typicus* and habitat structuring species like *Lanice conchilega* and *Owenia fusiformis* (Rabaut *et al.* 2007; Ropert & Dauvin 2000; Van Hoey *et al.* 2004). Some samples within the future impact area (FAR_14, FAR_17, FAR_20 and FAR_23) were quite distinct and showed no similarity with previously described communities by Van Hoey *et al.* (2004). These samples consisted of finer sands and high gravel fractions (> 20%), indicating the presence of mixed sediment substrate with boulders. These communities had very high total abundances (> 10,000 ind. m^{-2}), high diversity (> 40 spp. sample^{-1}), and were dominated by hard substrate-associated taxa such as *Monocorophium acherusicum*, *Monocorophium insidiosum* and the tanaid *Apseudopsis latreilli* (pers. comm. Francis Kerckhof & Gert Van Hoey). The majority of samples, however, consisted of medium sands and had total abundances between 1000-2000 ind. m^{-2}, probably representing a transitional community (Type II, SA3) between the rich *Abra alba-Kurtiella bidentata* community (Type I, SA1) and more impoverished communities (Type I, SA4 & SA6) found in offshore areas (Van Hoey *et al.* 2004).

The results gave a first insight into the Norther future impact site and the reference area, that have both been used as control for BACI tests to evaluate the future impacts of human-induced perturbations on the benthic ecosystem. Being situated in the coastal zone, different communities were described, therefore the validity of the chosen reference area as a whole can be questioned. In order to reduce the effects of this natural variation, it is proposed to classify the Norther and reference area into different habitat types and corresponding communities. This will allow to perform more reliable comparisons when testing for potential turbine effects in future studies. Despite the variability that was found in terms of granulometry and macrobenthic communities, it can be stated that sediments within the region are mainly composed of medium sands and receive a high amount of organic matter. In addition, high densities and diversity were found and communities were dominated by the common polychaetes *Spiophanes bombyx* and *Nephtys cirrosa*, while many other species contributed to a lesser extent.

5. Conclusion and future perspectives

This study confirms the effects of turbine presence on the surrounding sediment and associated macrobenthos. Refinement and organic enrichment were detected at very close distances (37.5 m) around jacket-based foundations on the TB. While the communities currently found closer to the turbines within TB cannot be described as true *A. alba – K. bidentata* communities, the increase in densities, diversity and the trends in species composition indicate an ongoing shift towards this fine-sediment associated community. Impacts were less pronounced around the monopiles at the BB, where only a difference in communities was detected between both distances from turbines. These contrasting results indicate that turbine-related effects can be site-specific and probably depend on several local-scale factors and/or on turbine foundation type.

Monopiles and jackets are completely different structures with distinct construction activities (scouring protection), shape and subtidal surface area, which in turn affects the colonization patterns of the fouling communities. Differences in epifauna in terms of abundance, diversity and zonation patterns probably influence the distance from where turbine-related enrichment is found. Furthermore, a site's dispersive capacity might also influence the spatial extent of enrichment to nearby sediments.

As the development of offshore wind farms is expanding in the North Sea (Baeye *et al.* 2005), continued monitoring is recommended to understand the impacts that are being found and to fill the current gap of long-term studies. In addition, this study highlights the importance of performing a targeted monitoring study that compares the effects of the three different turbine types (monopiles, jackets and gravity-based

foundations) found in the BPNS. Results found in this study and Coates *et al.* (2013) show that the spatial extent of enrichment effects differs between foundation types. Therefore, it would be more accurate and informative to perform future monitoring at several distances (gradient) from the turbines, with closest samples even closer than the distance used in this study (< 37.5 m). In addition to the established environmental parameters (MGS, sediment fraction > 2 mm and TOM), we propose to also incorporate Chl-*a* measurements, such that food availability can be assessed both in terms of quantity and quality. Moreover, it would be interesting to investigate the macrobenthic communities through the combination of taxon composition (distribution of taxa) – and functional traits analysis to translate community shifts to changes in specific ecosystem functioning rates.

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CHAPTER 6

A CLOSER LOOK AT THE FISH FAUNA
OF ARTIFICIAL HARD SUBSTRATA
OF OFFSHORE RENEWABLES IN BELGIAN WATERS

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Abstract

Artificial hard substrata are known to attract many marine species, among which several highly mobile species. In this contribution, we examined the species composition and uniqueness of the fish fauna around offshore wind turbine foundations in Belgian waters. These offshore structures provide shelter, suitable habitat and a source of food for several fish species. A total of 25 fish species were observed around the turbine foundations, 15 of which are also known to dwell around wrecks in the same area. Four species, the Tadpole Fish (*Raniceps raninus*), the Tompot Blenny (*Parablennius gattorugine*) and the Longspined Bullhead (*Taurulus bulbalis*) were previously rarely or, in the case of the Ballan Wrasse (*Labrys bergylta*), only once reported from Belgian waters. This, however, does not necessarily mean that they are rare. We show that, in order to obtain a good insight into the fish fauna, the use of a suite of varied sampling techniques is necessary. Most of the obligate hard substrata fish species that were observed are frequently recorded in the oyster beds and boulder fields of the nearby Eastern Scheldt estuary. We expect that hard substrata-frequenting fish species will increasingly benefit from the continued expansion of offshore wind farms in the Southern North Sea.

1. Introduction

Natural hard substrata in the North Sea include gravel beds and oyster banks, both of which have been degraded either by over-exploitation or disturbance through bottom disturbing fisheries. As a result, they are characterised by relatively low habitat diversity, and very few hard substrata dwelling fish are encountered (RBINS, unpublished data). Since the latter half of the 20th century, artificial hard substrata have become a common habitat type in the North Sea (Zintzen *et al.* 2008; Coolen 2017). They include seawalls, wrecks, oil and gas platforms and, more recently, offshore renewable developments. The increasing demand for marine renewable energy in the Southern North Sea has resulted in a rapid expansion of artificial hard substrata (*e.g.*, Mineur *et al.* 2012). All these offshore structures provide suitable habitat for a fouling community (Bohnsack 1989; Bull & Kendall 1994; Fabi *et al.* 2006; Leitao *et al.* 2007; Krone *et al.* 2013). The large biomass of invertebrates present in the fouling community on the structures

represents a potentially valuable food resource that attracts many hard substrata-dwelling fish (Pike & Lindquist 1994; Fabi *et al.* 2006; Leitao *et al.* 2007). The artificial structures also provide meeting points for fish and can serve as spawning and nursery sites (Bull & Kendall 1994). Additionally, the crevices and nooks provide refuge against currents and predators (Jessee *et al.* 1985; Bohnsack 1989; Reubens *et al.* 2013).

In this contribution, we perform a preliminary analysis of the composition and uniqueness of the fish fauna around offshore wind turbine foundations in Belgian waters.

2. Material and methods

2.1. Study site

The Belgian part of the North Sea (BPNS) is located in the Southern Bight of the North Sea and is characterised by shallow waters with a complex system of sandbanks. The seafloor consists of mostly sandy and muddy habitats with some smaller natural hard substrata comprised of shell hash, gravels and boulders (Kerckhof & Houziaux 2003). In the BPNS, artificial hard substrata consist of coastal defence structures, wrecks, buoys and offshore wind turbines. There are over 200 wrecks in the BPNS and these have been estimated to increase the total biomass of the BPNS by a maximum of 4% (Zintzen 2007). There are around 150 navigational buoys deployed on the BPNS (data: Agentschap

Maritieme Dienstverlening en Kust, dab Vloot). These floating structures usually provide a habitat for a typical fouling community (Kerckhof 2005). Although these buoys may serve as fish aggregating devices (*e.g.*, Relini *et al.* 2000), we could not find any published data on this for the Southern North Sea. By the end of 2016, 4 wind farms were built in a specially designated zone for renewable energy consisting of 232 turbines and their scour protection resulting in a scattered artificial reef of 0.09 km². By 2020, 5 more wind farms are expected to be constructed (totalling a capacity of 2 GW) and current planning envisions an additional 2 GW between 2020 and 2030 (see chapter 1).

2.2. Species list and categorisation

We compiled a list of fish species observed at offshore wind turbine foundations and wrecks in Belgian waters by combining data collected in the frame work of various projects on the artificial hard substrata in which a range of techniques to collect fish fauna were deployed, supplemented with our own observations (table 1). We compared this list to the dataset of 224 fish species observed by the Institute for Agricultural and Fisheries Research (ILVO), Animal Sciences Unit – Fisheries, Oostende, in the BPNS in their long term fisheries monitoring. ILVO has an epifauna and demersal fish monitoring programme in the BPNS running since 1979. Given the vast differences in both sampling effort and techniques we only used presence/absence data.

Table 1. Overview of datasets used with indication of sampling period, sampling techniques used and habitats sampled

Dataset	Period	Technique	Offshore turbines	Wrecks	Soft substrata
BEWREMABI	2001-2003	Dive transects		X	
ILVO	1979-2017	Beam trawl			X
Reubens PhD	2009-2012	Dive transects, line fishing	X	X	X
Own data	2016-2017	Dive transects, line fishing	X		

Based on the habitat preferences recorded in literature (*e.g.*, Camphuysen *et al.* 2017; Froese & Pauly 2018), we scored the substratum preference of the reported fish species. On the one hand, there are species that permanently live on and in the vicinity of hard substrata (obligate hard substrata species) such as the Tompot Blenny and the Tadpole Fish (*Raniceps raninus*). Other species including many gadoid fish prefer to swim closely to hard substrata (hard substrata associated species). In addition, there are accidental passers-by and indifferent species, *i.e.*, species that are found around hard substrata but do not have a specific substratum preference.

Based on literature (*e.g.*, Camphuysen *et al.* 2017; Froese & Pauly 2018), we further assigned the different fish species to relevant habitat groups according to the zone in the water column they inhabit: (1) benthic fish, *i.e.*, fish living on the bottom of the sea, examples are Dab (*Limanda limanda*) and Common Sole (*Solea solea*), (2) benthopelagic fish, *i.e.*, fish that live in close association with the bottom of the sea but do not rest on the bottom, examples are Pouting (*Trisopterus luscus*), Atlantic Cod (*Gadus morhua*) and Sea Bass (*Dicentrarchus labrax*), and (3) pelagic fish, *i.e.*, those fish living at mid-water or surface levels, examples are Mackerel (*Scomber scombrus*) and Horse Mackerel (*Trachurus trachurus*).

3. Results

3.1. Species richness

Within the framework of the different projects, a total of 25 fish species were observed at the different man-made offshore hard substrata (table 2). Of these 25 species, 15 are related to hard substrata and 5 have an outspoken hard substratum preference. Fifteen species were observed both near the turbines and the wrecks. Only bony fish were observed in the various studies, no sharks nor rays. One species, the Tadpole Fish, was not at all listed in the ILVO database.

Two other species Tompot Blenny and Longspined Bullhead were previously to this research rarely recorded and the Ballan Wrasse (*Labrys bergylta*) was only once reported in the past. None of the species solely occurs on artificial hard substrata within their distribution area.

4. Discussion

4.1. Remarkable species

Four species were previously tot this research rarely or only once reported in Belgian waters: Tadpole Fish, Ballan Wrasse, Tompot Blenny and Longspined Bullhead. For the first three species, we here provide some further knowledge on their habitat, ecology and geographic distribution. We do not discuss the Longspined Bullhead into detail, as it in fact is a known common inhabitant along the Dutch and Belgian inshore waters (Nijssen & De Groot 1987; Rappé & Eneman 1988) and hence seems to be underrepresented in our database.

Tadpole Fish Raniceps raninus (Linnaeus, 1758)

The Tadpole Fish (fig. 1) lives solitary in areas of rocks and boulders, and hides in crevices during the day. It is most often found in shallow water (1-20 m) but occurs up to a depth of 100 m. It is active at night and localizes prey with a sensitive probe wire.

Figure 1. Tadpole Fish (*Raniceps raninus*), Eastern Scheldt estuary. Photograph by S. Jansens.

Table 2. Fish species observed at the offshore turbines with their substratum preference (obligate hard substrata species: (2) hard substrata associated species: (1) accidental passers-by and indifferent species: (0) and habitat group

Latin name	Common name	Habitat group	Substratum preference	Turbines	Wrecks	Soft substrata
Blenniidae spec.	Blenny	benthopelagic	1	X		X
Callionymus lyra	Dragonfish	benthopelagic	0	X		X
Dicentrarchus labrax	Sea Bass	benthopelagic	1	X	X	X
Gadus morhua	Atlantic Cod	benthopelagic	1	X	X	X
Gobiinae	gobies	benthopelagic	1	X	X	X
Labridae spec.	Wrasse	benthopelagic	1	X		X
Labrus bergylta	Ballan Wrasse	benthopelagic	1	X		X
Limanda limanda	Dab	benthic	0	X		X
Merlangius merlangus	Whiting	benthopelagic	0	X	X	X
Microstomus kitt	Lemon Sole	benthic	0	X		X
Mullus spec.	Goatfish	benthopelagic	0	X		X
Myoxocephalus scorpius	Bull Rout	benthopelagic	2	X	X	X
Parablennius gattorugine	Tompot Blenny	benthopelagic	2	X	X	X
Pleuronectes platessa	European Plaice	benthopelagic	0	X	X	X
Pollachius pollachius	Pollack	benthopelagic	1	X	X	X
Pollachius virens	Saithe	benthopelagic	1	X	X	X
Raniceps raninus	Tadpole Fish	benthopelagic	2	X		
Scomber scombrus	Mackerel	pelagic	0	X	X	X
Solea solea	Common Sole	benthic	0	X		X
Spondyliosoma cantharus	Black Seabream	benthopelagic	2	X	X	X
Taurulus bubalis	Longspined Bullhead	benthopelagic	2	X		X
Trachinus vipera	Lesser Weever	benthopelagic	0	X	X	X
Trachurus trachurus	Horse Mackerel	pelagic	0	X	X	X
Trisopterus luscus	Pouting	benthopelagic	1	X	X	X
Trisopterus minutus	Poor Cod	benthopelagic	1	X	X	X

Its diet consists of crustaceans such as shrimps, molluscs and smaller bottom fish (Middeldorp 1978; Nijssen & De Groot 1987). In the turbine scour protection it can find both suitable shelter and food.

This species is known in suitable habitats from Norway and Iceland through the North Sea and along the Atlantic coast down to Portugal. It is sporadically caught along the Dutch and Belgian coast (Nijssen & De Groot 1987; Rappé & Eneman 1988) and although there have been a few recorded landings by Belgian fishermen, their exact origin remains unclear.

Ballan Wrasse Labrus bergylta Ascanius, 1767

The Ballan Wrasse (fig. 2) is an omnivorous fish and its diet mainly consists of crustaceans, molluscs and ophiurids although it can also feed on algae (Dipper *et al.* 1977). It is long-lived (up to 29 years), with slow growth and a protogynous hermaphrodite without sexual dimorphism in colour (Dipper *et al.* 1977). They can be found at depths from 1 to 50 m amongst rocks, seaweed and reefs. The rocky turbine scour protection forms a suitable habitat with abundant food.

This species of wrasse is native to the northeastern Atlantic Ocean from Norway to Morocco, including the islands of Madeira, the Azores and the Canary Islands (Quignard & Pras 1986). Despite being locally quite rare, likely due to the scarcity of suitable habitat in the Southern North Sea (Redeke 1941; Poll 1943), this is the most common wrasse in our waters (Rappé & Eneman 1988).

Tompot Blenny Parablennius gattorugine (Linnaeus, 1758)

Tompot Blennies (fig. 3) live in rocky areas of mostly shallow waters, in areas with a lot of shelters where they guard a territory and where the females lay eggs in rock crevices during the spawning season (March-May) (Dunne & Byrne 1979; Picton & Morrow 2016). This territory is fiercely defended and the eggs are also guarded by the male (brood care) (Naylor & Jacoby 2016). Tompot Blennies can also be found on wrecks, at depths of 30 m or more and in the tidal zone along rocky coasts. The fish are mainly hunting at dusk and at night. The animals are curious and often appear to observe divers (Holstein & Ates 1999). The food consists of crustaceans, worms, echinoderms and other invertebrates, but also sea anemones and seaweeds (algae) would be eaten (Dunne & Byrne 1979; Milton 1983). In addition to providing suitable habitat, shelter and abundant food, the turbine scour protection likely serves as a location to deposit eggs.

Figure 2. Ballan Wrasse (*Labrys bergylta*) caught in the Belgian part of the North Sea. Photograph by H. Hillewaert.

Figure 3. Tompot Blenny (*Parablennius gattorugine*). Photograph by F. Pointel.

This species is uncommon in the Southern Bight of the North Sea and occurs further along the East Atlantic from Ireland to West Africa (Morocco) and in the Mediterranean. The species is known from the Wadden Sea and especially from the Zeeland waters, where the species is mainly seen on stony dikes and on mussel and oyster beds, but is very rare along the Dutch and Belgian coast (Camphuysen & Henderson 2017; Nijssen & De Groot 1987; Rappé & Eneman 1988). However, the number of sightings over the 20th century is increasing. Along the Belgian coast there are recent findings on groynes and in coastal waters (waarnemingen.be). Dutch reports area mostly from the Delta area (Holsteijn & Ates 1999). The species probably benefits from rising water temperatures in winter and the increasing availability of the desired rocky habitat.

4.2. Fish diversity at hard substrata of offshore renewables in Belgium and beyond

The turbine foundations were quickly colonized by a diverse fouling community (Kerckhof *et al.* 2010) that provided a source of food for several fish species (Reubens *et al.* 2011; 2013). Our results show that at least 25 fish species are present near the turbine foundations. We remark that none of the species solely occurs on artificial hard substrata within their total distribution area, because even obligate hard substrata species can occasionally also be found on soft sediments, *e.g.*, when moving from one discrete hard substratum area to another one.

As the size and number of the suitable habitat increases because of the continued expansion of the offshore renewable developments, both further offshore as well as nearer to the shore, we can expect that in the future several additional fish species with affinities to hard substrata will be observed, such as Butterfish (*Pholis gunnellus*), Five- and Four-Bearded Rockling (*Ciliata mustela* and *Enchelyopus cimbrius*), Sea Horses

(*Hippocampus hippocampus* and *H. ramulosus*), European Conger (*Conger conger*) and, in particular, wrasses that are the most conspicuous and characteristic species of fish associated with reef habitats, especially the Goldsinny Wrasse (*Ctenolabrus rupestris*) a species already commonly reported in other studies (Krone *et al.* 2016; van Hal *et al.* 2017). Besides the effect of the proliferation of artificial hard substrata in the North Sea, also the increased sampling effort in that habitat may contribute to the discovery of new (hard substratum) species. The numerous planned environmental monitoring programmes targeting the effects of offshore renewables will hence increase the likelihood of detection of those fish and will undoubtedly complete the species list. These artificial substrata may also serve as a nursery for certain species and thus, in part, offset the degradation that has occurred in the natural hard substrata (Veer *et al.* 2015).

The faunal composition around wrecks and other artificial hard substrata is comparable with our findings, with an assemblage consisting of numerically dominant gadoids and a limited number of other species of which the obligate hard substrata species and hard substrata associated species are typical. Similar patterns in the fish assemblage with the dominance of gadoids and some typical rock associated species, including some 'rare' species, are also reported in other studies on the fish fauna of artificial hard substrata elsewhere in the North Sea, *e.g.*, around oil platforms in the northern North Sea Guerin (2010) and, wind turbine foundations in the Netherlands (van Hal *et al.* 2017) and the German Bight (Krone *et al.* 2016). These studies report several rare species, associated with hard substrata that we also observed and some others not yet reported from artificial hard substrata in the BPNS. All these studies, including ours, reveal the presence of several species such as Ballan Wrasse, Longspined Bullhead and the Tompot Blenny.

Published data on the fish fauna of natural hard substrata such as boulder and stony reefs are rare for the BPNS. However, studies on the natural hard substrata elsewhere show a similar species pool that can benefit from natural hard substrata restoration efforts (Støttrup *et al.* 2014). Additionally, most of the obligate hard substrata fish species that we observed are frequently recorded in the oyster beds and boulder fields in the nearby Eastern Scheldt estuary where many additional fish species are observed by divers (Bob Rumes,RBINS, personal communication). However, within the limited part of the North Sea in which this study was conducted, offshore wind farms do represent artificial hard substrata that are unique for certain habitat features. The fact that offshore wind turbines introduce huge vertical surfaces indeed is new to the offshore waters in the southern North Sea and may hence attract unique fish species not encountered in natural hard substrata in that area.

Many factors influence the number of reported species for the different habitat types. The species assemblage may depend upon environmental variables such as the design and the material of the structures (Bohnsack & Sutherland 1985; Relini *et al.* 2007). The hard substrata from artificial reefs provide shelter for predation and/or prevailing currents for fish species (Langhamer 2012; Reubens *et al.* 2014). The fouling community on the structures in its turn further increases the structural complexity and also provides food and shelter, and may hence itself influence the nature of the fish community (Hueckel & Buckley 1989; Hueckel *et al.* 1989). It is expected that the number of fish species associated with a certain habitat will – in part – be determined by the variation in prevailing aforementioned ecosystem features of this habitat. We could thus rank the different offshore artificial hard substrata in the BPNS in broad categories. Buoys would provide meeting points with only minor food resources. Turbines without scour protection would provide meeting points with better food resources given the longer life cycle of the turbines, the fact that they stretch the entire water column, and their larger scale. Wrecks and turbines with scour protection may finally also serve as spawning and nursery sites and, their crevices and nooks provide refuge against currents and predators.

On the other hand, the number of fish species reported is positively related to the use of various types of sampling techniques as well as to the sampling effort. Each sampling method has a specific catch selectivity (at species level) and will hence render its own specificity to the dataset. The deployment of scuba divers, for example, is very limited, both in space and time, and requires the ability to recognise the species. Line fishing is known to be a selective fishing method and is influenced by type and size of baits, hook design, hook size, fishing strategy and fish ecology (Erzini *et al.* 1996; Løkkeborg & Bjordal 1992; McClanahan & Mangi 2004; Ralston 1990), which may explain the low number of species observed by this technique. This sampling technique – and effort – dependent representation of fish species in the databases most likely explains the underrepresentation of the Longspined Bullhead in our database. A combination of sampling methods hence is expected to yield the best impression of the ecosystem and biodiversity, but such an approach is rare. Furthermore, also variation in time cannot be ignored. Seasonal and year-to-year patterns in fish presence may be observed, certainly in temperate waters, as many species migrate towards deeper water when temperature drops (Fabi & Fiorentini 1994). The deployment and combination of different techniques over a longer time frame hence are advised to obtain a comprehensive view of the fish communities associated with hard substrata. The findings presented in this study should thus be interpreted with care: it has to be considered as a minimum estimate of the total species number in the different hard substrata. Nevertheless, the

unique availability of fish data collected with different techniques targeting different habitat types in a restricted area (*i.e.*, BPNS), may be considered reliable for preliminary assessing the contribution of artificial hard substrata to the local fish communities.

5. Conclusions

Artificial hard substrata offer habitat to hard substrate fish among which some species previously rarely observed from the BPNS. We show that the deployment of offshore wind turbines alters the habitat for fish due to the introduction of hard substrata in an otherwise sandy area. This has proven an advantage for several special species that previously could not survive in this area. Where scour protection is present, it provides similar functions to the natural boulder and gravel fields thus increasing the surface area available for species of hard substrata. Additionally, we demonstrated that the use of a suite of varied sampling techniques is necessary to gain a proper insight in the fish biodiversity.

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CHAPTER 7

LESSER BLACK–BACKED GULL DISTRIBUTION IN AND AROUND THE THORNTON BANK OFFSHORE WIND FARM USING GPS LOGGER DATA

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Abstract

We analysed GPS data of lesser black-backed gulls (*Larus fuscus*) caught and tagged in the colonies at Ostend and Zeebrugge. After exploring general patterns in at-sea presence and behaviour, we performed three modelling exercises to study the response of lesser black-backed gulls towards the C-Power turbines at the Thornton Bank offshore wind farm (OWF) in more detail. These exercises confirmed that much more time was spent roosting on outer than on inner turbines. Next, we found a significant and gradual increase in the number of logs of flying birds going from the centre of the wind farm up to 2000 m from the wind farm edge, beyond which the response seemed to stabilise. For non-flying birds too, the model predicted a minimum number of logs in the centre of the wind farm and a flattening of the smoother at about 2000 m, yet with a highly increased presence right at the wind farm's edge, representing birds roosting on the outer turbine foundations. The last model, aiming to assess temporal variation in the presence of lesser black-backed gulls in and around the Thornton Bank OWF, showed that the birds were increasingly wary of entering the wind farm during times of strong winds with fast moving rotor blades. The results of this study illustrate that the response of lesser black-backed gulls towards OWFs can be subject to both temporal and (within-OWF) spatial variation, which in turn can be of high value in refining collision risk modelling.

1. Introduction

In this chapter we will analyse GPS data of lesser black-backed gulls (*Larus fuscus*) caught on the nest and tagged in the colonies at Ostend and Zeebrugge. GPS data have the major advantage of providing detailed information on the movements of individual birds, without being limited to specific time frames or environmental conditions as is the case with seabirds-at-sea monitoring. Moreover, when enough individuals of a specific colony or population are included, the cumulative data no longer reflect individual birds' preferences but allow for a general characterisation of their behaviour and distribution. First we will explore the dataset looking for general patterns in at-sea presence and behaviour, comparing these with the patterns observed in and around the C-Power offshore wind farm (OWF) at the Thornton Bank. Next, we will turn to three modelling exercises for a detailed study of the response of the tagged lesser black-backed gulls towards the Thornton Bank OWF.

2. Material and methods

2.1. Overall data selection

From 2013 to 2017, 133 lesser black-backed gulls breeding in Zeebrugge (77 birds), Ostend (6 birds) and Vlissingen (50 birds) have been equipped with an UvA-BiTS tracker (Bouten *et al.* 2013). As the colony of Vlissingen is located over 40 km from the Thornton Bank OWF, we only considered birds tagged in Zeebrugge and Ostend, and further selected all at-sea GPS logs at least 1 km from the shoreline and within 80 km from the colony of origin (fig. 1). Because the Thornton Bank OWF was only fully operational from the summer of 2013 onward, we further discarded all 2013 data from the analyses.

Resulting from differing needs and priorities of the GPS data end users, tracking resolution varied strongly from 10 to 3600 seconds. To obtain a balanced dataset, we selected one data point per 20 minutes for tracks with a higher resolution and deleted tracks with a resolution lower than 20 minutes. The choice for this 20-minute boundary was based on the fact that it is the original resolution for about half of the total tracking time in the regarded dataset. Meanwhile, by bringing down the tracking resolution, we avoided temporal correlation between records (Ross-Smith *et al.* 2016). This data selection was applied in all calculations except when assessing the actual time spent in a certain area.

About 40% of the birds from Ostend and Zeebrugge (34 individuals) were recorded inside the Thornton Bank OWF at least once (fig. 2), allowing for a characterisation of their presence and behaviour inside the wind farm compared to the surrounding or wider area. The tagged birds visited the further offshore Northwind and Belwind wind farms to a far lesser extent and interaction with these wind farms was therefore not considered in this chapter.

Furthermore, 48 herring gulls were equipped with a UvA-BiTS tracker, 37 in Ostend and 11 in Zeebrugge. Herring gulls generally stayed closer to the shore compared to lesser black-backed gulls, and never ventured far enough offshore to encounter the wind farms currently present in the Belgian part of the North Sea (BPNS) (fig. 3). Because of this lack of interaction with offshore turbines, herring gulls were not considered further on in this report.
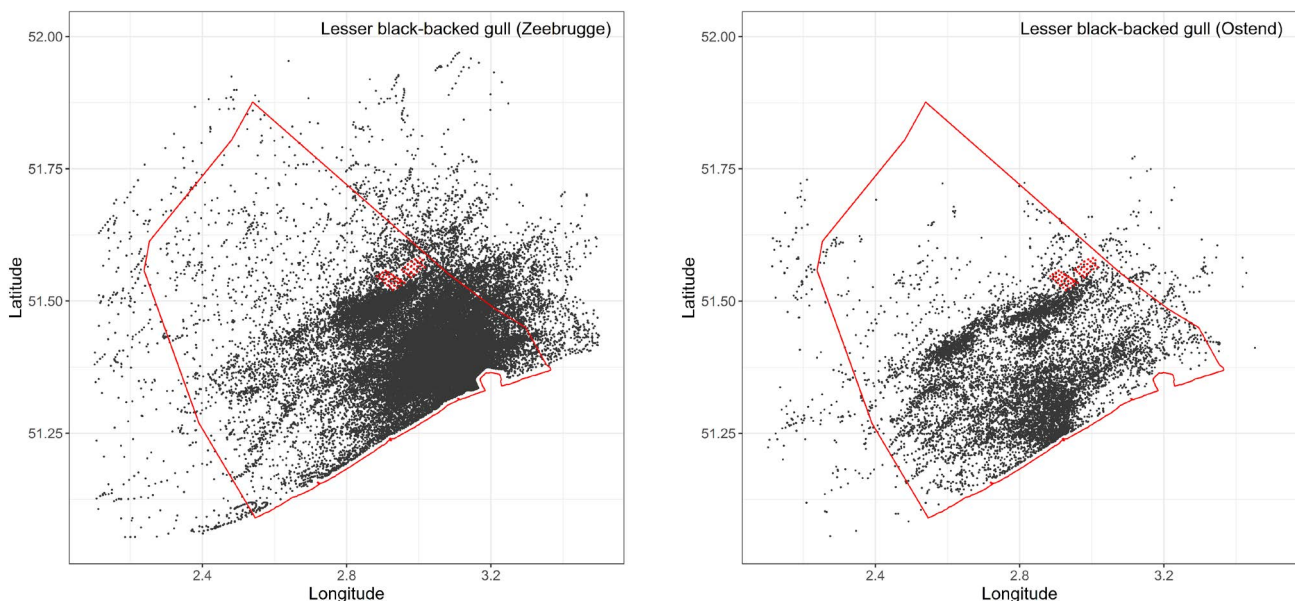

Figure 1. Twenty-minute interval GPS logs of lesser black-backed gulls (period 2014-2017) originating from Zeebrugge and Ostend; the Belgian North Sea border and the turbines at the Thornton Bank are indicated in red.
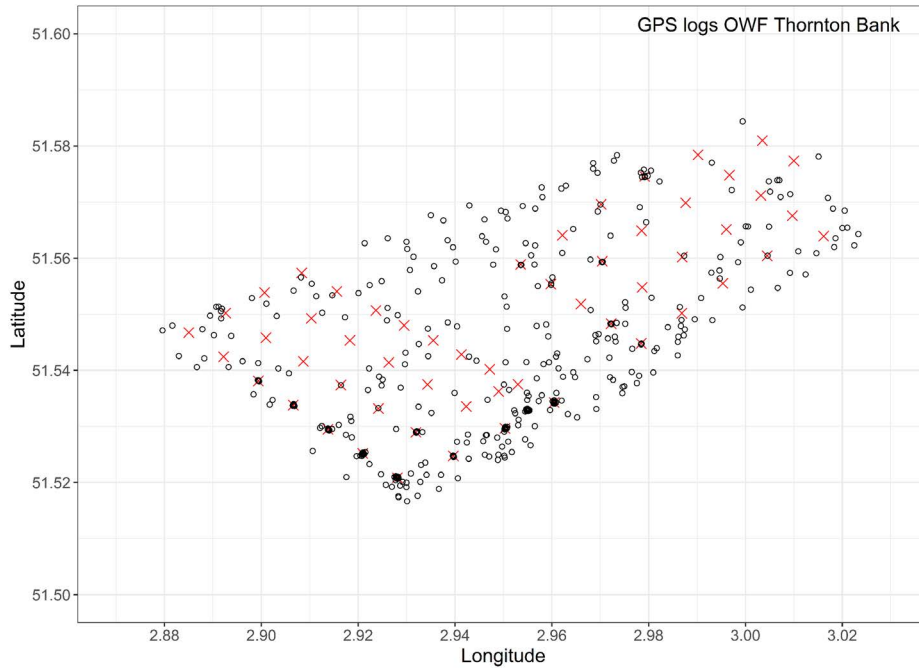
Figure 2. Twenty-minute interval GPS logs of lesser black-backed gulls (period 2014-2017) inside the Thornton Bank OWF; the turbines are indicated in red.
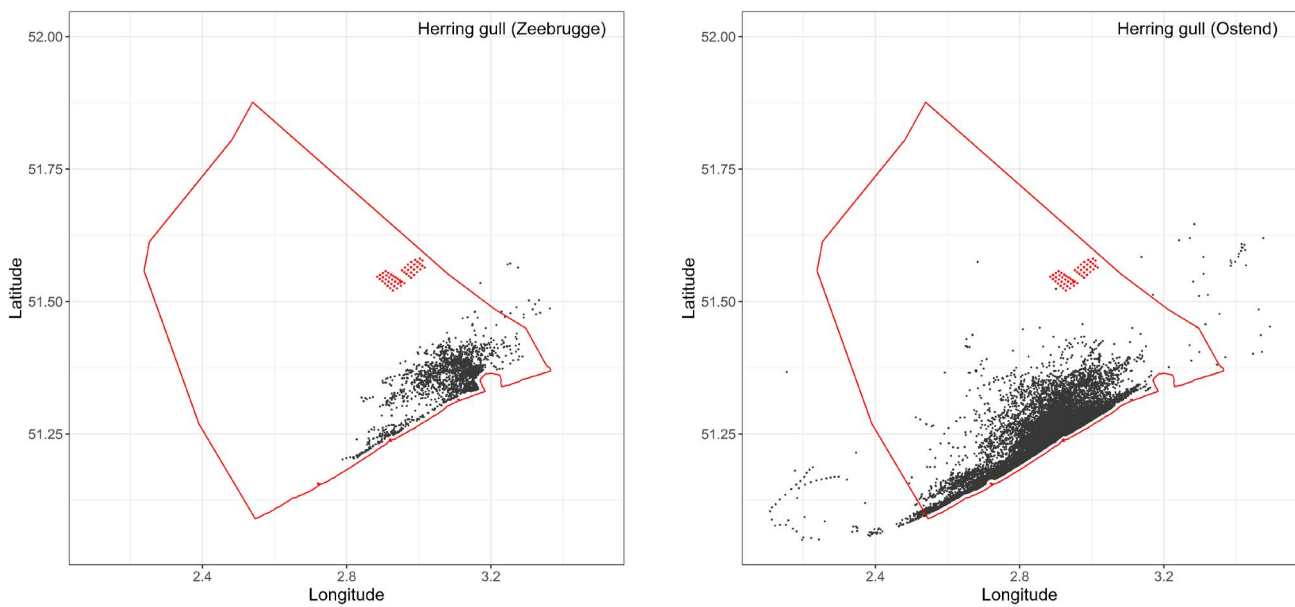

Figure 3. Twenty-minute interval GPS logs of herring gulls (period 2014-2017) originating from Zeebrugge and Ostend; the Belgian North Sea border and the turbines at the Thornton Bank are indicated in red.
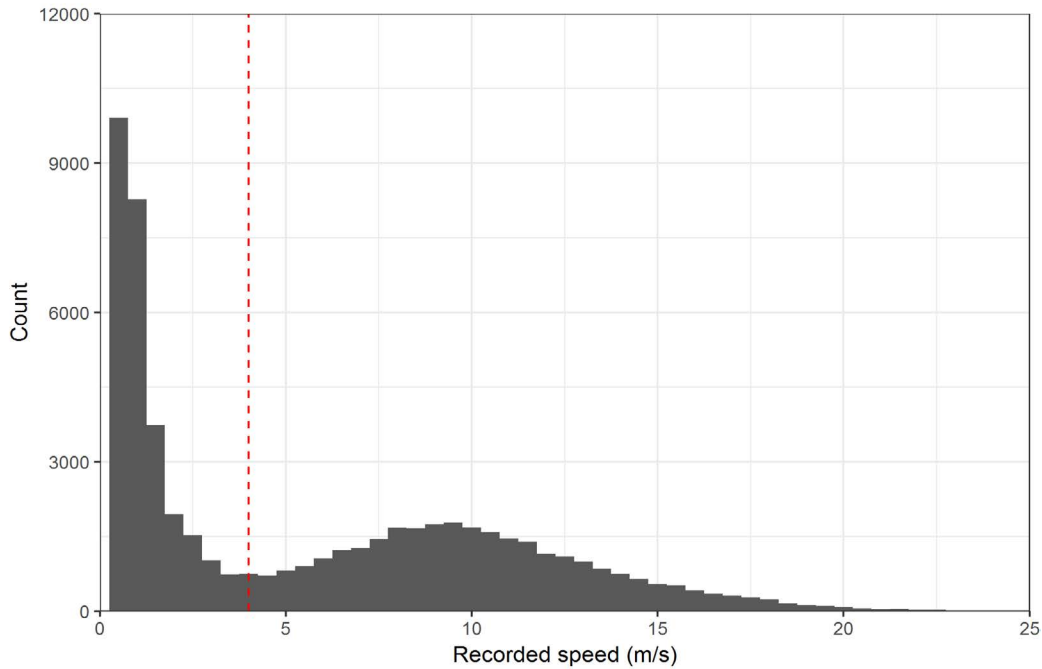
Figure 4. Histogram of the recorded ground speeds.

2.2. Data exploration

Based on the resulting dataset of 59,493 GPS logs, we explored how flight height, direction and activity related to the time of day, tidal height, wind velocity and wind direction, and investigated whether the patterns found varied across the BPNS. Birds were considered flying when their recorded ground speed exceeded 4 m/s, a cut-off speed coinciding with the minimum indicated in the bimodal histogram displayed in fig. 4. This value appears to be on the high side, as Gyimesi *et al.* (2017) applied a cut-off of 2.5 m/s, while Ross-Smith *et al.* (2016) used a value of only 1.1 m/s (4 km/h). Because tidal currents in the BPNS may already reach 1 m/s or more (Ruddick & Lacroix 2006), the latter seems to be an absolute minimum for birds logged at sea. On the other hand, Baert *et al.* (2018) used a measured ground speed of 4.5 m/s to discern active flight from a variety of behaviours (standing, resting, walking, floating, soaring and tortuous flight). Anyhow, knowing that the GPS speed measurements are subject to considerable error (Bouten *et al.* 2013) and based on the strongly bimodal pattern in fig. 4, the value of 4 m/s seemed to

be the best possible guess in the framework of this study.

Data on tide and wind conditions were queried from the Monitoring Network Flemish Banks by means of the LifeWatch Data Explorer (http://rshiny.lifewatch.be/MVB%20data/). The variables 'mean wind velocity' and 'mean wind direction' are based on offshore measurements at the Westhinder station, while 'tidal height TAW (cm)' measurements originate from the Ostend station, all with a sample period of 60 minutes.

When comparing general at-sea gull behaviour to the behaviour recorded in or around the Thornton Bank wind farm, we often made subsets of data as illustrated in fig. 5. These selections were based on the same before-after control-impact (BACI) polygons used in the displacement analyses in previous reports (*e.g.*, Vanermen *et al.* 2017), being a wind farm area (the turbine-built zone surrounded by an initial 0.5 km buffer), a buffer zone (the area 0.5-3.0 km from the nearest turbines) and a control area at a comparable distance to the shore and including the SW part of the Thornton Bank as well as the Goote Bank.
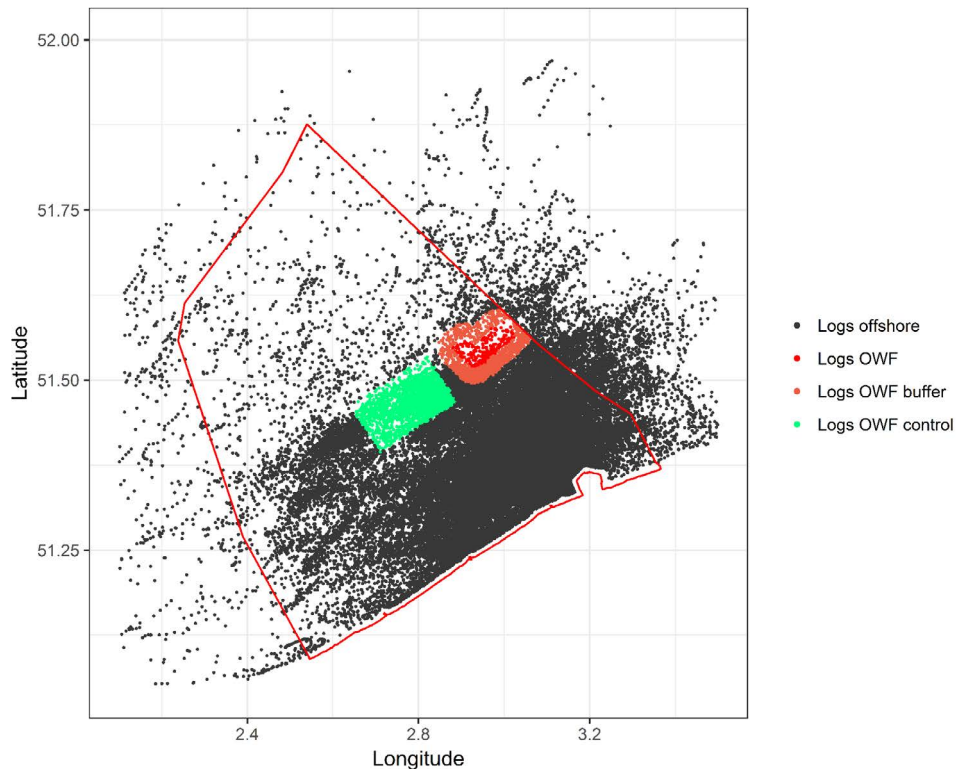
Figure 5. Data selection for comparison of bird behaviour inside versus outside the Thornton Bank OWF.

2.3. Modelling exercises

2.3.1. Association with turbine foundations

In order to explore the gulls' roosting behaviour on turbine foundations, we calculated the time spent in (1) 100 m wide buffer areas around the turbines and (2) the OWF as a whole by summing the time intervals between the first and last GPS log of each visit to these respective areas. This implies that single 'isolated' logs were not taken into calculation, but also that we assume birds to stay within the area boundaries between two subsequent logs inside these areas. Next, we modelled whether the time spent on a turbine foundation is affected by the distance from that turbine to the wind farm edge.

2.3.2. Modelling the effect of distance

To study the effect of distance to the wind farm on the presence of lesser black-backed gulls, we built a grid of 500x500 m cells up to a distance of 3250 m to the nearest turbine (fig. 6). Extending this distance was not

feasible due to the presence of the Northwind wind farm just north of the Thornton Bank. For each grid cell, the distance from its centroid to the wind farm edge was calculated. We then modelled the effect of distance to the wind farm edge on the number of logs per grid cell applying a smoother, both for flying and swimming/resting birds.

2.3.3. Modelling temporal variation

To study the temporal variation in the presence of lesser black-backed gulls in and around the Thornton Bank OWF, we defined eight 5x5 km grid cells aligned with the Thornton Bank. The two middle cells include the wind farm (the 'impact' cells), the other six cells being 'control' cells (fig. 7). We then generated a dataset with one line for each hour and each grid cell in the months of March to August in the years 2014 to 2017, resulting in 141,128 rows. The dataset was thus limited to the spring and summer period, but note that only 0.3% of our 59,493 at-sea GPS logs within 80 km from the colony (2.1) was logged in the months of
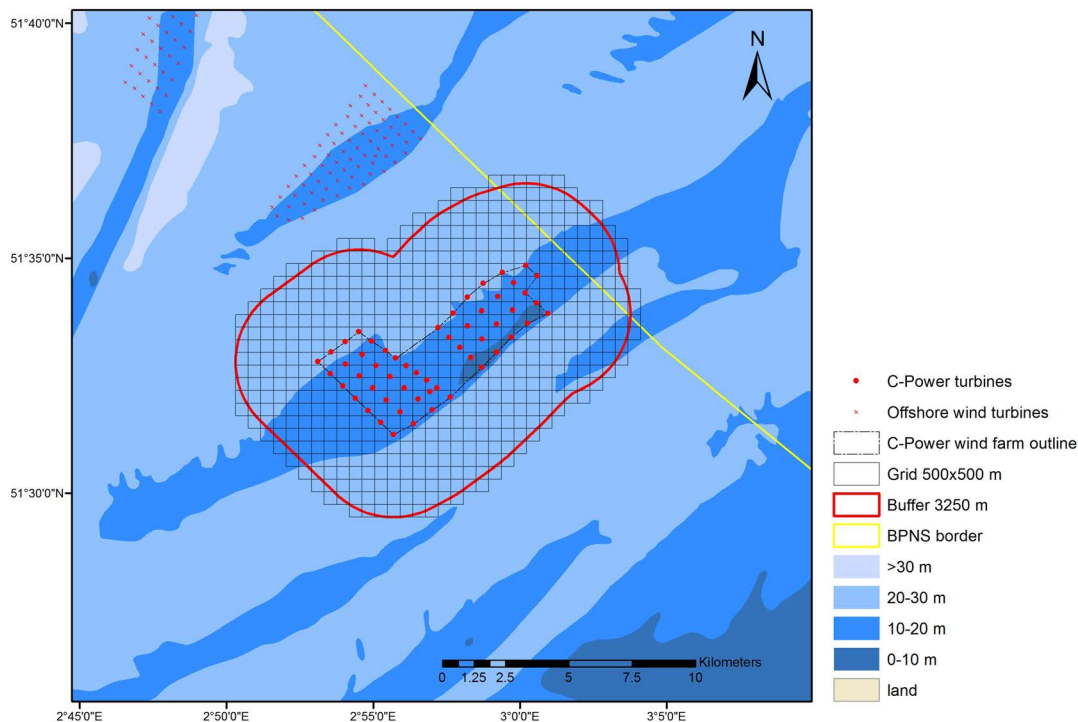
Figure 6. Grid of 500x500 m cells for modelling the effect of distance to the OWF edge on the number of lesser black-backed gull logs per grid cell.
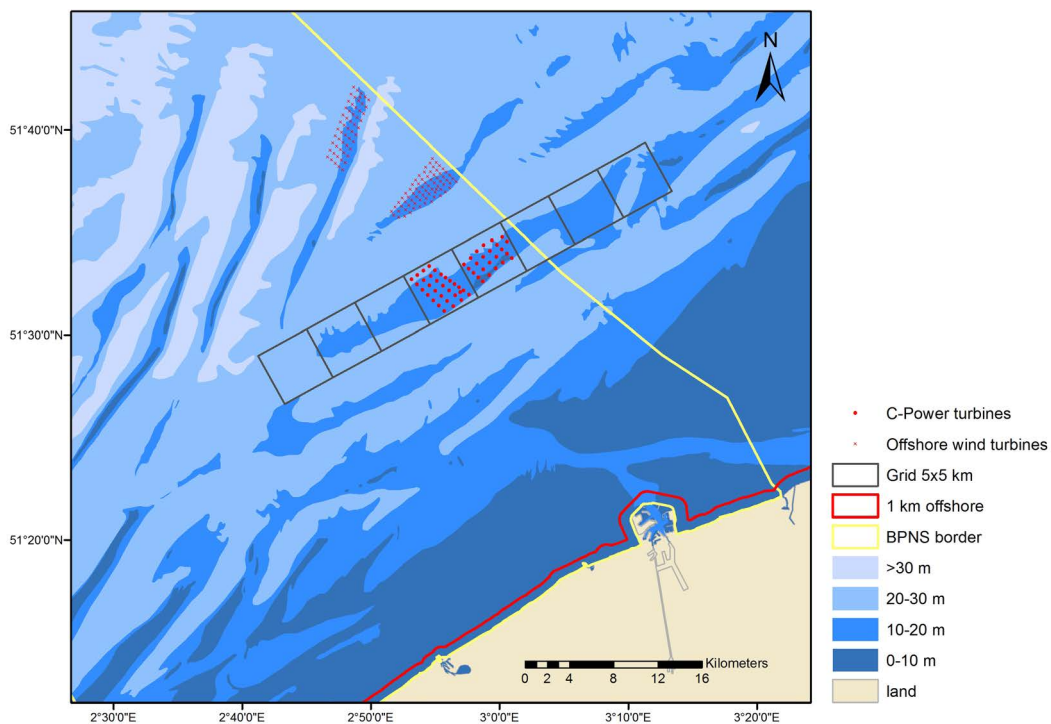

Figure 7. Grid of eight 5x5 km cells for modelling temporal variation in the presence of tagged lesser black-backed gulls in and around the OWF at the Thornton Bank.

September to February. Next, the response variable was calculated by aggregating the number of GPS logs per grid cell and per hour. The explanatory variables included in the dataset were wind speed, tidal height and hour of the day, next to the factor variables weekend/week and control/impact. Finally, this dataset was modelled using an information-theoretic approach, first to select the appropriate data distribution, and next to perform a backward model selection.

2.4. Statistics

All data processing and analyses were performed in R version 3.4.2 (R Core Team 2017) using RStudio (RStudio Team 2016) and the following packages (in alphabetical order):

- data.table (Dowle & Srinivasan 2017)
- ggplot2 (Wickham 2009)
- MASS (Venables & Ripley 2002)
- mgcv (Wood 2017)
- plyr (Wickham 2011)
- pscl (Zeileis *et al.* 2008)
- reshape (Wickham 2007)
- rgdal (Bivand *et al.* 2017)
- rgeos (Bivand & Rundel 2017)
- spatialEco (Evans 2017)
- sp (Pebesma & Bivand 2005)

3. Results

3.1. Data exploration

3.1.1. Distribution patterns

Figure 1 already illustrated that the Thornton Bank OWF falls within the normal at-sea distribution of lesser black-backed gulls breeding in Zeebrugge and Ostend. We can further illustrate this by plotting the cumulative proportion of the number of logs against the distance to the colony of origin. This shows that for both colonies, 95% of the offshore records occurred within about 38 km from the colony (fig. 8), while the OWF at the Thornton Bank is located at respectively 25 and 32 km from Zeebrugge and Ostend.

Up to a distance of 10-15 km, flight directions of lesser black-backed gulls breeding in Zeebrugge are mostly oriented perpendicular to the shoreline, either straight to the sea (-60°) or directed towards land (120°) (fig. 9, panel at the top). Gradually, flight orientations become more evenly spread over all directions, indicating a shift from directed commuting flights from and to the colony to less oriented flights in search of food. Accordingly, the proportion of birds flying is strongly affected by the distance to the colony, dropping steeply from about 90% close to the colony to 50% at a distance of 10 km. From there on, the proportion of birds flying decreases more
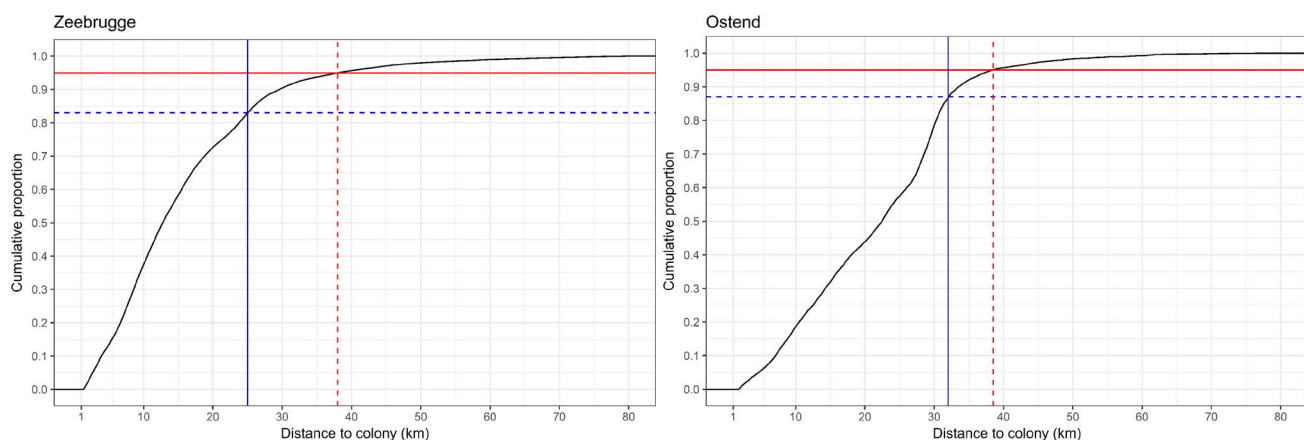

Figure 8. Cumulative proportion of the number of at-sea GPS logs against the distance to the colonies of origin in Zeebrugge and Ostend; the distance to the nearest turbine at the Thornton Bank is indicated in blue, while the 95% boundary is indicated in red.
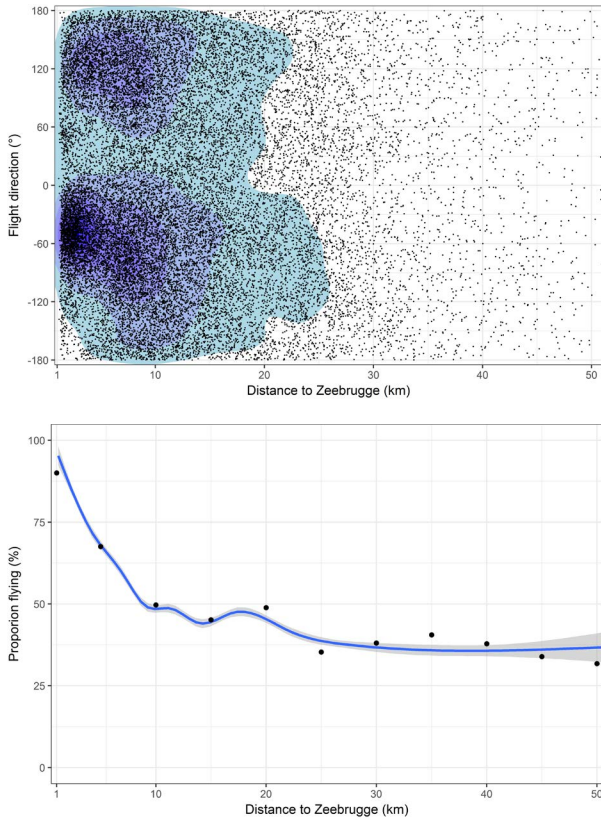
Figure 9. Flight directions in relation to the distance to Zeebrugge with a heat map in the background (panel at the top) and the relation between the proportion of birds flying and the distance to Zeebrugge (panel below).

gradually and stabilises at about 32% beyond a distance of 25 km.

Distribution patterns of birds from Zeebrugge appear to be very similar when comparing different years or seasons (see Annex 1). On the other hand, tagged lesser black-backed gulls showed a much lower at-sea presence during weekends compared to weekdays (fig. 10). By means of example and accounting for birds from Zeebrugge, our dataset holds 17,510 at-sea logs recorded on Wednesdays and Thursdays, compared to only 8423 records on Saturdays and Sundays. Birds thus appear to be present at sea twice more likely during weekdays, which is probably related to reduced fishery activities during the weekend, as was already reported for lesser black-backed gulls breeding on Texel in the Netherlands (Tyson *et al.* 2015).
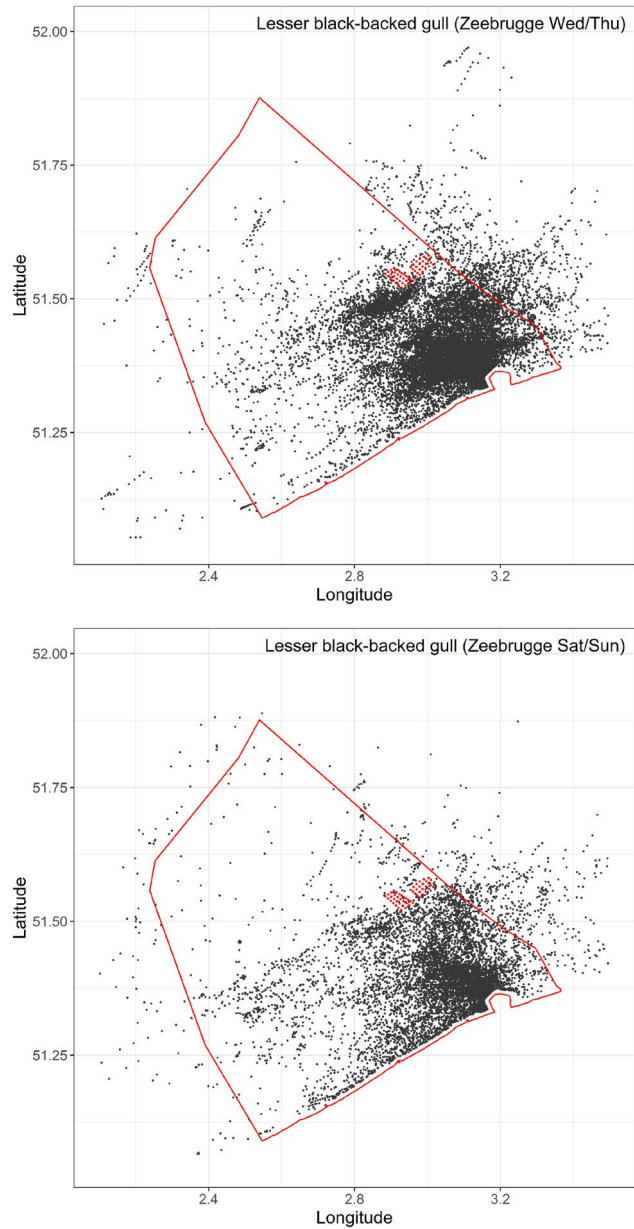

Figure 10. Distribution patterns of birds from Zeebrugge during weekdays (Wednesday/Thursday) and weekend days (Saturday/Sunday).

3.1.2. Diurnal patterns

The diurnal presence of lesser black-backed gulls at sea is characterised by a double-peaked pattern, with lowest numbers around midnight, highest numbers in early morning (3 am) and a secondary peak in the evening (7 pm) following a somewhat lower presence in between (fig. 11, panel at the top). However, when splitting up the data per month, there appear to be considerable differences between months. In March and April
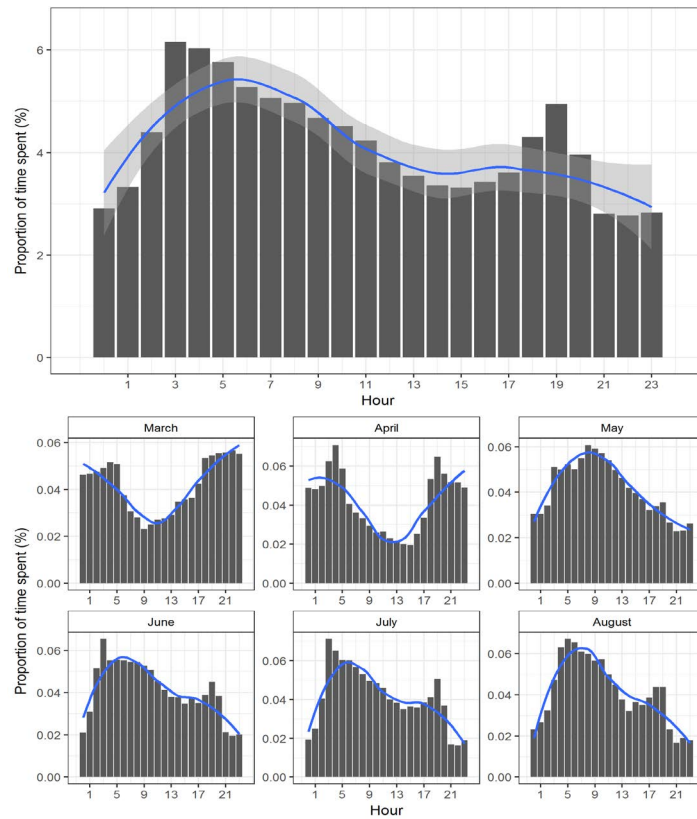
Figure 11. Diurnal rhythm in the time spent at sea by tagged lesser black-backed gulls for all data (panel at the top) and split up per month (panel below).
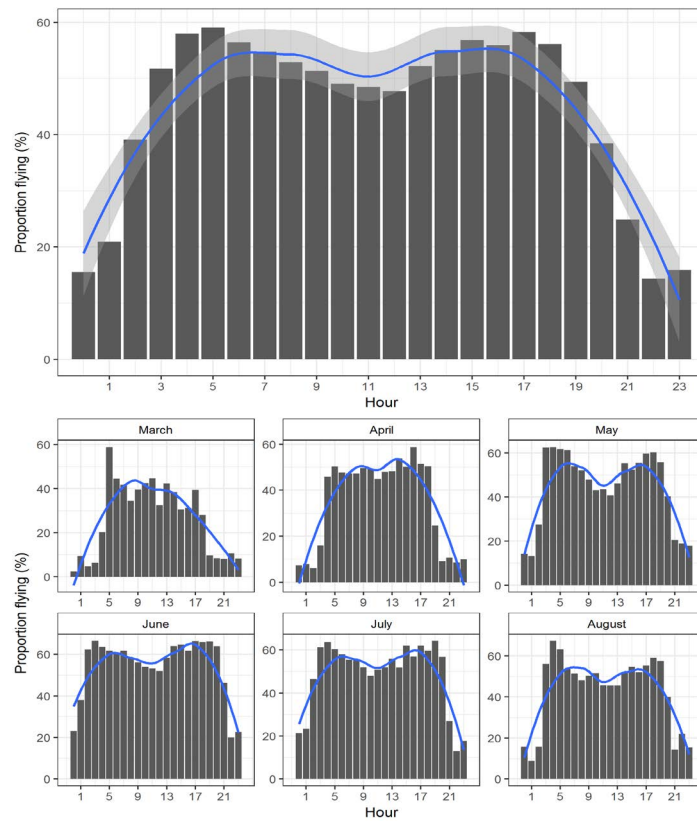

Figure 12. Diurnal rhythm in the proportion of birds flying for all data (panel at the top) and split up per month (panel below).

(pre-breeding period), highest presence rates occur from 6 pm to 5 am, with a considerably lower presence during the day. In contrast, from May to August (incubation, chick rearing and early post-breeding periods), the gulls spend most of their time at sea roughly between 3 am and midday, with a moderate secondary peak around 7 pm, while comparatively much less time is spent at sea around midnight (fig. 11, panel below).

Looking at the diurnal rhythm in the proportion of birds flying, we see a highly symmetrical and double-peaked pattern, with highest flight activity occurring at 5 am and again at 5 pm, and a slight dip in flight activity in between. Around midnight, flight activity drops below 20% (fig. 12, panel at the top) and birds are mainly resting. Again, there is a seasonal aspect to this, in the sense that flight activity is lowest from March to April, highest in the period May to July and decreasing again in August. When, for example, comparing the flight activity in March to that in May, the period of low flight activity during the night is longer, while the proportion of birds flying throughout the day is lower. It seems that during the pre-breeding period, relatively more lesser black-backed gulls prefer to spend the night resting at sea.

The diurnal presence of birds in the wind farm study area differs from the overall at-sea pattern. The morning peaks in presence inside the OWF and its control area come later, respectively at 7 and 10 am, compared to 3 am for all at-sea data compiled. In accordance to the pattern for all data compiled, there is a moderate evening peak in the OWF at 6 pm, yet at the same time there is no increased evening presence in the control area (fig. 13).

In the wind farm control area, there is less flight activity (35%) compared to the wider offshore area (52%) (fig. 14, panel at the top), which is in line with the decreasing flight activity with increasing distance to the coast as illustrated in fig. 9. Flight activity is lower still in the OWF (25%), and thus below
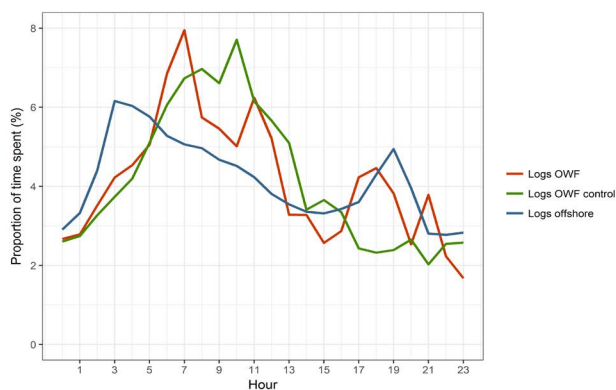
Figure 13. Comparison of the diurnal rhythms in presence of tagged lesser black-backed gulls in the OWF at the Thornton Bank, the nearby control area and at sea in general.
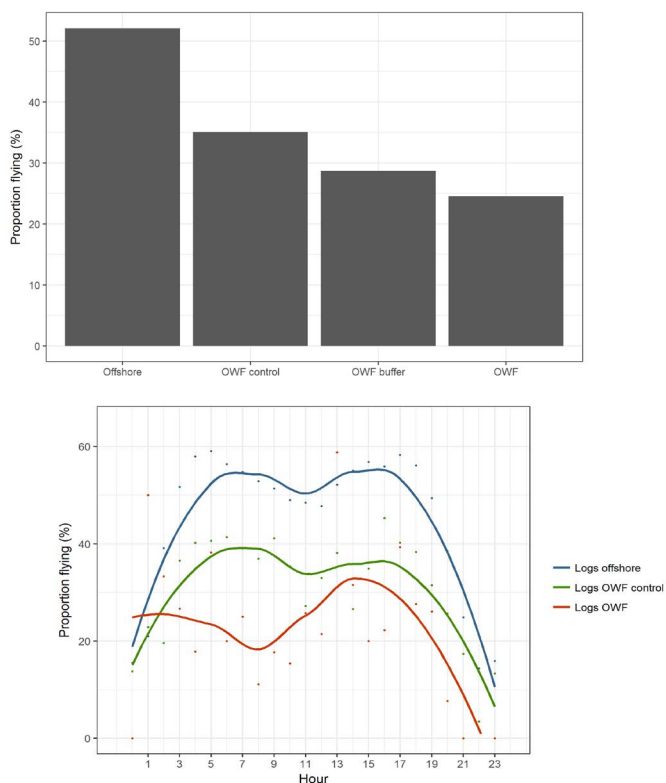
Figure 14. Bar plot of the proportion of birds flying per sub-area (panel at the top) and diurnal rhythm in flight activity in the OWF and its control area compared to the pattern observed in all data (panel below).

what could be expected based on the aforementioned general inshore-offshore pattern. This comparatively low flight activity inside the wind farm might indicate that birds come to the area to rest rather than to forage.

3.1.3. Flight heights

The UvA-BiTS trackers measure altitude, which allowed studying the proportion of birds flying at rotor height under different circumstances. Unfortunately, the altitude measurements are not without error, especially at higher measurement intervals (Bouten *et al.* 2013; Ross-Smith *et al.* 2016), illustrated by the rather large amount of negative altitudes (see figs 15 & 16). Nevertheless, they do give a good indication of overall height. A good example hereof is the difference between the altitude histograms of non-flying birds in the OWF and the control area. Next to the expected and most dominant cohort of birds logged at altitude zero (birds on the water) which is present in both histograms, we see an obvious second cohort of birds at altitudes of about 20 m in the OWF histogram, representing birds roosting on the turbine foundations (fig. 15).
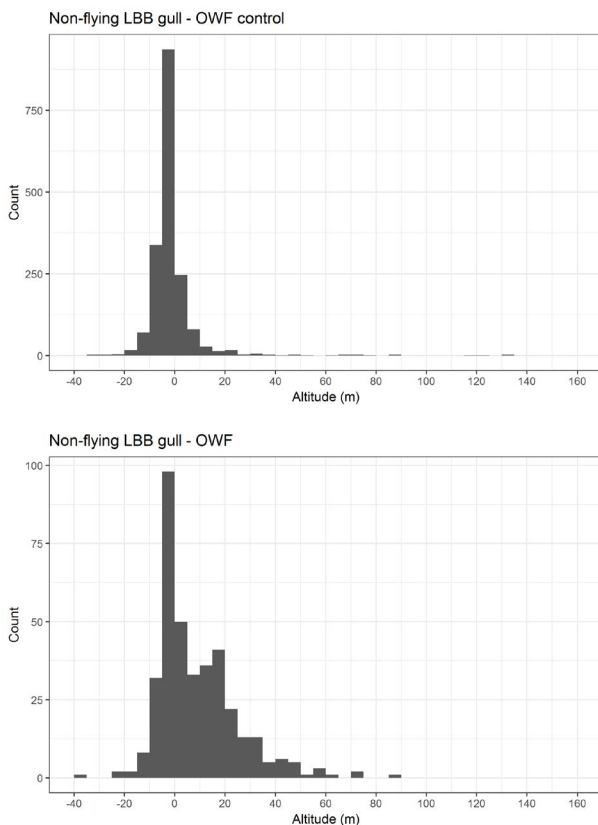
Figure 15. Histograms of recorded altitudes of non-flying lesser black-backed gulls in the control area (panel at the top) and the wind farm at the Thornton Bank (panel below).
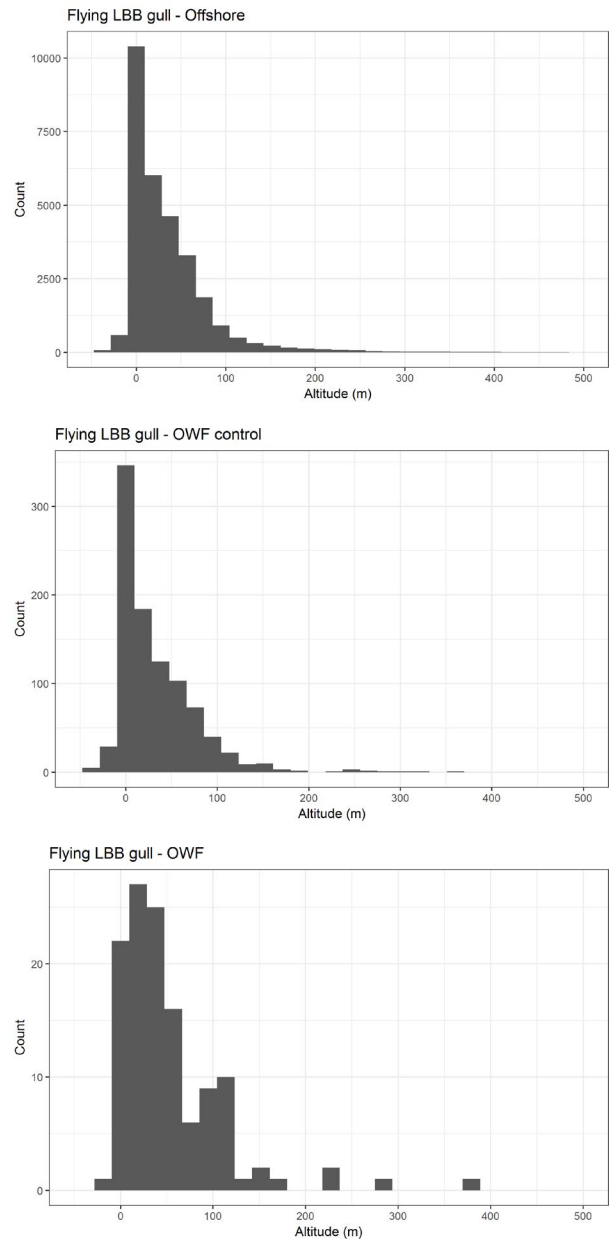
Figure 16. Histograms of recorded altitudes of flying lesser black-backed gulls in the offshore range under consideration (first panel), the OWF control area (second panel) and the Thornton Bank OWF area (third panel).

For flying birds, altitude histograms for all at-sea data and the control area are highly similar, with a peak occurrence of logs with a measured altitude of around zero (fig. 16, first and second panels). Flight altitudes recorded inside the OWF, on the other hand, show a clearly different histogram, with comparatively fewer measurements around zero, a higher weight for altitudes between

10 and 70 m and a secondary cohort of altitudes around 100 m (fig. 16, third panel). Note that this latter histogram is based on a limited amount of logs (n = 126) and may therefore not be fully representative.

When calculating the proportion of birds flying at rotor height (between 31 and 157 m), this leads to a percentage of 37% for both the full dataset and the logs inside the control area. In the wind farm area this percentage increases to 49% (fig. 17). Again, this difference might be coincidence due to the limited number of records involved. But on the other hand, logs recorded inside the OWF coincide with lower wind speeds than in the control area (median 5.3 m/s versus 6.2 m/s respectively) and, as we will see in 3.1.4, low wind speeds typically induce higher flight heights. It would thus be interesting to investigate whether the response of lesser black-backed gulls towards wind turbines varies with wind conditions, indirectly inducing this deviating altitude proportioning.
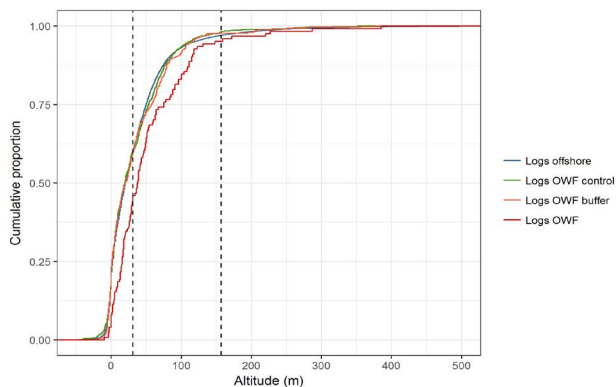

Figure 18. Boxplots of recorded flight heights for five wind velocity categories (panel at the top) and interaction between flight height, wind velocity and relative wind direction (180° representing back wind) (panel below).

Figure 17. Cumulative proportion of recorded altitudes for the full offshore dataset, next to the OWF and its buffer and control areas, with the rotor swept zone between 31 and 157 m indicated by the vertical dashed lines.

3.1.4. Effect of wind and tide

The overall presence of lesser black-backed gulls at sea seems to be largely unaffected by tidal height and wind direction, but is negatively affected by wind velocity. For example, while the median wind velocity in the regarded period was 8.3 m/s, the median wind velocity coinciding with birds GPS logs was only 6.4 m/s. Meanwhile, wind velocity clearly affects flight height, which drops from a 30 m median at low wind speeds (0-5 m/s) to close to sea level during wind speeds of over 15 m/s (fig. 18, panel at the top). Flight height is further determined by the relative wind direction, as birds tend to fly higher during back winds compared to head winds. This effect becomes more pronounced with increasing wind velocities (fig. 18, panel below).

Comparing the wind speeds coinciding with offshore GPS logs of lesser black-backed gulls (blue line in fig. 19, panel at the top) with the hourly wind speed measurements in the months of March to August in the years 2014 to 2017 (grey bars in fig. 19,

panel at the top), low wind speeds (< 5 m/s) occurred more often than expected, while the opposite is true for high wind speeds (> 10 m/s). The gulls thus clearly favour calm conditions when going offshore, and seem to prefer to stay on land during strong winds. These preferences are even more pronounced in the Thornton Bank OWF. Standardising the number of logs in the wind farm with the total number of logs recorded at sea per wind velocity category, we see the chance of visiting the wind farm decreasing linearly with increasing wind velocity (fig. 19, panel below). The latter also applies for the OWF control area, yet to a lesser extent, suggesting that this pattern is only partly related to the presence of the wind farm.

When doing the same exercise for tidal height categories, we see that there is no

Figure 20. The proportion of GPS logs (coloured lines) and hourly tidal height measurements (grey bars) per tidal height category (panel at the top) and the effect of tidal height on the probability of lesser black-backed gulls visiting the Thornton Bank OWF (panel below).

clear pattern of preference in case of all offshore logs neither for logs inside the control area, yet a clearly deviating pattern for logs inside the OWF (fig. 20, panel at the top). Coinciding with the latter, tidal heights below 300 cm occurred more often than expected, opposed to an under-representation of tidal heights above 300 cm. Resulting, the chance of lesser black-backed gulls visiting the wind farm at the Thornton Bank decreases with increasing tidal height (fig. 20, panel below).

3.2. Modelling exercises

3.2.1. Association with turbine foundations

As could already be deduced from figures 2 and 15, lesser black-backed gulls were often logged on or near the turbine foundations in the Thornton Bank OWF, with a
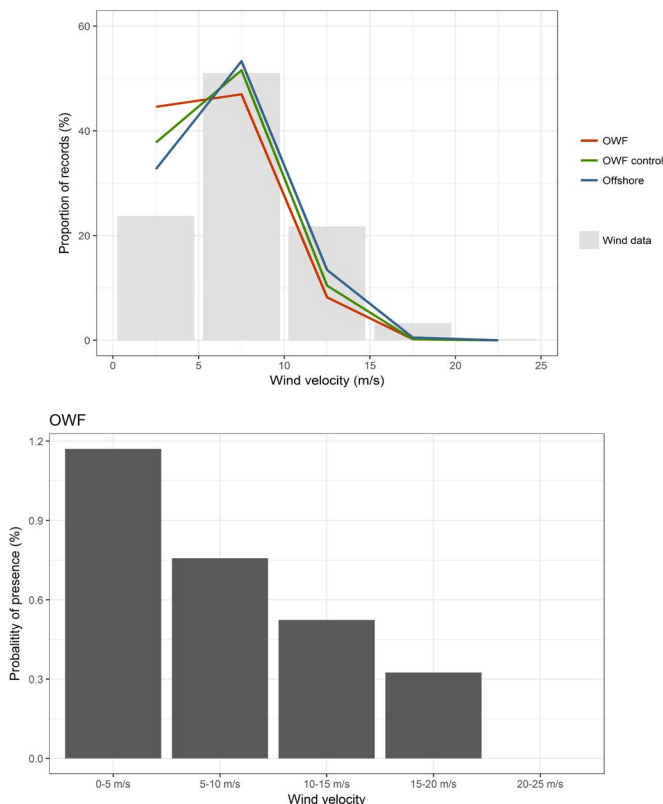

Figure 19. The proportion of GPS logs (coloured lines) and hourly wind speed measurements (grey bars) per wind velocity category (panel at the top) and the effect of wind velocity on the probability of lesser black-backed gulls visiting the Thornton Bank OWF (panel below).
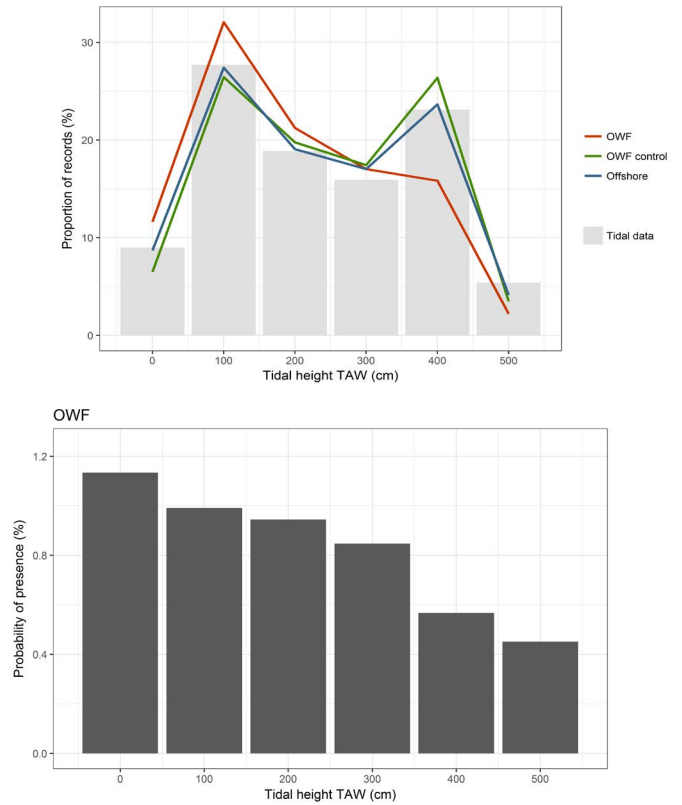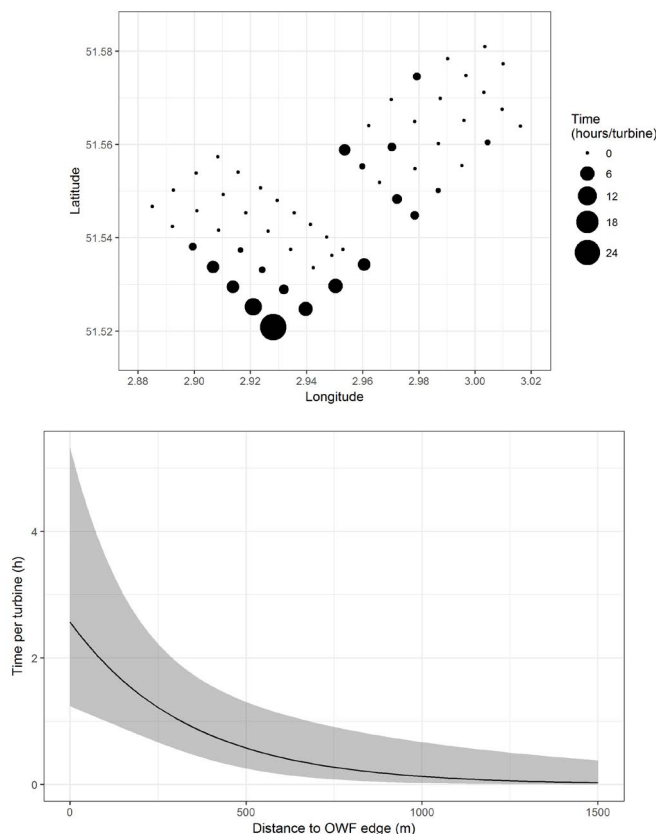
Figure 21. Actual time spent per turbine by lesser black-backed gulls tracked inside the Thornton Bank OWF (panel at the top), and the model prediction of the relation between the time per turbine and the distance to the edge of the wind farm (panel below).

clear preference for the south corner of the wind farm. The birds also seemed to prefer outer to inner turbines. Apart from this, they were more often found resting in the wind farm area compared to the nearby buffer and control areas, and the wider area in general (fig. 14).

In order to explore this apparent preference to turbine foundations, we first calculated the proportion of the time spent in (1) 100 m wide buffer areas around the turbines and (2) the OWF as a whole. Exploring the characteristics of the records comprised within these 100 m turbine buffer zones (n = 635), we see that most (96%) indeed refer to non-flying birds logged at a mean height of 14 m above sea level. Based on this exercise, we estimate that lesser black-backed gulls spend 49% of their time inside

the Thornton Bank wind farm resting on the jacket foundations. When simply calculating the proportion of the number of (20-minute resolution) logs within the 100 m turbine buffer areas *vs.* the number of logs inside the OWF as a whole, we obtain a very similar result of 48%. Considering the huge difference in surface between the OWF footprint area and the turbine buffer areas, it is clear that lesser black-backed gulls show high preference towards the turbine foundations. Figure 21 (panel at the top) illustrates the total time spent per turbine, a variable which was further used to test the hypothesis that birds prefer outer to inner turbines. Based on the Akaike information criterion (AIC), a negative binomial distribution performed better compared to the Poisson or zero-inflated alternatives, and distance modelled linearly was preferred over distance modelled as a smoother. In doing so, distance to the wind farm edge was found to have a significant and negative effect on the time spent per turbine (fig. 21, panel below). The model summary can be consulted in Annex 2.1.

3.2.2. Modelling the effect of distance

An overlay of the 595 grid cells of 500x500 m with the GPS records of lesser black-backed gulls from Zeebrugge and Ostend resulted in a selection of 2601 logs, 72% of which were categorised as non-flying and 28% as flying. The logs concentrate in the south corner of the study area (fig. 22), close to the most favoured turbines (fig. 21). For each grid cell we determined the distance from its centroid to the wind farm edge, and assigned negative distances to centroids that fall within the wind farm boundaries. Distance was used as a smoother to model its effect on the number of logs per grid cell, considering both a Poisson and negative binomial distribution. We performed separate aggregates for non-flying and flying birds, in order to model both categories.

Both for flying and non-flying birds, the AIC was in strong favour of a
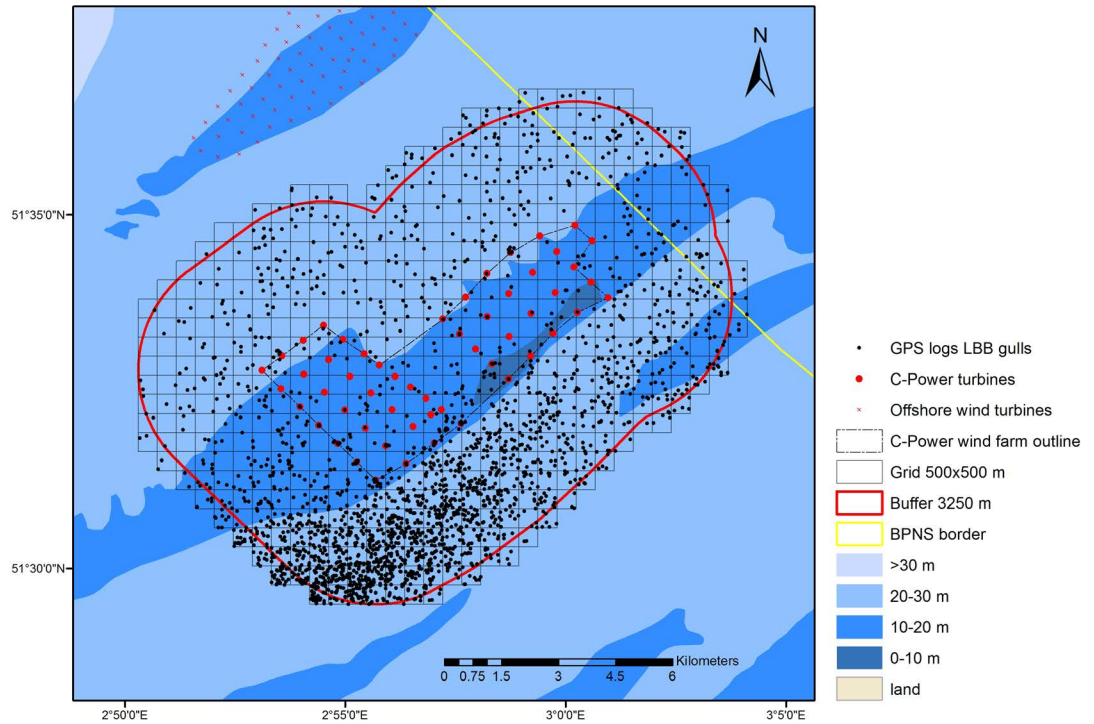
104

Figure 22. Selected GPS logs for the analysis of the effect of distance to the wind farm edge on the presence of lesser black-backed gulls.
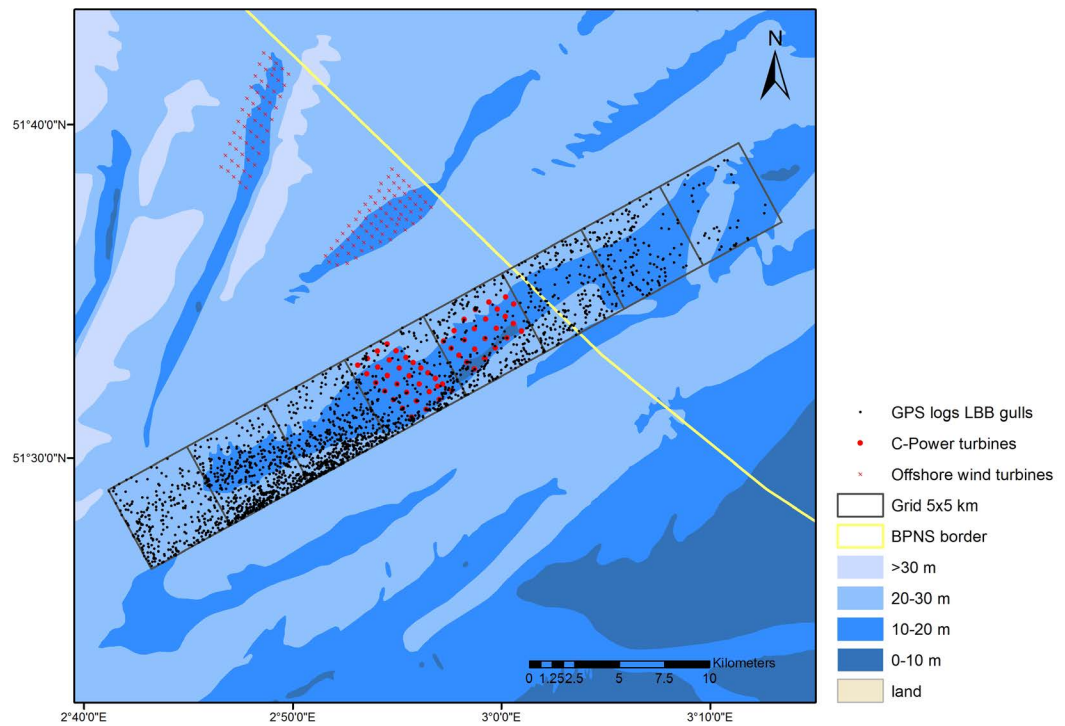

Figure 23. Selected GPS logs for modelling temporal variation in the presence of tagged lesser black-backed gulls in and around the OWF at the Thornton Bank.
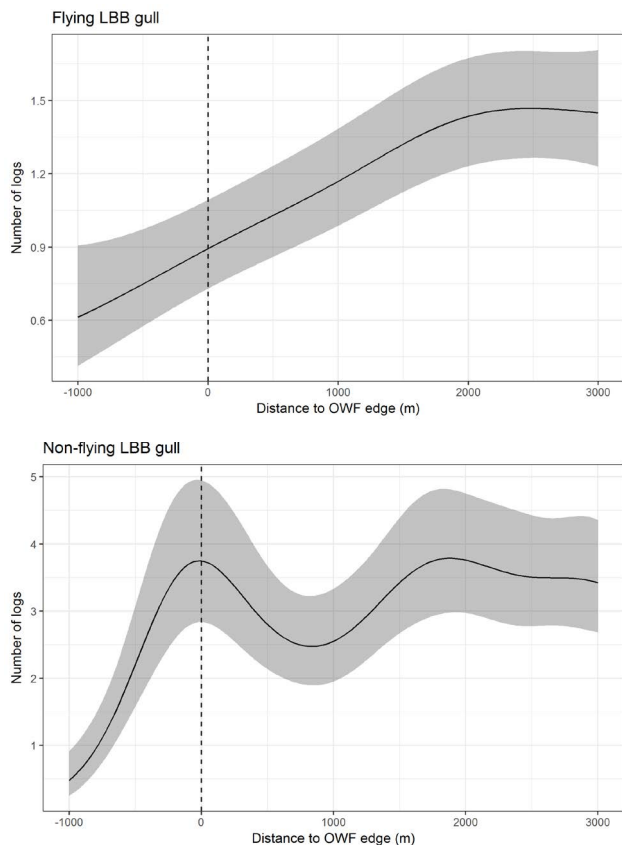
Figure 24. Model predictions of the effect of distance to the OWF edge on the number of logs per 500x500 m grid cells for flying birds (panel at the top) and non-flying birds (panel below).

Figure 25. Number of selected logs in the eight 5x5 km grid cells, with the mean for the control-impact groups indicated by the dashed horizontal lines.

negative binomial distribution, and the distance smoother appeared to be highly significant in both cases (P < 0.001, also see Annex 2.2). Moreover, the model predictions show interesting patterns (fig. 24). First of all, there is a clear positive effect of distance on the number of logs per grid cell, with the response variable reaching a ceiling at about 2000 m for both flying and non-flying birds, indicating avoidance of the wind farm and its immediate surroundings. In case of non-flying birds, however, there is a strong secondary peak in predicted numbers right at the edge of the wind farm, representing birds roosting on the outer turbine foundations.

3.2.3. Modelling temporal variation

We analysed the temporal variation in presence of tagged lesser black-backed gulls in and around the Thornton Bank OWF making

use of the generated dataset as described in Material and methods. Exploring the dataset learned that the eight 5x5 km grid cells include 2435 logs of tagged lesser black-backed gulls, which are quite unevenly distributed (fig. 23, see the previous page, and fig. 25). The four SW grid cells for example hold more than twice as many logs than the four NE grid cells, and based on the bar plot in fig. 25, one could suspect a SW-NE gradient in presence. Apart from this, the mean number of logs appears to be slightly higher in impact cells compared to control cells.

No fewer than 734 logs (30%) occurred in the relatively short period of June to July 2016, representing only 8% of the timeframe considered. Limiting the analysis to this specific timeframe offers some major advantages, considering the increased proportion of non-zero counts and the fact that there is less need to account for temporal random effects. Doing so, we investigated the effect of wind, tide and diurnal patterns on the presence of lesser black-backed gulls in the study area in general, and the wind farm in particular.

We regarded two types of distribution (Poisson and negative binomial) and chose following full models:

- N_records ~ CI * (WEEKEND + WIND + TIDE) + s(HOUR, bs = "cc", by = CI, k = 8)
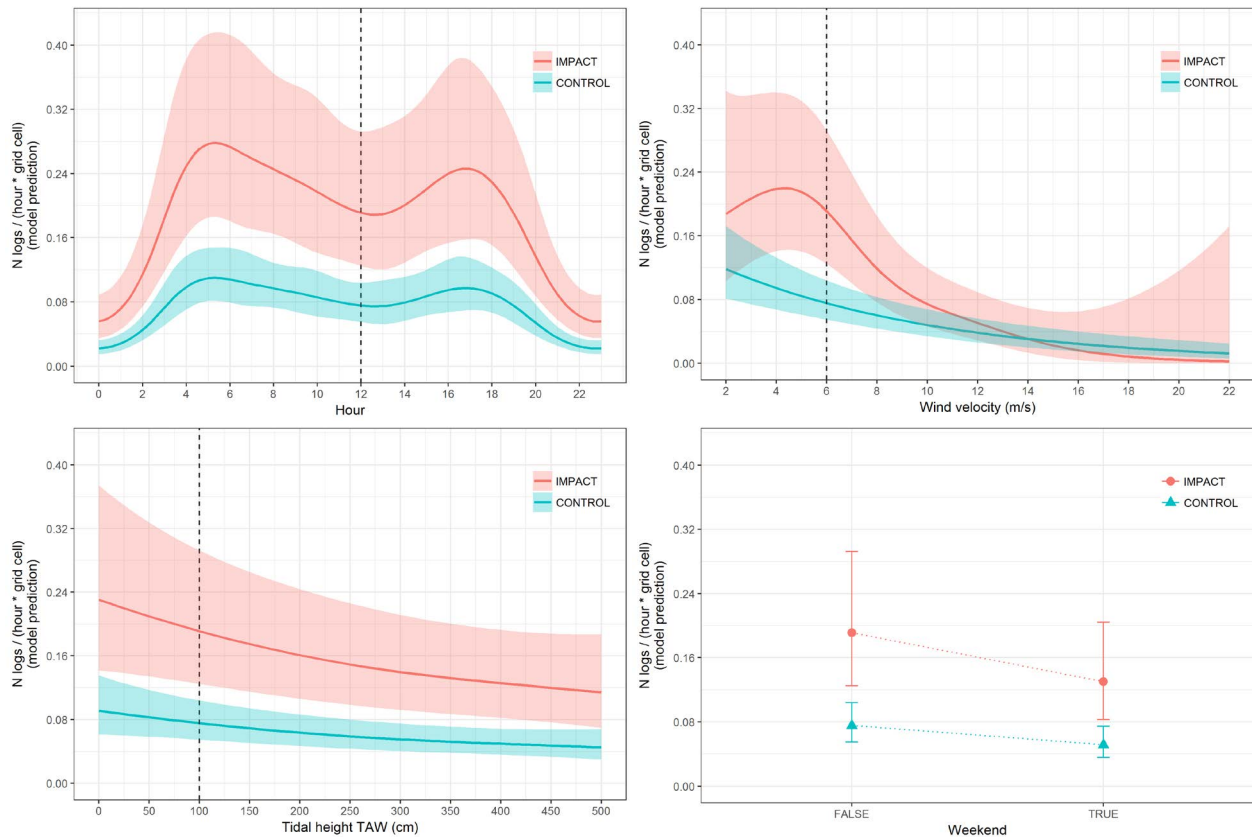
Figure 26. Model predictions of the number of logs per grid cell per hour for all coefficients included in the model (control/impact factor, true/false weekend factor, wind velocity, tidal height and hour of the day), each time keeping all other variables constant at the levels indicated by the vertical dashed lines.

- N_records ~ CI * WEEKEND + s(WIND, by = CI, k = 8) + s(TIDE, by = CI, k = 8) + s(HOUR, bs = "cc", by = CI, k = 8)

  In which:

- N_records = the number of records in a certain grid cell in a certain hour

- CI = a factor variable assigning the grid cells to 'control' (n = 6) or 'impact' (n = 2)

- WEEKEND = TRUE/FALSE factor variable for weekend versus weekdays

- WIND = the mean wind velocity (m/s) per hour as measured at the Westhinder station

- TIDE = the mean tidal height TAW (cm) per hour as measured at the Ostend station

- HOUR = hour of the day

The two full-model options only differ in the way we model TIDE and WIND effects, *i.e.,* either linearly (yet with a log link) or through smoothers. For HOUR, a cyclic smoother was applied in both full-model options. In all cases, the smoothers' basic dimension k was limited to 8 to avoid over-fitting. As the 'treatment' of the grid cells in terms of prevailing wind, tidal conditions and hour of the day is the same for all eight grid cells, we specifically looked for interactions between the factor CI, on the one hand, and WIND, TIDE, WEEKEND and HOUR on the other hand.

Based on AIC, a negative binomial distribution was by far the preferred distribution (ΔAIC's of 1183.0 and 1140.4), with a slight advantage for the option with WIND and TIDE modelled by smoothers (ΔAIC of 3.4). Applying a stepwise and backward model

selection based on the AIC led to the selection of the following model:

- N_records ~ CI + WEEKEND + s(WIND, by = CI, k = 8) + s(TIDE, k = 8) + s(HOUR, bs = "cc", k = 8)

All main terms were retained in the model, as well as the interaction between CI and WIND. Despite the high proportion of zero's in the dataset (96.4%), these were all accounted for by the included variables, as model overdispersion was estimated to be 0.99 by dividing the residual deviance by the residual degrees of freedom. The model summary is given in Annex 2.3.

Model predictions in relation to the included co-variables are shown in fig. 26. The results are in line with the patterns observed during the explorative analyses: a double-peaked diurnal pattern and negative correlations between presence and both tidal height and wind velocity. Following the positive CI and negative WEEKEND coefficients, the number of records is expected to be highest in the impact grid cells during the week under most circumstances, indicating a preference to the wind farm cells despite the local absence of fishery activities. Interestingly, the model predicts a highly different effect of wind velocity on the presence of lesser black-backed gulls between control and impact grid cells. Comparing the model prediction graphs in the right upper panel of fig. 26, we see a strong preference for impact cells during low wind speeds, followed by a steep decrease in the number of impact cell records at wind speeds of > 5 m/s, eventually resulting in a (slight) preference for control cells from wind speeds of > 14 m/s on. This is in line with the pattern observed in fig. 19, confirming that lesser black-backed gulls are increasingly wary of entering the wind farm during strong winds.

4. Discussion

Exploring the at-sea GPS data of tagged lesser black-backed gulls showed that most birds (95%) from Zeebrugge and Ostend were logged within a distance of about 38 km from the colony of origin. The proportion of gulls recorded in flight was found to decrease with distance to the coast, dropping from 90% close to Zeebrugge and stabilising at about 30% beyond a distance of 25 km from the colony. Flight activity was lower still in the OWF at the Thornton Bank (25%), which might be an indication of an increased preference for roosting between or on the turbines. Strikingly, the gulls were twice more likely to explore the offshore areas around the colonies during the week compared to the weekend, which is most probably related to the much reduced fishing activity during the weekend (Tyson *et al.* 2015). This immediately highlights our most important missing variable when aiming to study the observed distributional patterns of lesser black-backed gulls in relation to the Thornton Bank OWF. The inclusion of Vessel Monitoring System (VMS) data would therefore be a huge step forward.

In our dataset, the overall percentage of birds flying at rotor heights (set between 31 and 157 m) was 37%, but this percentage amounted to 49% inside the Thornton Bank OWF. The latter percentage, however, is based on a limited number of data, and should therefore be interpreted with care. Apart from this, the birds' flight height was strongly determined by wind velocity, with a median flight altitude of 30 m at wind speeds below 5 m/s, opposed to sea-level flight heights in wind speeds of over 15 m/s. Interestingly, there appears to be an over-representation of low wind speeds at times when lesser black-backed gulls were logged between the turbines, suggesting that the birds were more inclined to enter the turbine-built area during calm conditions. Nevertheless, this over-representation of low wind speeds (inducing increased flight heights) alone cannot explain the major difference in flight height proportioning inside compared to outside the wind farm, which might therefore indicate a behavioural response towards the

turbines. This has already been suggested by Camphuysen (2011) at Dutch OWFs and is something to keep an eye on when more GPS data of birds flying between offshore turbines become available.

Finally, we performed three modelling exercises. In the first exercise we analysed the time spent roosting on turbine foundations. Our results showed that lesser black-backed gulls strongly prefer the turbines along the edge of the wind farm to roost, especially those situated closest to the colonies. This result is different from the one presented in our previous report (no effect of distance to the wind farm edge was found by Vanermen *et al.* 2017), but note that the latter result was based on a dataset including the 2013 data, when all turbine foundations were already present, but not all of them were carrying turbines. The fact that gulls favour outer rather than inner turbines is interesting as it points towards a conflict of the opposing forces of avoidance and attraction right along the wind farm edge.

This is further illustrated in our second modelling exercise where we aimed to assess the effect of distance to the wind farm edge on the number of logs up to a distance of 3250 m. This showed that flying birds avoided the wind farm up to a distance of 2000 m, while the number of non-flying (roosting) birds peaked at the wind farm's edge, but also largely avoided the inner part of the OWF. Note that in previous reports and based on BACI analyses of at-sea survey results (Vanermen *et al.* 2016 & 2017), we could not detect an effect of the Thornton Bank wind farm on overall lesser black-backed gull densities, while a significant attraction effect was found in the more offshore Bligh Bank wind farm, despite the fact that the Bligh Bank turbines are installed on monopile foundations which offer much less roosting possibilities. The marked difference in response between both sites might be an illustration of the Bligh Bank OWF functioning as a stepping stone, allowing

birds to colonise areas that are otherwise off limits (Leopold *et al.* 2013). Such an effect is likely to be far more prominent outside the birds' normal distribution, as is the case for the Bligh Bank. Whatever the reason, all this at least shows that the response of birds can be subject to spatial variation, not only when comparing wind farms or regions, but also on a smaller 'within-OWF' scale.

In the third and last modelling exercise, we analysed the temporal variation in presence in eight 5x5 km² grid cells along the Thornton Bank, two of which include the wind farm and the other six representing control cells. This showed that the presence of lesser black-backed gulls in the area is driven by diurnal and tidal patterns as well as by wind velocity. The model further showed a decreased presence during the weekend and a preference for the impact cells including the wind farm. As there are no pre-construction GPS data available, we cannot assign this preference to an attraction effect. Yet, this preference for the wind farm is striking, considering the fishery activities in the surrounding area and the fact that the gulls seem to avoid the wind farm interior as illustrated in the previous modelling exercises. Interestingly, the wind farm cells were preferred at wind speeds below 5 m/s but avoided during wind velocities above 14 m/s. Lesser black-backed gulls thus seem to be increasingly wary to enter the wind farm during times of high winds with fast moving rotor blades, which is a clear indication of temporal variation in their response towards OWFs.

Gaining more knowledge on the spatio-temporal variation in the response of seabirds towards offshore wind turbines is considered to be a major challenge for reliable impact assessments. The results of this study can therefore be of high value in refining collision risk modelling for lesser black-backed gull, a species which may potentially suffer from population impact due to increased mortality following large-scale

exploitation of offshore wind in the North Sea region (Brabant *et al.* 2015).

Acknowledgements

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Annex 1

Distribution figures
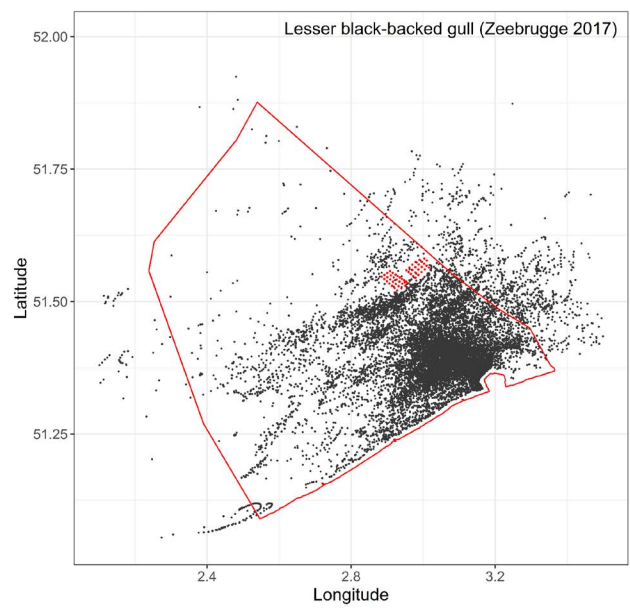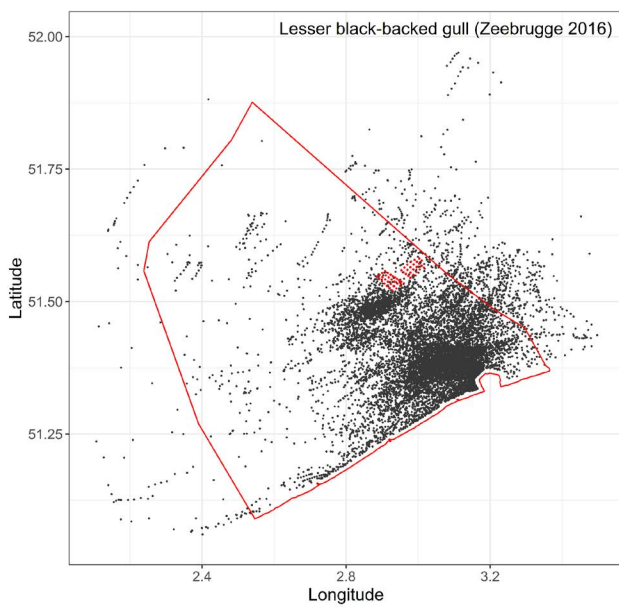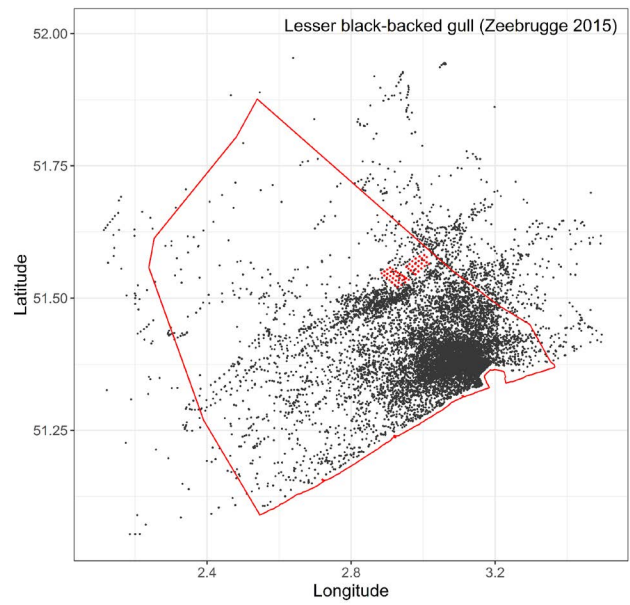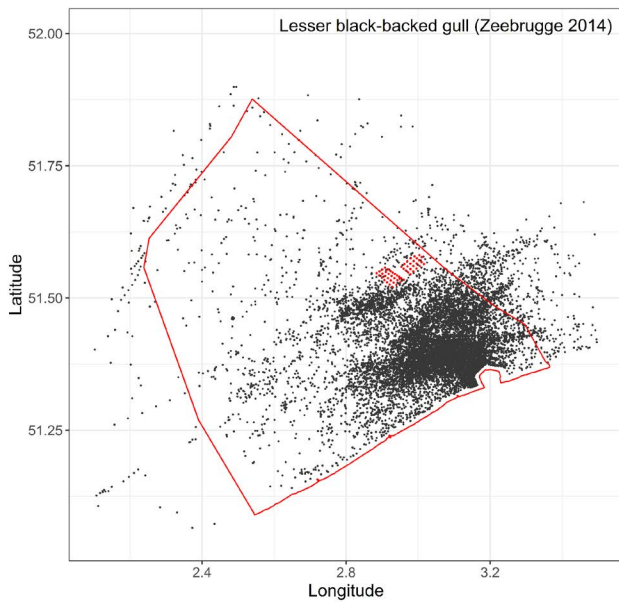
Yearly distribution patterns of birds from Zeebrugge.
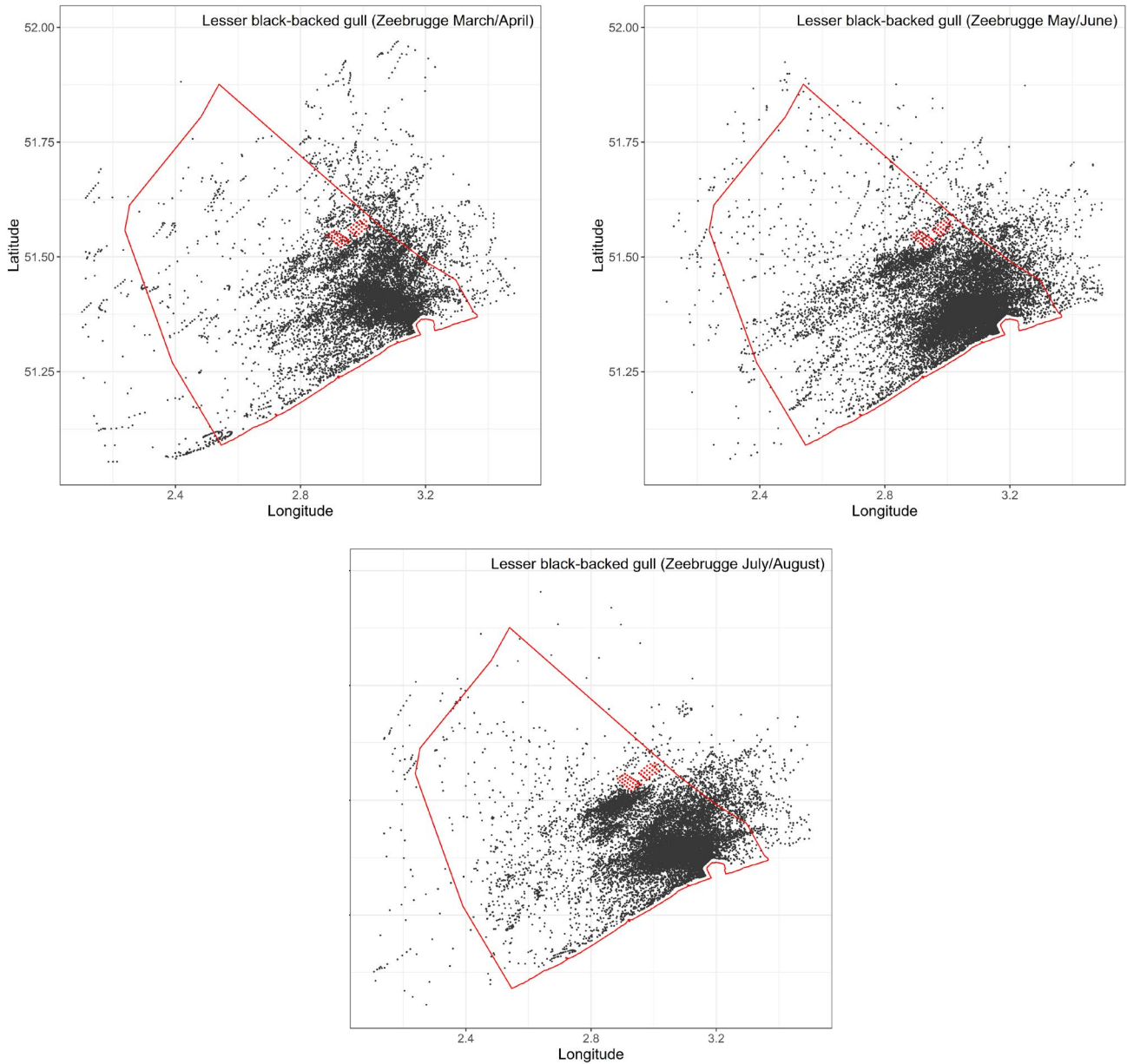
Seasonal distribution patterns of birds from Zeebrugge.

Annex 2

Model summaries

2.1. Association with turbine foundations

```
Deviance Residuals:
    Min        1Q     Median       3Q        Max
-1.11087  -1.11087  -0.66867   0.05869    1.86667

Coefficients:
             Estimate Std. Error z value Pr(>|z|)
(Intercept)  0.9427338  0.3717886   2.536  0.01122 *
distance    -0.0029789  0.0009926  -3.001  0.00269 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

(Dispersion parameter for Negative Binomial(0.2577) family taken to be 1)

    Null deviance: 48.118  on 54  degrees of freedom
Residual deviance: 39.524  on 53  degrees of freedom
AIC: 157.19

Number of Fisher Scoring iterations: 1

            Theta:  0.2577
         Std. Err.:  0.0860

 2 x log-likelihood:  -151.1910
```

2.2. Modelling the effect of distance

```
Family: Negative Binomial(0.694)
Link function: log

Formula:
number ~ s(Distance, k = 8)

Parametric coefficients:
            Estimate Std. Error z value Pr(>|z|)
(Intercept)  1.06831    0.05551   19.24   <2e-16 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:
              edf Ref.df Chi.sq  p-value
s(Distance) 5.731    6.5  38.97 1.59e-06 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) =  0.0147   Deviance explained = 6.59%
-REML = 1333.5  Scale est. = 1          n = 595
```

```
FLYING LBB
Family: Negative Binomial(1.155)
Link function: log

Formula:
number ~ s(Distance, k = 8)

Parametric coefficients:
            Estimate Std. Error z value Pr(>|z|)
(Intercept)  0.18140    0.05389   3.366 0.000762 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:
              edf Ref.df Chi.sq  p-value
s(Distance) 2.331  2.908  19.58 0.000234 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) =  0.0236   Deviance explained = 3.42%
-REML = 909.53  Scale est. = 1          n = 595
```

2.3. Modelling temporal variation

```
Family: Negative Binomial(0.053)
Link function: log

Formula:
records ~ CI + weekend + s(Mean.wind.velocity, by = CI, k = 8) + s(hour, bs = "cc", k = 8) +
s(Tidal.height.TAW, k = 8)

Parametric coefficients:
            Estimate Std. Error z value Pr(>|z|)
(Intercept) -3.12260    0.08248 -37.857  < 2e-16 ***
CIIMPACT     0.57746    0.13401   4.309 1.64e-05 ***
weekendTRUE -0.38418    0.13459  -2.854  0.00431 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Approximate significance of smooth terms:
                                  edf Ref.df Chi.sq  p-value
s(Mean.wind.velocity):CICONTROL 1.006  1.012 29.401 6.63e-08 ***
s(Mean.wind.velocity):CIIMPACT  2.916  3.661 27.528 1.09e-05 ***
s(hour)                         5.141  6.000 57.777 3.82e-12 ***
s(Tidal.height.TAW)             1.383  1.669  9.827  0.00351 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

R-sq.(adj) =  0.0197   Deviance explained =   12%
-REML = 2223.9  Scale est. = 1          n = 11600
```

CHAPTER 8

MODELLING THE IMPACT OF PILE DRIVING ON PORPOISE POPULATIONS IN THE BELGIAN PART OF THE NORTH SEA

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Abstract

In this study we used the interim Population Consequences of Disturbance model (iP-COD; Harwood & King 2014) to quantify how differences in regulatory regimes with regards to offshore wind farm construction impact a simulated harbour porpoise population. We modelled the likely construction schedules for the Rentel, Norther and Seastar wind farms and tested 17 scenarios with and without various mitigating measures.

The value in these simulations lies in the relative differences between the scenarios rather than in absolute outcomes of the model as there are some inherent issues both with the iPCOD model itself (*e.g.*, disturbance per day, not spatially explicit) and the assumptions that we made about the effectiveness of noise mitigation measures such as the big bubble curtain (BBC) and/or the noise mitigation screen (NMS).

Our results indicate that the impact of pile driving on the harbour porpoise population is strongly influenced by the timing of the activities, but that this effect is reduced when effective noise mitigation measures, *i.e.* BBC and/or NMS, is used. The combination of a seasonal pile driving restriction and an acoustic deterring device (ADD) was not enough to lower the impact on the porpoise population to acceptable values. In our simulation, building a wind farm every year affected the harbour porpoise population more than building two wind farms at the same time.

1. Introduction

It is of vital importance for both mankind and the natural environment to limit and mitigate anthropogenic climate change. However, the measures taken to mitigate climate change should not, by themselves, have a negative impact on the natural environment which endangers good environmental status. For offshore wind farms, the production of high levels of impulsive underwater sound, when large steel turbine foundations are hammered into the seabed, is one of these negative effects on the environment.

Potential effects on marine mammals caused by anthropogenic underwater sound can include physical injury, physiological dysfunction, behavioral modification and masking. For individual organisms, these effects and their secondary consequences vary in significance from negligible to fatal

(Marine Mammal Commission 2007). 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 noise pollution. Impulsive pile driving noise originating from the construction of offshore wind farms (OWF) has been shown to affect porpoises up to distances of 20 km from the noise source (Haelters *et al.* 2013; Brandt *et al.* 2016). As we have gained 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 (Rumes *et al.* 2017), 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 draw up the consequences of pile driving at (local) population scale using demography-based modelling, such as the interim Population Consequences of Disturbances model (PCoD, Harwoord *et al.* 2014). This model will be applied to estimate the cumulative effects of the planned piling in the BPNS and is expected to contribute to an informed choice of appropriate sound mitigation measures.

2. Material and methods

2.1. Study area

The Southern bight of the North Sea includes the Belgian continental shelf or BPNS with a surface of approximately 3457 km². The BPNS only covers 0.5% of the entire area of the North Sea. The Belgian continental shelf is characterised by shallow waters with a maximum depth of 45 m and a complex system of sandbanks.

The harbour porpoise (*Phocoena phocoena*) is by far the most common marine mammal in the BPNS, after several years of virtual absence (Haelters *et al.* 2011). The estimation of the harbour porpoise density ranges from 0.05 to 1.03 individuals per km², leading to an abundance of 186 to 3,697 animals (Haelters *et al.* 2011). The animals show a distinct spatial and temporal distribution in Belgian waters with relatively high densities from January to April and lower numbers from May to August, plus they tend to stay in more northerly and offshore waters (Haelters *et al.* 2011; 2016).

In the western part of the BPNS, a 238 km² zone has been designated for renewable energy. Nine projects have been granted permits to build and operate wind farms in this part of the BPNS (fig. 1). For this study we focused on the three wind farms that were to be built between 2017 and 2019, namely Rentel, Nother and Seastar, to construct the scenarios reflecting the impact of pile driving sound on the modelled population of harbour porpoises.

- Rentel NV was granted an environmental permit on 15 February 2013 to build and operate its offshore wind farm. The wind farm will be built at a distance of 31 km from the coastline in the north west of Thornton Bank and the south east of Lodewijk bank. The total capacity of this wind farm of 294 MW is provided by 42 turbines, each with an output of 7 MW.

- The second wind farm, NV Norther, will be placed at 21 km off the coast of Zeebrugge in the south east of Thornton Bank. It was granted an environmental permit on 18 January 2012. The planned capacity for the Norther wind farm of 378 MW is based on 45 wind turbines, each with a capacity of 8.4 MW.

- NV Seastar, the last wind farm in this simulation, was granted an environmental permit on 7 February 2014 to build and operate an offshore wind farm. This wind farm will be placed at a distance of 41 km from the coastline, to the north west of Lodewijk Bank and the south east of Bligh Bank. Seastar will contain 41 wind turbines with a total capacity of 246 MW.
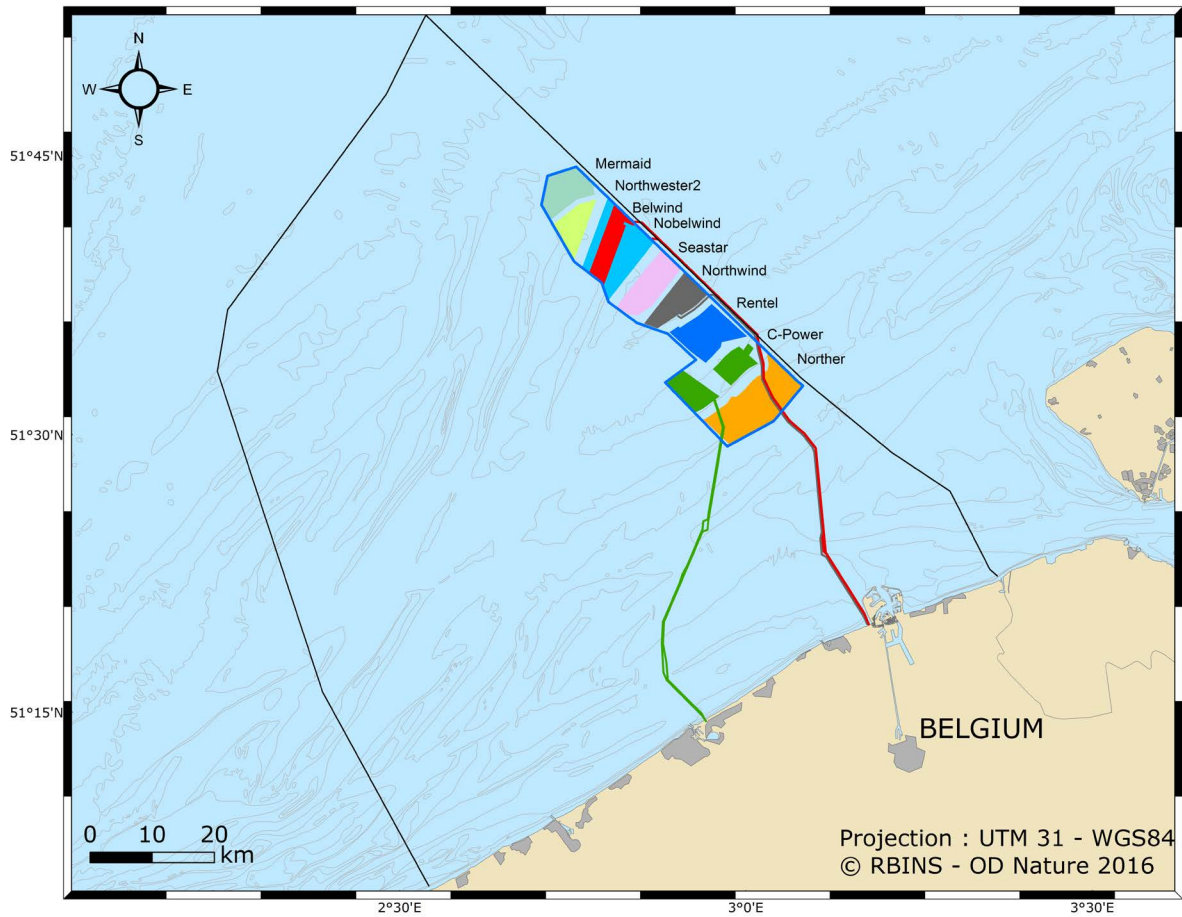
Figure 1. Position of offshore wind farms in the BPNS.

Based on feedback from the developers (Rentel/Seastar), construction was simulated under two different construction speeds, namely with either a slow piling calendar or a fast piling calendar. In case of a slow piling calendar, piling happens every other day (2 h of pile driving per foundation) and the next day there is no piling, over an eight-day period, per block of 14 days allowing for transit of the vessel carrying the foundations. Piling based on a fast piling calendar happens in 4 consecutive days per block of 10 days, starting from the 4th day.

2.2. Legal framework

The countries bordering the Southern part of the North Sea (*i.e.*, Belgium, the Netherlands, the United Kingdom, Germany and Denmark) are dealing differently with the uncertainties associated with the impacts

of high levels of impulsive sound associated with the installation of offshore wind farm foundations on marine mammals. This study will focus on three countries that span the range of legal regimes with regards to the mitigation of piling noise, namely the United Kingdom, Belgium and Germany (table 1).

In this study different scenarios for the construction of offshore wind farms will be modelled to determine the impact and thus usefulness of various mitigation measures such as seasonal piling restrictions, acoustic deterring devices (ADD) and noise mitigation systems on the modelled population of harbour porpoises. These were selected to reflect the impact of the different regulatory regimes shown in table 1.

These differences in regulatory regimes were used to perform the simulations. A baseline scenario was based on the regulatory

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Table 1. Summary of the regulatory regimes and required mitigation techniques for the production of underwater sound during pile driving of offshore foundations (Alstom *et al.* 2015)

	United Kingdom	Belgium	Germany
Noise thresholds	No	185 dB re µPa SPL at 750 m from piling event	160 dB SEL and 190 dB SPL at 750 m from piling event
OWF development forbidden in Natura 2000 areas	No	Yes[*]	Yes
Seasonal restrictions	No	Yes	No
Marine mammal observers (MMO)	Yes	No	No
Soft start	Yes	Yes	Yes
Acoustic deterring devices (ADD)	No	Yes	Yes
Obligatory noise mitigation systems (NMS)	No	Yes	Yes

* Based on the proposal for a new zone for marine renewable energy in the draft of the marine spatial plan 2020-2026; this is likely to change in the near future.

regime of the UK where no noise threshold nor seasonal restriction is enforced (scenario 1). The piling calendar for this scenario starts in March as the winter months of January and February are often characterised by adverse weather conditions.

The pre-Marine Strategy Framework Directive guidelines used by the Belgian government forms the base of the second scenario (scenario 2): it includes seasonal piling restrictions (*e.g.*, the start of the piling event is forbidden in a certain period, so the piling starts in May instead of March) and the environmental license obliges the use of acoustic deterring devices (ADD).

The third management scenario (scenario 3) is based on the current (2017-2018) environmental license conditions enforced by the Belgian government and comprise seasonal piling restrictions (*e.g.*, start in May), the use of ADD and a noise mitigation system, namely the big bubble curtain (BBC). For this scenario we assumed that a BBC reduces Sound Exposure Level (SEL) at 750 m by 10 dB re 1µPa/s. This was less than the a priori estimate of 17 dB re 1µPa/s provided by the developer (DEME 2017) but in line with data from literature (Lucke *et al.* 2011; Bellman *et al.* 2015). To determine the

impact of the seasonal pile driving restrictions, this scenario was also simulated with a start in March.

The regulatory regime of Germany gave inspiration for the final scenario (scenario 4), which included seasonal pile driving restrictions (*i.e.*, start in May), the use of ADD and a strict noise threshold, which in practice has resulted in the use of two combined noise mitigation systems. Here we simulated a combination of BBC and NMS. For this scenario we assumed that such a combination reduces SEL at 750 m by 20 dB re 1µPa/s. This is in line with data from literature (Rumes *et al.* 2016). Thus, the effect of an extra noise mitigation system (the noise mitigation screen) becomes visible when this scenario is compared with scenario 3.

An additional mitigation measure that was tested is the – no longer enforced – Dutch prohibition of piling activities by two nearby wind farms with overlapping construction periods. The influence of two wind farms built in one year was modelled with two new scenarios: a first scenario where two wind farms (here: Norther and Seastar) are built simultaneously (piling days overlap) and a second situation with serially built wind farms (no overlap in piling days).

Table 2. Overview of the 17 different scenarios simulated in iPCOD. When there is no seasonal pile driving restriction, construction is assumed to commence March 1st. Otherwise construction starts May 1st. Construction of the Norther and Seastar projects in the same year is either assumed to overlap (O) or to be serial (Se). Construction was simulated either with a fast (F) or slow (S) piling calendar

	Scenario 1				Scenario 2				Scenario 3					Scenario 4			
	A	B	C	D	A	B	C	D	A	B	C	D	E	A	B	C	D
Seasonal restriction					X	X	X	X	X	X	X	X		X	X	X	X
ADD					X	X	X	X	X	X	X	X	X	X	X	X	X
BBC									X	X	X	X	X	X	X	X	X
IHC														X	X	X	X
Norther & Seastar			O	Se			O	Se			O	Se				O	Se
Construction	S	F	S	S	S	F	S	S	S	F	S	S	S	S	F	S	S

Seventeen different scenarios were selected to investigate the impact of the range of noise mitigation measures on a harbour porpoise population (Table 2).

2.3. interim Population Consequences of Disturbance model (iPCOD model)

To assess the potential effects of anthropogenic noise, associated with offshore renewable energy developments, on harbour porpoise populations, the interim Population Consequences of Disturbance (iPCOD) model was developed (Nabe-Nielsen & Harwood 2016). In this model population dynamics are simulated based on the birth and average survival rates, derived from data from North Sea animals. iPCOD runs fast which makes it possible to compare many different scenarios and to take a wider range of uncertainties into account. Independent estimates of the number of animals, that may be disturbed by the offshore activity, combined with the results from an expert elicitation process (Donovan *et al.* 2016) are used in the iPCOD model. It is called an interim approach, as the values given by experts should be replaced with empirically derived values, when these become available (Nabe-Nielsen & Harwood 2016). iPCOD does not currently include density dependent population regulation. As a result, a population that is reduced in size as result of a disturbance activity will only be predicted to recover when the disturbance activity ceases if the population was increasing in size before the disturbance. Please note that the iPCOD model is not spatially explicit. Every scenario was simulated 500 times.

To parameterise the model the following data is required (Nabe-Nielsen & Harwood 2016):

- basic life-history parameters (*e.g.*, birth rate, calf, juvenile and adult survival, age of maturity);
- timing and spatial distribution of activities likely to cause disturbance (see 2.1. and 2.2.);
- for each of the developments being modelled, an estimation of the number of animals predicted to be disturbed by one day of piling;
- number of animals that experience permanent threshold shift (if any);
- residual days of disturbance;
- population size;
- years of disturbance;
- values for the parameters determining the relationship between the survival or birth rate of an individual and the number of days, the individual experiences disturbance;

- expected inter-annual variation in juvenile and adult survival and birth rate due to environmental variation.

For harbor porpoise, these latter two were obtained by expert elucidation at the time of the development of the iPCOD model.

2.3.1. Basic life history parameters

A script for harbour porpoise containing the basic life-history parameters is included in the iPCOD model (Harwood & King 2014). Here we applied the low adult survival rate as this was determined to be more representative for the North Sea harbour porpoise population (Winship & Hammond 2008).

2.3.2. Estimation of the number of animals predicted to be disturbed by one day of pile driving

As the three simulated wind farms are located in the same area and are expected to use the same techniques to install similar

monopiles, each wind farm was assumed to have both the same harbour porpoise density and (noise) effect radius. Based on 13 aerial surveys (from 2008 to 2016), we assumed the following seasonally fluctuating porpoise densities for the BPNS:

- March – April = 2.7 individuals/km²

- May – July = 0.9 individuals/km²

- August – September = 1.4 individuals/km²

- October – February = 0.9 individuals/km²

For these simulations, the number of animals disturbed by one day of pile driving was calculated by multiplying the density of harbour porpoise by the area affected by the pilling event. Brandt *et al.* (2016) indicate that all affected individuals are classified as disturbed when noise levels are above 160 dB re 1 μPa^2s or when porpoises are avoiding the pile driving event due to the use of acoustic deterrents. Robrecht Moelans (G-TEC) used an acoustic model to calcu-
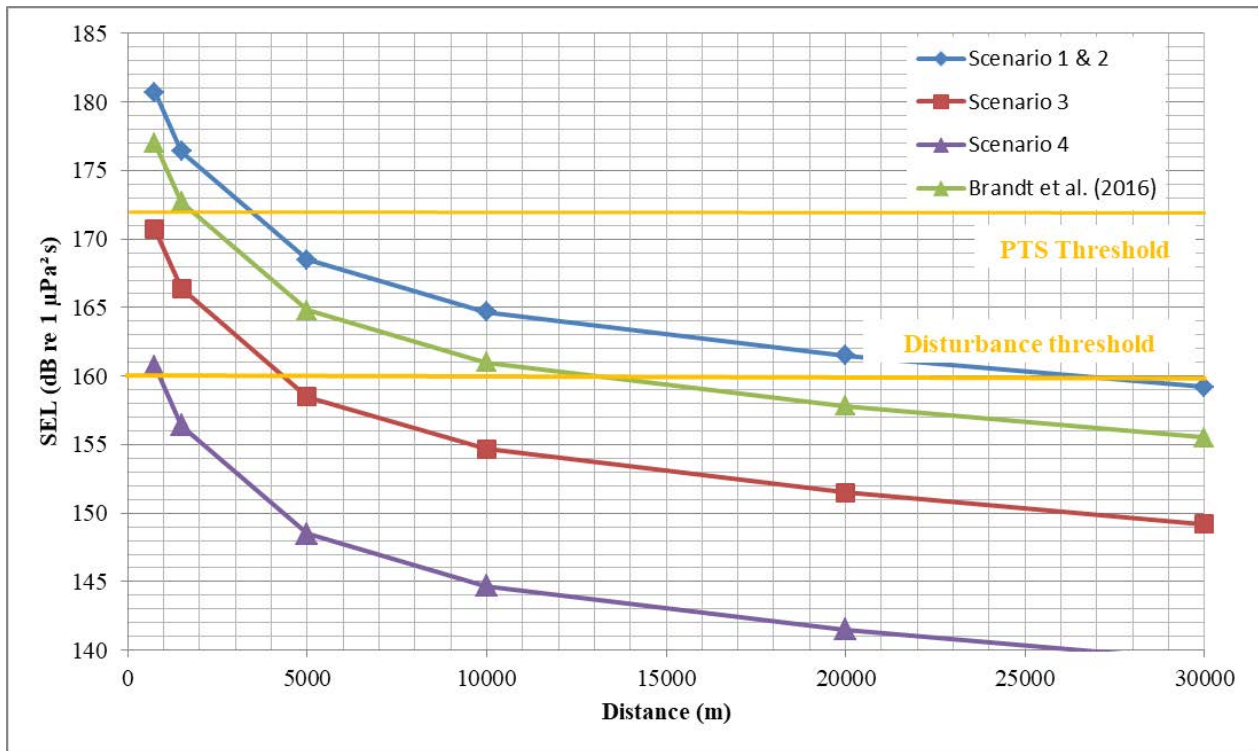

Figure 2. Anticipated Sound Exposure Level (SEL) re 1µPa.s. for the Rentel offshore wind farm under scenario 1 & 2 (blue), scenario 3 (red), scenario 4 (purple), and data from Brandt *et al.* (2016) (green) (data R Moelans, GTEC).

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Table 3. Illustration of the fast (top) and slow (piling) calendars showing the days when pile driving takes place (red) and residual days of disturbance (orange)

Day	1	2	3	4	5	6	7	8	9	10				
Fast piling calendar				P1	P2	P3	P4							
Day	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Slow piling calendar				P1		P2		P3		P4				

late the anticipated Sound Exposure Level (SEL) at various distances. From these simulations, the effect radius could be deduced for the different scenarios: scenarios 1 and 2 – 26 km, scenario 3 – 4.2 km, scenario 4 – 1 km (fig. 2) .

For the simulations with two wind farms built in the same year, the effect radius of both wind farms overlapped as the distance between these wind farms (Norther and Seastar: 15 km) was smaller than the impact radius. Under scenarios 1 and 2, an overlap of 1012 km² was calculated thus reducing the total area of disturbance for days when construction activities coincide.

2.3.3. Number of animals that experience permanent threshold shift (PTS)

The number of animals that experience permanent threshold shift is calculated in the same way as the number of disturbed animals (*i.e.*, density multiplied by the affected area). Under the assumption that PTS in harbour porpoise occurs at a SEL of 172 dB re 1µPa²s (see Brandt *et al.* 2016) and using the above-mentioned pile driving sound model, we computed an effect radius of 3.5 km for scenario 1. For scenario 2, a circular area with a radius of 1 km is subtracted from the affected circular area calculated for scenario 1, due to the assumption that the ADD scares the harbour porpoises away up to 1 km from the piling event (Brandt *et al.* 2012; 2013). Actual observed deterrence distance will be determined by the characteristics of deployed ADDs (source levels and

frequency) and local environmental conditions (Hermannsen *et al.* 2015). The number of animals that suffer from PTS is set to zero for scenarios 3 and 4, as due to noise mitigation measures a SEL > 172 dB re 1µPa²s is only exceeded in the first kilometer from the source and here the harbour porpoise were assumed to have been driven away (disturbed) by the ADD.

2.3.4. Residual days of disturbance

In theory each pile driving event could lead to two residual days of disturbance due to reduction of detection rates up to one day before as well as two days after piling (Brandt *et al.* 2016; Rumes *et al.* 2017). In practice this is dependent on the piling calendar. For the scenarios with a fast piling calendar, there were only three days of residual disturbance per set of four foundations (rounded up to one per pile to fit the structure of the iPCOD model) versus six days for the slow piling calendar (rounded up to two per pile – table 3). Animals were only vulnerable to PTS on the first day of disturbance, as piling only occurs on one day per piling event.

2.3.5. Population size

The North Sea porpoise population consists of 345,000 specimens as defined by SCANS III (with lower and upper 95% confidence limits of abundance of 246,000 and 496,000 – Hammond *et al.* 2017). However, for this study, we set the local porpoise population size on 9326 individuals *i.e.*, – at that time – the maximum number of porpoises

Table 4. Overview of the outcome of the 17 scenarios simulated with the iPCOD model, showing the median decrease (%) in porpoise population size and the added risk of a 1% and 5% decline in porpoise population between an undisturbed population (baseline *i.e.* no pile driving) and a disturbed population (construction of three wind farms) six years after the start of the piling calendar under four different regulatory regimes. When there is no seasonal pile driving restriction, construction is assumed to commence March 1st. Otherwise construction starts May 1st. Construction of the Norther and Seastar projects is the same year is either assumed to overlap (O) or to be serial (S). Construction was simulated either with a fast (F) or slow (S) piling calendar. Probabilities exceeding 50% or 10% are indicated in red or bold respectively

Regulatory regime	Scenario 1				Scenario 2				Scenario 3					Scenario 4			
	A	B	C	D	A	B	C	D	A	B	C	D	E	A	B	C	D
Seasonal restriction					X	X	X	X	X	X	X	X		X	X	X	X
ADD					X	X	X	X	X	X	X	X	X	X	X	X	X
BBC									X	X	X	X	X	X	X	X	X
IHC														X	X	X	X
Norther & Seastar			O	S			O	S			O	S				O	S
Construction speed	S	F	S	S	S	F	S	S	S	F	S	S	S	S	F	S	S
Median decrease in porpoise population (in %)	6.71	5.12	4.56	5.72	5.39	3.61	4.28	4.69	0.18	0.07	0.13	0.20	0.49	0.03	0.00	0.00	0.02
Added risk 1% decline	0.72	0.71	0.67	0.68	0.61	0.56	0.67	0.67	0.05	0.00	0.03	0.05	**0.10**	0.00	0.00	0.00	0.01
Added risk 5% decline	0.76	**0.49**	**0.46**	0.63	0.57	**0.30**	**0.38**	**0.47**	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00

reported from the BPNS and assumed that this entire population could potentially be affected by the pile driving (*i.e.*, could move into the impact radius). We chose to use this smaller population to emphasise the differences between management scenarios.

2.3.6. Years of disturbance

It takes three years to build the three wind farms (between 2017 and 2019) but the number of piling years is set on four. As the iPCOD model requires that the start date of the piling years should be the beginning of the breeding season *i.e.*, the first of June for harbour porpoise, the piling calendar starts in June 2016 and ends in May 2020.

3. Results

An overview of the outcome of the iPCOD model for the different scenarios is given in table 4. The median decrease in porpoise population after six years exceeded 1% for all scenarios where the pile driving sound was not reduced. For these scenarios, the additional risk of a 1% decline in porpoise population (due only to the effects of pile driving) was more than 50%.

Reducing the number of additional days of disturbance under the fast piling calendar, significantly reduced the risk of a porpoise population decline (*e.g.*, by 35 and 47% under scenario 1 and 2 respectively for a 5% decline).

The use of a seasonal piling restriction (start of works in May rather than March) and an acoustic deterrent device reduced the impact on the porpoise population, but this was minor compared to the effect of the reduction in excessive underwater sound of the simulated big bubble curtain (BBC). Here we anticipated that such a noise mitigation system would reduce pile driving sound by 10 dB re 1μPa/s which resulted in a decreased radius of disturbance from 26 to 4.2 km. The

further 10 dB re 1µPa/s reduction in pile driving sound, obtained by adding a second type of noise mitigation (IHC Screen), would theoretically reduce the impact radius to about 1 km, which is why there is hardly any impact on porpoise populations under scenario 3.

The effect of having two wind farms constructed in the same year is two-fold. On the one hand, more porpoises are disturbed during a single construction year, this is especially true for serial construction of two wind farms. On the other hand, the porpoise population is subjected to only two years with pile driving works. In these scenarios, having two of the three wind farms constructed in the same year was less detrimental to the porpoise population than three consecutive construction years. In this model, this is especially true for wind farms that are closely located as an overlap in space and time of disturbance will reduce the number of porpoises affected.

4. Discussion

The Belgian government plans to double the amount of operational offshore wind farms by 2020 (see Chapter 1). At a North Sea scale, construction of offshore wind farms is expected to increase for the next 30 years. How pile driving sound impacts harbour porpoise populations remains one of the major concerns identified by the intergovernmental Cumulative Environmental Assessment Framework (CEAF) working group. At present, the different North Sea countries have all defined different regulatory regimes with regards to offshore wind farm construction and anthropogenic underwater sound mitigation. In this study we used the iPCOD model to test how applying different management options to the construction of the same wind farms will influence the harbour porpoise population.

4.1. Effect of a seasonal pile driving restriction

In the iPCOD model, the main factor that will determine impact on the porpoise population is the number of animals that is (permanently or temporarily) affected by the construction works. This can be minimized by reducing or eliminating the number of foundations installed by pile driving. The first could be done by installing fewer, larger, foundations whereas the second would require the use of a different installation technique such as suction bucket. An alternative way to minimize impact is by avoiding construction works during periods of high porpoise density or when the animals are particularly sensitive to disturbance. In our simulations, shifting the start of construction by two months (from March 4th to May 4th) reduced the risk significantly. The main advantage of this measure is that it is easily enforceable. The main disadvantages are that it requires good knowledge of interannual variability in seasonal porpoise densities.

4.2. Effect of noise mitigation

In the iPCOD model, noise mitigation affects the impact on porpoise population by reducing the area of disturbance. A reduction of the Sound Exposure Level (SEL) by 10 dB re 1µPa resulted in a reduction of the area of disturbance by 97%. As a result, independently of the other factors (fast or slow piling/one or multiple parks constructed per year), no major impact on porpoise population was observed. Effective noise mitigation would thus seem to be the most promising way to reduce porpoise population level impacts. However, initial measurements from Rentel indicate that noise levels are higher than assumed, even with the BBC (chapter 2).

4.3. Effect of simultaneous construction

When construction of multiple wind farms in the same year overlapped in both space and time, then this resulted in a reduction of the number of porpoise disturbance days and thus a lower impact on the population than if the construction did not overlap in space and time.

4.4. Effect of time schedule and adverse weather

Weather conditions impact the speed of construction of a wind farm. Transport of the foundations and pile driving is not possible under adverse weather conditions and, in that case, the wind farm will be piled based on a piling calendar which will more closely resemble the slow piling calendar (*e.g.*, 56 foundations piled in 149 days at Belwind – winter 2009-2010). The fast piling calendar is only possible in good weather conditions (*e.g.*, 43 foundations piled in 64 days at Rentel – summer 2017), which – in the North Sea – are most frequent from late spring to early autumn.

4.5. Some words of caution

The values obtained in our scenarios should not be interpreted as absolute or even indicative of the magnitude of the changes to be expected following the construction of the three studied wind farms. They merely serve to identify the relative effectiveness of possible management measures. As explained in the methodology section, population size was set artificially low to exaggerate consequences and allow us to identify differences between the management scenarios. In addition, Marine Scotland emphasises the interim nature of the iPCOD model, which was developed to deal with the current situation, where there is limited data on how changes in behaviour and hearing sensitivity may affect the ability of individual marine mammals to survive and to reproduce. The values provided by experts should be replaced with empirically derived values as soon as they become available.

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CHAPTER 9

FIRST EVER DETECTIONS OF BATS MADE BY AN ACOUSTIC RECORDER INSTALLED ON THE NACELLE OF OFFSHORE WIND TURBINES IN THE NORH SEA

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Abstract

Several bat species are known to migrate long distances between summer and winter roosts. During this migration, a part of the population even crosses the North Sea. The development of offshore wind farms in the North Sea could therefore be a risk for migrating bats. The activity of bats at sea at turbine rotor height is unknown. We therefore installed eight acoustic bat detectors at four turbines in the Belgian part of the North Sea. Four were installed on the platform of the transition piece (17 m amsl) and four were installed on the nacelle of the turbines in the center of the rotor swept area (94 m amsl). A total of 98 recordings of bats were made by all eight Batcorders during 19 different nights during the entire study period (from the end of August 2017 until the end of November 2017). The detections at nacelle height were around 10% of the detections made at low altitude. The observations made by the detectors at nacelle height give an indication of the activity of bats at that altitude, but do not allow to make sound conclusions about the collision risk for bats, especially not in the lower part of the rotor swept zone.

1. Introduction

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

During migration, bats have been found regularly in the southern North Sea, *e.g.*, on oil rigs (Boshamer & Bekker 2008; Russ 2000; Skiba 2009; Walter 2007; Brabant *et al.*

2016). In 2013, a Nathusius' pipistrelle specimen banded in the UK was found in the Netherlands, proving that bats can cross the North Sea (Leopold *et al.* 2014). Lagerveld *et al.* (2014) report regular occurrences of bats in the Dutch offshore wind farms. The reported bat activity offshore was generally limited to periods with calm weather suitable for long-distance migration.

Most research on the spatio-temporal patterns of bats at sea was based on detections made by acoustic detectors, registering the echolocation calls of bats, installed well below rotor height *in casu* at an altitude between 15 and 26 m above mean sea level (amsl; *e.g.*, Lagerveld *et al.* 2014, 2017; Hüppop & Hill 2016). Hüppop & Hill (2016) state that migrating bats might be missed in such studies as they, presumably, fly at altitudes above 100 m under tailwind conditions. The activity of bats at sea at turbine rotor height hence remains unknown (Lagerveld *et al.* 2017).

Taking account of the increase of wind farms in the Belgian part of the North Sea (BPNS) and the entire North Sea, there is an urgent need to gain insight in the altitudinal distribution of bats at sea and the associated collision risk for bats, a taxon in global decline. Therefore, the Royal Belgian Institute of Natural Sciences studied the activity of bats at nacelle height on four turbines in the Belgian part of the North Sea (BPNS) in late Summer – Autumn 2017.

2. Material and methods

We installed eight ultrasonic recorders (Batcorder 3.0/3.1 EcoObs Ltd., Germany) on four different wind turbines in the C-Power wind farm on the Thornton Bank in the BPNS (fig. 1). Four batcorders were installed on the platform of the transition piece of the turbines, at approximately 17 m amsl, and four were installed on the helicopter winching platform at the back of the nacelle, at 94 m amsl (fig. 2). We made full spec-

trum recordings in .RAW format (sampling rate: 500 kHz; record quality: 20; threshold amplitude [sensitivity]: -27/-36 dB; post trigger: 400 ms; threshold frequency [sensitivity]: 16 kHz). The Batcorders were installed on 20 August 2017 and were operational until 30 November.

3. Results

Bats were registered throughout the entire study period, from the end of August until the end of November. A total of 98 recordings of bats were made by all eight Batcorders. All echolocation calls were identified as calls from the species Nathusius' pipistrelle *Pipistrellus nathusius*. Bat activity was recorded during 19 different nights (table 1 and fig. 3). All Batcorders recorded at least one bat, except for the recorder installed on the nacelle of turbine J1. Only nine recordings were made at nacelle height. The number of recordings made at nacelle height was significantly lower than the number of detections made at low altitude.

Table 1. Number of bat recordings per Batcorder from 22 August until 30 November 2017. Low, detections at 17 m amsl; high, detections at nacelle height (94 m amsl); records, number of bat recordings; DP10, detection positive 10 minutes

Turbine	Height	Records	DP10
G01	Low	23	11
	High	2	1
H02	Low	26	16
	High	6	4
I1	Low	17	10
	High	1	1
J1	Low	23	14
	High	0	0
	Total	**98**	**57**

To level off high numbers of recordings caused by one individual residing near the recorder, the recordings were converted to detection positive ten minutes (DP10) meaning that a ten-minute period is considered as positive if it contains at least one bat call (*e.g.*, a specimen producing 100 calls in 10 minutes and a specimen only calling
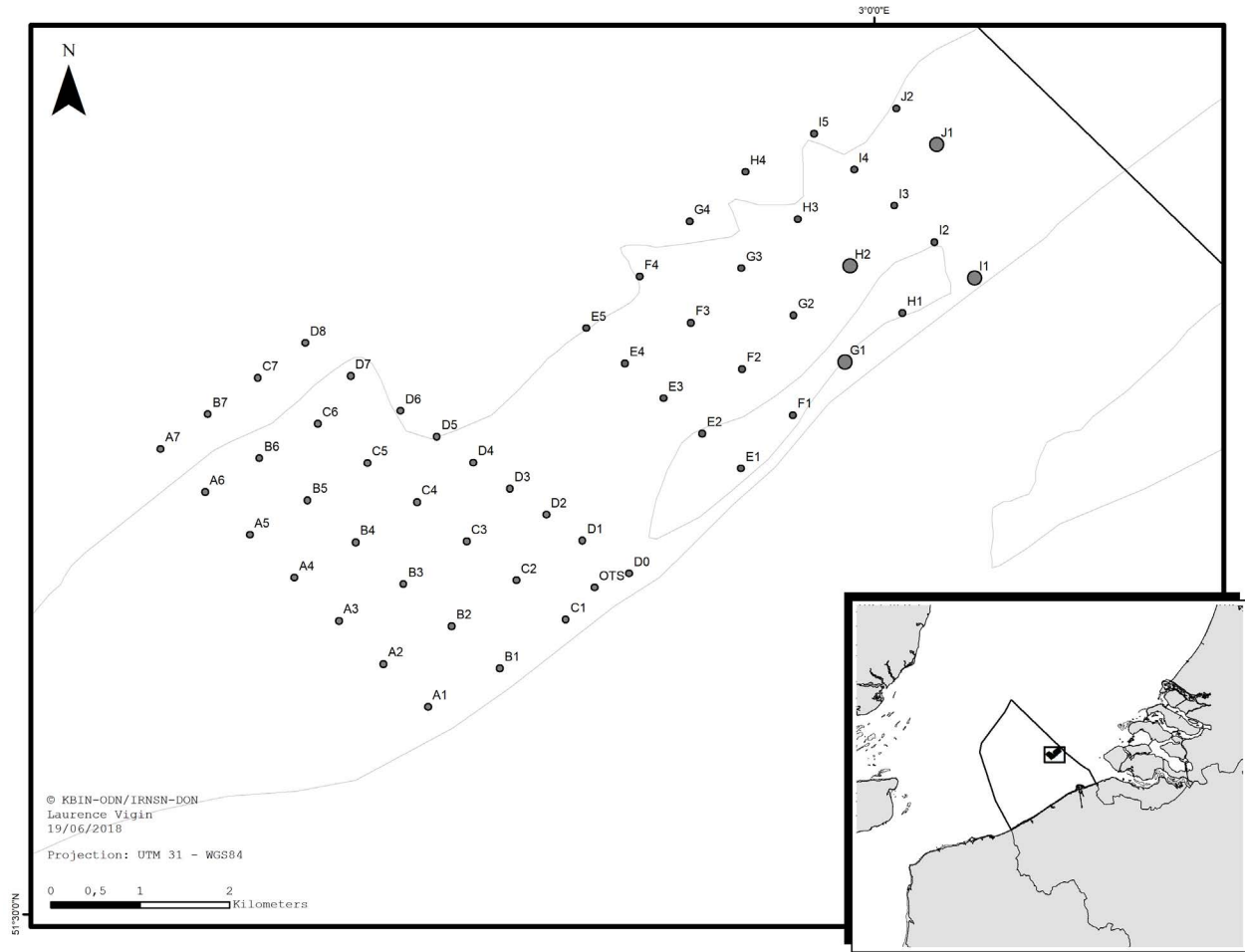
Figure 1. Lay-out of the C-Power wind farm on the Thornton Bank in the Belgian part of the North Sea. Each dot represents a wind turbine. Turbines G1, H2, I1, J1 (indicated by the large dots), in the North-East of the wind farm, were equipped with two Batcorders each (one on the transition piece – 17 m amsl and one on the nacelle – 94 m amsl).

Figure 2. Batcorder installed on the helicopter winching platform at the back of the nacelle of turbine I1.
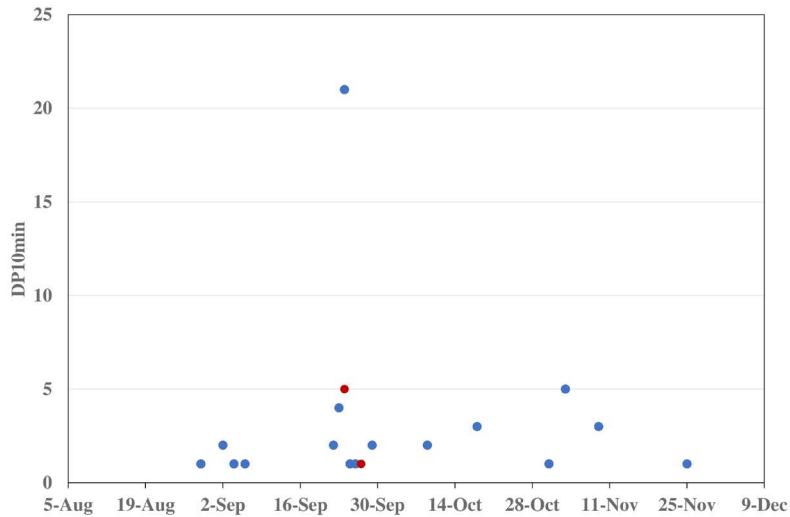
Figure 3. Detection positive 10 minutes (DP10) of the Batcorders at low altitude (blue) and at nacelle height (red), during the entire study period.

once are valued in the same way and render one DP10). The detections at nacelle height were around 10% of the detections made at low altitude (table 1). Figure 3 shows the occurrence of bats (DP10), as registered by the recorders at low altitude and the recorders at high altitude, throughout the monitoring season.

4. Discussion

All recordings made during this study concerned Nathusius' pipistrelle. This is in line with similar studies by Lagerveld *et al.* (2014; 2017, textbox 1). Our results confirm that the majority of migratory activity of Nathusius' pipistrelle takes place from mid-August until the end of September (Lagerveld *et al.* 2014). Further analyses of these data will focus on the impact of environmental conditions (*e.g.*, wind speed, wind direction) and turbine activity on the activity of bats.

From this preliminary study we can conclude that bats are active at high altitude at sea, but that this activity is significantly lower than the activity at low altitude. However, it would be premature to conclude that the risk of a collision of bats with turbine blades is therefore negligible. The detection range of small bats like Nathusius' pipistrelle with a Batcorder is rather limited (15-25 m; Lagerveld *et al.* 2017). This means that a Batcorder installed on a nacelle, at best, covers a range from 70 to 130 m amsl. The rotor of the C-Power turbines reaches from 31 m to 157 m amsl. The observations made by the detectors at nacelle height give an indication of the activity of bats at that altitude, but do not allow to make sound conclusions about the collision risk for bats, especially not in the lower part of the rotor swept zone. Therefore, there is a need for studies assessing bat activity at the lowest point of the rotor, *e.g.*, by installing acoustic detectors at different altitudes on wind turbine masts.

Textbox 1

Spatial and temporal occurrence of bats in the Southern North Sea area

This text is the summary of the report with reference: Sander Lagerveld, Daan Gerla, Jan Tjalling van der Wal, Pepijn de Vries, Robin Brabant, Eric Stienen, Klaas Deneudt, Jasper Manshanden & Michaela Scholl, 2017. Spatial and temporal occurrence of bats in the Southern North Sea area. Wageningen Marine Research (University & Research centre), Wageningen Marine Research report C090/17, 52 p.

The Royal Belgian Institute of Natural Sciences (RBINS), the Research Institute for Nature and Forest (INBO) and the Flanders Marine Institute (VLIZ) were involved in facilitating this study at the Belgian locations at sea and at the coast, and provided general ecological expertise.

The full report can be downloaded from https://doi.org/10.18174/426898

Since a few years, it is known that bats migrate over sea on a regular basis. As numerous land-based studies have shown that wind turbines can cause high fatality rates amongst bats, Rijkswaterstaat started a bat monitoring programme for 2015 and 2016 in order to reduce uncertainties about possible impacts. At the same time, Eneco commissioned a bat monitoring programme for 2015 and 2016 as part of the Monitoring and Evaluation Programme (MEP) for the offshore wind farm Luchterduinen. In 2016, Gemini conducted a bat monitoring campaign in wind farm Buitengaats and Wageningen Marine Research executed a bat monitoring programme at Wintershall platform P6-A and offshore research station FINO3 in the same year. The joint monitoring effort included 12 different offshore locations and 5 locations at the coast.

The specific aims of these monitoring programmes are an assessment of:

1. the species composition at sea and at the coast;

2. the spatiotemporal pattern of occurrence, including the flight height;

3. the relation between environmental conditions and the occurrence of bats;

4. the function of the Dutch Territorial Sea for bats;

The monitoring results at the coast showed that Nathusius' pipistrelle is very common during both spring and autumn migration, but is also regular throughout the summer. It is also the most frequently recorded species at sea, albeit much less frequently recorded in comparison to the coast. At sea, it was recorded from late August until late October (and one observation in November), and – to a lesser extent – from early April until the end of June. There were no records in July until mid-August. The observed pattern of occurrence

matches previous offshore monitoring studies in the German and Dutch North Sea. Due to a limited amount of data in spring, we analysed the presence/absence of Nathusius' pipistrelle per night from mid-August until late October. In this period bat activity was recorded during 11% of the nights at sea and during 66% of the nights at the coast. The higher number of nights at the coast may reflect the relative proportion of bats migrating at the coast and over sea, but the numbers at the coast are likely to be higher due to funneling, whereas migration over sea is likely to follow a broad front due to the absence of guiding landscape features. However, locally densities at sea may also be inflated as bats are likely to be attracted to offshore structures. Consequently, based on bat detector data alone, we cannot estimate the proportion of bats migrating along the coast and over sea.

Due to the differences in occurrences at sea and at the coast, we developed one statistical model for the offshore stations and one for the coastal stations. We modelled the presence/absence per night as a function of various weather parameters, the moon illumination, the spatial coordinates and the night in year in the period mid-August until late October. The most important predictor for the occurrence of Nathusius' pipistrelle in autumn at sea and at the coast are low to moderate wind speeds, followed by night in year (the date). At the coast their presence increases rapidly from mid-August and continues to be high subsequently. At sea the occurrence is strongly peaked. The first wave of migrating animals occurs late August/early September and the second late September. Next,

high temperatures increase significantly the presence of bats, both at the coast and at sea. Wind direction is also important; at sea wind directions between NE and SE (with a peak at 94 degrees) result in highest presence, whereas this is the case with wind directions between E and SW (with a peak at 170 degrees) at coastal locations. The observed optimal wind direction at sea (94 degrees) implies that bats crossing over sea choose tailwind conditions, whereas the presence at the coast seems to be shaped by funnelling. Therefore, it seems unlikely that wind drift or storms cause its presence off our western coastline. However, it has been suggested that wind drift is the main cause for the occurrence of bats north of the Wadden Islands. We also found a moon illumination effect in both models. Increasing moon illumination raised the probability of presence at sea and at the coast. Rain reduced probability of the presence of bats at the coast. In contrast, we did not find an effect for rain at sea; thus, bats were recorded with and without rain at sea. High cloud cover was negatively correlated with the presence of bats at sea, but was positively correlated with the presence of bats at the coast.

The sea model predicts a higher probability of presence in the northwestern corner of the study area. However, we think that this is an artefact caused by the relatively high number of nights with bat activity at the P6A platform, in comparison to the presence at the other offshore monitoring locations. This may be just be a coincidence, but it is also possible that a spatial pattern of occurrence at sea is actually present. For example, if bats follow their general

migration direction (WSW) after leaving the Afsluitdijk, they will pass closely to P6-A. The recorded bat activity at nearshore monitoring locations (between 22 and 25 km from the coast) peaks approximately 4 h after dusk. It seems likely that these animals departed the same night from the coast. However, bat activity at the locations further offshore (between 58 and 69 km from the coast) starts often close to dusk. This means that these animals must have spent the day at the monitoring location at sea, or in its vicinity. This pattern of occurrence means that the observed bat activity at a particular night may depend on their departure decision in the previous night, or even earlier. Other species recorded during this study included Common pipistrelle which was occasionally recorded offshore, but was common at the coast throughout the monitoring season. Nyctaloids were recorded uncommonly offshore from June until October and from May until late October at the coast. Nyctaloids identified to species level included Common noctule, Particoloured Bat, Leister's Bat, Northern Bat and Serotine Bat. Pond bats were not recorded offshore but were regular at the Afsluitdijk and rare elsewhere along the coast. Finally, there were some occasional records of Daubenton's bats and Soprano pipistrelles at the coast.

The results of this study show that the occurrence of bats at sea is highly seasonal which indicates that individuals recorded at sea are on migration. The peak period runs from late August until the end of September. After that it levels off throughout October. Spring migration is much less pronounced but the duration seems to be quite extensive; from late March until the end of June. Records of bats in July and early August are rare. At the coast bats are much more common in general and their presence is both shaped by migratory movements and the presence of foraging individuals from local populations. Therefore, the relevant period to consider the presence of bats at sea off the western coast of the Netherlands and Belgium seems to be from 15 March until 30 June and from 15 August until 31 October, whereas bats should be considered the entire active season at the coast. Based on the monitoring results of the 2012-2014 studies, a precautionary mitigation measure was issued using 5 m/s as cut-in wind speed for the wind farms in the Borssele area in the period 15 August until 1st October. The current study, however, shows that other environmental parameters, in addition to the wind speed, are important as well. The model developed in this study is likely to predict the presence of bats at sea more accurately, despite the fact the model can be improved. In order to improve the sea model, it is recommended to continue monitoring offshore to increase the number of observations in the dataset. The model can, furthermore, be improved by monitoring in a denser grid to reveal spatial patterns and include information on the availability of insects (bat migration fuel). In addition, we urgently need monitoring data from higher altitudes as bat migration may occur at altitudes beyond the detection range of the current monitoring network at sea.

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