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on the Marine Environment

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

MARKING A DECADE OF MONITORING,
RESEARCH AND INNOVATION

Edited by
Steven Degraer
Robin Brabant
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CONTEXT

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 to achieve that goal.

Within the Belgian part of the North Sea, a zone of 238 km² is reserved for the production of electricity from water, currents or wind. Six wind farms are already operational, two more are under construction. A second area for renewable energy of 284 km² is foreseen by the new Belgian marine spatial plan (2020-2026).

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 tackled through basic 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.

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 environmental monitoring programme (WinMon.BE), based on data collected up to and including 2018.

DEGRAER Steven, BRABANT Robin, RUMES Bob and VIGIN Laurence

PREFACE & EXECUTIVE SUMMARY

MARKING A DECADE OF MONITORING, RESEARCH AND INNOVATION

DEGRAER Steven, BRABANT Robin, BRAECKMAN Ulrike, COLSON Liesbet, COURTENS Wouter, DE BACKER Annelies, DE WITTE Bavo, HOSTENS Kris, JONGE POERINK Bob, KERCKHOF Francis, LAURENT Yves, LEFAIBLE Nene, MOENS Tom, MUTETI Jane, NORRO Alain, POLET Hans, RUMES Bob, STIENEN Eric W.M., SYS Klaas, VANAVERBEKE Jan, VANERMEN Nicolas, VAN DE WALLE Marc, VANELSLANDER Bart, VERSTRAETE Hilbran & VIGIN Laurence

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In 2008, the first six offshore wind turbines were constructed in the Belgian part of the North Sea (BPNS). The year 2008 hence also meant the start of the operational phase of the Belgian offshore wind farm environmental monitoring programme, WinMon.BE. At that time, environmental impact assessments stressed the significant uncertainty and knowledge gaps on potential impacts of offshore wind development in our waters. The WinMon.BE programme therefore kicked off by fine tuning the operational logistics and strategies of the monitoring programme, and observing some early effects onto a variety of ecosystem components, *i.e.* from benthic invertebrates over fish to birds and marine mammals. How to best make use of those data to steer the further development of offshore renewables in an environment-friendly way was still in its infancy.

About a decade later, much has changed. By the end of 2018, 318 offshore wind turbines with a total installed capacity of 1556 MW, were operational in the BPNS. In Belgian waters, we have witnessed *e.g.* (1) an evolution in foundation types, from gravity-based foundations and jacket-foundations to XL monopiles, (2) an expansion of the wind farms, from transitional waters to English Channel waters, (3) a marked increase in the size and capacity of the wind turbines (from 3 MW turbines with a 72 m rotor diameter to 8.4 MW turbines with a 164 m rotor diameter), (4) a new, shared way to transport the electricity generated by the wind farms to the shore with the development of a modular offshore grid and (5) the first attempts to widen offshore renewables to also make use of wave energy (Chapter 1).

Within and beyond Belgian waters, we observed this proliferation of offshore wind farms, from a continued construction of offshore wind farms and a delineation of a new offshore renewables zone in Belgian waters over an ever-increasing occupation of space by offshore wind farms in the Southern North Sea to the construction of the first offshore wind farm in the USA. In line with this offshore renewable energy revolution, we have also witnessed a substantial boost and maturation in dedicated environmental monitoring and research programmes (*e.g.*,

WOZEP, ORJIP). The multitude and variety of environmental monitoring programmes substantially and exponentially increased our understanding of how offshore wind farms impact the marine ecosystem and hence, our capability to steer environment-friendly offshore renewable developments.

In the meantime, the WinMon.BE programme evolved from cautiously observing early effects to the basis for an in depth understanding of longer-term effects. Milestones like the organisation of the international WinMon.BE symposium (2013), lifting the basic monitoring programme to a higher level of ecosystem component integration (2015) and the connection to research programmes (from 2016 onwards) paved the way to where we are now. This year's report on the environmental effect assessment of offshore wind farms in Belgian waters takes stock of what has been done and what we have learned so far. We particularly focus on the increased knowledge base through basic monitoring and targeted research, and zoom into a selection of innovative monitoring and impact mitigation techniques.

Increased knowledge base through basic monitoring and targeted research

A decade of the WinMon.BE monitoring and research allowed stepping back from observing only short-term and local effects to the benefit of longer-term and more regional insights into the environmental impacts. This progress substantially increased the knowledge base on offshore wind farm-related pressures and the consequent impacts onto the ecosystem.

Pressures

Because fishing is either prohibited or strictly limited within most European offshore wind farms, the overall surface area available for fisheries is decreasing as offshore wind farms are proliferating. Changes in fishing activity in the vicinity of offshore wind farms may hence be expected. We

demonstrated that the Belgian offshore wind farms (ca. 140 km² operational, not yet contiguous) only subtly changed the fishing activity (effort, landings and catch rate of target species sole Solea solea and plaice Pleuronectes platessa) of the Belgian and Dutch beam trawl fleet in Belgian waters over the period 2006-2017 (Chapter 4). In general, a business-as-usual scenario, comparable to the wider area, was seen in the vicinity of the offshore wind farms in both fishing effort and landings of the top 10 species. Evidently, since fishing is forbidden within operational offshore wind farms, a remarkable decrease in fishing effort was observed within the offshore wind farms. Our results suggest that local fishermen have adopted efforts to take into account the exclusion of the wind farm zone from their fishing grounds and have increased fishing efforts at the edges (especially those of the more offshore ones). While catch rates of sole in the vicinity of the operational offshore wind farms remained comparable to catch rates in the wider area, catch rates of plaice were higher around some operational wind farms as were landings from that area.

Recent studies have listed potential contamination by chemical emissions of metals and organic compounds related to offshore wind farms. Aluminium (Al), zinc (Zn) and indium (In) are metals used for corrosion protection in sacrificial anodes. As Zn concentrations have increased in the BPNS over the past decades, a first indicative and preliminary test was set up to measure Zn concentrations in the sediment from wind farms in the BPNS (Chapter 3). Zn concentrations in sediment samples collected nearby (37.5 m) and further away from (300-500 m) wind turbines on the Bligh Bank and Thornton Bank however were almost 60% lower (average 4.6 ± 1.0 mg.kg^{-1}) than those in the nearby reference zone on the Goote Bank, with no significant difference between nearby and far sediment samples. More extensive research is needed to further validate the results and to unravel the potential emission of other metals and organic compounds from offshore wind farms.

Ecosystem impacts

Two consecutive years of sediment sampling very close to (37.5 m) and further away from (350-500 m) wind turbines at the Thornton Bank (jacket-founded wind turbines) and Bligh Bank (monopile-founded wind turbines) revealed consistent foundation type-related impacts on the sediment composition and macrobenthic communities (Chapter 5). Sediment fining was only observed very close to the jacket foundations (21 ± 2% compared to 15 ± 1% further away), while no conclusive results were found in terms of organic enrichment. General trends in benthic responses were most pronounced at the Thornton Bank and comprised higher densities and diversity (species richness) in closer vicinity of the jacket-founded wind turbines. Macrobenthic assemblages closer to the turbines showed similarities with communities that are associated with hydrodynamically lower-energy environments. The recurrent trend of more pronounced results for jacket-founded wind turbines confirms the hypothesis that impacts are site and/or turbine-specific, which highlights the importance of a continued monitoring of the macrobenthos at the three different turbine types (gravity-based, jacket-founded and monopile-founded) in the BPNS.

A decade of monitoring of the macrofauna fouling the foundations revealed three succession stages at two types of offshore wind turbines (*i.e.*, gravity-based and monopile foundation) off the Belgian coast (Chapter 7). The installation of the turbine foundations was followed by rapid colonisation and a relatively short pioneer stage (~2 years) which differed between the two locations. At both locations, this was followed by a more diverse intermediate stage characterised by large numbers of suspension feeders, *e.g. Jassa herdmani*. A third, and possibly "climax" *Metridium senile*-dominated stage, was reached after ten years on the gravity-based foundations, while the assemblage on the steel monopiles at the more offshore site (nine years after construction) was described as an

M. senile-Mytilus edulis-co-dominated assemblage. We conclude that earlier reports on offshore wind turbines as biodiversity hotspots should be read with caution as these reports generally refer to the typical species-rich second stage of succession reached after a few years of colonisation but disappearing in a later stage (after about six years in this study). Our results further underline that artificial hard substrata hence differ greatly from the species-rich natural hard substrata and consequently cannot be considered as an alternative for the quantitatively and qualitatively declining natural hard substrata such as gravel beds.

Comparing three years of pre-construction seabird distribution data with six years of post-construction data at the Thornton Bank showed a significant avoidance of the wind farm area by northern gannet *Morus bassanus* (-98%), common guillemot *Uria aalge* (-60 to -63%) and razorbill *Alca torda* (-75 to -80%) (Chapter 8). In contrast, attraction to the wind farm could be demonstrated for herring gulls Larus argentatus and great black-backed gulls Larus marinus, for which a factorial change in densities of 3.8-4.9 and 5.3-6.6 was found, respectively. Importantly, most of these effects account for the offshore wind farm footprint area only and were no longer noticeable in the buffer area 0.5-3.0 km away from the wind farm edge. Great cormorants *Phalacrocorax carbo* showed major attraction effects too, amplified by the fact that the species was quasi-absent in the study area prior to wind farm construction. The effects at the Thornton Bank show striking parallels with those observed at the nearby Bligh Bank, and European studies in general show good consistency in the avoidance response of gannets and auks, as well as in the attraction effects observed for great cormorants and great black-backed gulls. How displacement effects impact individual fitness, reproductive success and survival remains yet unknown, hampering a reliable assessment of the actual and cumulative ecological consequences of extensive offshore wind farm installations.

Innovative monitoring and impact mitigation techniques

Innovative monitoring

During the last decade, new environmental concerns requesting new data collection arose and new analytical techniques allowing an in-depth analysis of existing data were introduced. The former necessitated extending the WinMon.BE programme to explore taking steps into the pelagic ecosystem (*in casu* hyperbenthos) and to incorporate the assessment of impacts on migrating bats. The latter allowed taking preliminary steps towards correlating marine mammal strandings data to piling activities.

Despite their important role in benthopelagic coupling and analogies with macrobenthic communities in terms of distribution patterns and seafloor dependency, hyperbenthos is often not included in current monitoring programs. It is proposed that turbine-related habitat changes such as altered hydrodynamics and organic enrichment could create more favourable conditions for the settlement of pelagic species and attract mobile species, resulting in richer hyperbenthic communities within the offshore wind farms. Therefore, an exploratory feasibility study on the sampling effort (design/strategy, processing time) needed to achieve reliable hyperbenthos data was performed (Chapter 6). The feasibility study revealed that short (*i.e.* 150 m) hyperbenthic sledge tracks parallel to sand ridges provide useful samples processable within a reasonable time frame (*i.e.* 1 week/sample). These results demonstrate the feasibility of including the hyperbenthos into the basic WinMon.BE programme.

Given the known impact of onshore wind turbines on bats, concerns were recently raised on whether offshore wind farms pose risks to bats. Bats undertaking seasonal migration between summer roosts and wintering areas can cross large areas of open sea. A better comprehension of the phenology and associated weather conditions of offshore bat migration will therefore provide a science base for mitigating the impact of offshore wind turbines on bats. We demonstrated that the Belgian offshore wind farms are predominantly visited by Nathusius' pipistrelles *Pipistrellus nathusii* (142 bat recordings during 23 nights in autumn 2017; Chapter 9). Wind speed seemed to have a large influence on the presence of bats during the study period, with 87% of the detections when the wind speed was maximally 5 m/s. The number of migrating bats may hence be negatively correlated with the wind turbine activity. Also, wind direction proved to be important, with a clear peak in occurrence when wind blew from the East and the South-East. Bat activity was further positively related to temperature and barometric pressure. This study shed a preliminary light on the meteorological conditions favouring bat activity in the Southern North Sea and the possible risk of colliding with offshore wind turbines.

The high sound levels produced during offshore wind farm construction result in displacement and disturbance of harbour porpoises *Phocoena phocoena,* the most common cetacean in the Southern North Sea. Prolonged exposure to high sound levels is likely to reduce fitness and may indirectly lead to increased mortality. We therefore examined, over a period of fourteen years, whether prolonged periods of intermittent high intensity impulsive sound influenced the temporal pattern of strandings of harbour porpoises on Belgian beaches (Chapter 10). Generalized Additive Mixed Modelling revealed a strong seasonal pattern in strandings, with a first peak in spring (March-May) and a second, less pronounced, in September. In addition, our analysis revealed a significantly higher occurrence of stranded harbour porpoise on Belgian beaches during months with prolonged periods of intermittent high intensity impulsive sound, suggesting an increased mortality of harbour porpoise

during those periods. A future in-depth analysis of age, sex, cause of death, and overall health (prior to death) of the stranded specimens should help determining the drivers (*i.e.* cause-effect relationships) for this additional mortality and reducing the uncertainty due to the biases inherently associated with the use of this strandings data.

Innovative mitigation techniques

The massive technical revolution in offshore renewables developments of the last decade has technically facilitated the mitigation of well-known unwanted pressures using innovative techniques.

Reducing the amount of underwater sound introduced into the marine environment during pile driving by using sound mitigation techniques has received a lot of attention in recent years and various techniques are now commercially available. During the pile driving at the Norther wind farm in 2018 for example, a Single Big Bubble Curtain (BBC) was applied to lower the sound pressure. In addition, sound mitigation experiments were conducted using the AdBm Noise Mitigation System, which is a stationary resonator system. AdBm was applied either alone or together with the BBC. *In situ* measured zero to peak sound levels (L_{z-p}) produced by piling 7.2 to 7.8 m diameter monopiles (max. possible hammer energy of 3500 kJ) ranged from 188 to 200 dB re 1μ Pa (normalized to a distance of 750 m from the source; Chapter 2). The higher values were measured when no sound mitigation measures were deployed, while the lower sound levels were achieved when the AdBm and BBC noise mitigation systems were applied simultaneously. The sound mitigation achieved by the BBC was within the single digit range (max. 7 dB re 1 Pa (L_{z-p}) reduction), while a simultaneous deployment of both systems showed a two-digit reduction of the sound level (max. 11 dB re 1 Pa (L_{z-p}) reduction). Accounting for local hydrodynamic conditions and an optimal use of the sound mitigation devices may further increase the sound mitigation efficiency.

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. At present, an installed capacity of 1.5 Gigawatt (GW), consisting of 318 offshore wind turbines, is operational in the Belgian part of the North Sea (BPNS). Two other projects, Seamade and Northwester 2, are scheduled to become operational in 2020. With the revision of the marine spatial plan, the federal government is looking to reserve an additional zone for 2 GW of offshore wind. With 522 km² reserved and planned 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 program focusing on various aspects of the marine ecosystem components, but also on the human appreciation of offshore wind farms. This introductory chapter provides an overview of the status of offshore renewable energy development in the BPNS.

1. Offshore wind energy development 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 was 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). A second area of 284 km² is reserved in the
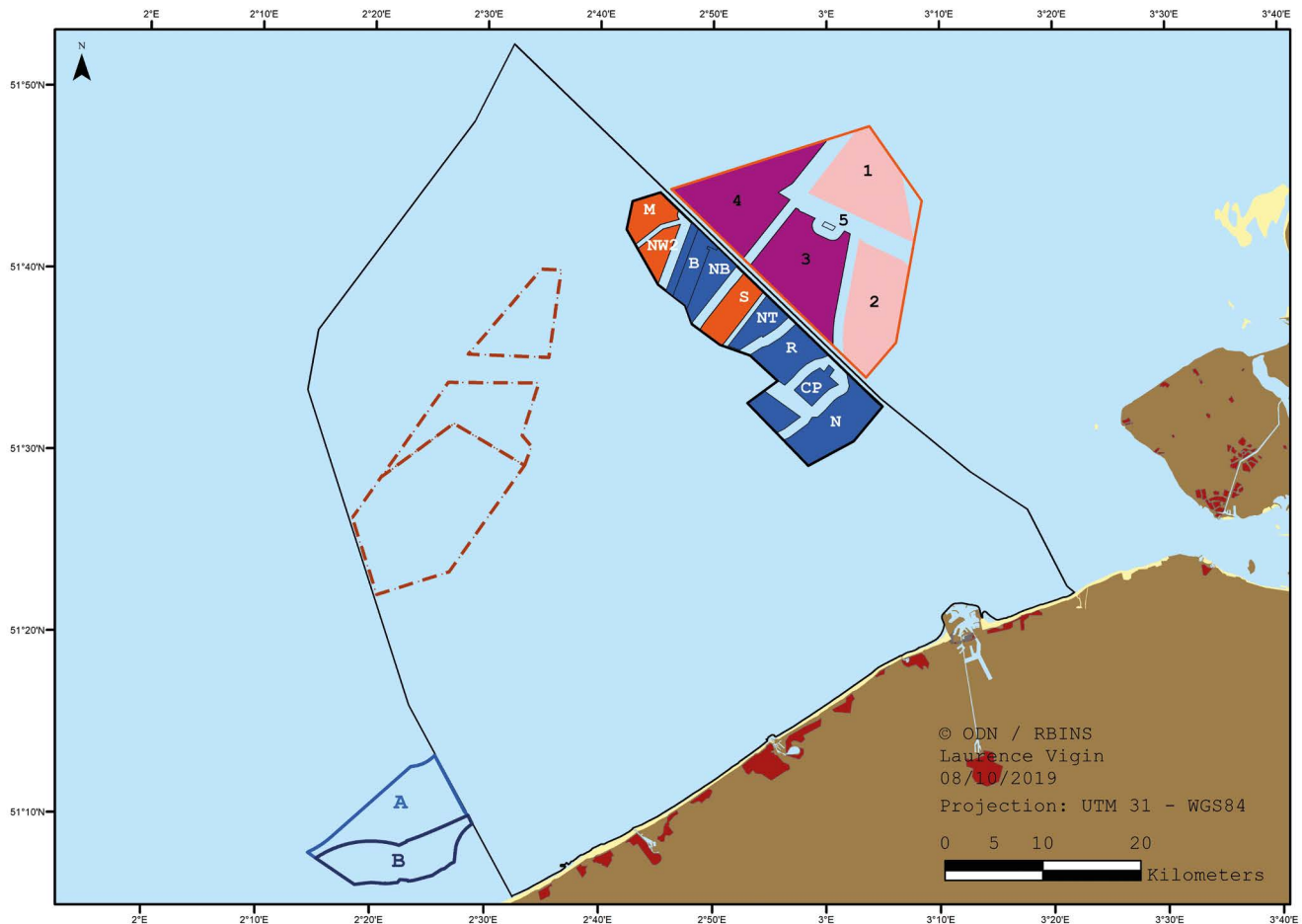
Figure 1. Current and planned zones for renewable energy in and around the Belgian part of the North Sea with indications of wind farms that are operational (blue), currently under construction (orange) or set to start construction end 2019 (purple) or 2020 (pink). A-B sites of proposed Dunkerque offshore wind farm. Dashed lines: locations of the new renewable energy zone as proposed in the draft of the marine spatial plan 2020-2026.
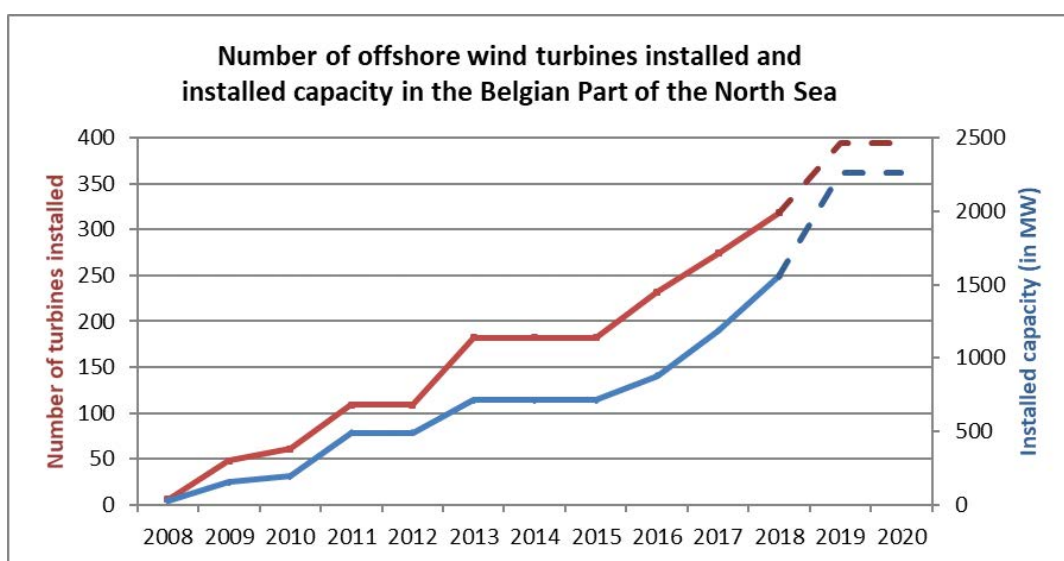
Figure 2. Number of offshore wind turbines installed and installed capacity in the Belgian part of the North Sea since 2008.

Table 1. Overview of wind farms in the Belgian part of the North Sea (situation on 20 April 2019)

Project		Number of turbines	Capacity (MW)	Total capacity (MW)	Status
C-Power	Phase 1	6	5	325	Phase 1 operational since 2009
	Phase 2 & 3	48	6.15		Phase 2 and 3 operational since 2013
Belwind	Phase 1	55	3	171	Phase 1 operational since 2011
	Alstom Demo project	1	6		Demo turbine operational since 2013
Nobelwind		50	3.3	165	Operational since 2017
Northwind		72	3	216	Operational since 2014
Rentel		42	7.35	309	Operational since 2019
Norther		44	8.4	370	Operational since mid-2019
SeaMade		58	8.4	487 + 5*	Construction foreseen to start in 2019
Northwester 2		23	9.5	219	Construction foreseen to start in 2019

*including 5 MW of wave energy

marine spatial plan that will come in force on 20 March 2020 (more information in §2).

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.

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). On 20 July 2018, the merger between the Seastar and Mermaid projects was finalised and the resulting merged project was named Seamade NV. In a rush to meet the 2020 goals, near-continuous pile driving activities can be expected from Mid-2019 to Mid-2020 in the Belgian and adjacent Dutch Borssele zone. By the end of 2020, when all Belgian wind farms are built, there will be a little less than 400 wind turbines in the Belgian part of the North Sea (fig. 2). The first entire area will have a capacity of 2262 MW and can 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.

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

On 20 March 2014, Belgium approved a marine spatial plan for the BPNS by Royal Decree. The 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. In this revision of the marine spatial plan (MRP 2020-2026), the Belgian federal government has delineated a second zone for renewable energy of 284 km² located at 35-40 km offshore (fig. 1). This second zone would be suitable for an additional 2 GW of installed capacity. The Belgian Offshore Platform, the association of investors and owners of wind farms in the BPNS, has recommended a density of 5 to 6 MW of installed capacity/km in this new zone in order to be able to realize maximum energy yields, and thereby reduce production costs. Storage of energy and grid reinforcement (see below) continue to be major hindrances to the further integration of renewables into the electricity grid and locations are foreseen for reinforcing the offshore electricity grid.

The second Belgian zone for marine renewable energy is partly located inside the Habitats Directive area "Vlaamse Banken". A targeted research programme was designed in order to determine whether and under what conditions renewable energy development is compatible with the natural values of this marine protected area. This programme commenced in 2019 and will last four years. The first results will become available from December 2019 onwards.

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, a comprehensive network upgrade was 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. However, further grid reinforcement is needed given the plans to expand wind capacity with a second concession area for offshore wind in the Belgian part of the North Sea. In this framework, Elia proposed the Ventilus project which includes a new power station near the port of Ostend and a high voltage network from Ostend to Brugge although the exact route still is to be determined. Ventilus will be a 380-kV high-voltage line with a capacity of 6 GW. In the long term, Ventilus will also make it possible to build a second subsea connection with the United Kingdom, alongside the existing Nemo Link project that became operational at the start of 2019.

The first five operational wind farms each ensure the export of their electricity to the onshore grid. Several proposals were 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
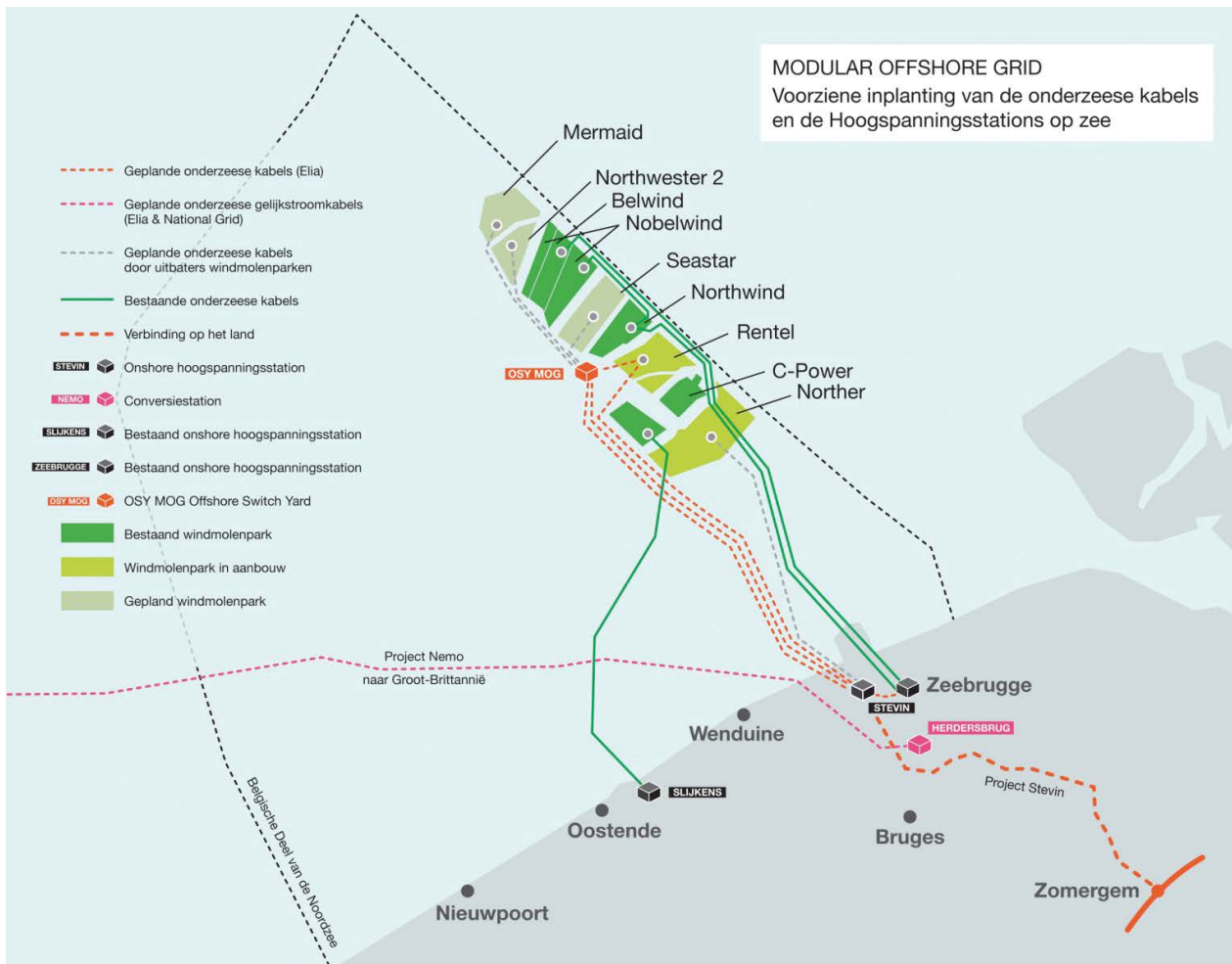
Figure 3. Design of the Modular Offshore Grid (MOG) (source: http://www.elia.be).

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, connects the remaining three wind farms to the grid (fig. 3). Construction of the MOG started in November 2018 and is expected to be operational by September 2019.

4. Wave energy in Belgium

Wave energy (or wave power) is the largest estimated global resource form of ocean energy. According to the World Energy Council (World Energy Council Netherlands 2017), the economically exploitable resource ranges from 140 to 750 TWh yr^{-1} for current designs of devices when fully mature and could rise to levels as high as 2000 TWh yr^{-1} if all the potential improvements to existing devices are realised. Wave energy converters (WEC) have been developed to extract energy and can be deployed from the shoreline out to the deeper offshore waters. In order to stimulate the development of wave energy in Belgium, the Mermaid project obtained its domain concession license only on condition that a certain amount of energy would be generated from waves as well as from wind. However, wave energy developments have not reached the anticipated level of commercial deployment and although the environmental permit of the Mermaid allows for an installed capacity of 5 MW of WECs no actual WEC deployment is foreseen in the near future.

Figure 4. The NEMOS Ostend Research Station.

Test sites are an essential element of any emerging technology developments including wave energy extraction. One such test site, for the NEMOS Wave Energy Converter, was constructed of the coast of Ostend in April 2019 (fig. 4). 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.

CHAPTER 2

ON THE EFFECTIVENESS OF UNDERWATER SOUND MITIGATION MEASURES USED DURING PILE DRIVING IN BELGIAN WATERS

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Abstract

During the construction work of the wind park Norther off the Belgian coast, underwater sound mitigation measures have been applied to lower the sound pressure generated during pile driving. The sound mitigation system (insertion loss) applied in this project was a Single Big Bubble Curtain (BBC). In addition, for five monopiles, experiments were conducted using the AdBm Noise Mitigation System, a stationary resonator system, either alone or together with the BBC. In this study, the underwater sound generated was recorded during five full pile driving events, including during three of the stationary resonator experiments. The diameter of the monopiles ranged from 7.2 to 7.8 m. The hammer used during this project was capable of a maximal energy of 3500 kJ. In situ measured zero to peak sound levels (L_{z-p}) showed values ranging from 188 to 200 dB re 1 µPa (normalised to a distance of 750 m from the source) respectively with the higher values when no sound mitigation measures were deployed at all, and with the lower sound levels when the AdBm and BBC noise mitigation systems were both active. Based on our measurements, the sound mitigation achieved by the BBC was in the single digit range, and the only two-digit reduction was achieved when both mitigation systems were working concurrently, achieving an 11 dB re 1 µPa (L_{z-p}) reduction. As previously observed, there was a lower-than-expected performance of the sound mitigation measures, which is likely to be due to local hydrodynamic conditions and or sub-optimal use of the devices.

1. Introduction

The construction of a new wind farm in the Belgian part of the North Sea (BPNS) nowadays relies on the installation of lower numbers of large steel monopiles (7 m diameter and bigger) accommodating generators of more than 7 MW, compared to a larger number of smaller monopiles (5 m diameter) equipped with 3 MW generators as was common practice in the BPNS from 2008 to 2016. Given the size of these monopiles, a large hydraulic hammer is required to drive these steel piles 26 to 47 m into the seafloor. Consequently, an important quantity of energy is introduced underwater in the form of sound that must be damped by noise mitigation systems (insertion loss) to comply with national legislation. In Belgium, impulsive sound should not exceed a zero to peak level of 185 dB re 1 µPa

at 750 m distance from the source Belgian State (2018).

In August 2018, Norther NV started the construction of an offshore wind farm at about 13 NM from Zeebrugge. This 370 MW wind farm consists of 44 Vestas V164-8.0 MW wind turbines and one offshore transformer station thus requiring the installation of a total of 45 foundation structures. The diameter of these monopiles ranges from 7.2 to 7.8 m. The pile driving was done using a 3500 kJ hydraulic hammer. Pile driving took place between 6 August and 12 November 2018.

In absence of direct sound mitigation, zero to peak sound levels (L_{z-p}) as high as 204 dB re 1 µPa at 750 m from the source were predicted (Bellmann *et al.* 2017). The concessioner proposed a sound mitigation system consisting of a single big bubble curtain (BBC). In addition, a field test of the experimental AdBm stationary resonator method was to be applied to five monopile installations.

This setup was accepted by the responsible administration and an underwater sound monitoring ensued.

The purpose of this report is first to quantify the emitted underwater sound measured *in situ* at sea and, second, to assess and evaluate the noise abatement achieved by the sound mitigation measures, *i.e.* BBC and AdBm static resonator systems.

2. Material and methods

2.1. Construction activities and local conditions

The first steel monopile of the Norther offshore wind farm was installed on 6 August 2018 (WTG-35-I2) and the last one was piled on 12 November 2018 (WTG-31-K4). A hydraulic hammer MHU 3500S from Menck Gmbh was deployed from the jacking-up platform Aeolus for every piling. With the exception of the Adbm field tests, sound mitigation was always in place during pile driving in the form of a BBC of 660 m long (table 2). The flow of air was provided by seven oil-free compressors (AC PTS 916) at the rate of 43 $m^3 min^{-1}$ each.

For the Adbm field tests, a series of four periods of pile driving were undertaken at constant energy of 1750 kJ for the monopiles WTG 30-J2, WTG 21-F1 and with an energy of 2250 kJ, for WTG 27-K2. Those four phases, that have a duration of 5 to 6 minutes each, are:

- piling reference period with no sound mitigation applied;

- piling period with the AdBm stationary resonator deployed alone;

- piling period with AdBm and BBC applied together;

- piling period with BBC only.

These piling sequences are clearly identifiable in the underwater sound recordings because of the long period of no activity of about 20 minutes separating the different phases of the experiment.

The BPNS is the seat of strong semi-diurnal tides. At the Norther construction site, the tidal current can be in excess of 1.5 m/s at a given time during the moon cycle (Belgian nautical chart D11). In this zone of the North Sea, the semi-diurnal tidal current is changing speed and direction all along the 12h25 tidal cycle.

2.2. Research strategy

Underwater sound generated by driving 7.2 to 7.8 m diameter steel monopiles into the seabed was measured *in situ* during construction. Five full pile driving events were recorded from 12 September 2018 to 18 September 2018 (tables 1 & 3). Two events were measured when only the BBC was operational. The three other events were measured during the field test of the experimental AdBm stationary resonator method. Various metrics like Level zero to peak (L_{z-p}), Sound Pressure Level (SPL) or the

sound exposure levels of a single stroke 95 percentile (SEL_{95}) were considered. The effectiveness of the sound mitigation measure was assessed comparing the measured value *in situ* on site during the various phases of pile driving with the reference value of the pile driving during the insertion loss experiment.

2.3. Underwater sound measurement equipment

Underwater sound was recorded from two moored stations (fig. 1). Each mooring was equipped with a measuring chain consisting of an acoustic release (Benthos 866 A/P), one underwater sound recorder (RTsys EA-SDA14), one hydrophone (Brüel & Kjær – B&K – 8104 or HTI-96-MIN), 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

Figure 1. Mooring design of the underwater sound measurement equipment. Here on the rear deck of Zeetijger prior deployment.

block afterwards. The pop-up buoy is attached rigidly to the concrete block to avoid perturbing sound. The sound recorder 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 deployment and further verified using a ½ inch precision microphone B&K 4191-l in the laboratory.

The moorings were deployed on 28 August 2018 from the work vessel Zeetijger at (WGS84) N 51° 32,503; E 003° 00,967 (mooring 1) and N 51° 31,010; E 3° 00,849 (mooring 2). The distance between the measuring equipment and the piling locations ranged from 439 m to 3758 m. No surface marker was left on site to reduce risks to navigation inside the construction zone as well as to avoid any perturbing sound originating from a line linking a surface buoy to the mooring.

Table 1. Position of the monopiles and instruments in UTM31, distance from the monopile to the measuring equipment (RTsys1 and RTsys2)

Names	Center Point Position (Coordinates)		Distance to RTSys1 (m)	Distance to RTsys2 (m)
	Easting	Northing		
WTG-21-F1	501316.93	5707017.97	3124	439
WTG-27-K2	499778.32	5709684.00	1305	2670
WTG-28-I1	501743.17	5709164.60	1211	2013
WTG-30-J2	501610.29	5710025.85	613	2796
WTG-33-F3	504098.43	5709401.95	3176	3758
RTsys1	501006	5710126		
RTsys2	500982	5707301		

2.4. Underwater sound measurements and post-treatment

Sound pressure was recorded continuously at a sampling rate of 78,125 Hz and stored on hard drives coded on WAV format. One instrument did not work properly and no data could be exploited from mooring 1. Mooring 2 was dedicated more specifically to the AdBm tests and was recording five full piling events (table 1).

MATLAB was used for the post treatment of the records. SEL$_{95}$, as well as the normalisation of the sound levels to the reference distance of 750m was computed following the Material & Method section of Norro *et al.* (2013).

2.5. Evaluation of noise mitigation measures

For assessing the broadband efficiency of the sound mitigation system or insertion loss, a direct comparison is made here between the measured peak sound levels during the different phases of the AdBm field test. For simplification of the interpretation, the same energy was assumed to be used by the hammer during the insertion loss experiment.

Spectral analysis of the sound pressure levels in 1/3 octave was conducted to examine the efficiency of the insertion loss as a function of the frequency of the generated sound.

2.6. Single big bubble curtain specificities

Mitigation measures introduce an "insertion loss" between the sound source and the surrounding environment.

Table 2. Specificities of the single big bubble curtain, nozzle hose (as provided by the concessioner)

FAD available	> 0,4 m³/m/min
Diameter holes	1,5 mm every 200-300 mm

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). When optimised (table 2), 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). Norro (2018) measured efficiency of a BBC during the Rentel piling work operated in 2017 in the Belgian North Sea. The sound reduction achieved during operations in the field for L$_{z-p}$ was ranging from 11 to 13 dB re 1 µPa. Based on a study from iTap a sound reduction of 19 dB re 1 µPa (L$_{z-p}$) using the BBC only was predicted prior to construction.

3. Results

In the absence of any noise mitigation, L$_{z-p}$ at 750 m from the source ranged between 198 and 200 dB re 1 µPa. This L$_{z-p}$ was reduced to 189 to 194 dB re 1 µPa when only the BBC was applied. When both noise mitigation systems, AdBm and BBC, were operating concurrently L$_{z-p}$ at 750 m from the source ranged from 188 to 190 dB re 1 µPa (table 3).

SEL$_{95}$ normalised at 750 m ranged for the reference values between 172 and 177 dB re 1 µPa^2 s while a reduction to a level ranging from 164 to 166 dB re 1 µPa^2 s was obtained for the combination of AdBm and BBC. For the most frequently used sound mitigation system (BBC only), SEL$_{95}$ ranged from 163 to 174 dB re 1 µPa^2 s.

The broadband insertion loss (table 3 in bold) of the various mitigation measures applied ranged from 3 to 6 dB re 1 µPa (L$_{z-p}$) when applying AdBm alone, from 6 to 11 dB re 1 µPa (L$_{z-p}$) when applying BBC alone and from 10 to 11 dB re 1 µPa (L$_{z-p}$) when both AdBm and BBC are applied.

The spectral analysis presented at figure 2 shows that the insertion loss
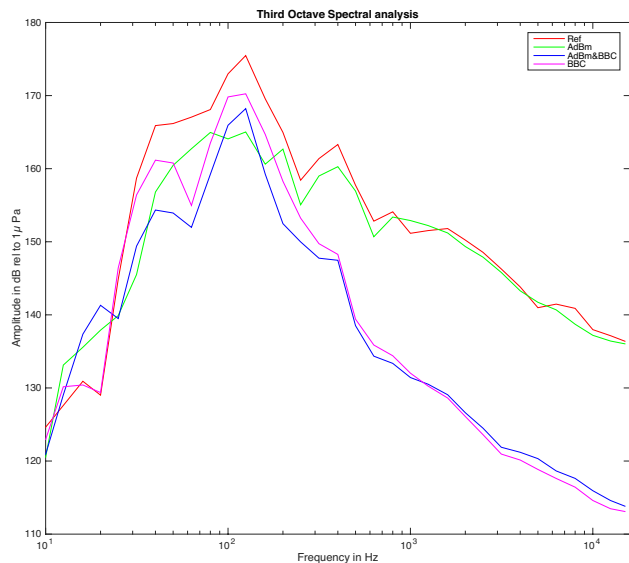
Figure 2. Spectral analysis of the underwater sound pressure level measured during the reference, AdBm, AdBm & BBC and BBC phases of the insertion loss experiment for pile WTG 21-F1 (measured at 439 m distance).

Table 3. Zero to peak sound pressure levels in dB re 1 µPa as measured *in situ* and normalised at 750 m distance from the pile driving location for five piling events measured at the Norther site. Reference is measured during the insertion loss experiment without any direct sound mitigation while Ref-Bellmann is taken as reference for those pile driving events where no period without any mitigation is available. Broadband efficiency estimate of the mitigation measures is based on the difference between measured zero to peak sound pressure level (L_{z-p}) on site during the various phases of the piling during the insertion loss experiments. Normalised at 750 m values are also provided

Norther 2018	Reference		AdBm			AdBm & BBC			Flow BBC	E Hammer	BBC			Max Observed		Distance from source
Name	L_{z-p}	SEL$_{95}$	L_{z-p}	SEL$_{95}$	Eff AdBm	L_{z-p}	SEL$_{95}$	Eff AdBm & BBC	m³/m/min	kJ	L_{z-p}	SEL$_{95}$	Eff BBC	L_{z-p}	SEL$_{95}$	m
WTG-30-J2	189	163	186	159	3	179	155	10	0,42	1750	182	157	7			2796
	198	172	195	168	3	188	164	10			191	166	7			Norm at 750 m
WTG-27-K2	192	166	187	164	5	181	159	11	0,42	2250	186	161	6			2670
	200	174	195	172	5	189	167	11			194	169	6			Norm at 750 m
WTG-21-F1	206	183	200	176	6	195	172	11	0,42	1750	200	176	6			439
	200	177	194	170	6	190	166	10			194	170	6			Norm at 750 m
WTG-28-I1	Ref-Bellmann								0,42	2612				191	168	2013
	204													193	174	Norm at 750 m
WTG-33-F3	204								0,36	2053				178	157	3758
														189	167	Norm at 750 m

efficiency is increasing at higher frequencies than those featuring the maximal transfer of energy into the water column. The latter is located from 100 to 150 Hz. Sound mitigation appears to be more efficient as the frequency of the sound increases above 300 Hz (fig. 2). Both BBC and, BBC and AdBm showed an important decrease in measured sound pressure level above 300 Hz. AdBm showed the best attenuation (about 10 dB re 1 µPa) of the sound pressure level at 63 and 125 Hz while above 300 Hz the attenuation of about 20 dB re 1 µPa is mainly achieved by the BBC.

4. Discussion

During the five complete piling events measured in this study, L_{z-p} was generally higher than the 185 dB re 1 µPa at 750 m distance from the source which is the MSFD limit in Belgium. The predicted (by the concessioner) reduction of 19 dB re 1µ Pa (broadband) was never obtained during our measurements and this even when both the AdBm static resonator and the BBC were used concurrently. Furthermore, the value obtained for the efficiency of the BBC is below what can be found in literature (Bellmann *et al.* 2017; OSPAR 2014). Several reasons can be at the basis of this lower-than-expected performance of the sound mitigation measures.

The usual cause for lower than expected performance of BBC lies in its sub-optimal set up (Bellmann *et al.* 2017; OSPAR 2014).

Airflow as well as hole diameter and spacing on the nozzle hose play an important role in the sound mitigation efficiency (Bellmann *et al.* 2017; OSPAR 2014). Any leakage in the system reduces the effective flow into the nozzle hose. For this project, it is impossible to assess whether such flow could be the reason for the lower performance because exact airflow passing throughout the nozzle hose was not measured during the operation.

Another reason for lower performance may be linked to strong tidal currents (see below). Indeed, the Belgian part of the North Sea is the seat of strong semi-diurnal tidal currents (Nautical chart D11). If one takes into account that the smallest bubbles of air have an ascent speed of 10 to 15 m.min⁻¹.
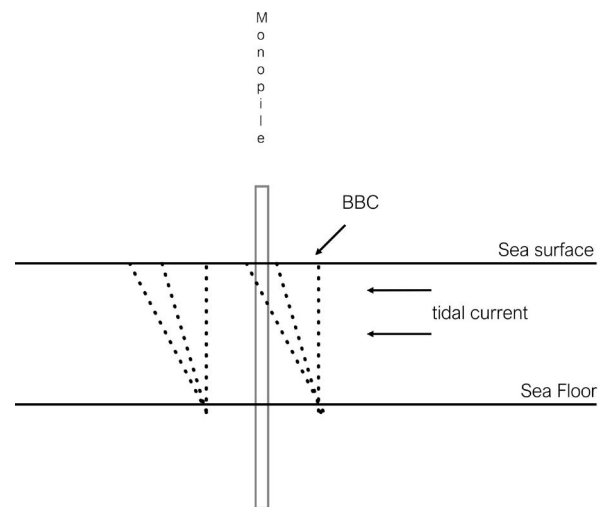
Figure 3. Sketch of the effect of current on the bubble curtain (BBC). Not to scale.

From a depth of about 25 m, 2 minutes are needed to reach the surface and therefore, in presence of a 1 m.s^{-1} current, a drift of about 120 m may be expected.

Since the BBC is deployed featuring a round or sometime elliptical shape diameter of maximum 200 m (cf. nozzle hose of 660 m) it is possible that the bubble curtain will lose part or most of its efficiency and possibly "overtake" the monopile location during piling operations (fig. 3).

A possible optimisation of BBC efficiency in high tidal current situation could be to pile only when the tidal current speed is minimal or at least below a threshold to be determined as a function of depth and nozzle hose diameter.

For two piling events (WTG-28-I1 and WTG-33-F3), the BBC efficiency seems to be higher than for the other measured piling events. This can possibly be explained by the relatively high reference value of 204 dB re 1 µPa that had to be assumed here in absence of direct measurements of the reference value. The reference value was taken from the ITAP model by Bellmann *et al.* (2017). In comparison with the piling events for which we were able to directly measure reference values, we may assume the 204 dB re 1 µPa value to probably be too high by 5 dB re 1 µPa. When applying that difference, the efficiency of the BBC alone is the same as for the other cases presented.

Another point to discuss is the difference in the efficiency of the sound mitigation system in relation to the frequency of the sound. Figure 2 shows that the insertion loss is not equal for every frequency. The efficiency is higher for frequencies above 300 Hz while the higher levels of

underwater sound are produced below that frequency and that considering the exposure time to piling works that is limited to 2 h by pile and occurs all in a period of few weeks only for the construction of a wind farm.

In order to comply with Belgian MSFD regulation Belgian State (2018), it is advisable to, at least, test on site the combination of sound mitigation measures before the construction works start and not rely only on predicted efficiency. Another option could also be to reduce the size of the monopiles used or even to come back to using jacket or tripod designs. Using smaller monopiles will reduce the maximum emitted underwater sound to levels that can be efficiently reduced by today's mitigation systems and end up with levels below the Belgian MSFD limits. Another possibility could be to use new methods like blue piling technology or to reconsider using suction bucket foundations.

5. Conclusion

With a level zero to peak (L$_{z-p}$) in excess of 185 dB re 1 µPa at 750 m (189 to 194) even with both AdBm and BBC sound mitigation measures in place, the emitted sound surpassed the Belgian MSFD limits. Values obtained for the efficiency of noise mitigation systems may be more site specific than previously thought and developers should state under which environmental conditions these reductions in underwater sound were achieved. For future construction activities of large monopiles, it is advisable to test the anticipated sound mitigation system efficiency prior to the start of the work on site in order to be certain to be able to comply with the Belgian MSFD thresholds for impulsive underwater sound.

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CHAPTER 3

PRELIMINARY ZINC ANALYSIS AT OFFSHORE WIND FARMS

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Abstract

Recent studies have listed potential contamination by chemical emissions of metals and organic compounds related to offshore wind farms. Aluminium (Al), zinc (Zn) and indium (In) are metals used for corrosion protection in sacrificial anodes. As Zn concentrations have increased at the Belgian part of the North Sea (BPNS) over the past decades, a first indicative and preliminary test was set up to measure Zn concentrations in the sediment from wind farms in the BPNS. Zn concentrations from nearby (37.5 m) and far (300-500 m) samples near four wind turbines on the Bligh Bank and Thornton Bank were lower than those in the nearby reference zone on the Goote Bank, with no significant difference between nearby and far sediment samples. More extensive research is needed to further validate the results and to unravel the potential emission of other metals and organic compounds from wind farms.

1. Introduction

Although wind energy is considered a green energy source, environmental impact may not be excluded. Whereas many studies have focused on the impact of offshore wind farms on the biodiversity, chemical emissions are less studied (Kirchgeorg *et al.* 2018). Chemical emissions can be divided in contamination by metals and by organic compounds. Metal emissions may originate from corrosion protection systems, such as sacrificial anodes, and include aluminium (Al), zinc (Zn) and indium (In) (Kirchgeorg *et al.* 2018; Tornero & Hanke 2016). Contamination of organic compounds may originate from increased vessel traffic and potential spillovers or from routine and maintenance activities, including leaching from antifouling paints or from hydraulic fluids and lubricant oils. Potential organic contaminants related to these activities are booster biocides, bisphenols, hydrocarbons such as BTEX and PAHs, silicone fluids, mineral oils, biodiesel, natural vegetable oils and synthetic esters, coolants and electrolytes (Tornero & Hanke 2016; Kirchgeorg *et al.* 2018). Although chemical emissions from offshore wind farms are probably low compared to other offshore activities, they can become relevant with increasing number of wind farms. Available data is scarce, making it currently difficult to assess the impact of these chemical emissions on the marine environment (Kirchgeorg *et al.* 2018).

Zn concentrations have increased in the Belgian part of the North Sea (BPNS) over the past decades (Lagring *et al.* 2018). Therefore, a first indicative and preliminary test was set up to measure Zn concentrations in and around offshore wind farms at the BPNS.

Table 1. Sediment sample coordinates with zone (NEAR) at about 37.5 m from the specified wind turbine, zone (FAR) at about 350-500 m from the specified wind turbine and zone (REF) reference zone

Sample	Wind turbine	Zone	Latitude	Longitude	Location
BB9_ZEERDICHT	BB9	NEAR	51°39.79357'N	2°47.88500'E	Bligh Bank
BB9_VER	BB9	FAR	51°39.89838'N	2°48.15348'E	Bligh Bank
TB7_ZEERDICHT	TB7	NEAR	51°33.12130'N	2°57.98891'E	Thornton Bank
TB13_VER	TB13	FAR	51°33.30576'N	2°58.26396'E	Thornton Bank
TB15_ZEERDICHT	TB15	NEAR	51°32.00005'N	2°55.47882'E	Thornton Bank
TB24_VER	TB24	FAR	51°32.24592'N	2°55.48956'E	Thornton Bank
BGR 2	-	REF	51°27.51944'N	2°54.51424'E	Goote Bank
BGR 19	-	REF	51°27.76585'N	2°50.84924'E	Goote Bank

2. Material and methods

Sediment samples were taken by the Marine Biology Research Group of Ghent University in autumn 2017 with the RV Simon Stevin (VLIZ) and Aquatrot (OMS) by using a Van Veen grab. For heavy metal analysis, an insertion tube of about 4-5 cm inner diameter was used to take a cross-section of the Van Veen grab sample. By this sampling technique, Zn concentrations were measured in the total sediment sample, supposing chemical equilibrium between the sediment and the interstitial water. Samples were taken about 37.5 m (NEAR) and 350-500 m (FAR) away from five specified wind turbines at the Bligh Bank (Belwind) and Thornton Bank (C-power) (table 1). Two reference locations (REF) were selected at the nearby Goote Bank, suggested as a useful reference area in De Maersschalck *et al.* (2005).

Samples were analysed by Sciensano, Service Trace Elements and Nanomaterials, for their Zn concentration. Total freeze-dried sediment samples were digested in Teflon bombs with a mixture of concentrated $HClO_4$, HNO_3 and HF at 170° C for 16 h. Dry residues were dissolved in HNO_3 prior to analysis with ICP-OES (De Witte *et al.* 2016). In contrast to the OSPAR assessment approach (OSPAR, 2011), Zn data is not normalised to aluminium, as Al can be a contaminant at offshore wind farms (Kirchgeorg *et al.* 2018).

3. Results

First indicative results do not seem to reveal higher Zn values in the direct neighbourhood of the wind turbines. On the contrary, concentrations inside both wind farms (average 4.6 ± 1.0 mg.kg^{-1}) are almost 60% lower than those in the reference zone on the Goote Bank (average 11.1 ± 1.1 mg.kg^{-1}). No clear difference was noted between nearby and far samples (on average 4.4 ± 1.2 mg.kg^{-1} at 37.5 m *vs.* 4.9 ± 0.8 mg.kg^{-1} at 500 m distance from the wind turbines).

4. Discussion

In this experimental design, Zn contamination measurements were performed as an indicative and preliminary test. Zn was selected as target compound as it is applied at wind farms for corrosion protection, and Zn values have been shown to rise over the last decades in the BPNS (Lagring *et al.* 2018).

The studies of Kirchgeorg *et al.* (2018) and Tornero & Hanke (2016) indicate that chemical emissions may be relevant at offshore wind farms. This first preliminary test does not reveal high(er) Zn concentrations at wind farms in the BPNS. However, the set-up of this study was too limited to conclude that Zn concentrations have not increased by offshore wind farms, as it is not investigated why concentrations at the Goote Bank are higher. More research on Zn contamination
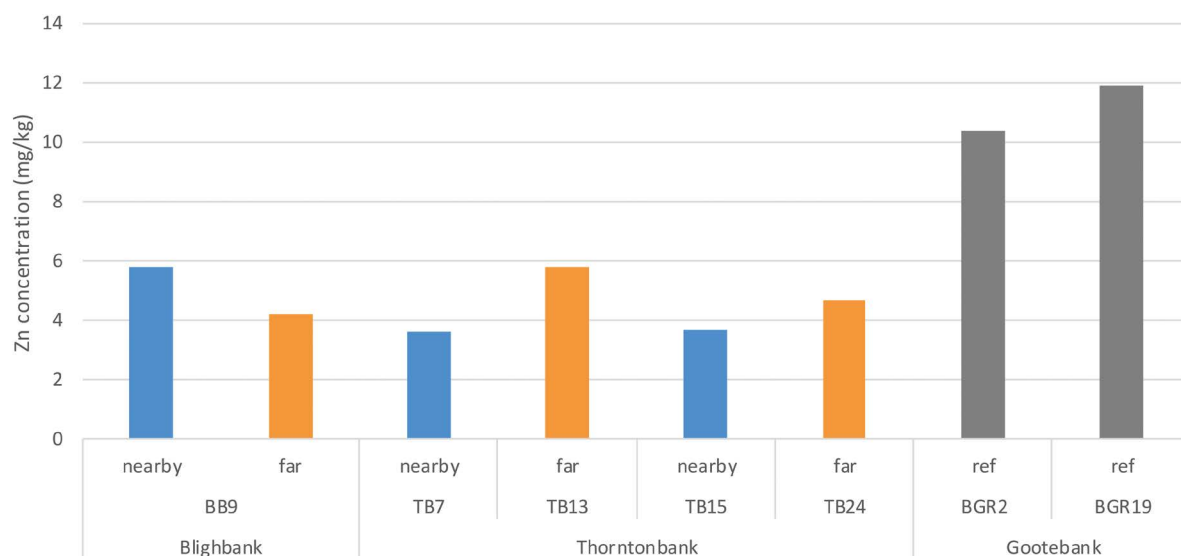
Figure 1. Zn concentrations in sediment samples at wind farms.

at the BPNS is needed. Moreover, further studies should also handle the normalisation issue. As Al can be a contaminant at offshore wind farms, currently applied OSPAR normalisation with Al does not seem appropriate. Evaluation of different normalisation methods, based on sediment granulometry, mineralogy or a combination of both will be essential in future research.

A more in-depth study on a broad selection of metals, including In, and organic contaminants seems appropriate to further investigate pollution by other wind farm operation contaminants and/or to unravel the potential emissions of other chemical compounds from wind farms in the BPNS.

Acknowledgements

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CHAPTER 4

FISHING ACTIVITIES IN AND AROUND BELGIAN OFFSHORE WIND FARMS:

TRENDS IN EFFORT
AND LANDINGS OVER THE PERIOD 2006–2017

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Abstract

Fishing is prohibited within most offshore wind farms (OWFs) within Europe. With a European target set to reach 20% renewable energy by 2020, the number of OWFs in the North Sea is increasing fast, and thus fishing grounds are decreasing. This could cause changes in fishing activity in the vicinity of OWFs. Using the 238 km² area dedicated for offshore energy production in the Belgian part of the North Sea as a case, this study aimed to investigate whether beam trawl fishing activity (effort, landings and catch rate of target species sole and plaice) changed over the period 2006-2017 in relation to the presence of OWFs. To our knowledge, this is the first study investigating the effect of operational wind farms, closed for shipping, on fishing activity.

To this end, trends in fishing activity within and around the Belgian OWF area were compared to the wider area surrounding the OWF. The active Belgian OWF area (now ca. 140 km² operational, not yet contiguous) only subtly changed the fishing activity of the Belgian and Dutch beam trawl fleet. Mostly, a "business as usual" scenario, comparable to the wider area, was noted in the vicinity of the OWF concessions in both fishing effort and landings of the top 10 species. Of course, since fishing is forbidden within operational OWFs, a remarkable decrease in fishing effort was observed. However, with the current design and size of the operational OWF area, the fishermen seem to have adapted to the new situation, and are not avoiding the areas around the OWFs, they even seem to be attracted to the edges (especially of the more offshore concessions). Catch rates of target species sole in the vicinity of the operational OWFs remain comparable to catch rates in the wider area, but for plaice, catch rates and landings seem to be even higher around some operational wind farms.

1. Introduction

Within European Directive 2009/28/EC, Europe has set a target to reach 20% renewable energy by 2020. Installation of offshore wind farms (OWFs) is a major component to reach this target. Currently, 4543 wind turbines produce 18,499 MW of renewable energy in Europe, of which 70% in the North

Sea (Wind Europe 2019). In 2018, Belgium had the fourth highest offshore wind capacity (6.4%), after the UK, Germany and Denmark (44; 34 and 7% respectively) (Wind Europe 2019). In 2004, the Belgian government delineated an offshore area of 238 km² for the production of electricity. At the moment, 5 OWFs are operational in the Belgian part of the North Sea (BPNS), good for an installed capacity of 1186 MW. Another one, which will produce another 370 MW, is under construction, and 3 others are in the pre-construction phase.

For safety reasons, shipping is prohibited within most OWFs in Europe. Consequently, commercial (and artisanal) fishing activities are excluded as well. The North Sea is one of the most heavily fished regions in the world, so with the further expansion of OWFs in the North Sea region, fishermen see their fishing grounds disappear, which makes them feel afraid to lose valuable areas and to see their catches and profit decline (Gray *et al.* 2016; Bolongaro 2017). On the other hand, OWFs, as fisheries exclusion zones, might provide shelter, spawning or nursing grounds, and higher food biomass (*e.g.* Leitao *et al.* 2007; Reubens *et al.* 2013 a & b; Stenberg *et al.* 2015), at least for certain fish species, which may then benefit the fisheries for these specific fish stocks. Closure of OWF areas for commercial fisheries will most probably result in changes in fishing intensity in the vicinity of the OWFs and/or in displacement of the fishing activity.

The Belgian OWF area is predominantly situated outside the 12 Nmile area in the southern North Sea. The fishery in this area is mainly dominated by Dutch and Belgian beam trawlers (incl. pulse trawlers since 2011), targeting sole and plaice (Gillis *et al.* 2008; Eigaard *et al.* 2017). The first Belgian turbines were operational in 2009, and since then increasingly more are built, enlarging the fisheries exclusion zone over the past decade. As such, the Belgian OWF area forms an interesting case to study whether the presence of OWFs has an effect on the fishing activity of Dutch and Belgian beam trawlers.

Our first objective was to study whether or not the effort, the landings and the landings per unit effort (LPUE) changed over the period 2006-2017 in relation to the presence of OWFs. The second objective was to investigate whether spatial changes in fishing activity took place due to OWFs by comparing the period 2006-2007 (pre-turbines) with 2016-2017 (232 operational turbines).

2. Material and methods

2.1. Study area

To investigate changes in fisheries activities, it is important to include a wider area to deduct general trends over time. The Belgian OWF area is covered by ICES rectangles 31F2, 31F3, 32F2 and 32F3, which were selected to represent the general trends within the wider area (fig. 1). VMS and landing data for these four rectangles were collected for the Belgian and Dutch beam trawl fleet (including both small and large beam trawls, with vessels respectively $\leq$ 221 kW and > 221 kW, including also the Dutch pulse trawlers operational since 2011). The ICES rectangles were gridded with a resolution of 0.05 x 0.025 decimal degrees (further referred to as "grid cells"), a trade-off between having a fine enough resolution for analyses in the Belgian OWF area, and still being adequate for VMS registrations that are recorded at a 2 hour ping rate (fig. 1).

The spatial coverage, construction and operation dates and number of turbines for each OWF concession area are summarised in table 1. Three wind farms (Seastar, Northwester2 and Mermaid) still have to be built (dashed lines in fig. 1). The other ones have been constructed between 2009 and 2019 (with the exception of 6 gravity-based turbines that were already constructed in 2007-2008 in the C-Power A concession zone).
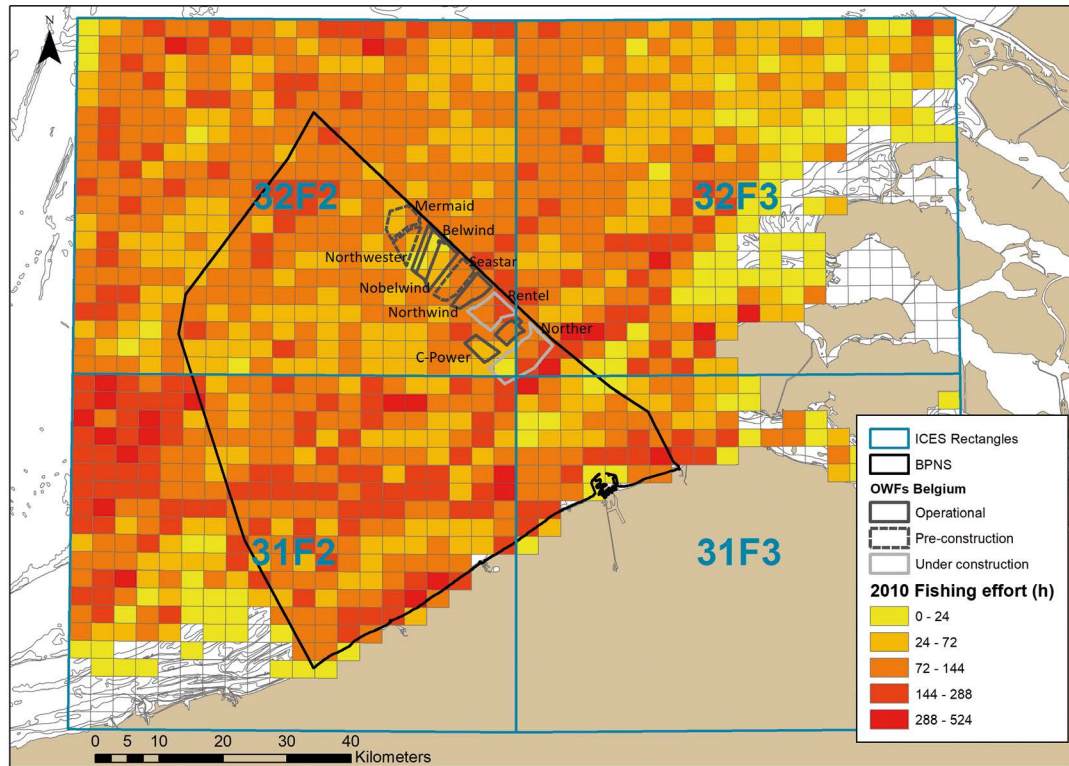
Figure 1. Overview of the study area with indication of ICES rectangles, used grid, Belgian territorial area, Belgian OWF concessions, and 2010 fishing effort of Belgian and Dutch beam trawlers (both = < 221 kW and > 221 kW).

Table 1. Summary of the Belgian OWF concession areas (ordered in relation to distance from the coast, *i.e.* from east to west, see fig. 1) with indication of surface area, number of turbines, start of construction and moment when first power is generated

Windfarm	Area (km²)	Turbines	Start construction	First power generated
Norther	44	44	2017	Foreseen in 2019
C-Power A	10.68	6 + 24	May 2007 & November 2010	January 2009 & September 2013
C-Power B	9.16	24	November 2010	September 2013
Rentel	23	42	July 2017	January 2019
Northwind	13.8	72	April 2013	May 2014
Seastar	18.4	30	Planned summer 2019	
Belwind	17	56	September 2009	January 2011
Nobelwind (S & N)	19.8	50	April 2016	December 2017
Northwester 2	15.2	23	Planned summer 2019	
Mermaid	16.3	28	Planned summer 2019	

2.2. VMS and logbook data

Fishing intensities, based on VMS (Vessel Monitoring by Satellite) data, and landed catches, based on logbook data, of Dutch and Belgian fishing vessels were calculated and combined for ICES statistical rectangles 31F2, 31F3, 32F2 and 32F3 per gear class. We distinguished fishing activity of vessels with different engine classes (engine power = < 221 kW and > 221 kW). Fishing intensities and landings were calculated on a yearly basis from 2006 until 2017.

VMS is a satellite-based monitoring system which, at regular intervals (mainly every 2 hours), provides data to the fisheries authorities on the location, date

Table 2. Overview of the top 10 species landed by the Dutch and Belgian beam trawl fleets

Common name	Scientific name	FAO code
Sole	*Solea solea*	SOL
Plaice	*Pleuronectes platessa*	PLE
Turbot	*Psetta maxima*	TUR
Brill	*Scophthalmus rhombus*	BLL
Dab	*Limanda limanda*	DAB
Flounder	*Platichthys flesus*	FLE
Cod	*Gadus morhua*	COD
Whiting	*Merlangius merlangus*	WHG
Tub gurnard	*Chelidonichthys lucerna*	GUU
Bib/Pouting	*Trisopterus luscus*	BIB

time, course and speed of vessels. Logbook data is compulsory for almost all commercial fishing vessels and contains information on daily catch composition, fishing gear usage, engine power and departure, and arrival harbours. Belgian VMS and logbook data are collected by dienst Zeevisserij (Departement Landbouw en Visserij; Afdeling landbouw- en visserijbeleid) and analysed by ILVO. The processed Dutch data were provided by Wageningen Marine Research.

All data processing of VMS and logbook data was made in R using the vmstools package (Hintzen *et al.* 2012). Before we analysed the Belgian fishing activities, an extensive quality control of the data was performed. We checked for duplicated data, locations inside the harbours, impossible time, dates, headings and locations. Only VMS records with speeds that correspond with fishing activity were selected. VMS and logbook data were linked based on vessel identity and date time. Using this link, we combined data on fishing location, date and time, fishing speed, fishing gear and landed catch. To derive the number of times fishing vessels have actively fished inside a grid cell, and to calculate their corresponding catches, we performed a spatial overlay analysis using routines in the R package sp (Bivand *et al.* 2013).

Given the low frequency of VMS pings (every two hours) and the determination of fishing activity based on speed, these results are only an estimation of the actual fishing activity in these areas.

2.3. Temporal trends in effort, landings and LPUE

In order to be able to link trends in fishing activities in and around the OWF concession area to real wind farm effects, the ICES rectangles covering the Belgian OWF area (31F2, 31F3, 32F2 and 32F3) were selected for a general trend analysis. Therefore, total yearly fishing effort (in hours fished) and landings (kg) of the top 10 species of the Dutch and Belgian beam trawl fleets are presented for the period 2006 to 2017 for the selected ICES rectangles. Similarly, landings per unit effort (LPUE) of the target species sole and plaice were calculated as total landings (kg) divided by fishing effort for the selected ICES rectangles.

To calculate fishing effort (hours fished) and landings of the top 10 species within each OWF concession area, the Geofish platform (developed by ILVO for the Geofish project (https://geofish.be/ – still in test environment) was used, which allows to sum fishing effort or landings within each OWF concession polygon on a yearly basis. For grid cells that are partly inside a specific polygon, a percentage of the value in that grid cell is taken relative to the surface area of the grid cell in the polygon. Afterwards LPUE of sole and plaice per OWF concession were calculated by dividing total landings within the concession by total hours fished (fishing effort) within the same concession.

This implies that the values that are presented for the operational wind farms,

actually include edge effects, since fishing activities occurring just outside the OWF polygon are allocated to the OWF. This can be seen as a drawback, but it especially provides insight in the "whether or not" overall avoidance of or attraction to the areas surrounding the active concession areas. As such, fishing activity within operational concessions should not be interpreted as ongoing fishing within the wind farm, but it is mainly representing edge effects.

To get an idea on the real intrusions of fishing vessels in the closed area, and thus the actual fishing inside the OWFs, information on intrusions of commercial fishing vessels was requested to and provided by Parkwind (representing the OWFs Belwind, Nobelwind and Northwind) and C-Power. These figures are proxies, since especially in the earlier years, when no AIS detection tools were available, intrusions might have happened unnoticed (pers. comm. Kristof Verlinden).

All data processing and analyses were made in R version 3.5.1 (R Core Team 2018) using R Studio (R Studio Team 2016) using the packages reshape2, dplyr and ggplot2.

2.4. Spatial changes in effort and LPUE distribution

We compared changes in effort and LPUE distribution for the period 2006-2007 (pre-OWF period) to the period 2016-2017 (3 operational OWFs and 3 OWFs under construction) for the combined Dutch and Belgian fishing effort data of the large beam trawl segment (> 221 kW). We excluded the small beam trawl segment (= < 221 kW) from this analysis, since these were only fishing within the 12 Nmile zone before 2009 due to high fuel prices, and thus not inside the OWF area. Including the effort data of the small beam trawl segment in this type of analysis would therefore distort the outcome in and

around the OWF area, since it was zero in the pre-OWF period.

The change in effort and LPUE distribution was calculated as a deviation of proportional effort or LPUE in the period 2016-2017 compared to the reference period 2006-2007. Since effort and LPUE can vary over years, effort and LPUE are expressed as a proportion of the average effort and LPUE (within the study area *i.e.* four statistical ICES rectangles) in each grid cell for the defined period. In this way, the general temporal trend was excluded.

Thus, for each grid cell, proportional effort and LPUE (relative to the average effort and LPUE for the wider ICES area) were calculated for both periods. Afterwards, the deviation was calculated as the difference in proportional effort or LPUE between period 2016-2017 and period 2006-2007. A negative deviation means that there is relatively less effort or lower LPUE in the grid cell in 2016-2017 compared to 2006-2007, while a positive deviation reveals a relative increased fishing effort or a higher LPUE.

Analyses were done in R version 3.5.1 (R Core Team 2018) using R Studio (R Studio Team 2016) using the package reshape2. Visualisation of calculated deviations was done using ArcMap 10.4.

3. Results

3.1. Temporal trends in fishing activity within the wider ICES area and the OWF concessions

3.1.1. Fishing effort

Over the period 2006-2017, Dutch and Belgian beam trawls spent on average around 108,000 (± SD 13,680) hours at sea in the selected ICES rectangles. Total fishing effort within the wider ICES area surrounding the OWF area showed a general decreasing trend over the years (fig. 2).
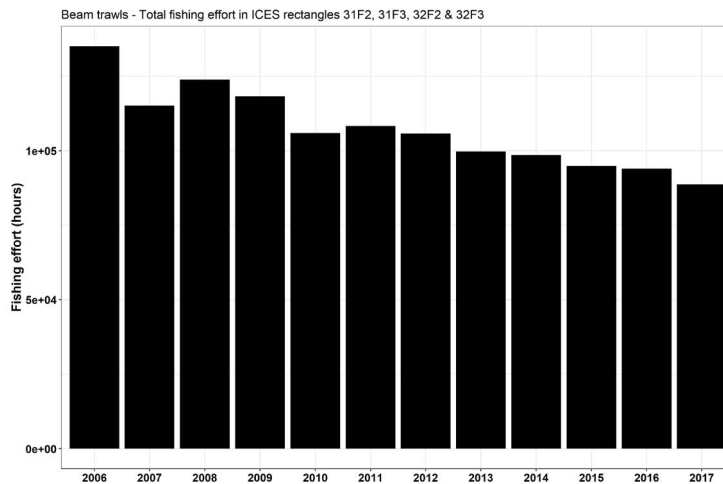
Figure 2. Time series of total fishing effort in hours of Dutch and Belgian beam trawlers in ICES rectangles 31F2, 31F3, 32F2, 32F3.

For the OWF area, a clear drop (up to 50%) in fishing effort is observed during the construction phase for all concessions where turbines have been built (fig. 3). Once in the operational phase, the reduced fishing effort continues to exist in C-Power A&B and Northwind. While in Belwind, fishing effort gives the impression to increase again in the operational phase, this is due to the resolution of the grid cells, since grid cells are only partially inside the Belwind concession, which means that fishing along the edges is (proportionally) seen as fishing inside the wind farm, but this actually is mainly an edge effect. In 2016, with the construction of Nobelwind (OWF surrounding Belwind),
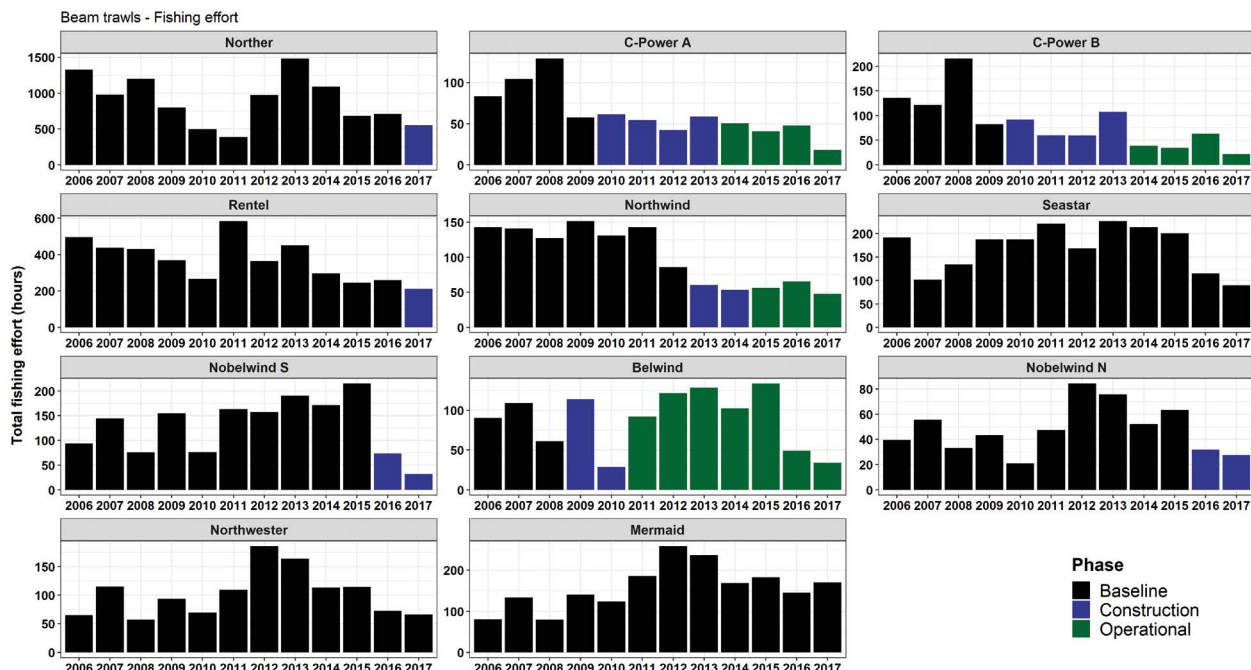
Figure 3. Time series of fishing effort of Dutch and Belgian beam trawlers for the different OWF concession areas in the BPNS with indication of the baseline, construction and operational phase. ! Nuanced interpretation of trends is needed: ongoing fishing effort during construction and operational phase (where fishing is forbidden) is mainly due to grid cells overlapping the edges of concessions, as such fishing activity along the edge of the concession is allocated to the concession.
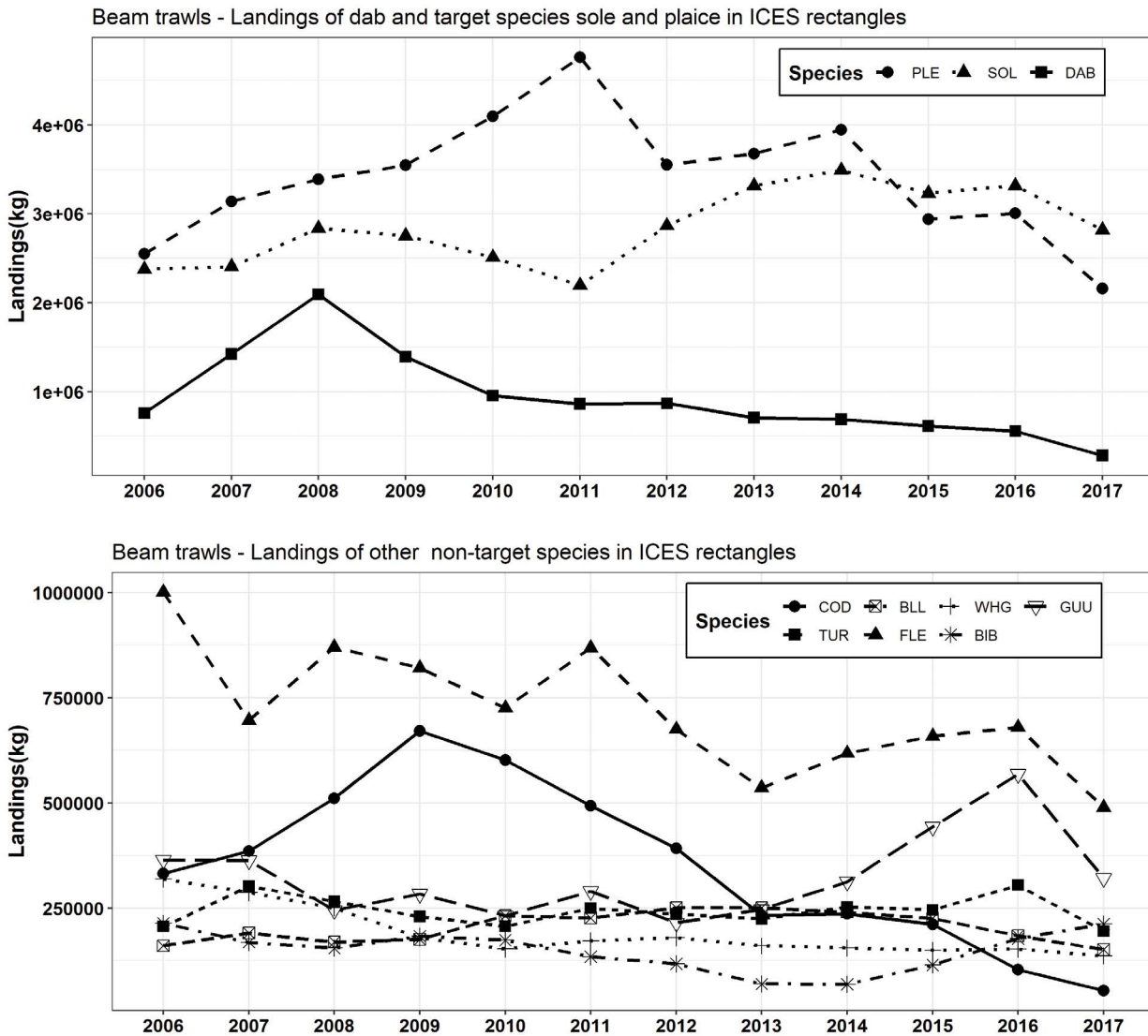
Figure 4. Time series of landings in kg of top 10 species for Dutch and Belgian beam trawlers in ICES rectangles 31F2, 31F3, 32F2, 32F3. Top: landings of dab and target species, plaice and sole; bottom: non-target species.

and thus expansion of the none-fishing area, fishing effort drops again, since grid cells are now predominantly situated entirely inside either Belwind or Nobelwind, where fishing is prohibited. In the OWFs without turbines yet (Seastar, Northwester and Mermaid), temporal variation in fishing effort is observed mainly in the earlier years, while in the later years (from 2012-2013), a decreasing trend is found, similar to the general trend in the selected ICES rectangles. For Northwester and Seastar, OWFs bordering Nobelwind, a drop in fishing effort in 2016 and 2017 was observed when construction started at Nobelwind.

3.1.2. Landings of top 10 species

Within the wider ICES area, Dutch and Belgian beam trawlers landed on average 2.8 MM (± SD 0.4 MM) kg of sole and 3.4 MM (± SD 0.7 MM) kg of plaice, their target species. Landings are in general well related to fishing effort. Landings of plaice, dab, cod and flounder increased in the earlier years, but decreased later on (fig. 4). Sole is the only species with a clear increase (26%) in landings after 2012. Although, gurnard landings were higher as well in 2015 and 2016. Landings of the other species (turbot, brill, whiting and bib) remained more or less stable (fig. 4).
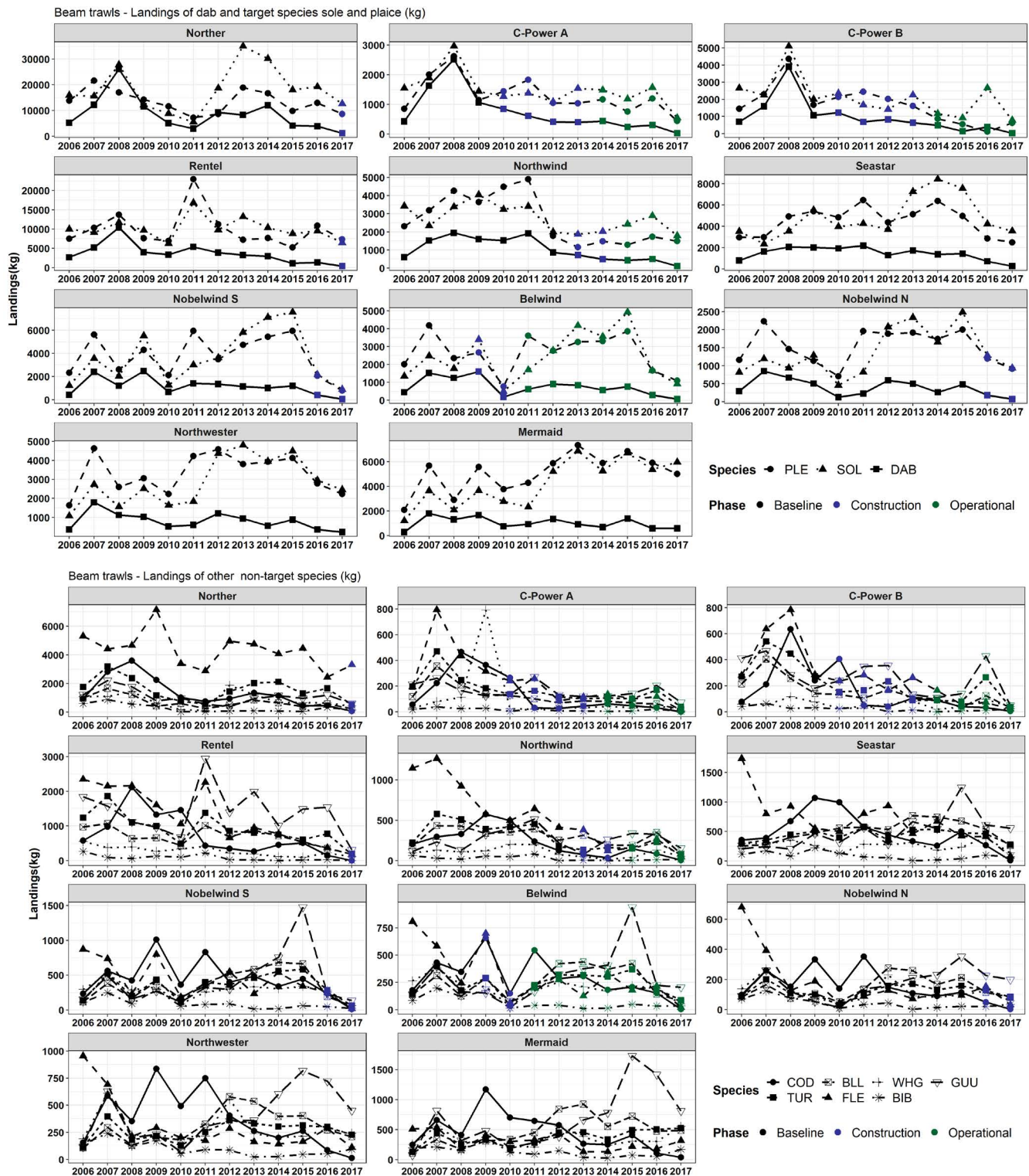
Figure 5. Time series of landings of top 10 species of the Dutch and Belgian beam trawlers for the different OWF concession areas in the BPNS. Top: landings of dab and target species, plaice and sole; bottom: non-target species.

! Nuanced interpretation of trends is needed: landings during construction and operational phase (where fishing is forbidden), is due to grid cells overlapping the edges of concessions, as such the landings of fishing activities along the edge of the concession are allocated to the concession.
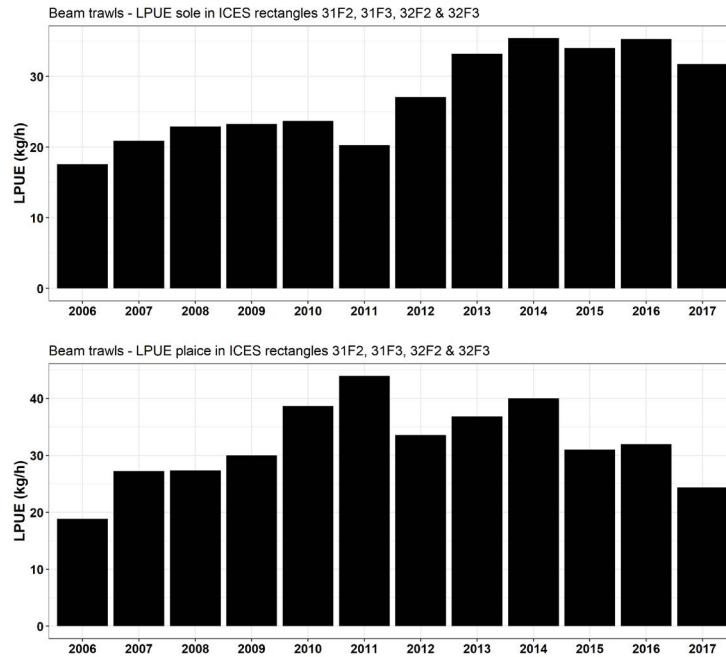
Figure 6. Time series of LPUE for sole (top) and plaice (bottom) of Dutch and Belgian beam trawlers in ICES rectangles 31F2, 31F3, 32F2, 32F3.

For the OWF area, trends in total landings for the top 10 species are highly correlated with fishing effort (r > 0.8 for all concessions, except for Mermaid r = 0.72), so similarly as for fishing effort a drop in the landings is observed for all species during construction and remains lower once operational (except for Belwind) (fig. 5). The target species sole and plaice are caught in the highest volumes in all concession areas. Especially in the earlier years and the areas closer to shore (Norther, Rentel and C-Power), dab is caught in higher volumes, but shows a decreasing trend, just as in the wider ICES area (fig. 5). Also the other species show very similar trends in the OWF concession areas (especially in OWFs without turbines) compared to the general ICES area trend: increased landings of sole after 2012 and of gurnards in later years, decreased landings for flounder and stable for brill, turbot, bib and whiting (fig. 5). Only for plaice, trends in landings in most OWF concessions seem to deviate a bit from the wider general trend, not really showing a decrease in landings after 2011, as seen in the general trend (fig. 5).

3.1.3. Landings per unit effort for sole and plaice

Landings per unit effort (LPUE) of the target species sole and plaice over the years are shown in fig. 6. A remarkable increase (almost 50%) for sole LPUE is visible from 2012 onwards going from 20-25 kg/h to 30-35 kg/h indicating increased catch rates. LPUE for plaice shows more yearly variation and is on average 32 (± SD 7) kg/h (fig. 6). LPUE of plaice follows more or less the landing trend, increasing towards 2011 and decreasing again in the later years (fig. 6).

For LPUE of sole within the concession areas, we found an increase in LPUE from 2012 onwards similar to the selected ICES rectangles surrounding the OWFs. Construction of wind turbines or presence of an operational wind farm do not affect the catch rate of sole in the vicinity (fig. 7). For LPUE of plaice, catch rates are highest in the wind farms furthest offshore compared to the more coastal ones (avg. LPUE of plaice in Mermaid is 33 [± SD 7] kg/h while in Norther it is 16 [± SD 4] kg/h). The time trend in LPUE of plaice shows some

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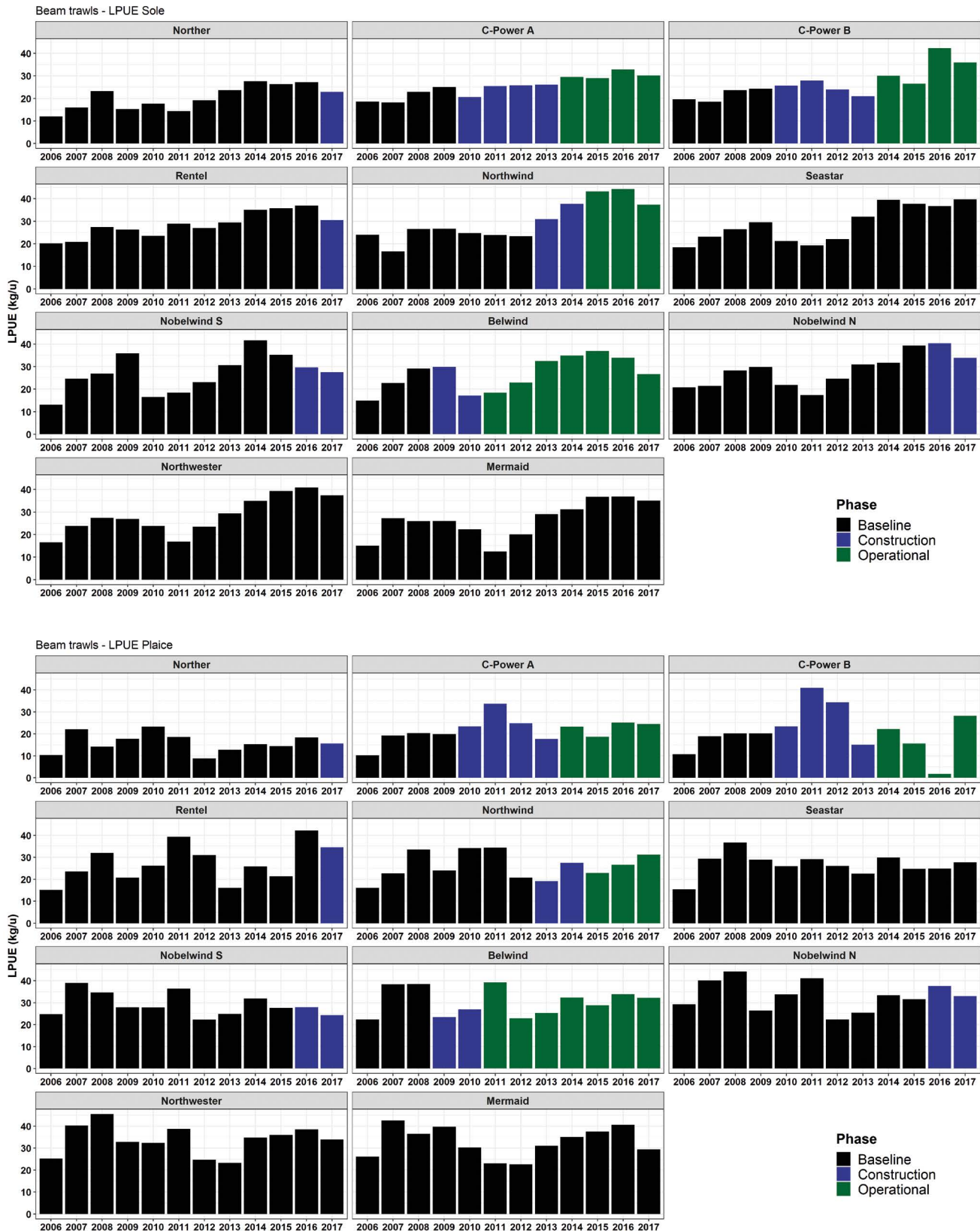
Figure 7. Time series of LPUE for sole (top) and plaice (bottom) of Dutch and Belgian beam trawlers for the different OWF concession areas in the BPNS with indication of baseline, construction and operational phase. ! Nuanced interpretation of trends is needed: LPUE during construction and operational phase (where fishing is forbidden), is due to grid cells overlapping the edges of concessions, as such values of LPUE just outside the concession are allocated to the concession.

Table 3. Overview of registered intrusions by commercial fishing vessels for C-Power, Northwind, Belwind and Nobelwind concessions

Wind farm	2012	2013	2014	2015	2016	2017
C-Power	12 intrusion by fishing vessels registered over entire period					
Northwind			0	0	7	6
Belwind & Nobelwind	3	6	0	3	14	17

year-to-year variation within most concession areas, and it seems to deviate a bit from the general trend, not really showing a decrease in the later years (fig. 7).

3.2. Registered intrusions of fishing vessels inside OWFs

Based on the intrusion reports received from Parkwind and the information provided by C-Power, we have an idea of the violations of fishing vessels within or inside the safety perimeter of the 4 operational concession areas with the longest history in Belgian waters (C-Power, Northwind, Belwind and Nobelwind). Table 3 presents the intrusions by commercial fishing vessels inside the "no-shipping" area of 3 operational concession areas. Some fishing vessels ignored the no-shipping area, and intruded concessions or their safety perimeter. The data suggests intrusions have increased in the later years.
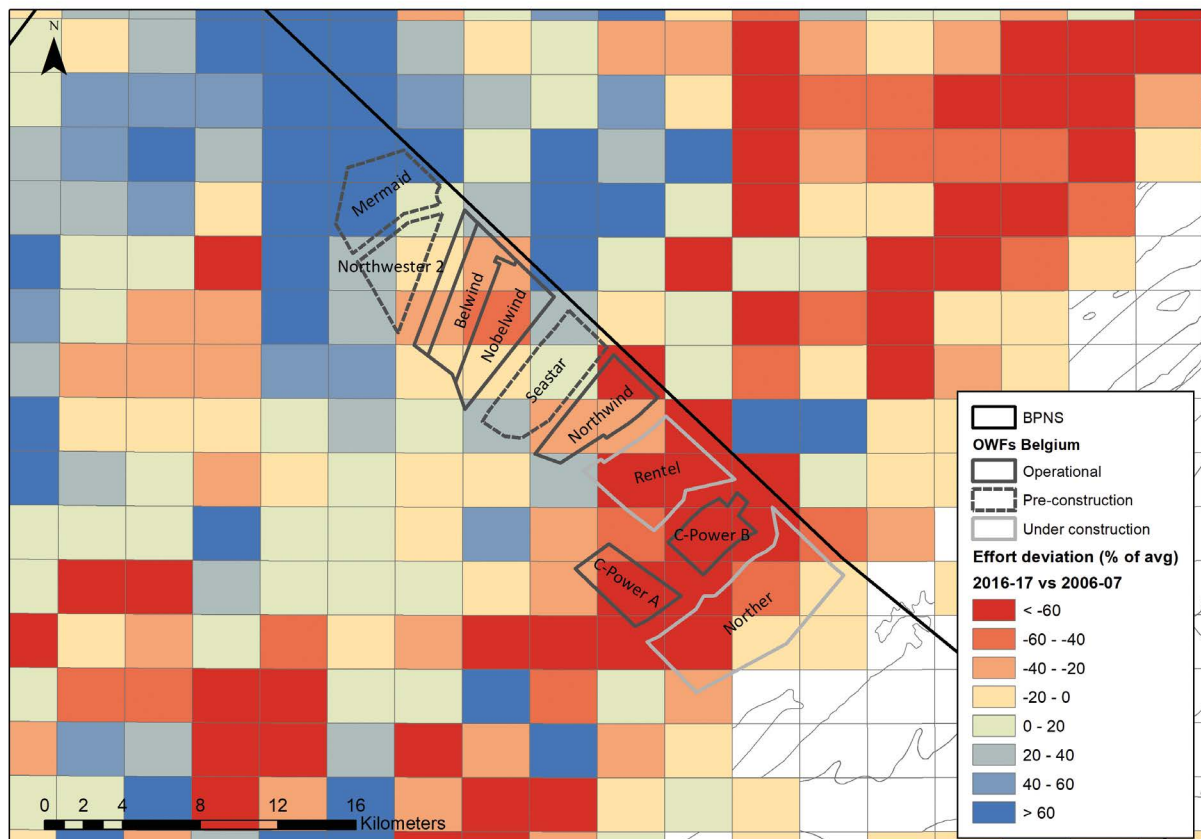
Figure 8. Spatial changes in fishing effort based on the deviation in proportional effort for Dutch and Belgian large beam trawls in 2016-2017 compared to 2006-2007.
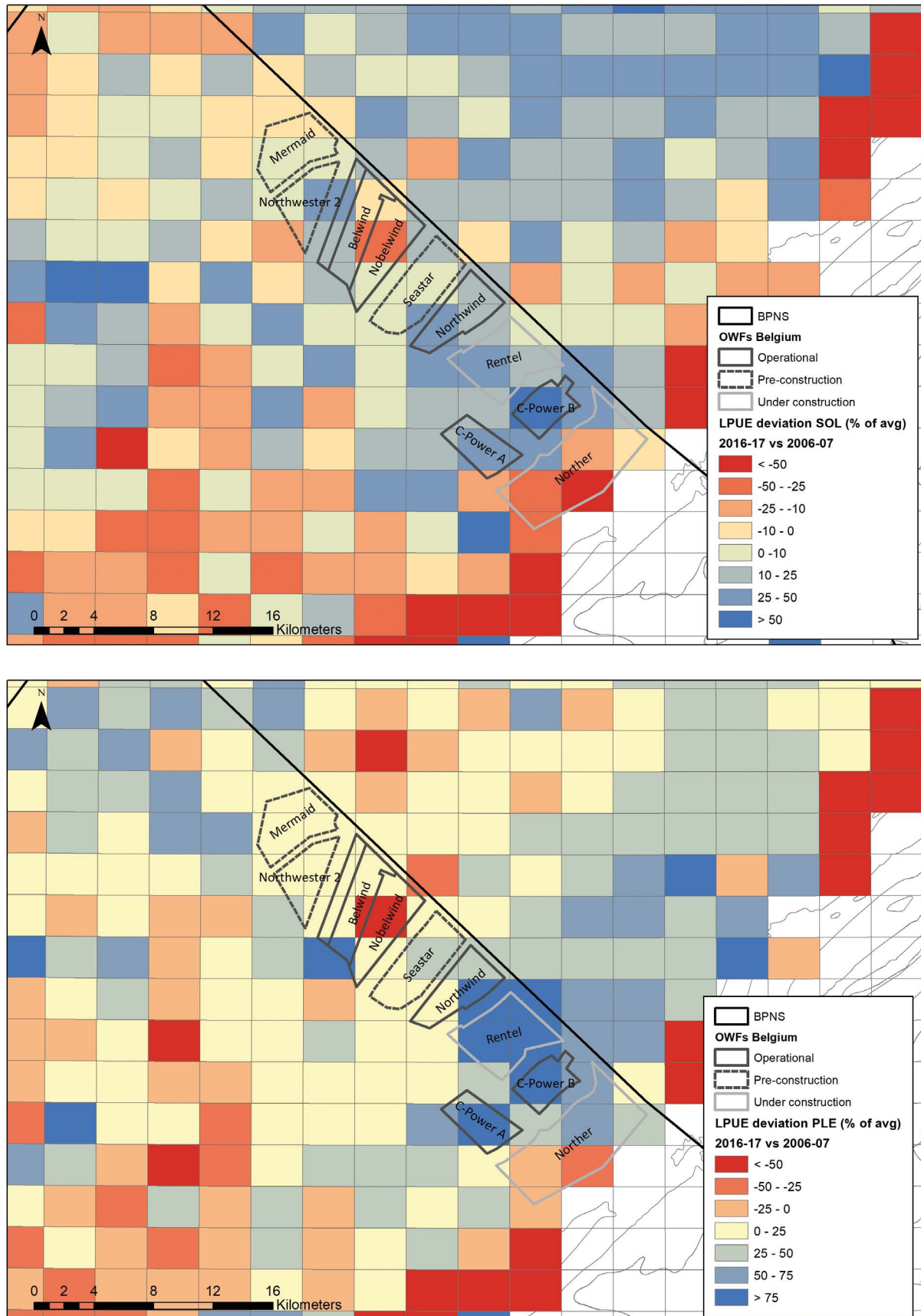
Figure 9. Spatial changes in LPUE of sole (top) and plaice (bottom) based on the deviation in proportional LPUE for Dutch and Belgian large beam trawls in 2016-2017 compared to 2006-2007.

This, however, may be considered an artefact because most likely intrusions might have been missed in the earlier years in absence of proper AIS tools (pers. comm. Kristof Verlinden).

3.3. Spatial changes in fishing effort and LPUE of sole and plaice

Within the grid cells covering the operational wind farms C-Power, Northwind, Nobelwind and Belwind, a large reduction in proportional fishing effort (up to more than 60%) was observed when comparing the period 2006-2007 with 2016-2017. However, a similar trend was also observed in the wind farms Norther and Rentel, where construction only started in 2017 (fig. 8). In fact, a general shift towards more offshore fishing activity away from the 12 Nmile line within the selected ICES rectangles can be observed (fig. 8). Around the operational OWFs situated further offshore (Belwind, Nobelwind and also Northwind), proportional fishing effort increased slightly, especially towards the NE for Belwind and Nobelwind, and SW for Northwind, which suggests that fishermen might be attracted to the edges of these more offshore wind farms. On the other hand, fishing effort in general increased in these more offshore areas as well by over 60%, as seen in the wider area north of Belwind (fig. 8). The proportional effort distribution around the wind farm area thus more or less mirrored the fishing effort in the wider area.

Spatial changes in proportional LPUE of sole do not indicate a clear wind farm effect. Proportional increases in catch rate of sole are observed around the wind farms, especially around C-Power, Rentel and Northwind when comparing the period 2006-2007 with 2016-2017. However, similar increases in proportional sole LPUE are seen in the wider area to the NE (fig. 9). For plaice however, a remarkable increase of more than 75% in proportional LPUE was observed in C-Power and Rentel and SW of Belwind, indicating an increased catch rate and a deviation of the general proportional

trend around these wind farms, where lower increases are observed (fig. 9).

4. Discussion and conclusions

This chapter provides an overview of the spatio-temporal fishing patterns of the Belgian and Dutch beam trawl fleets (both small = < 221 kW and large > 221 kW engine power) in and around the Belgian OWF area over the period 2006-2017. To our knowledge, this is the first study investigating the effect of OWF concessions closed for shipping, on fishing activity.

An important thing to note is that the data seem to imply that fishing is still ongoing in operational OWFs and OWFs under construction, although this is actually forbidden. Some violations have indeed been registered during the studied period. However, the observed fishing activity inside active OWFs is mainly a consequence of the necessary aggregation of VMS and landings data on grid cell level (both due to the low frequency of VMS pings and due to confidentiality reasons). Grid cells are not belonging entirely to a concession area, which makes that fishing activity along the edge of an OWF may be counted for (on a percentage basis) as being located inside the concession zone. This forms a potential drawback on the results and edge effects might be missed in this way, but it does provide clear insights in the "whether or not" overall avoidance of or attraction to the immediate vicinity of the active concession areas. In fact, the observed fishing activity within the operational concessions partly represents the edge effects. Despite the aggregation at grid cell level, the VMS and landing data do give a good idea on the trends in the OWFs and the surrounding area, indicating that beam trawlers are still fishing very close to the edges of the OWFs.

Apart from the expected and observed decreases in fishing effort in the Belgian OWF concession areas once construction of the turbines has started, we noted

mostly a "business as usual" in both fishing effort and landings for the Belgian and Dutch beam trawl fleets in the four selected ICES rectangles in the southern North Sea. No clear avoidance nor attraction towards the onshore operational wind farm edges could be noticed. Though this could be an artefact of working with aggregated grid cell making it harder to reveal edge effects. Nevertheless, for the more offshore situated wind farms, the deviation maps (comparing the pre-turbine with the post-turbine period) seem to suggest that fishermen are slightly attracted to the edges. Furthermore, we found an indication that during construction of a certain OWF also the immediate surroundings are avoided by fishermen; for example, a decreased fishing effort was observed in the Seastar concession (bordering Nobelwind) during the construction of Nobelwind.

Catch rate of the target species sole (expressed as LPUE) showed the same time trends as in the wider ICES area, indicating that the presence of the Belgian wind farms did not affect "efficiency" of the beam trawl fleet in catching soles. On the other hand, we found an indication of increased catch rates for plaice in the presence of OWFs, C-Power and Rentel and also to the SW of Belwind. This is supported by the monitoring results investigating potential effects of OWFs on demersal fish, where increased densities of plaice have been found inside OWFs compared to the reference areas (De Backer & Hostens 2017). More in depth research, on potential attraction of plaice towards wind farms is needed to confirm these results.

During the studied time period (2006-2017), some notable changes in beam trawl fishing in general took place, which may obscure potential real wind farm effects. Between 2011 and 2017, 80 pulse trawl vessels were introduced in the Dutch beam trawl fleet (largely replacing the original beam trawlers). This caused a displacement towards the English Channel, but also further offshore the Dutch and Belgian coasts (Turenhout *et al.* 2016). Furthermore, pulse

trawlers focus more on sole, and are clearly more efficient in catching this species (Turenhout *et al.* 2016). This effect was picked up as an increase in LPUE for sole since 2012, both in the OWF area and the wider ICES area, indicating that OWFs as such do not (yet) affect sole catches. Distribution of fishing activity is also dependent on catch composition, available quotas, fish prices and oil prices. High oil prices make that fishermen stay closer to port (Bastardie *et al.* 2010; Poos *et al.* 2013). During our study period, oil prices raised rapidly from 2006 to a peak in 2008 but decreased fast again afterwards. We noted a tendency in the distribution data that fishermen were fishing further offshore in the later years. Especially for the small beam trawl segment, this pattern was very prominent: in the years 2006 to 2009, they all stayed within the 12 Nmile zone, with a shift more offshore in the later years. Based on the overall distribution and deviation maps, a similar but smaller offshore shift may be observed in the later years for the larger beam trawlers as well.

When filtering out the general trends in fishing effort and landings in the deviation maps, our results indicate that the active Belgian OWF area (now ca. 140 km² operational not yet contiguous) only subtly changed the fishing activity of Belgian and Dutch beam trawlers (including the pulse trawlers). With the current design and size of the operational OWF area, fishermen seem to have adapted to the new situation, and are certainly not avoiding the areas around the OWFs, they even seem to be attracted to the edges. They keep fishing in the surroundings of the OWFs, still with comparable LPUE, at least for the two main target species sole and plaice. For plaice, LPUE seems even higher around some operational wind farms. As such, the relatively small loss of potential fishing grounds did not yet result in a real decrease of catches in the region or thus of the potential profit of the fishermen. However, this conclusion can change when more OWFs will be operational in the North Sea

and larger "connected" areas will be excluded for fisheries. Also, the possible effect of increased nursery or feeding grounds for the targeted commercial fish species, especially plaice, potentially offered by the OWFs still needs to be further investigated.

Acknowledgements

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CHAPTER 5

EVALUATION OF TURBINE–RELATED IMPACTS ON MACROBENTHIC COMMUNITIES WITHIN TWO OFFSHORE WIND FARMS DURING THE OPERATIONAL PHASE

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Abstract

Two consecutive years of monitoring at different distances (far: 350-500 m *vs*. very close: 37.5 m) within the offshore wind farms at Thornton Bank (TB; jacket foundations) and Bligh Bank (BB; monopile foundations) revealed consistent turbine-related impacts on the surrounding sediments and macrobenthic communities. Sediment fining was only observed around the jacket foundations at TB, while no conclusive results were found in terms of organic enrichment. General trends of benthic responses included higher densities and diversity (species richness) in closer vicinity of the turbines, but effects were most pronounced at TB. Community composition differed consistently at both banks and several recurring species were responsible for between-group (far *vs*. very close samples) differences during both years. Macrobenthic assemblages closer to the turbines showed similarities with communities that are associated with lower-energy environments. An in-depth community analysis was used to describe the typical within-group assemblages, but also revealed some small-scale variation in terms of densities, richness and community composition. At last, the recurrent trend of more pronounced results at TB confirm the hypothesis that impacts are site-specific (dispersive capacities, turbine-type) and can vary over different spatial scales, which highlights the importance of a targeted monitoring at the three different turbine types (gravity-based, jackets and monopiles) found in the Belgian part of the North Sea.

1. Introduction

Considering the 2020 Belgian targets for renewable energy, there has been an expansion of offshore wind farms (OWFs) and licences in the Belgian part of the North Sea (BPNS) since 2008 (Rumes *et al.* 2017). Several projects are currently at different stages of development (planned construction, under construction or operational), but it is expected that by 2020 a total of nine OWFs will be operational within the eastern area (Rumes *et al.* 2018). In addition, the government aims to double the capacity of electricity outputs from wind energy and has planned to designate a new concession area after 2020 (Rumes & Brabant 2018).

Because every stage (pre-construction, construction, operational and decommissioning phase) of development has the potential to influence the surrounding macrobenthic communities, consistent monitoring remains of high importance (Gill *et al.* 2018). Specific impacts will, however, depend on the local habitat type and community characteristics in which the OWFs are constructed (Byers *et al.* 2013; Gill *et al.* 2018).
The OWFs under study (C-power and Belwind) are situated offshore and the naturally occurring habitats can be categorised as high-energy environments. The seafloor within these areas is usually devoid of topographic structures with mobile, medium-coarse sediments and low organic matter content (Van Hoey *et al.* 2004; Byers *et al.* 2013). Macrobenthic communities within these rather homogenous soft sediments are mainly dominated by relatively "poor" assemblages (densities and diversity) with high resilience such as the *Nephtys cirrosa* assemblage (Van Hoey *et al.* 2004; Reubens *et al.* 2009; Coates *et al.* 2014; Breine *et al.* 2018). However, a recent study by Breine *et al.* (2018) revealed that assemblages with moderate densities and high diversity (*Hesionura elongata* community) are also found in this dynamic area. The permanent presence of the turbines during the operational phase results in a modification of the habitat, by means of altered local environmental conditions (hydrology, sediment type, water column stratification) and infaunal community structures (De Backer *et al.* 2014; Maar *et al.* 2009; Danheim *et al.* 2019; Gill *et al.* 2018; Coates *et al.* 2014). In addition, these vertical structures provide surface area for colonising epifouling communities, which in turn might intensify these changes by influencing particle and organic matter fluxes and local biodiversity (De De Backer *et al.* 2014; Maar *et al.* 2009).

These predictions, together with the results from a targeted study around one gravity-based foundation at TB, led to the

hypothesis that in closer vicinity of the turbines, sediment fining and organic matter enrichment could result in a shift towards richer macrobenthic assemblages that are associated with finer sediments (Coates *et al.* 2014; Wilding *et al.* 2012). Testing this hypothesis was implemented within the large scale monitoring from 2015 onwards by sampling at two distances (close: 50 m; far: 350-500 m) from the turbines to allow a spatial comparison within the OWFs under study. Findings from the first years of monitoring (2015-2016) did, however, not coincide with the expected results. It was proposed that effects could be restricted to distances closer (< 50 m) to the turbines and that impacts might differ between turbine types (gravity-based, jacket, monopiles). Consequently, the sampling strategy was adjusted by comparing far (350-500 m) to very close (37.5 m) locations, while also taking into account differences between turbine foundations.

Results from the 2017 monitoring campaign indeed revealed sediment fining, organic enrichment and changes in macrobenthic communities (higher densities, diversity and different compositions) at very close distances around the jacket-based foundations at TB (Lefaible *et al.* 2018). Impacts were less pronounced around the monopiles at BB, where only a difference in community composition was detected between both distances from the turbines (Lefaible *et al.* 2018). It was suggested that these contrasting results could be related to a combination of the site's flushing potential (dispersive capacity) and structural differences between foundation types and the associated "epifouling capacity" (Lefaible *et al.* 2018). While these findings confirm the "positive" effects of turbine presence on nearby local macrobenthic communities, it also highlights that these impacts appear to be site-specific and can vary over different spatial scales, which partially explains the discrepancy that is found in current literature in terms of effects of OWFs on benthos (Dannheim *et al.*

2019; Jak & Glorius 2017). Moreover, a recurrent finding is that within the same wind park, there appears to be a high inter-turbine variability both in terms of epifouling communities and infaunal benthic communities (Jak & Glorius 2017). As a result, describing general conclusions on turbine-related impacts remains a challenge within ongoing monitoring programs.

During the 2018 campaign, a similar sampling strategy as in 2017 (far samples: 350-500 m *vs*. very close samples: 37.5 m) was applied and results within this report were used to assess the strength and consistency of the distance-based differences that were observed in 2017. In addition, an in-depth community analysis was performed to determine the small-scale variability within the communities living very close to and far from the turbine foundations.

2. Material and methods

2.1. Study area

Sampling was conducted in the concession area of two wind farms within the BPNS: C-Power and Belwind (fig. 1). Both parks are built offshore on naturally occurring sandbanks, but they differ in terms of distance from the coastline, timing of construction and turbine type. C-power is situated on the Thornton Bank (TB) at approximately 30 km from the coastline. The park became fully operational in 2013 and is
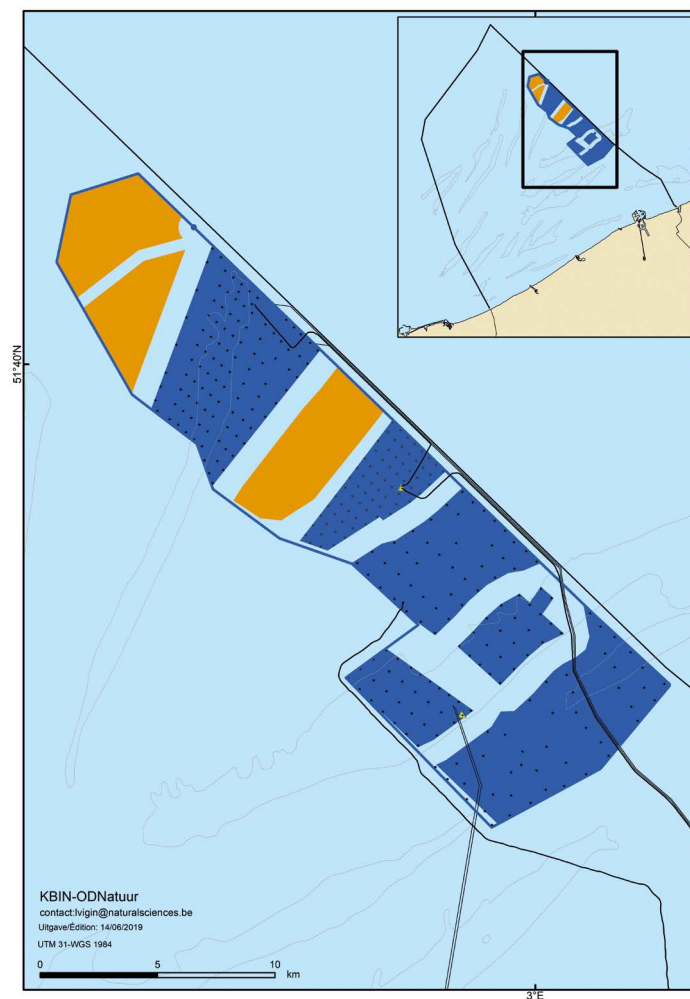
Figure 1. Wind farm concession area in the Belgian part of the North Sea. Blue areas represent operational offshore windfarms, while orange areas are domains for which concessions have been granted and building is expected to start in 2019.

composed of six gravity-based foundations and 48 jacket foundations (Rumes *et al.* 2017). Belwind was constructed on the Bligh Bank (BB), which currently represents the most offshore wind park at 46 km from the port of Zeebrugge and consists of 55 monopile foundations which have been operational since 2010 (Rumes *et al.* 2017).

2.2. Sampling design, sample collection and treatment

The potential effects of turbine presence on macrobenthic communities were tested by conducting spatial comparisons. Therefore, samples were collected at two distances from the turbines in autumn 2018 on board the vessels Aquatrot and RV Belgica. Very close samples (TB:16, BB:15) were taken at approximately 37.5 m from the centre of the turbine, whereas far samples (TB:32, BB:31) were collected in the middle between the four wind turbines (*i.e.*, farthest possible distance), *i.e.*, at distances between 350 and 500 m from any wind turbine (fig. 2).

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 mm). 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. In addition, results from the grain size distributions were also used to determine the fine sand fraction (125-250 μm) within each sample. Total organic matter (TOM) content was calculated 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.
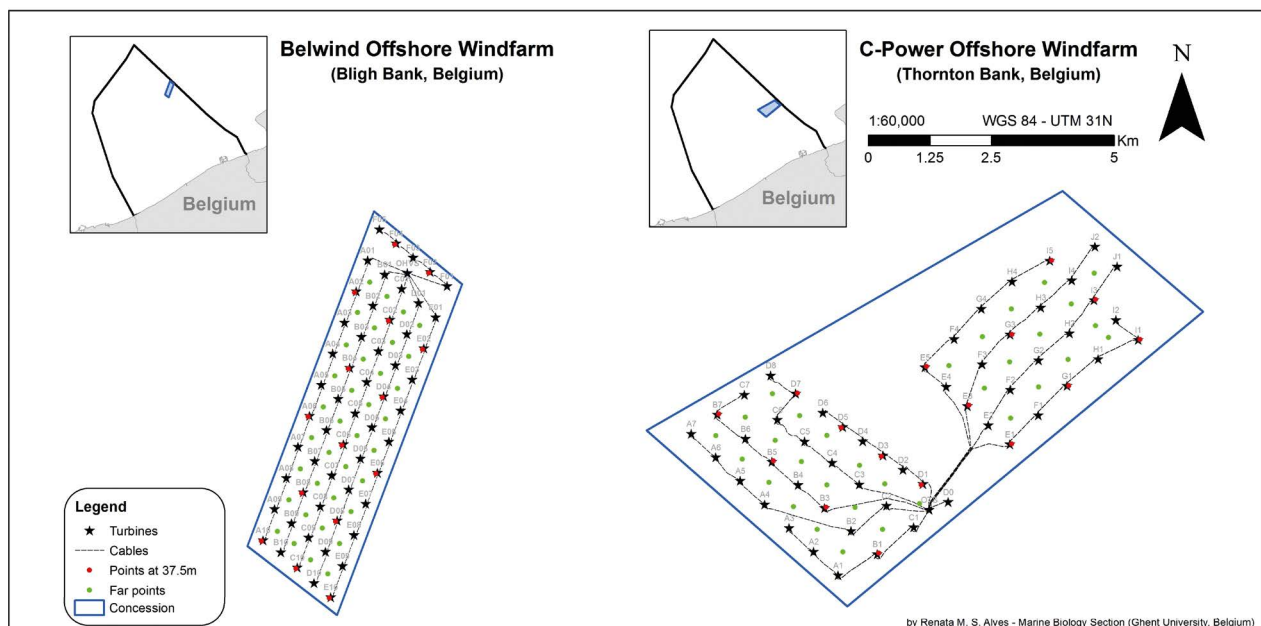
Figure 2. Overview of far and close samples at the Bligh Bank (left) and Thornton Bank (right).

2.3. Data analysis

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) in terms of the above-mentioned biological parameters and the sediment parameters MGS, fine sand fraction, fraction > 2 mm and TOM. Assumptions of normality and homogeneity of variances were tested by Shapiro-Wilk and Levene 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 *et al.* 2008), 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 MGS, the fine sand fraction, sediment fraction > 2 mm and TOM. Potential multicollinearity was verified using a Variance Inflation Factor (VIF). When the final model was found, residuals were inspected to detect outliers, which were subsequently removed from the models. Shapiro-Wilk tests were used to verify whether residuals were normally distributed.

Multivariate analysis was performed in PRIMER (version 6.1.11) with PERMANOVA add-on to investigate the potential effects of distance 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). Homogeneity of multivariate dispersions was tested using the PERMDISP routine (distances among centroids). Principal coordinates analysis (PCO) was used to visualise the data with additional vector overlay that was based on multiple correlations (Spearman correlation, R > 0.65). Similarity percentages (SIMPER) routine analysis was done to specify the contributions of individual species to the distinction between groups of samples and/or to the similarity of samples within a group (Clarke & Gorley 2006). These SIMPER results together with CLUSTER analysis were also used to describe the assemblages within each group (far and very close samples). 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, type "III" sums of squares were used for every analysis of variance test, and a significance level of $p << 0.05$ was applied. Quantitative results are expressed as mean values and corresponding standard error (mean ± SE).

3. Results

3.1. Effects of turbine presence

Quantitative results for the spatial comparison between both distances are summarised in table 1, fig. 3 and fig. 4. Sediments within both sandbanks consisted of medium sands (250-500 µm), except for two far samples at BB with very coarse sands (BB17_Far: 572 µm, BB22_Far: 715 µm). Median grain size was on average lower in the very close samples at both sites, but no significant difference with the far samples was found. The average percentage of fine sand (125-250 µm) was, however, significantly higher (table 1, $p << 0.01$) within the very close samples (21 ± 2%) compared to the far samples (15 ± 1%) at TB, whereas it was generally lower (~ 9%) and comparable for both distances at BB (table 1). Sediment

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Table 1. Overview of calculated environmental and community descriptors (mean ± SE) for the spatial comparison between both distances from a turbine in two operational wind farms at TB and BB

Univariate results	TB Very Close	TB Far	BB Very Close	BB Far
Median grain size (MGS, µm)	346 ± 10	362 ± 7	394 ± 11	410 ± 13
Fine sand fraction (125-250 µm, %)	**21 ± 2** **	**15 ± 1**	9 ± 1	9 ± 1
Total organic matter (TOM, %)	0.58 ± 0.04	0.63 ± 0.03	**0.42 ± 0.02**	**0.63 ± 0.04** ***
Sed. fraction > 2 mm (> 2 mm, %)	3 ± 0	3 ± 0	4 ± 1	5 ± 1
Total abundance (N, ind. m^{-2})	**6955 ± 5888** ***	**334 ± 42**	351 ± 86	205 ± 24
Biomass (BM, mg WW m^{-2})	**265 ± 95** **	**134 ± 54**	**34 ± 19** *	**7 ± 2**
Number of species S	**15 ± 2** **	**9 ± 1**	11 ± 2	8 ± 1
Shannon-Wiener H'	1.74 ± 0.17	1.71 ± 0.08	1.92 ± 0.15	1.76 ± 0.09
Evenness J'	**0.68 ± 0.06** **	**0.82 ± 0.02**	0.86 ± 0.02	0.90 ± 0.02
Multivariate results	**TB Very Close**	**TB Far**	**BB Very Close**	**BB Far**
Community composition		***		**
Permdisp analysis		NS		NS

Signif. codes: "***" 0.001; "**" 0.01; "*" 0.05; NS = not significant. "*" represents significant differences that were also found in 2017.
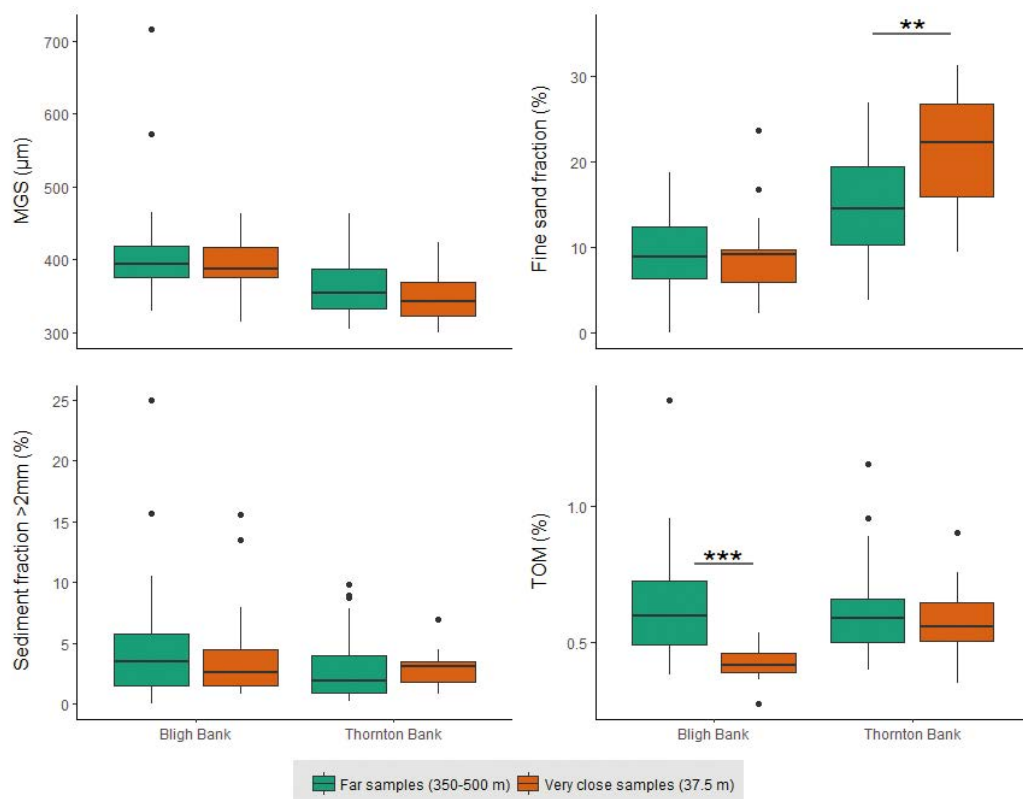
Figure 3. Overview boxplots of the abiotic variables: median grain size (MGS), fine sand fraction (125-250 µm, %), sediment fraction above 2 mm (> 2 mm) and total organic matter (TOM) per sampling site (TB and BB) for the very close and far samples. Black dots represent outliers.
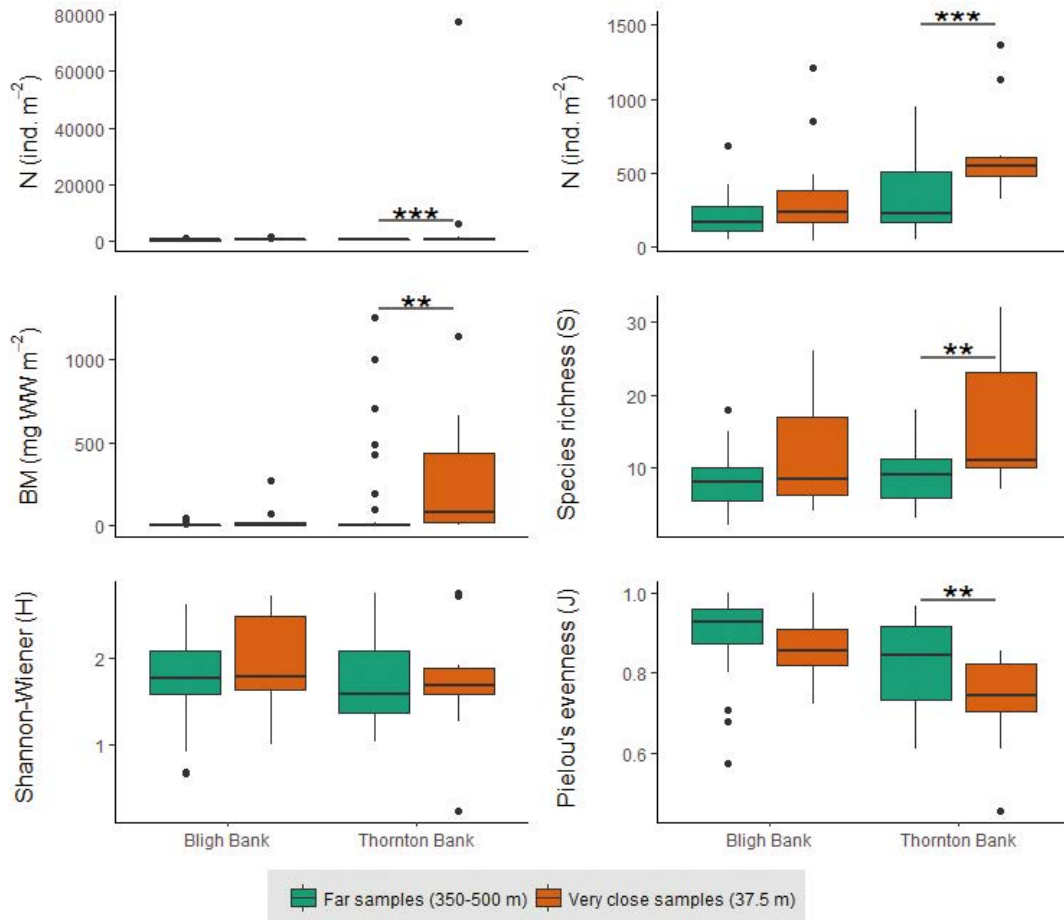
Figure 4. Overview boxplots of the biotic variables: abundance (N), abundance (N) without extreme values TB13_VC and TB16_VC, biomass (BM), Species richness (S), Shannon-wiener diversity (H'), Piélou's evenness (J') per sampling site (TB and BB) for the very close and far samples. Black dots represent outliers.

fractions > 2 mm were around 3% at TB and 4.5% at BB but within each sand bank, values proved similar at both distances from the turbines (table 1). Average total organic matter contents varied around 0.60% for all the samples at TB. A similar average value was found for the far samples ($0.63 \pm 0.04\%$) at BB, while the average TOM was only $0.42 \pm 0.02\%$ at very close samples at BB and this difference proved to be significant (table 1; $p << 0.001$).

At TB, samples closer to the turbines displayed significantly higher macrobenthic densities and biomass (1 way-Anova, $p << 0.001$ and $p << 0.05$). Two very close samples showed extremely high total densities (TB13_VC: 6020 ind.m⁻², TB16_VC: 77430 ind.m⁻²) due to the dominant presence

of the amphipod *Monocorophium acherusicum*. When these samples were removed from the analysis, macrobenthic densities remained significantly higher within very close samples. In terms of diversity indices, species richness and Shannon-Wiener diversity tended to be higher at the very close samples, along with a lower evenness closer to the turbines. Except for the Shannon-Wiener diversity, all these differences proved to be significant (table 1). Multiple regression showed that only certain granulometric descriptors (MGS, fine sand fraction and > 2 mm fraction) were significant predictors for the univariate macrobenthic community descriptors at TB, while TOM was never included in the best models. The fine sand fraction was the only significant

predictor (R²adj = 0.10) for macrobenthic densities. In terms of macrobentic biomass, a model containing only the > 2 mm fraction explained about 8% of the variation (R²adj = 0.08). Both MGS and the fine sand fraction were significant predictors for species richness (R²adj = 0.42), while Piélou's evenness was only explained by the fine sand fraction and the > 2 mm fraction (R²adj = 0.19).

At BB, density, biomass and diversity indices (S, H' and J') exhibited similar trends with distance from the turbine, but only macrobenthic biomass differed significantly between both distances. Very close samples had a 5 times higher average biomass (34 ± 19 mg WW m-²) compared to the far samples (7 ± 2 mg WW m-²). This difference was attributed to the high biomass of *Echinocardium cordatum* in the samples BBVC_01, BBVC_12 and BBVC_14, a species that was only found at very close distances. When these samples were removed from the dataset, average biomass was no longer significantly different between distances.

Macrobenthic community structure differed significantly between distances at both banks (table 1, fig. 5), but these

differences were more pronounced at TB (Permanova, p < 0.001, fig. 5) compared to BB (Permanova, p < 0.01, fig. 5). Permdisp tests were not significant, indicating true turbine distance effects. SIMPER analysis showed that within TB, the average dissimilarity between both distances was 62.54%. *Urothoe brevicornis* (6.67%), *Spiophanes bombyx* (5.48%), *Monocorophium acherusicum* (4.56%), *Ophiura* juv. (4.35%) and *Nemertea* sp. (4.31%) together contributed more than 25% to this dissimilarity and all these species were more abundant in the very close samples. Many other species contributed to a lesser extent (table 2). Within BB, the average dissimilarity between both distances was slightly higher (65.27%). *Bathyporeia elegans* (6.27%), *Nemertea* sp. (6.07%), *Urothoe brevicornis* (5.94%) and *Ophelia borealis* (5.38%) together contributed almost 25% of this dissimilarity. All these species, except for *Ophelia borealis*, were more abundant in the very close samples. Comparable to the results at TB, many other species contributed to a lesser extent (table 2). A DistLM was carried out to investigate the relationship between the macrobenthic community and the environmental variables. At the TB, MGS, the fine sand fraction and TOM had a significant
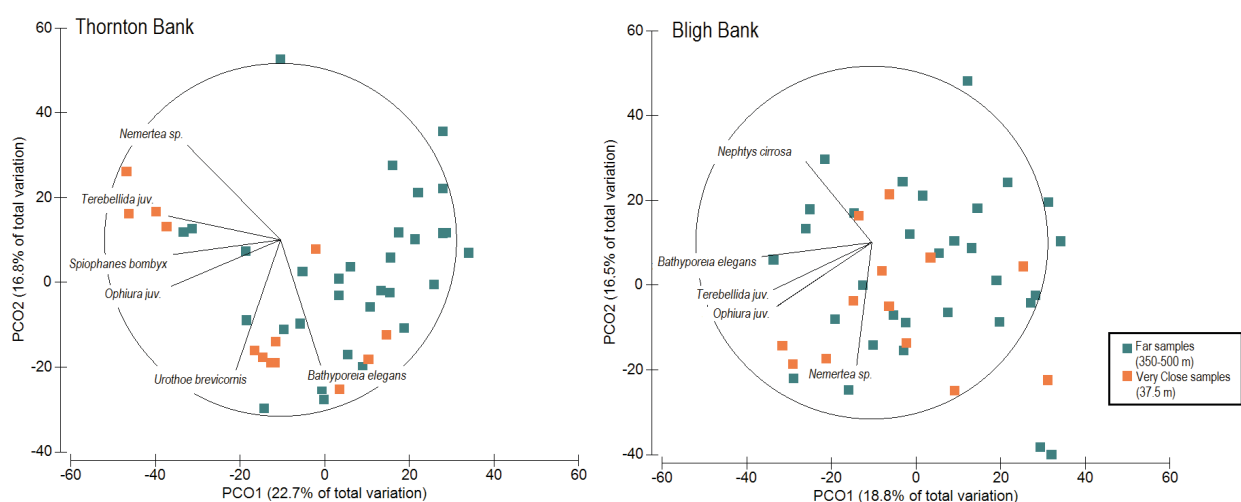
Figure 5. PCO (Principal coordinates analysis) plots based on Bray-Curtis resemblance matrix of fourth root transformed macrobenthic density data at two sandbanks (TB and BB) at two distances from the turbines (very close – far). Vector overlay was based on Pearson correlations (> 0.65).

Table 2. 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 (TB and BB)

Thornton Bank	Group Far	Group Very close	Average dissimilarity between groups 62.54%	
Species	Avg. abundance	Avg. abundance	Contribution (%)	Cumulative contr. (%)
Urothoe brevicornis	2.11	3.38	6.67	6.67
Spiophanes bombyx	0.72	2.33	5.48	12.15
Monocorophium acherusicum	0.41	2.26	4.56	16.72
Ophiura juv.	0.79	1.64	4.35	21.06
Nemertea sp.	0.98	1.60	4.31	25.37
Bathyporeia elegans	1.99	1.37	4.18	29.55
Lanice conchilega	0.41	1.30	3.73	33.28
Terebellida juv.	0.37	1.39	3.48	36.76
Nephtys juv.	2.16	2.37	3.06	39.83
Spisula sp.	0.35	0.74	2.97	42.80
Echinocardium cordatum	0.38	0.96	2.95	45.75
Bathyporeia guilliamsoniana	0.48	0.57	2.61	48.35
Spio sp.	0.76	0.27	2.52	**50.88**

Bligh Bank	Group Far	Group Very close	Average dissimilarity between groups 65.27%	
Species	Avg. abundance	Avg. abundance	Contribution (%)	Cumulative contr. (%)
Bathyporeia elegans	1.51	1.70	6.27	6.27
Nemertea sp.	1.04	1.92	6.07	12.34
Urothoe brevicornis	0.16	1.55	5.94	18.27
Ophelia borealis	1.78	1.29	5.38	23.65
Glycera sp.	0.88	1.14	4.62	28.27
Nephtys cirrosa	1.79	1.79	4.58	32.85
Terebellida juv.	0.55	1.19	4.42	37.27
Nephtys juv.	2.05	2.08	4.09	41.36
Spisula juv.	0.95	0.29	3.80	45.16
Spio sp.	0.54	0.25	2.90	48.07
Echinocyamus pusillus	0.39	0.60	2.84	**50.91**

relationship with the multivariate data structure and together explained about 16% of the total variation. All abiotic variables (MGS, fine sand fraction, TOM and > 2 mm fraction) together explained 13% of the total variation in the macrobenthic community structure of BB.

3.2. In-depth analysis of community composition

Multivariate analysis of the macrobenthic community structure revealed significant differences between distances at both banks, but high residual variances and trends on the PCO (fig. 5) also suggest some variability within groups (far and very close samples). Therefore, we conduct a more in-depth analysis of the small-scale variability within the communities living very close to, and far from the turbine foundations. Results from the CLUSTER and SIMPER analysis revealed that for both banks, a typical assemblage could be identified within each distance group (table 3). These assemblage descriptions were established by identifying the truly characterising species (~ 65% of total densities) and the supporting species that also contributed a significant part to the overall assemblage composition. While the communities in most of the samples were similar to these typical assemblages, certain variations and distinctive assemblages were also found within each group.

3.2.1. TB (C-power)

Communities within the far group at TB had an average similarity of 40.62% and most of the samples were similar to the described typical assemblage, which can be regarded as a polychaete-amphipod dominated community. Within these samples, there was, however, a separation between poorer and richer assemblages. Poorer samples showed lower average densities for the truly characterising species and were devoid of some of the supporting species resulting in lower total densities and richness, while richer samples had occurrences (1-2 individuals) of some extra species in addition to the typical assemblage.

Four out of 32 far samples (TB04_Far, TB05_Far, TB22_Far, TB23_Far) were distinct from most of the samples due to their low species richness (S = 4) and low densities (N = 50-70 ind. m^{-2}). In addition, these "impoverished" assemblages usually lacked one or more of the truly characterising species (*Nephtys cirrosa*, *Bathyporeia elegans*). TB23_Far only consisted of *Ophelia borealis*, *Glycera* sp. and *Nemertea* sp. which separated this sample clearly from the others.

Table 3. Overview of the typical assemblage compositions (within group SIMPER analysis) for each group (far and very close) at both banks (TB and BB)

Thornton Bank (C-power)	
Typical far assemblage	**Typical very close assemblage**
Characterising species: *Nephtys cirrosa* (+juv.), *Bathyporeia elegans*	Characterising species: *Urothoe brevicornis*, *Nephtys cirrosa* (+juv.), *Spiophanes bombyx*
Supporting species: *Urothoe brevicornis*, *Nemertea* sp., *Spio* sp.	Supporting species: *Bathyporeia elegans*, *Ophiura* juv., *Lanice conchilega*, *Nemertea* sp., *Terebellidae* juv.

Bligh Bank (Belwind)	
Typical far assemblage	**Typical very close assemblage**
Characterising species: *Nephtys cirrosa* (+juv.), *Ophelia borealis*	Characterising species: *Nephtys cirrosa* (+juv.), *Nemertea* sp., *Bathyporeia elegans*
Supporting species: *Bathyporeia elegans, Nemertea* sp., *Glycera* sp.	Supporting species: *Ophelia borealis, Urothoe brevicornis, Glycera* sp., *Terebellidae* juv.

In contrast, two far samples (TB09_Far and TB17_Far) showed higher densities (N = 500-950 ind. m^{-2}) and species richness (S > 15) than most of the samples and represented "enriched" typical far assemblages. While both samples had relatively high densities for *Nephtys cirrosa* and were devoid of *Bathyporeia elegans*, they revealed quite different compositions. TB09_Far had high densities of *Magelona* sp. and *Spiophanes bombyx*, while many other species contributed to a lesser extent (*Echinocardium cordatum*, *Tellimya ferruginosa* and *Urothoe brevicornis*). Within TB17_Far, high densities were found for *Monocorophium acherusicum*, *Lanice conchilega* and *Terebellidae* juv. together with the polychaetes *Eumida sanguinea* and *Spiophanes bombyx*.

Very close assemblages showed slightly higher average similarities (42.96%) compared to the far samples due to the fact that all three characterising species from the typical very close assemblage were found in almost every sample. In addition, no truly "impoverished" samples were found at very close distances, but some samples were also considered to be poorer due to lower densities of the characterising species and the absence of some supporting species. Comparable to the findings at far distances, four out of the 16 very close samples (TB04_VC, TB06_VC, TB13_VC and TB16_VC) showed very high densities (N > 1000 ind. m^{-2}) and species richness (S > 20). These "enriched" assemblages were devoid of *Bathyporeia elegans* and had occurrences of *Monocorophium acherusicum* and *Actinaria* sp., two species that were not found in any other very close samples. Especially TB13_VC and TB16_VC were dominated by the amphipods *Monocorophium acherusicum* and *Jassa hermani* while many other species contributed to a lesser extent. During the processing of both samples, a high amount of medium-large blue mussels (*Mytilus edulis*) were encountered, with associated *Actinaria* growing on the mussel shells (personal observation). These samples are therefore considered to represent hard substrate ("mussel-bed") associated communities. TB04_VC and TB06_VC had lower amounts of *Mytilus edulis* and fewer individuals of *Monocorophium acherusicum* and *Actinaria* sp. were encountered. These samples were quite comparable to the "typical" assemblages found at very close distances but were distinct due to the relatively high amounts of *Lanice conchilega* and *Terebellidae* juv. and the recurring counts of *Tellimya ferruginosa* and *Urothoe poseidonis*.

3.2.2. BB (Belwind)

Far samples at BB revealed an average similarity of 40.14%, but SIMPER and CLUSTER analysis indicated that despite small differences in terms of supporting species and sporadic counts of additional species, the vast majority of the samples were relatively similar to the typical far assemblage described for sediments around the monopiles. Comparable to the results at TB, four out of 31 samples (BB02_Far, BB05_Far, BB17_Far, BB22_Far) were clearly "impoverished" due to the low amount of species (S: 2-4 species/sample) and total densities (N: 70-120 ind. m^{-2}), and these communities were mainly composed of polychaetes. Communities at BB02_Far and BB05_Far were dominated by *Nephtys cirrosa*, while BB17_Far and BB22_Far mainly consisted of *Nemertea* sp. and *Ophelia borealis*. The latter two samples also showed the highest MGS found across both banks (572 μm, 715 μm).

Very close samples had a lower average similarity (34.26%) compared to the far samples, but the majority of the 15 samples were composed of the four characterising species with some minor differentiations for the supporting species. BB14_VC and BB15_VC proved to be "impoverished" assemblages both in terms of density (N: 40-90 ind. m^{-2}) and diversity (S: 4 species/sample) and showed a completely different composition from the typical very close

assemblage. In contrast, four "enriched" samples (BB01_VC, BB02_VC, BB08_VC, BB13_VC) displayed high densities (N: 380-1210 ind. m^{-2}) and diversity (S > 20 species/sample). Besides the truly characterising species, most of the supporting species were also found and especially *Terebellidae* juv. were encountered frequently within these samples. Additional recurrent species included: *Monocorophium acherusicum*, *Ophiura* juv., *Echinocyamus pusillus*, *Aora typica*, *Spisula* sp. and *Spiophanes bombyx*.

4. Discussion

Changes in soft sediment macrobenthic communities during the operational phase of OWFs are a result of complex interactions between the abiotic and biotic components that are being affected (Dannheim *et al.* 2019). In addition, feedback-loops between both components and the fact that effects might be restricted to different spatial scales, make it difficult to understand the true cause-effect relationships that drive these changes (Dannheim *et al.* 2019). In particular, the influence of the turbine-associated epifouling communities on the surrounding sediment composition and macrobenthic communities remains challenging to quantify within the current monitoring program. To address these impediments, findings from two consecutive years of monitoring are being used to describe general patterns that were observed and to assess certain hypothesis-based questions related to turbine presence.

4.1. Turbine-related impacts on habitat characteristics

Increasing evidence is emerging that the continual presence of wind turbines in naturally homogeneous soft sediments can affect local abiotic components and processes (Coates *et al.* 2014; Lefaible *et al.* 2018; Dannheim *et al.* 2019). The introduction of vertical structures in these well-flushed environments changes local hydrodynamics,

which largely determines the sediment composition around the turbines (Byers *et al.* 2004). The induced alterations of bottom currents and sedimentation rates might allow the creation of sheltered areas and the deposition of finer particles in the wake of the turbine (Leonard & Pedersen 2005). Consequently, the sediment refinement and associated decrease in permeability will facilitate the retention of deposited organic matter (De Backer *et al.* 2014), leading to a potential shift from high-energy areas with coarser sediments and low organic matter to lower-energy areas with the accumulation of fine sediment and higher organic content (Leonard & Pedersen 2005; Byers *et al.* 2004).

In 2017, refinement and increased food availability were clearly found around the jacket foundations at TB (Lefaible *et al.* 2018). Samples very close to the turbines displayed lower average median grain size, a higher fine sand fraction (125-250 μm) and higher organic matter content (Lefaible *et al.* 2018). While similar trends were found in 2018 for the median grain size and the fine sand fraction, only the latter proved to be significantly different between both distances in both years. Within the far samples at TB, a decrease in average MGS was observed compared to 2017, and this was also accompanied by an increase in average TOM leading to more similar values between distances in 2018. Comparable to last year's results, turbine-related changes in sedimentology (grain size distributions) were not observed around the monopiles at BB, and TOM was even significantly lower at very close samples.

It can be concluded that results from the environmental parameters support the fining hypothesis to a certain degree (fine sand fraction) at TB. A potential explanation for the lack of significantly lower MGS at the very close samples in 2018 could be that around the jacket foundations, the spatial extent of refinement exceeds the distance

of 37.5 m, resulting in non-existent or only minor differences between both sampling locations. If true, a declining trend in MGS should also be observed at far samples since the installation of the turbines. However, a temporal comparison (since 2015) of the average MGS for the far samples at TB did not confirm this hypothesis. The absence of a consistent temporal trend in terms of MGS affirms that distant changes of altered current flow on particulate transport and organic enrichment might be difficult to measure, especially within dynamic environments subject to high natural variability (Wilding *et al.* 2014; Dannheim *et al.* 2019; Jak & Glorius 2017). Additionally, the complete lack of granulometric differences at BB reinforces the proposed hypothesis that impacts of artificial structures can vary over different spatial scales and might be dependent on local factors such as a site's flushing potential (dispersive capacities) and foundation type (Lefaible *et al.* 2018).

4.2. Turbine-related impacts on biodiversity and community structure

The areas in which TB and BB are located represent environments that are subject to strong physical disturbance where the seafloor typically consists of well-sorted, medium-coarse and mobile sediments (Van Hoey *et al.* 2004; Breine *et al.* 2018; Byers *et al.* 2004). Within these habitats, relatively "poor" communities such as the *Nephtys cirrosa* assemblage are usually found, which are dominated by mobile species with opportunistic life strategies (Van Hoey *et al.* 2004; Breine *et al.* 2018). Impacts from the presence of the wind turbines, such as the above-mentioned seafloor alterations, are likely to induce changes in the surrounding macrobenthic communities, which are strongly associated with local physical properties (Coates *et al.* 2014; Van Hoey *et al.* 2004).Within the BPNS, richer communities such as the *Abra alba* community, are generally found in coastal areas with fine to medium sandy (< 300 μm) sediments (Van Hoey *et al.* 2004). It was

therefore suggested that near the artificial structures, macrobenthic communities with a higher density, biomass and diversity could develop (Coates *et al.* 2014; Lefaible *et al.* 2018).

At TB, relatively similar results for the univariate biological parameters were found compared to 2017, which implies a consistent trend of higher densities (N), species richness (S') and lower evenness (J') at very close distances from the jacket foundations. Correspondingly, community composition also revealed persistent differences between both distances. The in-depth community and SIMPER analysis indicated that despite some divergent samples, most far samples within TB could be categorised as the widely occurring *Nephtys cirrosa* community. This assemblage is mainly composed of the polychaetes *Nepthys cirrosa*, *Nemertea* sp. and *Spio* sp. together with the amphipods *Urothoe brevicornis* and *Bathyporeia elegans* (Van Hoey *et al.* 2004; Breine *et al.* 2018). In general, very close samples were also composed of these species but showed higher relative abundances, especially for the species *Urothoe brevicornis* and *Nemertea* sp., whereas *Bathyporeia elegans* had lower average abundances. *Spiophanes bombyx* also proved to be a characterising species within the very close samples and consistently contributed to between-group differences. While this polychaete is commonly found within the BPNS, it also comprises a significant share of the *Abra alba* community along the Northern French and Belgian coast (Van Hoey *et al.* 2004; Van Hoey *et al.* 2005; Desroy *et al.* 2002). The occurrence of *Spiophanes bombyx* is also often positively associated with the habitat structuring polychaete *Lanice conchilega*, which has beneficial effects on local faunal abundance and richness (Rabaut *et al.* 2007; De Backer *et al.* 2014). In contrast with last year's results, *Lanice conchilega* was also frequently encountered closer to the jackets. The higher relative abundances of the sediment modifying polychaetes described

above and of species such as *Ophiura* juv. and the deep-burrowing *Echinocardium cordatum* confirm a shift towards richer assemblages that are usually found in lower-energy environments.

Trends for the univariate community descriptors were similar at BB with higher average densities, biomass, richness (S', H') and a lower evenness at very close distances from the monopiles. However, comparable to the findings in 2017, none of these differences proved to be significant. Community composition did differ between distances, but dissimilarities were less pronounced compared to the results at TB. Whereas average densities were generally lower, assemblages at very close distances from the monopiles showed relatively similar compositions as the *Nephtys cirrosa* assemblages found at TB (far samples), except for the low occurrences of *Urothoe brevicornis* and the presence of the supporting species *Ophelia borealis*. The latter seems to be an indicator species of "poorer" communities (Van Hoey *et al.* 2004) and also contributed to between-group differences with higher relative abundance at far distances. While the expected turbine-related impacts seem to be less prominent at BB, SIMPER results did show some similarities with the findings at TB, and four very close samples showed considerably higher densities and diversity. Comparable to the results at TB, species such as *Terebellidae* juv., *Spiophanes bombyx*, *Ophiura* juv. and *Echinocyamus pusillus* were frequently encountered in these assemblages. This indicates that also at BB, turbine-based enrichment of infaunal communities is taking place, but on a smaller scale.

4.3. Artificial reef-effect and potential impacts of epifouling communities

Besides the increased habitat complexity that is provided by the permanent presence of the wind turbines, the structures (and scour protection) are also rapidly colonised by hard-bottom assemblages with sessile – and

mobile fauna (Dannheim *et al.* 2019). These combined effects create so-called "artificial reefs" which are considered to increase overall biodiversity and have the potential to turn exposed soft bottom systems into rich ecosystems (De Backer *et al.* 2014; Maar *et al.* 2009; Dannheim *et al.* 2019). In addition, the epifouling communities that establish on the turbines might indirectly intensify the described turbine-related impacts on local habitat characteristics by altering organic matter fluxes to the surrounding sediments (De Backer *et al.* 2014; Maar *et al.* 2009; Coates *et al.* 2014; Dannheim *et al.* 2019). *Mytilus edulis* is known to be a rapid coloniser of newly available surface-area and constitutes an important share of epifouling biomass on wind turbines (Maar *et al.* 2009; Krone *et al.* 2013). These bivalves are strong habitat modifiers and their shells provide secondary hard substrate, thereby enhancing local spatial heterogeneity and associated biodiversity (Maar *et al.* 2009; Krone *et al.* 2014).

As stated in the beginning of the discussion section, the mechanisms by which the epifouling communities might influence their environment remain challenging to quantify within the current monitoring program. However, the "enriched" assemblages (TB04_VC, TB06_VC, TB13_VC, TB16_VC) found at very close distances at TB could provide direct evidence of their potential effects. Wind turbines and especially, jacket-like foundations (oil rigs, gas platforms) seem to form a very favourable substrate for *Mytilus edulis* colonisation (Maar *et al.* 2009; Krone *et al.* 2014; Dannheim *et al.* 2019). In all four samples, significant amounts of blue mussels or their empty shells were found and they were clearly distinct from the majority of the very close samples due their high macrofaunal abundances and species richness. In addition, finer and organically enriched sediment was found at the very close distances. While these results confirm the proposed impacts of blue mussels, it also shows that the

presence of these organisms has the potential to expand the artificial reef-effect to areas beyond (> 30 m) the construction itself.

Bivalve shells can be knocked off from the structures by heavy weather or as a result of their own weight, thereby creating additional habitats or "secondary reefs" at close distances from the turbines (Krone *et al.* 2013). Krone *et al.* (2013) already described that around the piles of an oil-rig, high accumulations of deposited shells resulted in secondary hard substrate habitats (shell mounds) with high diversity and attachment sites for sessile reefs forming organisms. The samples TB04_VC and TB06_VC seem to concur with this description as they were characterised by high densities and diversity (N > 1000, S > 20) and contained significant amounts of the tube-building polychaete *Lanice conchilega*. In addition, complexes with many connected (abyssal threads) mussels were found in the samples TB13_VC and TB16_VC. Therefore, these samples might represent mussel-bed associated communities that occur around the turbines, where densities are dominated by smaller crustaceans such as the amphipods *Monocorophium acherusicum* and *Jassa herdmani*.

It is most likely that the described habitats (shell mounds and mussel-beds) were initially introduced as "drop-offs" from the turbines and impacts are therefore expected to act within an ephemeral time scale. Nevertheless, the abundant presence of *Mytilus edulis* on the turbines (source populations) and the altered seafloor conditions (sheltered areas) might allow these types of secondary/biogenic reefs to expand over time and establish permanently within the sediments surrounding the OWFs.

5. Conclusion and future perspectives

In two consecutive years of monitoring, turbine-related impacts on habitat characteristics such as sediment refining were found, with higher fine sand fractions at very close distances (*i.e.* < 50 m) around the jacket foundations at TB. Organic enrichment was also observed around the jackets in 2017, but not in 2018. In contrast, an opposite trend of lower average organic matter content was observed at very close distances around the monopiles at BB. In terms of benthic responses, several analogies were found with last year's results. General trends include higher densities and diversity in close vicinity of the turbines, where effects seem to be most pronounced around the jackets at TB, but there are indications that a similar process is taking place around the monopiles at BB. Furthermore, community composition between distances differed consistently at both sandbanks, with several recurring species that were responsible for between-group differences. While the hypothesis of a shift towards the *Abra alba* community could not be validated, assemblages closer to the turbines did show similarities with macrobenthic communities that are associated with finer sediment and low-energy environments (Breine *et al.* 2018; Byers *et al.* 2004). The observed changes in macrobenthic assemblage structure might in turn induce alterations in terms of functioning of the local ecosystem and a study by Breine *et al.* (2018) already revealed that physical factors such as grain size (coarse *vs.* fine sediment) were responsible for differences in trait modality compositions. It is therefore suggested that in addition to the structural biological changes that were found, functional properties might also be altered within the macrobenthic communities closer to the turbines. Consequently, results from 2017 and 2018 will be used to assess potential changes in functional diversity and trait modalities within the assemblages found around the turbines.

An in-depth community analysis provided valuable results to describe typical within-group (far and very close) assemblages.

However, considerable variation was also found in terms of densities, richness and assemblage structure. While this indicates that in these mobile sands assemblages can vary on a small spatial scale, it also emphasises that inter-turbine variability should be incorporated into future statistical analysis. Additionally, some hard substrate associated assemblages were found at very close distances around the jackets at TB, which provided insights in the potential effects of epifouling communities on the surrounding infaunal macrobenthos.

Finally, the recurrent trend of more pronounced responses at TB confirms the hypothesis that impacts can be site-specific and may differ between turbine types (jackets *vs*. monopiles). These results highlight 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.

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CHAPTER 6

MONITORING IMPACTS OF OFFSHORE WIND FARMS ON HYPERBENTHOS: A FEASIBILITY STUDY

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Abstract

Despite their important role in bentho-pelagic coupling and analogies with macrobenthic communities in terms of distribution patterns and seafloor dependency, hyperbenthos is often not included in current monitoring programs. It is proposed that turbine-related habitat changes such as altered hydrodynamics and organic enrichment could create more favourable conditions for the settlement of pelagic species and attract mobile species, resulting in richer hyperbenthic communities within the offshore wind farms (OWFs). Therefore, the 2019 monitoring campaign was used to perform a feasibility study concerning sampling effort (design/strategy, processing time) and to explore the obtained data (inside *vs*. outside tracks) within two OFWs (C-power and Belwind). The unsuccessful initial sampling showed that efficient sampling strongly depends on appropriate weather conditions and local topography. Eventually, an adjusted sampling strategy (tracks parallel to sand ridges and shorter towing distance) resulted in two representative samples (reference – and impact tracks) at C-power, which were processed within a reasonable time frame. While no actual conclusions can be made from the obtained dataset, this study did reveal some results that show the relevance of including this group within future monitoring to allow a more comprehensive view of turbine-related impacts on seafloor associated communities within the Belgian part of the North Sea (BPNS).

1. Introduction

Hyperbenthos can be described as the group of animals that occur in the lower part of the water column and are dependent on their association with the seafloor (Dewicke 2001; Hamerlynck *et al.* 1991). These communities are usually divided into two groups: the holohyperbenthos, which contains animals that are permanently present, and a temporary group, the merohyperbenthos, which only spend a part of their life cycle in the hyperbenthic zone. Comparable to the infaunal macrobenthos, hyperbenthic communities are believed to play a major role in bentho-pelagic coupling (Dewicke 2001). Being characterised by good swimming capacities and vertical migrations, these species contribute to the modification and transport of organic matter within the water column (Dewicke 2001). In addition, the mobility of the hyperbenthos at the seafloor-water

Table 1. Overview of hyperbenthos sampling campaigns and sample status (NR = non-representative)

Date of sampling	Vessel	Location	Impact / reference	Track name	Sample status
22-26/10/2018	RV Belgica	BB	Impact	WBB06a	Not sampled
		BB	Reference	WBB02	Not sampled
		TB	Impact	ftTrack2	Sampled but NR
		TB	Reference	ftwT2triss	Sampled but NR
06/02/2019	Simon Stevin	TB	Impact	ftTrack2	Sampled and processed
		TB	Reference	ftHyper	Sampled and processed

column interface could influence important processes such as bioturbation and bioresuspension, thereby changing overall particle fluxes (Dewicke 2001). At last, hyperbenthic communities are considered an important food source for higher trophic levels such as juvenile demersal fish (Dewicke 2001).

Large-scale studies on the distribution of hyperbenthic communities were performed throughout the 90's and revealed two gradients in community structure which consisted of a principal onshore-offshore gradient (abundance and biomass) perpendicular to the coastline and a less pronounced east-west gradient parallel to the coastline reflecting differences in species richness and diversity (Vincx *et al.* 2004). The latter gradient seems to be most pronounced in the onshore area, with more diverse communities at the western Coastal banks compared to the eastern Coastal banks (Vincx *et al.* 2004). In general, hyperbenthic abundance and diversity were significantly lower at the offshore Hinder Banks compared to the Flemish and Zeeland Banks closer to the coast (Vincx *et al.* 2004; Dewicke 2001). These findings are in accordance with described distribution patterns of macrobenthic communities, indicating that scarcer food supply and stronger water flow at offshore locations provide a less favourable environment for bottom-dwelling animals (Vincx *et al.* 2004). Due to these analogies with macrobenthic communities in terms of

distribution patterns and seafloor dependency, it is very likely that the permanent presence of wind turbines would also affect this part of the bentho-pelagic community within the Belgian part of the North Sea (BPNS). Alteration of hydrodynamics around the turbines and the creation of sheltered areas with lower current velocities might promote the settlement of planktonic species with passive transport mechanisms that comprise a substantial portion of the hyperbenthic community (Hamerlynck & Mees 1991). In addition, the proposed organic enrichment around the turbines due to the combined effects of sediment fining and the deposition of organic matter from the epifouling communities (Lefaible *et al.* 2018) may also influence the occurrence of mobile organisms such as mysids which can actively migrate to areas with higher food availability (Hamerlynck & Mees 1991). A potential hypothesis could therefore be that wind turbines within the offshore areas create more favourable conditions for the settlement of several species and attract mobile species, resulting in richer hyperbenthic communities.

Therefore, the main objectives of this feasibility study were to i) assess the suitability of the sampled tracks for hyperbenthos research, ii) estimate the time needed to process the obtained samples, and iii) explore the abundance and diversity data of the sampled hyperbenthos inside versus outside of offshore wind farms.

2. Material and methods

The sampling of the hyperbenthos was done at two sandbanks, the Bligh Bank (BB) and the Thornton Bank (TB). The proposed sampling sites consisted of one track within each offshore wind farm (Belwind at the BB, C-power at the TB), together with a reference track. The locations of the tracks were based on existing tracks that are sampled during the epifauna research (fig. 1, table 1).

Due to adverse weather conditions, hyperbenthos sampling at BB had to be cancelled. The two remaining sites at TB were sampled through the use of a hyperbenthic sledge containing 4 nets: two pairs of nets were mounted next to each other with mesh sizes of 0.5 and 1 mm of the two superimposed (upper and lower) nets on the left and right side of the sledge. Collectors were placed at an angle of 45° and a flowmeter was installed to calculate the volume of water filtered during each sampling event. Tows were made during daytime conditions for 1 km against the current at a speed

of 1.5 knots. Sampling at TB resulted in a non-representative (NR) reference sample due to the high accumulation of sediment within the collectors and the near absence of organisms.

One potential explanation for the lack of successful sampling at the reference site could be a combination of the position of the track relative to the sandbank ridges and the long towing distance. Therefore, an alternative sampling strategy was proposed with shorter (150 m) tracks that run parallel to the ridges. During a one-day sampling cruise (06/02/2019) on board the Simon Stevin, this new sampling method was tested in C-power where two tracks were performed during daytime conditions: one impact track and one adjusted reference track just outside the offshore wind farm (fig. 1, table 1). The hyperbenthic sledge on the Simon Stevin only consists of two nets: the lower-net samples from 0.2 to 0.5 m, and the upper-net from 0.5 to 1 m above the bottom; both nets have a 1 mm mesh size. Comparable to the sledge
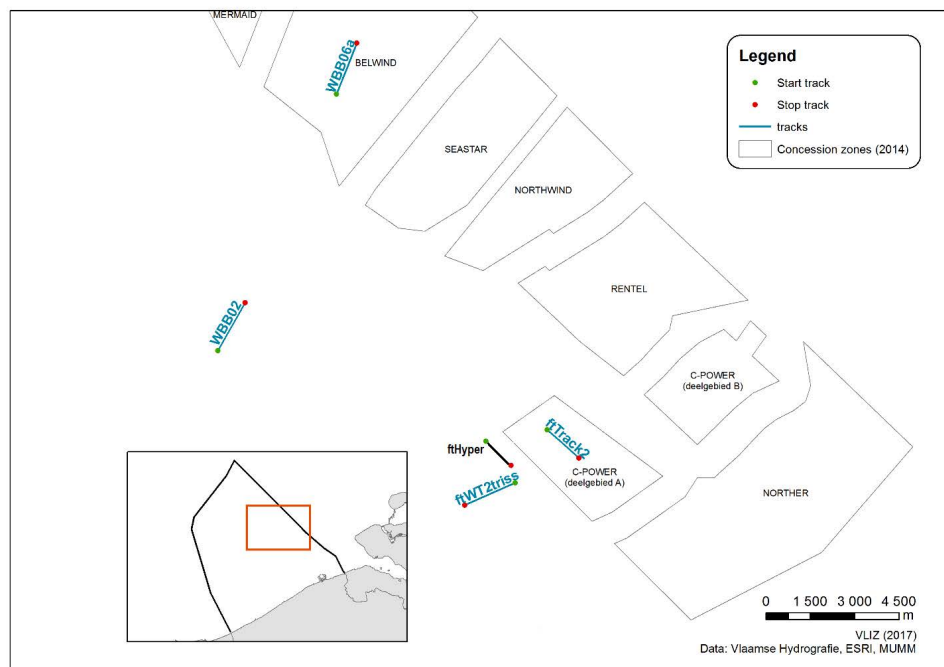
Figure 1. Wind farm concession areas in the Belgian part of the North Sea and the hyperbenthos sampling tracks (blue lines). The black line represents the newly proposed hyperbenthos track for the reference site at C-power.

of the RV Belgica, collectors were placed at an angle of 45° and a flowmeter was installed within the lower-net. Sampling proved to be successful and catches were rinsed separately (upper- and lower-nets) over a 1 mm sieve on board and preserved in an 8% formaldehyde solution.

Organisms were sorted, counted and identified to the lowest possible taxonomic level. In case of uncertainty, organisms were identified to a higher taxonomic level (genus, family or order) and typical macrobenthic species were removed from the dataset. Structural community characteristics within TB were explored by calculating total densities and diversity indices (species richness and Shannon-Wiener diversity) within each sample (REF upper, REF lower, IMPACT upper and IMPACT lower). Diversity indices were calculated using raw count data. Species reported on a higher taxonomic level were considered as "unique" if no other representative of the same taxon level was present or if they were distinctly different (morphospecies). Densities were obtained by standardising the data to individuals per 100 m³ through the following formula:

ind/100 m³ = number individuals / (surface net * amount of turns flowmeter) * 100. In addition, the Bottom Association (BA) index was also calculated to assess the vertical distribution of the animals in the hyperbenthic layer. The BA index was calculated by the total numbers caught in the lower net divided by the total number of individuals caught in both nets. Finally, differences in community structure were analysed through CLUSTER analysis (based on the Bray-Curtis dissimilarity index) and SIMPER analysis was performed both on untransformed and transformed (fourth root) data to specify the contributions of individual species to the distinction between groups of samples and/or to the similarity of samples within a group (Clarke & Gorley 2006).

3. Results

Total densities (upper- and lower-nets combined) were higher at the REF site (91 ind. 100 m⁻³) compared to the impact site (66 ind. 100 m⁻³), but strongest differences were found within each site between both nets, with higher total densities in the
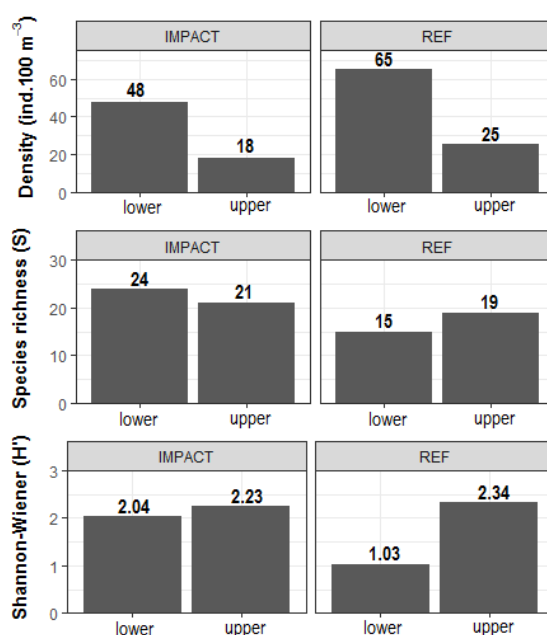

Figure 2. Overview barplots for calculated structural indices: total density (ind. 100 m⁻³), species richness (S) and Shannon-Wiener diversity (H') per sample (upper- and lower-nets) within the impact and reference site at C-power. Total values were added per sample above the corresponding barplot.

lower-net samples (fig. 2). It must be mentioned, however, that in the impact sample a significant amount of hydromedusae (upper sample) and mysids (lower sample) were too damaged for identification and were thus excluded from all analyses. Therefore, total densities in this sample might be underestimated. Total densities were highest for the sample Cref_lower (65 ind. 100 m^{-3}), which could be attributed to the high abundances of hydromedusae (48 ind.) and the amphipod *Atylus swammerdamei* (10 ind.). This finding of higher densities in the lower samples was also confirmed by the Bottom Association index, which revealed that about 70% of all individuals were caught in the lower-nets, indicating a vertical segregation within the hyperbenthic zone at both sites. In contrast, Cref_lower had the lowest diversity (S, H'), while these values were rather comparable among the other samples (fig. 2). In general, it can be concluded that for the structural indices, the most pronounced differences were found between both nets within the two sites in terms of total densities, while diversity differed mostly between the lower-net samples of the reference and impact site.

Hyperbenthic communities were mainly composed of hydromedusae (47%), Amphipoda (12%), fish larvae (11%), Chaetognatha (10%) and Mysida (8%). A total of 33 species were recorded and mysids proved to be the most species-rich order. Almost 1/3 of these species were unique for the impact site, and most of them represented smaller crustaceans such as mysids (*Siriella* sp., *Praunus inermis* and three mysid morphospecies), amphipods (*Corophium volutator*, *Orchomene* sp.) and cumaceans (*Bodotria* sp., *Pseudocuma* sp.). Species that were only found at the reference site included: *Gastrosaccus spinifer*, *Schistomysis kervillei*, *Ammodytes* larvae and two morphospecies of the orders Ctenophora and hydromedusae.

Comparable to the univariate results, CLUSTER analysis on untransformed density data revealed a clear separation between the lower and upper net samples in terms of community composition (fig. 3), and SIMPER results showed that average dissimilarity between nets was higher at the reference site (85%) compared to the impact site (66%). Lower samples were
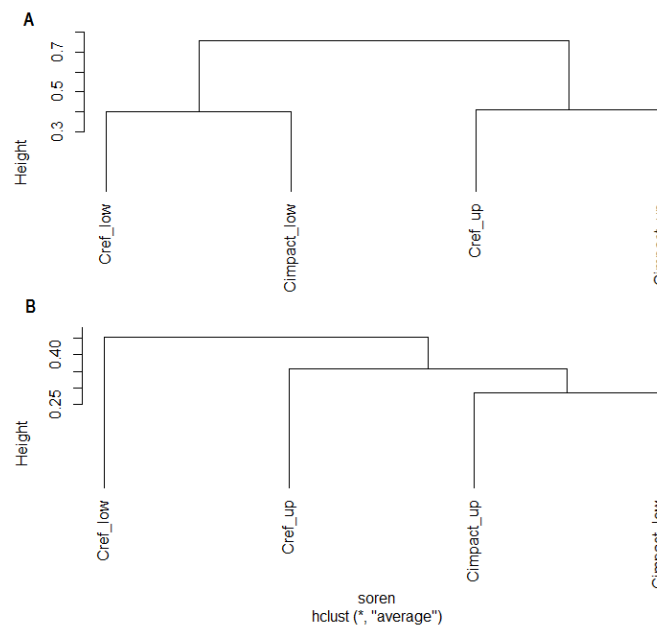
Figure 3. CLUSTER dendrograms based on untransformed density data (top) and fourth root transformed density data (bottom) per samples (upper- and lower-nets) within the reference and impact site at C-power.

dominated by hydromedusae (74%), *Atylus swammerdamei* (10%) and *Syngnathus rostellatus* (3%). The most important species contributing to about 90% of total densities for the upper samples included: *Sagitta elegans* (40%), *Atylus swammerdamei* (16%), *Mesodopsis slabberi* (15%), *Beroe cucumis* (6%), *Jassa* sp. (4%) and *Calanoida* sp. (3%).

A similar cluster analysis on fourth-root transformed density data showed a different trend in which the sample Cref_low was distinguished from the other samples (fig. 3). Despite the high total densities, this sample was dominated by hydromedusae and had the lowest diversity (S, H'). SIMPER analysis showed that strongest dissimilarities were found between Cref_low – Cref_up (46%) and Cref_low – Cimpact_low (40%). Differences between both nets at the reference site were mainly due to the higher average abundances of hydromedusae, Ctenophora and *Gastrosaccus spinifer* in the lower-net sample, while average abundances of *Mesodopsis slabberi*, *Neomysis integer* and *Jassa* sp. were higher within the upper-net sample. Shared species between the lower-net samples at the reference and impact site included: Hydromedusae, *Atylus swammerdamei* and *Sagittus elegans* where only the latter had higher average abundances at the impact site. Strongest between-site differences were, however, due to the unique presence of Ctenophora, *Gastrosaccus spinifer* and *Ammodytes* larvae at the reference site, while a high species richness (12) was only found at the impact site. Most of these unique species were representatives of mysids such as *Praunus inermis*, amphipods (*Corophium volutator*, *Jassa* sp.) or cumaceans such as *Bodotria* sp. and *Pseudocuma* sp.

4. Conclusions and future perspectives

The unsuccessful initial sampling showed that efficient hyperbenthos sampling strongly depends on favourable weather conditions and local topography, leading to potential practical difficulties in terms of sample collection. It appears that the relative position of the sampled tracks to the sandbank ridges is of great importance. Implementation of this prerequisite of tracks that run parallel to the sandbank ridges, together with a shorter towing distance (150 m), resulted in representative hyperbenthos samples. The obtained samples were processed in a reasonable time frame (one week), but it must be stated that there are no baseline studies for hyperbenthos communities within these areas of the wind farms, so it is not clear whether the processed samples can be considered as poor/rich in terms of abundance and diversity. As a result, future samples with higher abundances and the occurrence of many rare species might substantially prolong the sample processing.

Due to the small dataset (4 samples within C-power), it is difficult to draw firm conclusions with regard to turbine-related impacts on hyperbenthic communities. Nevertheless, this feasibility study did reveal some results that show the relevance of including this group of the bentho-pelagic community in future monitoring. Firstly, the strong association with the seafloor and the differences (densities, species composition) between upper- and lower-net samples indicate a vertical segregation within the hyperbenthic zone at both sites. The sediment-bound species that comprise an important component of total hyperbenthos densities could therefore be influenced by the altered sediment characteristics that are created by the presence of wind turbines. Secondly, the high amount of unique species at the impact site and the tendency of a higher diversity (S, H') at Cimpact_low compared to Cref_low suggest that structural differences might be established between the reference and impact site, but a more elaborate sampling design is necessary to confirm or discard this idea.

We therefore propose to perform several (2-3 replicates) shorter (150 m) reference and impact tracks for each wind farm.

When tracks are chosen, the position of the sand ridges should be taken into account and lower- and upper-net samples should be separated during sampling. In order to validate the proposed hypothesis of enriched hyperbenthic communities due to turbine-related habitat changes like decreased current velocities (improved settlement) and organic enrichment (attraction mobile fauna), environmental variables (grain size, bottom-current flow and food availability) should also be incorporated within the sampling strategy. If done correctly, the study of hyperbenthos may provide new insights and result in a more comprehensive view of wind turbine-related impacts on seafloor associated communities.

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CHAPTER 7

ABOUT "MYTILISATION" AND "SLIMEIFICATION": A DECADE OF SUCCESSION OF THE FOULING ASSEMBLAGES ON WIND TURBINES OFF THE BELGIAN COAST

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Abstract

In this contribution we describe the three succession stages of the subtidal fouling assemblages at two types of offshore wind turbines (*i.e.* gravity-based and monopile foundation) off the Belgian coast in the first decade after the installation. Installation of the turbine foundations was followed by rapid colonisation and a relatively short pioneer stage (~2 years) which differed between locations. At both locations, this was followed by a more diverse intermediate stage characterised by large numbers of suspension feeders. A third, and possibly "climax" *Metridium senile*-dominated stage, was reached after 10 years on the gravity-based foundations, while the assemblage on the steel monopiles of the more offshore site (9 years after construction) was described as an *M. senile-Mytilus edulis*-co-dominated assemblage. This study suggests that earlier reports on offshore wind turbines as biodiversity hotspots should be considered with caution as these reports generally refer to the typical species-rich second stage of succession reached after a few years of colonisation but disappearing later (after about 6 years in this study). Our results underline that artificial hard substrata differ greatly from the species-rich natural hard substrata and hence cannot be considered as an alternative for the quantitatively and qualitatively declining natural hard substrata such as gravel beds.

1. Introduction

Aside from natural hard substrata, there are numerous man-made hard substrata in the North Sea, such as wrecks, oil rigs and wind farms (*e.g.*, Zintzen *et al.* 2008; Coolen 2018). Today, these substrata are ever more prominent features in shelf ecosystems. That evolution, known as the hardening of the coast (Wolff 1999), started in the XVIth century with the construction of coastal defence and harbour structures. Initially, the hardening of the coast remained largely restricted to the coastal zone. That changed with the construction of oil and gas rigs further offshore when semi-permanent man-made structures were installed in the open sea. More recently, the open sea witnessed a further increase in man-made structures due to the proliferation of offshore wind farms (OWFs) and other marine infrastructures in response to the increasing demand of renewable energy (*e.g.*, Mineur *et al.* 2012; Causon & Gill 2018).

All these submerged man-made structures are rapidly colonised by fouling organisms (Horn 1974; Connell & Slatyer 1977; Kerckhof *et al.* 2009, 2010, 2012) that successively develop assemblages which may or may not resemble epibioses of natural hard substrata (Connell 2001; Wilhelmsson & Malm 2008; Svane & Petersen 2011; Kerckhof *et al.* 2017). In the case of OWFs, both the foundations of the turbines and the erosion protection surrounding them form hard permanently submerged substrata on which species can settle. Thus, in the Southern North Sea – a region that is mostly characterised by sandy sediments –, the wind turbines introduced a new habitat of artificial hard substratum that has enhanced the habitat heterogeneity of the region (Kerckhof *et al.* 2009; 2010). The effects of the introduction of these hard substrata – the so-called reef effect – is regarded as one of the most important changes of the marine environment caused by the construction of offshore wind farms (Petersen & Malm 2006).

Like natural hard substrata, also artificial hard substrata are often reported as hosting a lush epifouling community. The creation of new habitats increases the habitat diversity, which in turn increases species diversity, and artificial hard substrata are thus often considered hotspots of biodiversity (Wolff 1999). In fact, artificial hard substrata have recently been put forward as a possible alternative for the loss of natural hard substrata habitat and their installation has been proposed as a measure to strengthen biodiversity *e.g.*, plan Zeehond (Vande Lanotte *et al.* 2012) or to offset the negative environmental impacts of OWF construction (Veraart *et al.* 2017).

Contrary to newly exposed landforms where succession may take several hundreds of years, the succession of animals and algae on recently denuded rock walls in the marine subtidal zone takes less time, approximately a decade (Hill *et al.* 2002; Whomersley & Picken 2003). However, few studies of the development of fouling assemblages on large

hard substrata over a longer time period (*i.e.* a decade or more) exist. Much of the information forming the documenting biodiversity at artificial hard substrata in the North Sea hence is derived from one-off sampling events or short-term time series (Vanagt *et al.* 2013, 2014) often providing contradicting views on biodiversity and hence focusing on the high species richness compared to surrounding soft sediments (Leonhard & Pedersen 2006; Lindeboom *et al.* 2011). Short-term time series indeed typically reference the high species richness at *e.g.* OWFs (Kerckhof *et al.* 2009, 2012; De Mesel *et al.* 2015).

The aim of this paper is to study the longer-term dynamics of the macrobenthic fouling assemblage on the first offshore wind turbines that were installed in Belgian waters. This study provides context for one-off sampling events and short-time series results.

2. Material and methods

We investigated the early stage, short-term and longer-term dynamics of the macrobenthic fouling assemblage at two OWFs in the Belgian part of the North Sea (BPNS), focusing on the establishment of new species and the successional trend of those species that, at any stage of the succession, were observed to be superabundant on the Marine Nature Conservation Review SACFOR abundance scale (Hiscock & Connor 1991).

2.1. Study site and sampling locations

We collected hard substrata subtidal macrofauna from the foundations of two OWFs, C-Power and Belwind, both located in the Belgian offshore renewable energy zone (see chapter 1). In the Belwind wind farm, situated on the Bligh Bank at about 50 km offshore and entirely located in the clear English Channel's water (M'harzi *et al.* 1998), wind turbines were installed on steel monopile foundations. The C-Power wind

farm (6 concrete gravity-based foundations (GBFs) and 48 steel jacket foundations) is located on the Thornton Bank, some 30 km off the Belgian coast in waters transitional between the more turbid coastal and clear offshore waters. Both banks belong to the Zeeland Banks system (Cattrijsse & Vincx 2001).

The samples were collected from a selected set of wind turbine foundations: 2 GBFs (D5 and D4; C-Power wind farm, installed respectively on the 30th and 24th of May 2008) and 3 steel monopile foundations (BBB8, BBC8 and BBC2; Belwind wind farm, installed respectively on the 21st and 24th of October and on 14 December 2009), this because access to the same foundation was not always possible due to *e.g.* maintenance works. In this study, the data from both GBF foundations and all three monopile foundations were considered equally representative for location and type of foundation, and were hence pooled in the database to GBF foundation and monopile foundation.

2.2. Sampling and sample processing

Sampling started shortly after the installation of the turbines *i.e.* first sampling in September 2008 for the GBFs (start: 164 days after installation; 10 years' time series) and in February 2010 for the monopiles (start: 181 days after installation; 9 years' time series). The last sampling event included in this analysis took place in August 2018. Samples are missing for 2012, 2016 and 2017.

Scuba divers collected the subtidal scrape samples from the foundations by scraping three replicas of the fouling organisms from a square sampling surface area of 6.3 dm² (Kerckhof *et al.* 2010, fig. 1). Samples were collected at -15 m, which is considered representative for the assemblages in most of the subtidal part of the foundations (Kerckhof *et al.* 2010). In order to exclude seasonal variability in the data and hence focus on the year-to-year variability in the fouling assemblages, only samples
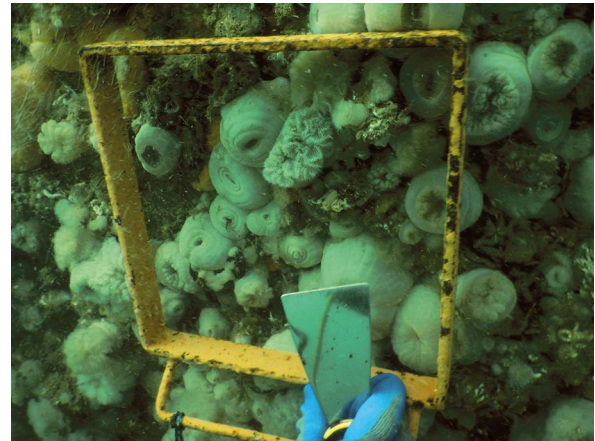
Figure 1. Sampling square and scraper at the BBC02 monopile (August 2019). Note the presence of *Mytilus edulis* clumps in between and under *Metridium senile*.

that were collected in late summer and autumn (August-November) were included in the analysis.

The scraped material was collected in plastic bags that were sealed, preserved in buffered formalin 10% and transported to the laboratory for further processing: sieving over a 1 mm sieve and sorting. Individual organisms were sorted and identified to the lowest taxonomic level possible (mostly species level; further called "species") using a stereoscopic binocular microscope. Density was determined for countable species, while percentage cover was assessed for the uncountable colonial crust forming and erect (bushy) epifaunal species such as hydrozoans, bryozoans and sponges. All data were transformed to the SACFOR scale to allow for integrating relative abundances of countable and uncountable species (Connor & Hiscock 1996).

2.3. Data analysis

For the sake of investigating trends in successful coloniser arrival, we considered a new species established (*i.e.* successful new colonisation) when the species was first detected in year x and also found in subsequent years. As such, we maximally exclude bias caused by the erratic and hence less reliable

observations of rare species. Both countable and uncountable species were included.

To investigate species turnover in the assemblages, we focused on the most numerically abundant species, being those that have been scored superabundant (cf. SACFOR scale) at least once in the time series; this either for GBFs and monopile foundations. This resulted in a list of 13 species, eleven of which being countable, and only two being uncountable. Among the countable species, the Echinodermata dominated with four species (*Asterias rubens*, *Amphipholis squamatus*, *Ophiothrix fragilis*, *Psammechinus miliaris*) followed by three Amphipoda (*Jassa herdmani*, *Stenothoe* sp., *Monocorophium acherusicum*), two Polychaeta (*Phyllodoce mucosa*, *Spirobranchus triqueter*) and finally, one bivalve mollusc (*Mytilus edulis*) and one hydrozoan (*Metridium senile*). The uncountable species were *Electra pilosa*, a bryozoan, and *Tubularia indivisa*, a hydrozoan. Per countable species and per type of foundation, absolute densities were relativised to the maximum density recorded for that species in any of the samples (*i.e.* normalised density); this to be able to visualise the successional patterns of the different species irrespective of their absolute densities. The succession of uncountable species was presented using the SACFOR scale.

3. Results

On both GBFs and monopiles, successful new colonisers clearly dominated over non-successful newcomers only in the first 3-4 years (fig. 2), with year 5 being a transition year, because successful new arrivals did not (or hardly) occurred in years 6 and beyond. For both foundation types, overall species numbers declined after the fifth year. After the fifth year, the number of species per sample was higher on the monopiles than on the GBFs, where the increasingly dominating presence of . *M. senile* coincided with a species-poor assemblage. On the monopiles, large clumps of *M. edulis* and a few other species coexisted with . *M. senile*.

The colonisation of the most numerically abundant countable species shows a similar pattern on both foundation types, with a similar suite of species involved. Looking at those species that were observed to be superabundant, we can distinguish three stages in the succession, each characterised by certain specific species (figs 3 & 4, table 1). A pioneer stage of about two years was characterised by a number of opportunistic species and differed markedly between foundation types. In the intermediate stage between years 3 and 6, several types of suspension feeders became superabundant including
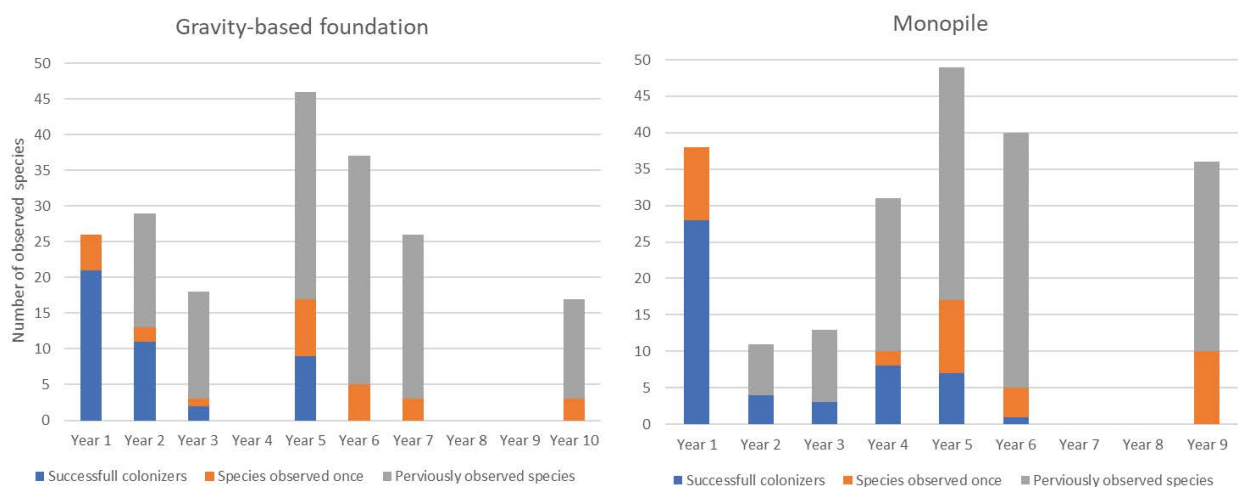
Figure 2. Total number of species observed per year, distinguishing between successful new coloniser arrivals (*i.e.* species first detected in year x and also found the subsequent years: blue), non-successful new arrivals (*i.e.* species first detected in year x and not found in subsequent years: orange), and previously observed species (*i.e.* species detected in year x and also found in a previous year: grey).
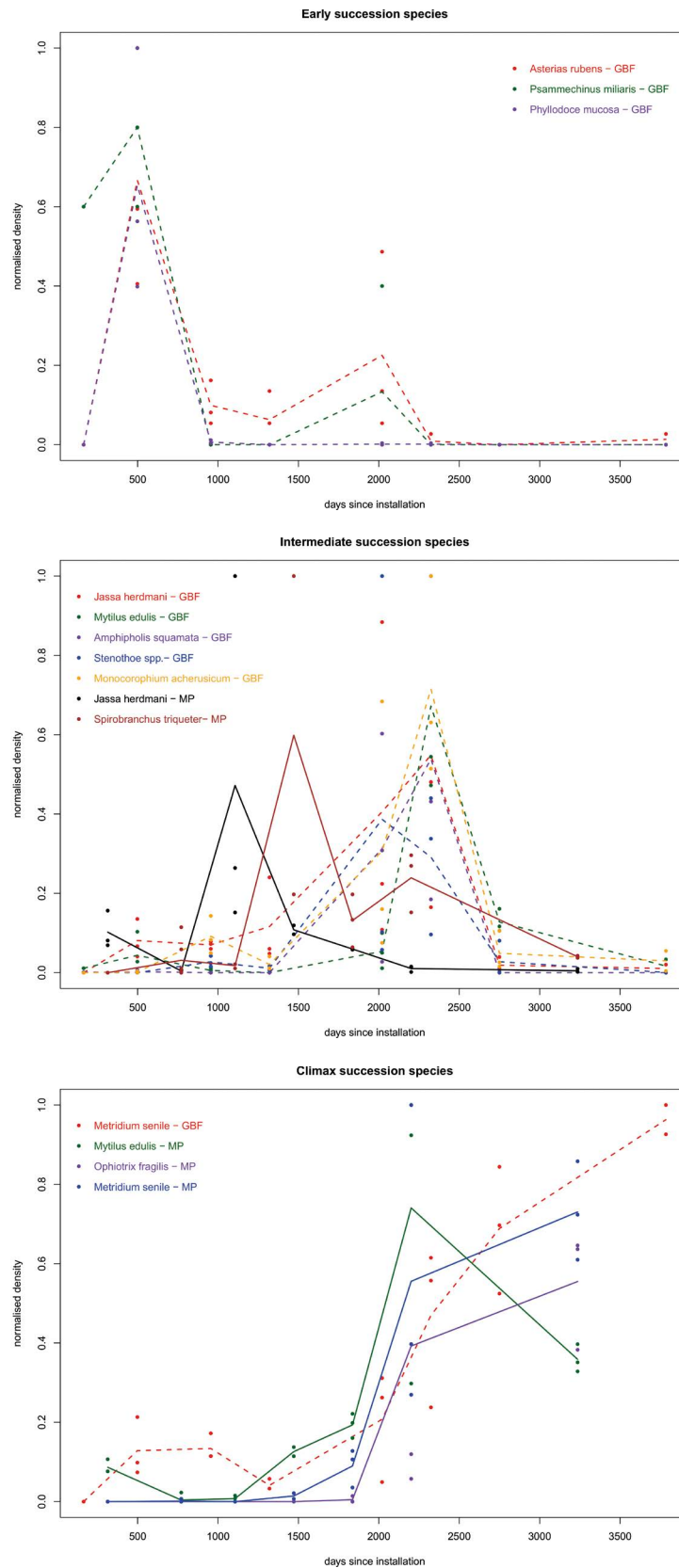
Figure 3. Succession stages and species turnover on the gravity-based foundations (GBF; dashed lines, ~ 10 years) and the monopile foundations (MP; full lines, ~ 9 years). Normalised density: per species and per type of foundation, absolute densities were relativised to the maximum density recorded for that species in our time series.

Table 1. List of superabundant epifouling species per foundation type with an indication of when these epifouling species were superabundant on the turbine foundations

Succession stage	Monopile	GBF
Stage 1: Pioneer (y 1 & 2)		
	Jassa herdmani	*Jassa herdmani*
		Electra pilosa
		Psammechinus miliaris
		Asterias rubens
		Phyllodoce mucosa
Stage 2: Intermediate (y 3-5)		
	Jassa herdmani	*Jassa herdmani*
	Tubularia indivisa	*Tubularia indivisa*
	Spirobranchus triqueter	*Mytilus edulis*
		Monocorophium acherusicum
		Amphipholis squamata
		Stenothoe sp.
Stage 3: "Climax" (from y 6 onwards)		
	Metridium senile	*Metridium senile*
	Mytilus edulis	
	Ophiothrix fragilis	

amphipods, bivalves, hydrozoans and polychaetes. From approximately the 6th year onwards, there is a third stage in which only a few species dominate (table 1).

4. Discussion

4.1. Successional stages

The establishment of a biofouling community follows a clear successional development, as the newly available structures are gradually colonised by several species. These organisms will each influence the environment in a species-specific way, as such preventing other organisms to get established (*i.e.* inhibition, *e.g. M. senile*) or creating the right circumstances for other species to settle (*i.e.* facilitation, *e.g. Tubularia* sp., *M. edulis*; Connell & Slatyer 1977). During our survey, new species arrivals and the disappearance of other species continuously modified the
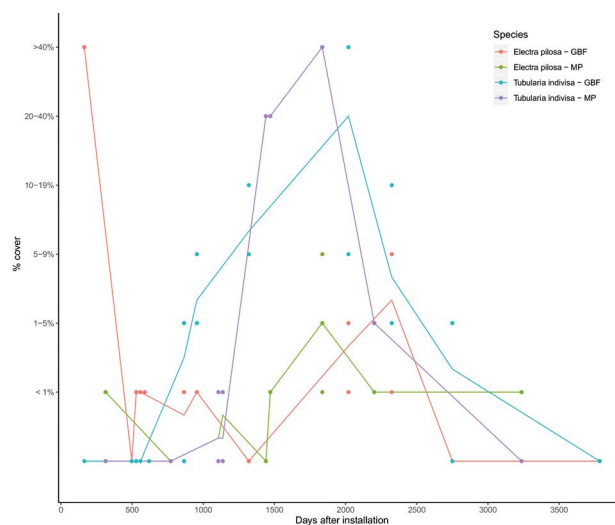
Figure 4. Percentage cover per sample of *Electra pilosa*, a bryozoan, and *Tubularia indivisa*, a hydrozoan, on the gravity-based foundations (GBF; ~ 10 years) and the monopile foundations (MP; ~ 9 years).

assemblage, as suggested by Underwood and Chapman (2006). Three stages could be identified: an early colonisation stage, a species-rich intermediate stage and a . *M. senile*-dominated or *M. senile*-*M. edulis*-co-dominated mature stage here called "climax" stage.

Succession started with a swift colonisation by early colonisers (first stage of succession). As previously discussed by Kerckhof *et al.* (2010, 2012), this pioneer stage, of about two years, was characterised by a number of opportunistic species. Its species composition differed markedly between foundation types which may be explained by both the difference in timing of installation (GBF: late spring; monopile: autumn-winter) and geographical location of the foundations (with the GBF located in coastal waters and the monopiles located in more clear Channel waters) resulting in a different species pool of spores and larvae available for colonisation.

Despite initial differences in the intermediate stage, a convergence towards a common assemblage dominated by the same suspension feeders was achieved. This conforms with the results of Pacheco *et al.* (2011) who studied succession of sessile biota in sublittoral rocky shore communities and demonstrated that, although seasonality generates high variability during early colonisation process, this effect diminishing as succession proceeds.

Most of the first colonisers persisted during the subsequent years, but with a decreasing trend in their abundance as succession proceeded. However, several of these *e.g. Spirobanchus, Jassa herdmani, Electra pilosa* showed a second peak (figs 3 & 4), likely related to the presence of keystone habitat modifiers species *sensu* Coolen *et al.* (2018), *in casu M. edulis* and *Tubularia* sp. *Jassa herdmani* could indeed have been facilitated by the presence of *Tubularia* sp. offering complex 3-dimensional structures, home to many associated species by offering addition

substrata to settle (Zintzen 2007; De Mesel *et al.* 2015; Coolen *et al.* 2018). This facilitation effect could also apply to *Stenothoe* sp. and, to a lesser extent, *Monocorophium acherusicum*.

In the last stage, and especially apparent during the latest sampling event, the plumose anemone. *M. senile* dominated the species assemblage on both GBF and monopiles; this at the expense of most other fouling species. Because the prominent presence of *M. senile* on the GBFs was already apparent in the earlier stage of the GBF succession series, the development towards a species-poor *M. senile*-dominated biotope was already suggested before (De Mesel *et al.* 2015).

4.2. *Metridium senile* effect

Due to its large body size, the plumose anemone constitutes a conspicuous part of the subtidal community everywhere in the Southern North Sea where it appears, which is especially noticeable on artificial hard substrata such as wrecks (Van Moorsel *et al.* 1991; Zintzen 2007) and buoys, but also on (disturbed) natural hard substrata (*e.g.* Coolen *et al.* 2015; own observations).

Metridium senile is a strong competitor for space that can have a strong structuring effect onto the fouling community by rapidly colonising new substrata, covering large areas, preventing other species' propagules to settle, consuming free-living larvae and smothering new recruits (Nelson & Craig 2011; Kaplan 1984). With other predators lacking or occurring in low numbers (such as the sea spider *Pycnogonum littorale* (Wilhelm *et al.* 1997) or *Epitonium clathratulum*, respectively), *M. senile* can be seen as both a keystone predator and a keystone modifier on artificial hard substrates in the North Sea (Coolen *et al.* 2018).

The establishment of a species-poor assemblage dominated by Actiniaria, as it was the case on the GBFs, is a common feature on man-made substrata in the North Sea.

The *M. senile* biotope, *sensu* Connor *et al.* (2004), is often the characteristic final state for this type of substrata in the North Sea.

4.3. The *Metridium* – *Mytilus* stage

There was a difference in the "climax" assemblages between GBFs and monopiles. When the samples came on board during the last sampling event, we observed that the sea anemones all lived attached to adult blue mussels *M. edulis*, often occupying both valves of the shells. They were not directly attached to the surface of the monopiles, while on the GBFs the sea anemones lived directly attached to the concrete surface of the piles. The association of mussels with *M. senile*, although not commonly observed in the North Sea (see above), is also known from areas beyond the North Sea, *e.g.* from Shark River along the New Jersey US East coast (Kaplan 1984).

Due to the presence of aggregates of large mussels, the species diversity on the monopiles was higher in the "climax" stage than the species-poor *M. senile*-dominated GBFs. Mussels *M. edulis* create complex aggregates that increase habitat heterogeneity resulting in increased species richness (Drent & Dekker 2013; Coolen *et al.* 2018). Indeed, many *Ophiothrix fragilis* lived between the mussels on the monopiles while the shells on their turn added additional hard substratum surface suitable for other species to settle and live, such as *M. acherusicum*, *J. herdmani* and *S. triqueter*, but also *M. senile*. *Mytilus edulis* can therefore be considered as another keystone habitat modifier (Coolen *et al.* 2018).

During the initial succession stages, mussels were almost absent from the subtidal (Kerckhof *et al.* 2009, 2010). In contrast, a conspicuous blue mussel *M. edulis* belt had developed in the intertidal and shallow subtidal zone already during the early stages of colonisation (*e.g.* De Mesel *et al.* 2015). Observations of blue mussels in subtidal samples gradually increased throughout the

years, not as spat nor young mussels but already as adults (Kerckhof, unpublished data). Likely, the adult mussels – being semi-mobile organisms – had moved down from the intertidal mussel belt. Subsequently, this mussel belt gradually expanded to greater depths and extended to the deepestr subtidal as individuals moved downwards in a natural thinning process (*e.g.* Hughes & Griffiths 1988).

A similar situation of co-dominance of *M. edulis* and *M. senile* was seen at the monopile foundation after about nine years of colonisation and was also detected in GBF samples collected in the years prior to the *M. senile*-dominated stage, which corresponds to about nine years of colonisation at the GBFs. We argue that the co-dominance of *M. edulis* and *M. senile* may hence represent a transitional stage that will ultimately lead to a pure *M. senile*-dominated assemblage. Once these large mussels die off, *M. senile* may be able to fully occupy the free space and hence block further colonisation opportunities for adult mussels and propagules of other organisms. The mytilisation *sensu* Krone *et al.* (2013) and observed in other studies might hence represent only a transitional stage towards a pure *M. senile*-dominated stage common in the deeper waters on man-made structures. In the short-term, however, the presence of *M. senile* could be considered beneficial to *M. edulis* as the presence of *M. senile* on the valves of *M. edulis* has been shown to significantly reduce mussel predation by starfish (Kaplan 1984).

Why artificial hard substrate epifouling communities tend to evolve towards a species-poor anemone-dominated community is yet to be proven. As for shipwrecks, specific features of the wind turbines such as verticality, spatial patchiness, unnatural substratum composition and/or the lack of microhabitats may explain the specific community structure typified by an impoverished epifauna compared to the surrounding natural hard substrata (Svane & Petersen 2001; Chapman 2003).

5. Conclusion

We revealed that, as suggested by De Mesel *et al.* (2015), the subtidal community on the wind turbine foundations in the BPNS develops towards a species-poor *M. senile*-dominated biotope *sensu* Connor *et al.* (2004). In our study, a relatively short pioneer stage (~ 2 years) was followed by a more diverse intermediate stage characterised by large numbers of suspension feeders. A third and possibly "climax" stage has been reached after about 10 years on the GBFs of the C-Power site, while the assemblage on the monopiles of the Belwind site after only 9 years of colonisation may be seen as a transient *M. senile-M. edulis*-co-dominated assemblage that may ultimately evolve to the same species-poor *M. senile*-dominated biotope as on the GBFs. This study suggests that earlier reports on offshore wind turbines as biodiversity hotspots may be considered premature; this because the reports generally refer to the typical species-rich intermediate stage of succession reached after a few years of colonisation but disappearing later on (about 6 years in this study). Our results underline that artificial hard substrata strongly differ from the species-rich natural hard substrata and hence cannot be considered an alternative for the quantitatively and qualitatively declining natural hard substrata such as the gravel beds in Belgian waters.

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CHAPTER 8

SEABIRD MONITORING AT THE THORNTON BANK OFFSHORE WIND FARM

FINAL DISPLACEMENT RESULTS AFTER 6 YEARS OF POST-CONSTRUCTION MONITORING AND AN EXPLORATIVE BAYESIAN ANALYSIS OF COMMON GUILLEMOT DISPLACEMENT USING INLA

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Abstract

Since 2005, the Research Institute for Nature and Forest (INBO) has been engaged in a monitoring program aiming to study seabird displacement effects caused by offshore wind farms in the Belgian part of the North Sea. Here we report our findings for the C-Power wind farm at the Thornton Bank after six years of post-construction monitoring. Results showed significant avoidance of the wind farm area by northern gannet, common guillemot and razorbill, these species having dropped in numbers by 98%, 60-63% and 75-80% respectively. In contrast, attraction to the wind farm could be demonstrated for herring and great black-backed gulls, for which our BACI models predicted a factorial change in densities of 3.8-4.9 and 5.3-6.6 respectively. Importantly, most of these effects account for the footprint area only, and were no longer noticeable in the buffer area 0.5-3 km away from the wind farm edge. Great cormorants too showed major attraction effects, amplified by the fact that the species was quasi-absent in the study area prior to wind farm construction. The effects at the Thornton Bank show striking parallels with those observed at the nearby Bligh Bank, and among European studies in general there seems good consistency in the avoidance response of gannets and auks, as well as in the attraction effects observed for great cormorants and great black-backed gulls. A major lack in understanding, however, persists when it comes to the translation of these behavioural responses into displacement impact, considered as any change in individual fitness, reproductive success and survival. Clearly, filling in this wide knowledge gap is crucial for a reliable assessment of the actual and cumulative ecological consequences of extensive offshore wind farm installations, and should therefore be the primary goal of future research.

1. Introduction

In aiming to meet the targets set by the European Directive 2009/28/EG on renewable energy, no fewer than 4,543 offshore wind turbines were fully grid-connected in European waters by the end of 2018, totalling 18.5 GW. In the Belgian part of the North Sea (BPNS), 5 wind farms are currently operational, representing a capacity of 1.2 GW (EWEA 2019).

Since 2005, the Research Institute for Nature and Forest (INBO) has been in charge of studying seabird displacement caused by offshore wind farms (OWFs), applying a before-after control-impact (BACI) strategy. Results after 5 years of post-construction monitoring at the Bligh Bank were presented in Vanermen *et al.* (2016), and this report will discuss the results after 6 years of post-construction data collection at the Thornton Bank. The results will be compared with those obtained at the nearby Bligh Bank and other European OWF sites, and we will continue the discussion by defining important knowledge gaps on wind farm displacement research.

The results presented here can be regarded as final because it is no longer feasible to continue or repeat the monitoring as performed up until now, in which we focused on single isolated wind farm sites surrounded by 3 km buffer zones. However, these buffer areas are now increasingly occupied by wind turbines of newly constructed, adjacent wind farms. From an environmental monitoring perspective, the Belgian offshore wind farm concession zone will soon have to be considered as one large wind farm cluster. Therefore, a new seabird displacement monitoring scheme will start once the concession zone is fully occupied by turbines (expected by 2020), demanding a different and more advanced analysis method. We will explore the possibilities of using an INLA (integrated nested Laplace approximation) approach to detect displacement in seabirds-at-sea monitoring data, based on a post-construction subset of common guillemot data collected at the Thornton Bank. Importantly, this Bayesian analysis method allows to take into account fine-scaled spatial correlation, which has been avoided in our BACI analyses by aggregating the count data to day totals per area, yet hereby strongly limiting the sample size as well as statistical power.

2. Material and methods

2.1. Thornton Bank offshore wind farm

The Thornton Bank wind farm is located 27 km off the coast of Zeebrugge. The wind farm consists of 2 subareas of 10.7 and 9.2 km² occupied with 30 and 24 wind turbines respectively and separated by a 1.7-2.0 km corridor (fig. 1). The water depth of the turbine-built area ranges between 12 and 28 m (C-Power 2019). Distances between the turbines range from 500 up to 900 m. The wind farm was built in three phases:

- Phase 1: 6 x 5 MW turbines (gravity-based foundations), operational since May 2009

- Phase 2: 30 x 6.15 MW turbines (jacket foundations), operational since October 2012

- Phase 3: 18 x 6.15 MW turbines (jacket foundations), operational since September 2013

2.2. Displacement study

2.2.1. Seabird counting

Ship-based seabird counts were conducted according to a standardised and internationally applied method, combining a "transect count" for birds on the water and repeated "snapshot counts" for flying birds (Tasker *et al.* 1984). We focused on a 300 m wide transect along one side of the ship's track, and while steaming at a speed of about 10 knots, all birds in touch with the water (swimming, dipping, diving) observed within this transect were counted ("transect count"). Importantly, the distance of each observed bird (group) to the ship was estimated, allowing to correct for decreasing detectability with increasing distance afterwards (through distance analysis, see §2.2.2). The transect was thus divided in four distance categories (A = 0-50 m; B = 50-100 m; C = 100-200 m;

D = 200-300 m). Counting all flying birds inside this transect, however, would cause an overestimation and would be a measure of bird flux rather than bird density (Tasker *et al.* 1984). The density of flying birds was therefore assessed through one-minute interval counts of birds flying within a quadrant of 300 by 300 m inside the transect ("snapshot counts"). As the ship covers a distance of approximately 300 m per minute when sailing the prescribed speed of 10 knots, the full transect was covered by means of these subsequent "snapshots".

For data processing, observation time was linked to the corresponding GPS coordinates registered by the ship's board computer. Taking into account the transect width and distance travelled during a certain time interval, the cumulative result of the coinciding transect and snapshot counts could be transformed to a number of birds per km², *i.e.* a seabird density. Up to 2012, observations were aggregated in ten-minute bouts, which were cut off to the nearest minute at waypoints. In 2013, resolution was increased and seabird observations were pooled in two-minute bouts ever since, again cut off to the nearest minute at waypoints.

It should be highlighted that in practice we recorded all birds (and sea mammals) observed, but those not satisfying above conditions (*i.e.* not observed inside the transect nor during snapshots) were given another code and could not be included in any analysis using seabird densities (*i.e.* all analyses in this study, except for great cormorants). On board, we further noted as much information as possible regarding the birds' age, plumage, behaviour, flight direction and association with objects, vessels or other birds. Whenever fishing vessels were in view, distance (perpendicular to the monitoring track whenever possible), type and activity of the vessel were assessed and recorded. Finally, observation conditions (wind force, wave height and visibility) were noted at the start of each survey and updated continuously.

2.2.2. Distance analysis

We corrected our transect count numbers for the fact that the probability of detecting birds decreases with distance to the ship (Buckland *et al.* 2001; Thomas *et al.* 2010). The exact relation between distance and detection probability is expected to be species-specific, and further likely to depend on bird group size and observation conditions (Marques & Buckland 2003). Observation conditions were included in the detection models as "wind force" (Beaufort scale) or "wave height" (categorised as 0-0.5 m / 0.5-1.0 m / 1.0-2.0 m / 2.0-3.0 m…), both variables being assessed continuously throughout the surveys.

To look for suitable species-specific detection models, we fitted each of the following "full models" with a half-normal as well as a hazard-rate detection function:

- P(detection) ~ group size + wind force
- P(detection) ~ group size + wave height
- P(detection) ~ log(group size) + wind force
- P(detection) ~ log(group size) + wave height

We did not add cosine or polynomial adjustments to the models as doing so sometimes appeared to result in non-monotonic functions. This would imply that the detection probability *increases* with distance, which is assumed to be highly improbable. The best fitting full model was chosen based on the "Akaike Information Criterion" (AIC), and backward covariate selection was then performed to obtain a parsimonious detection model. In the end, the resulting models were used to predict species-specific detection probabilities varying with the selected covariates, and the counted numbers were corrected accordingly.

Table 1. Definition of the reference, construction and impact periods at the Thornton Bank study area; only the "reference period" and "impact period (phase I, II & III)" were applied in the analyses

| Phase	Period
Reference period	< 04/2008
1st construction period	04/2008 > 05/2009 (highly restricted access)
Impact period (phase I)	06/2009 > 04/2011 (6 turbines)
2nd construction period	05/2011 > 09/2012 (variable access)
Impact period (phase I, II & III)	10/2012 > 12/2018 (54 turbines)

2.2.3. Monitoring set-up

The seabird displacement monitoring was performed according to a before-after control-impact (BACI) set-up (fig. 1). The OWF footprint area was surrounded by a buffer zone of 3 km to define the impact area, further distinguishing between the "footprint + 0.5 km" and the "buffer 0.5-3 km" area. Next, we delineated a control area harbouring comparable numbers of seabirds before OWF construction and showing a similar range in water depth and distance to the coast. Meanwhile, the distance between the control and impact area was kept small enough to be able to survey both on the same day by means of a research vessel (Vanermen *et al.* 2005).

Following fixed monitoring tracks, the Thornton Bank study area was counted on a highly regular basis from 2005 until present (figs 1 & 2). During this dedicated monitoring program, the study area should have been visited monthly, but research vessels were not always available and planned trips were regularly cancelled due to adverse weather conditions (characterised by significant wave heights higher than 2 m and/or poor visibility). Prior to this dedicated monitoring program, the study area was counted on a much more irregular base, but we did include surveys dating back to 1993 provided that the control and impact area were visited on the same day.

For our displacement analyses, only data falling within the "reference period" and "impact period (phase I, II & III)" were used (table 1). Note that phase III was not yet operational before September 2013 (§2.1), while the impact period defined in table 1 and as used in the analyses already starts in October 2012 (when phase II became operational). This is justified by the fact that access for monitoring was not allowed where active construction activities of phase III were going on, and data collected between October 2012 and September 2013 thus account for the operational part of the OWF only.

Compared to the previous monitoring report (Vanermen *et al.* 2017), data from 12 more surveys could be added to the dataset. Our dataset now includes 52 post-construction opposed to 65 pre-construction surveys (fig. 2).

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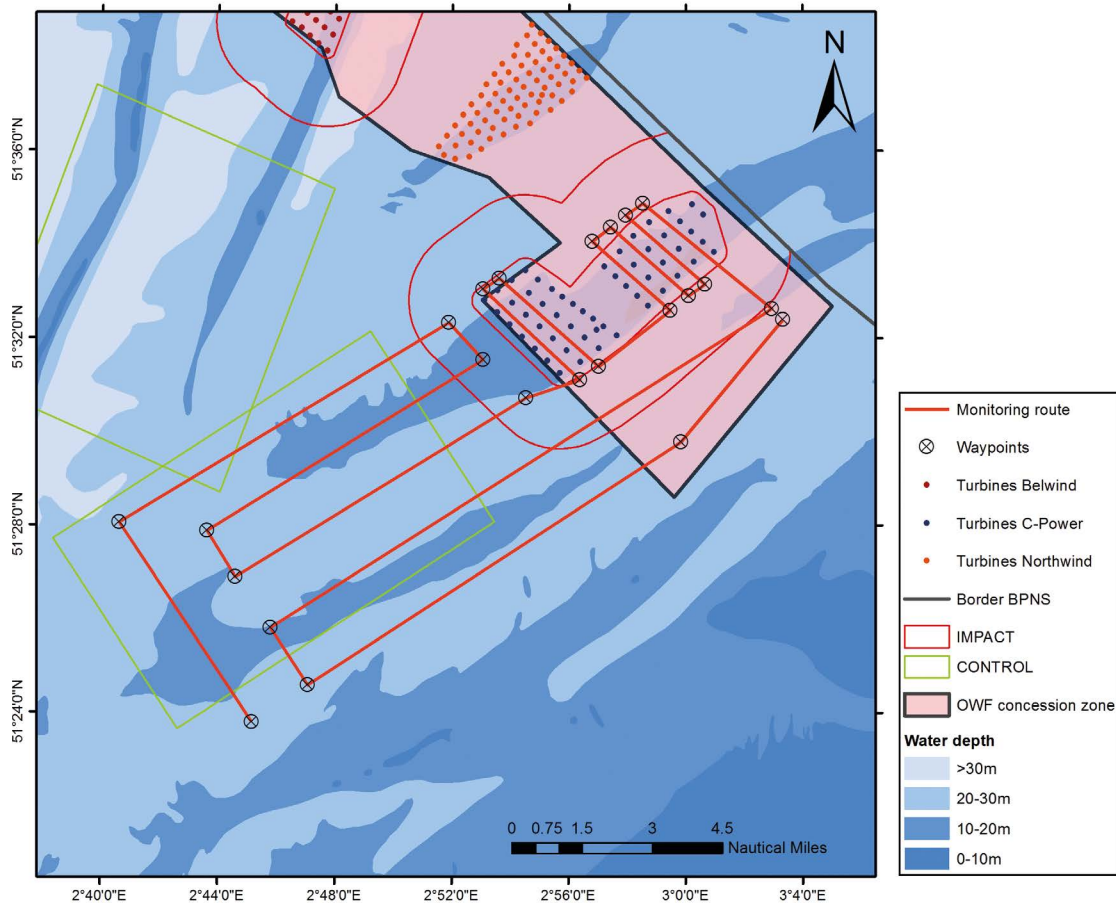
Figure 1. Post-construction monitoring route across the Thornton Bank OWF study area.
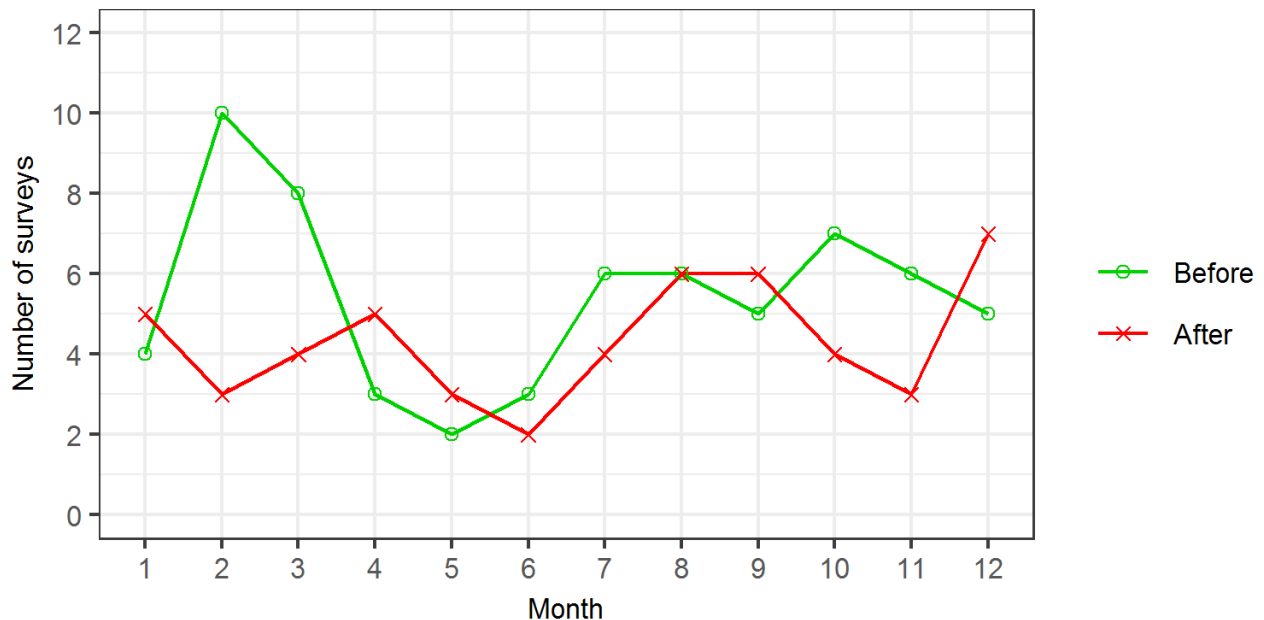
Figure 2. Count effort in the Thornton Bank study area indicated by the number of surveys performed before the construction of the phase I turbines (< April 2008) and after the construction of the phase II turbines (> September 2012).

2.2.4. BACI analysis

Data selection

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

Modelling was performed for thirteen seabird species occurring regularly in the study area, *i.e.* northern fulmar (*Fulmarus glacialis*), northern gannet (*Morus bassanus*), great cormorant (*Phalacrocorax carbo*), great skua (*Stercorarius skua*), little gull (*Hydrocoloeus minutus*), common gull (*Larus canus*), lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), black-legged kittiwake (*Rissa tridactyla*), Sandwich tern (*Thalasseus sandvicensis*), common guillemot (*Uria aalge*) and razorbill (*Alca torda*). For each of these species, we modelled three different impact datasets differing in the post-construction impact data selection (OWF footprint + 0.5 km, OWF footprint + 3 km, OWF buffer 0.5-3 km, see fig. 3).

Response variable

The response variable (Y) of our displacement models was the number of birds observed inside the transect and during snapshot counts, aggregated per area and per monitoring day. For great cormorants, a different response variable was applied, *i.e.* the number observed per kilometre. This was done because analysing cormorant densities caused analytical problems resulting from extremely low presence rates in the control as well as the impact area before wind farm construction. For the large gull species (herring, lesser black-backed and great black-backed gull), we modelled an "adjusted response variable"

on top of the standard response. Because the corridors between the C-Power turbines used during seabird monitoring (fig. 1) vary in width between 650 and 900 m, combined with the fact that the research vessels always aim to sail in the middle of these corridors for security reasons, birds associated with the turbines were by definition outside our 300 m wide transect. To counter this, an adjusted response variable was calculated by adding (i) the number of birds that would have been present inside the transect if the turbine-associated birds had occurred homogenously spread across the area to (ii) the number of birds counted inside the transect and during snapshot counts (*i.e.* the original response variable). This is most easily illustrated with an example: on 28 August 2015 we counted 161 great black-backed gulls resting on the jacket foundations, as opposed to only 1 bird present inside our transect (the original response) despite a survey effort of 7.4 km² inside the impact area. As we checked 43 turbines out of a total of 54 turbines, we estimated the number of turbine-associated great black-backed gulls in the OWF as a whole at 202 birds. The wind farm area surrounded by the 0.5 km wide buffer zone measures 36 km², and the density of turbine-associated great black-backed gulls was thus 5.6 birds/km². If these birds would have occurred homogenously spread across the area, and knowing we counted 7.4 km², the number of birds inside the transect would have been (about) 42, which is our adjusted response. The original and adjusted response variables were always analysed both, and the difference is clearly indicated in all graphs and tables.

Explanatory variables

To account for varying monitoring effort, the number of km² counted was included in the models as an offset-variable (only in case of the great cormorant we used the number of kilometres sailed). The explanatory variables used were (i) a time factor BA (before/after construction), (ii) an area
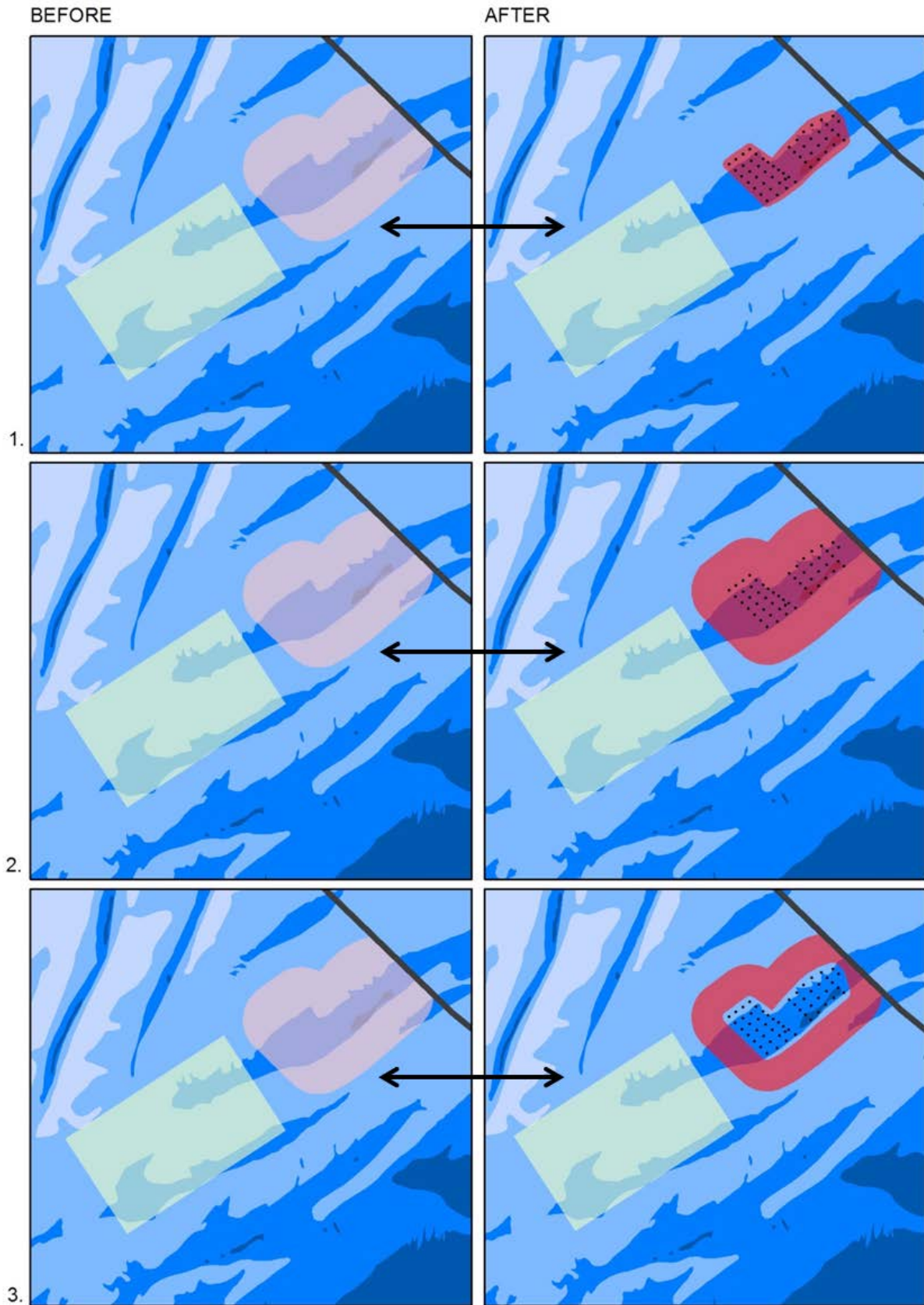
Figure 3. Overview of the BACI polygons used for data selection to study OWF induced seabird displacement at the Thornton Bank (green = control area / red = impact area; 1 = "OWF footprint + 0.5 km", 2 = "OWF footprint + 3 km", 3 = "OWF buffer 0.5-3 km").

factor CI (control/impact area), (iii) an off-shore wind farm factor OWF (wind farm present/absent) and (iv) a fishery factor F (fishing vessels present/absent in the wider area). The latter excluded fishing vessels observed beyond 3 km from the monitoring track, and was considered only for those species known to aggregate around fishing vessels (*i.e.* not used for great cormorant, little gull, sandwich tern, common guillemot and razorbill). Finally, the continuous variable month (m) was included to model seasonal fluctuations by fitting a cyclic smoother or alternatively a cyclic sine curve, the latter described through a linear sum of sine and cosine terms (Stewart-Oaten & Bence 2001, Onkelinx *et al.* 2008). Seasonal patterns can often be successfully modelled applying a single sine curve with a period of 12 months, but in some cases adding another sine curve with a period of 6 or 4 months is needed, thus allowing to fit more than one peak in density per year and/or an asymmetric seasonal pattern. Eventually, we considered five different "full" models:

- no seasonal variation
  Y ~ BA + CI + F + OWF

- 12 month period sine curve
  Y ~ BA + CI + F + OWF
  + sin(2*π*m/12) + cos(2*π*m/12)

- 12 + 6 month period sine curve
  Y ~ BA + CI + F + OWF
  + sin(2*π*m/12) + cos(2*π*m/12)
  + sin(2*π*m/6) + cos(2*π*m/6)

- 12 + 4 month period sine curve
  Y ~ BA + CI + F + OWF
  + sin(2*π*m/12) + cos(2*π*m/12)
  + sin(2*π*m/4) + cos(2*π*m/4)

- cyclic smoother
  Y ~ BA + CI + F + OWF + s(m)

Model selection

In case of a randomly dispersed subject, count results tend to be Poisson-distributed, in which the mean equals the variance (McCullagh & Nelder 1989). Seabirds,

however, mostly occur strongly aggregated in (multi-species) flocks, resulting in "over-dispersed" count data which may demand the use of a negative binomial (NB) distribution (Ver Hoef & Boveng 2007; Zuur *et al.* 2009). When the data still exhibit more zeros than can be predicted through a NB distribution, it may be necessary to apply a zero-inflated (ZI) distribution (Potts & Elith 2006; Zeileis *et al.* 2008) which consists of two parts: (i) a "count component" modelling the data according to a Poisson or NB distribution and (ii) a "zero component" modelling the excess in zero counts.

As such, the five different full models were fitted to the "OWF footprint + 3 km" dataset (see fig. 3) applying these four different distributions (Poisson, NB, ZI Poisson, ZI NB). Based on the 20 resulting AIC values, the best fitting full model was selected, which was subsequently fitted to the "OWF footprint + 0.5 km" and "OWF buffer 0.5-3 km" datasets. In the results section we often refer to (i) the OWF coefficient, being the model coefficient of the OWF factor variable and the estimator of the displacement effect, and (ii) the estimated density, being the model prediction for a specific month and factor combination with the number of km^2 fixed at one. Densities in the impact area are further often referred to as X% lower or Y times higher than the "expected value". The latter should be regarded as the post-construction density that would have occurred in the impact area if the numbers had shown the same trend as observed in the control area. Also note that in all models used, the response is related to the linear sum of covariates through a log-link, implying that the OWF coefficient represents a factorial effect after transformation. A coefficient of 0 for example is transformed by calculating the exponential function *e* to the power 0, representing a factorial effect of 1, *i.e.* no effect. On the other hand, a coefficient of 1 is transformed by doing *e* to the power 1, equalling 2.718, implying that (according to the model) post-construction numbers inside

the OWF area are almost three times higher than expected based on the trend observed in the control area.

In this report we moved away from further model selection as applied in the previous reports (Vanermen *et al.* 2016, 2017), in which a specific set of nested models was considered and the best-fitting covariate combination was ultimately chosen. Here we used the full model OWF coefficient as a first estimator of the OWF displacement effect. But based on the model selections performed in aforementioned reports, we know that AIC differences between the best-fitting model on the one hand, and the full model or any nested model on the other hand may be very small (< 1), implying that there is actually more than one "good" model, each of them estimating the wind farm effect somewhat (or sometimes quite) differently. Interestingly, the differences in AIC values among a set of candidate models can be used to calculate relative model probabilities ("Akaike weights"), which in turn can be used to calculate a weighted average of any targeted model coefficient. In this study we used the full model and the nested alternatives for the "BA + CI + F" part of the model (always keeping the OWF factor and seasonality part into the model) to obtain a so-called multi-model inferred (MMI) OWF coefficient, as a "second opinion" displacement effect estimator (Burnham & Anderson 2002). For the MMI coefficient, no P-values were provided, but statistical significance could be deducted from the 95% confidence interval and whether or not this included zero (a coefficient of zero implying no effect).

2.2.5. Explorative INLA analysis

Finally, we explored the possibility of detecting displacement in post-construction data making use of the Bayesian "integrated nested Laplace approximation" (INLA) approach. In contrast to the BACI analyses above, we used raw count data (*i.e.* in their original resolution of geo-referenced one- to two-minute counts) allowing to account for spatial correlation. We built a non-convex hull mesh with both maximum edge and cut-off values set at 1.0 km and used a subset of (post-construction) count results of common guillemot, the most common and homogenously distributed seabird in our study area. Survey date was included in the models as a random intercept, and the area surveyed was log-transformed and included as an offset by defining a high-precision prior for its coefficient (a mean of 1 with a precision of 10,000). Another prior was given for the range of spatial correlation, which based on expert judgment was considered unlikely (P = 0.01) to be below 3 km.

Next, we ran Poisson models without spatial correlation, with spatial correlation and with spatial correlation replicated for each survey. Building on the latter, a fourth model included a three-level factor variable assigning counts to either the control, buffer or impact area. Unlike the polygons shown in fig. 3, the impact area in this analysis was confined to the footprint of the wind farm and the buffer area to a 2 km zone outside this footprint. All counts outside these polygons were regarded as control counts.

The resulting models were compared based on the Watanabe-AIC (wAIC) and interpreted by plotting the spatial random field or evaluating the fixed model coefficients. For further model evaluation, simulation exercises were performed to compare the models' data distribution with the original data distribution.

2.3. Statistical software

All data handling and modelling was performed through R.3.5.2 (R Core Team 2018a) in RStudio version 1.1.383 (RStudio Team 2016), making use of the following (randomly ordered) packages: RODBC (Ripley & Lapsley 2017), foreign (R Core Team 2018b), date (Therneau *et al.* 2018), ggplot2 (Wickham 2016), compare (Murrell 2015), reshape (Wickham 2007), MASS (Venables

& Ripley 2002), mgcv (Wood 2011), glm-mADMB (Skaug *et al.* 2016), Distance (Miller 2017), mrds (Laake *et al.* 2018), gridExtra (Auguie 2017), MuMIn (Barton 2018), rgdal (Bivand *et al.* 2018), spatialEco (Evans *et al.* 2017), lattice (Sarkar 2008), INLA (Lindgren & Rue 2015), tidyverse (Wickham 2017), plyr (Wickham 2011) and RColorBrewer (Neuwirth 2014).

3. Results

3.1. General observations

Since the Thornton Bank OWF became operational, most of the birds observed inside the OWF footprint area were gulls (91% of all non-passerines – see table 2). Most of these (82%) belong to one of the three "large gull" species, *i.e.* herring, lesser black-backed and great black-backed gull. With over 1800 individuals observed, great black-backed gull was by far the most numerous of all. This species showed a higher preference to the turbine foundations compared to the other two large gulls (89% *versus* 28% and 64% for lesser black-backed and herring gull, respectively). Cormorants too showed strong preference to the turbines, as 95% of the great cormorants and 76% of the European shags were observed roosting on the jacket or gravity-based foundations.

Although gannets and auks generally avoid the wind farm (Vanermen *et al.* 2017), these birds were regularly observed within the OWF boundaries. In total, we observed 47 northern gannets, 104 common guillemots and 59 razorbills.

3.2. Distance analysis

For all species except for great skua, hazard-rate detection models fitted our data better than the half-normal alternative (table 3). In general, either wave height or wind force proved to affect the detectability of seabirds significantly, except in great skua and both tern species. The (natural logarithm of) group size was retained for all species except for great skua. Detection probabilities of single birds during moderate observation conditions (wave height between 0.5 and 1 m or wind force of 4 Beaufort) were predicted to be highest and above 80% for conspicuous birds like great skua and northern gannet, and ranged between 0.54 and 0.71 for all other species. For great cormorants, the numbers observed per kilometre were not distance-corrected as for this species the response also included observations outside the 300 m transect for which detailed distance information was unavailable (see §2.2.1).

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Table 2. Numbers of birds and sea mammals observed inside the Thornton Bank wind farm during 873 km of surveying from October 2012 until December 2018

		Total number observed	Number present on turbines	Percentage present on turbines
BIRDS				
Northern fulmar	*Fulmarus glacialis*	1	0	
Northern gannet	*Morus bassanus*	47	0	
Great cormorant	*Phalacrocorax carbo*	150	143	95%
European shag	*Phalacrocorax aristotelis*	17	13	76%
Unidentified cormorant	*Phalacrocorax* sp.	3	1	33%
Eurasian sparrowhawk	*Accipiter nisus*	1	0	
Bar-tailed godwit	*Limosa lapponica*	1	0	
Arctic skua	*Stercorarius parasiticus*	1	0	
Little gull	*Hydrocoloeus minutus*	35	0	
Black-headed gull	*Chroicocephalus ridibundus*	18	0	
Common gull	*Larus canus*	154	3	2%
Lesser black-backed gull	*Larus fuscus*	732	206	28%
Herring gull	*Larus argentatus*	210	135	64%
Yellow-legged gull	*Larus michahellis*	2	0	
Great black-backed gull	*Larus marinus*	1819	1610	89%
Unidentified large gull	*Larus* sp.	577	498	86%
Black-legged kittiwake	*Rissa tridactyla*	528	3	1%
Sandwich tern	*Thalasseus sandvicensis*	19	0	
Common tern	*Sterna hirundo*	1	0	
Common guillemot	*Uria aalge*	104	0	
Unidentified auk	*Alca torda* or *Uria aalge*	20	0	
Razorbill	*Alca torda*	59	0	
Domestic pigeon	*Columba livia "domestica"*	1	0	
Common starling	*Sturnus vulgaris*	122	3	2%
Passerines		28	3	11%
MAMMALS				
Harbour porpoise	*Phocoena phocoena*	6	0	
Grey seal	*Halichoerus grypus*	1	0	

Table 3. Results of the multi-covariate distance analysis for the seabird species targeted in this study; the species-specific detection probabilities account for single birds during moderate observation conditions (wave height between 0.5 and 1 m or wind force of 4 Beaufort)

Species	Detection function	Covariates	Detection probability (single bird)
Northern fulmar	Hazard-rate	log(group size) + wave height	0.54
Northern gannet	Hazard-rate	log(group size) + wave height	0.85
Great skua	Half-normal	/	0.84
Little gull	Hazard-rate	log(group size) + wind force	0.58
Common gull	Hazard-rate	log(group size) + wind force	0.56
Lesser black-backed gull	Hazard-rate	log(group size) + wind force	0.60
Herring gull	Hazard-rate	log(group size) + wind force	0.58
Great black-backed gull	Hazard-rate	log(group size) + wind force	0.71
Black-legged kittiwake	Hazard-rate	log(group size) + wave height	0.55
Sandwich tern	Hazard-rate	log(group size)	0.67
Common tern	Hazard-rate	log(group size)	0.59
Common guillemot	Hazard-rate	group size + wind force	0.54
Razorbill	Hazard-rate	log(group size) + wind force	0.58

3.3. BACI modelling results

3.3.1. Northern fulmar

During the operational phase of the Thornton Bank OWF, the number of northern fulmars was low both in the control and impact area (fig. 4, right panel), in line with an overall decrease in densities in the BPNS. Within the "OWF footprint + 0.5 km" area, no birds were observed at all after wind farm construction, explaining the empty space in the left panel of fig. 4 and the extreme values in table 4 (a strongly negative OWF coefficient of -28.65 combined with a

rounded P-value of 1.000). For both the "OWF footprint + 3 km" and "buffer 0.5 - 3 km" areas, the OWF coefficients were negative (-1.44 and -0.77) yet not significantly different from zero, and the same accounted for the MMI coefficient of -1.49 for the buffer area. In conclusion, despite good indications of avoidance of at least the wind farm footprint area no statistically significant effect of the Thornton Bank OWF on the numbers of northern fulmar could be detected.
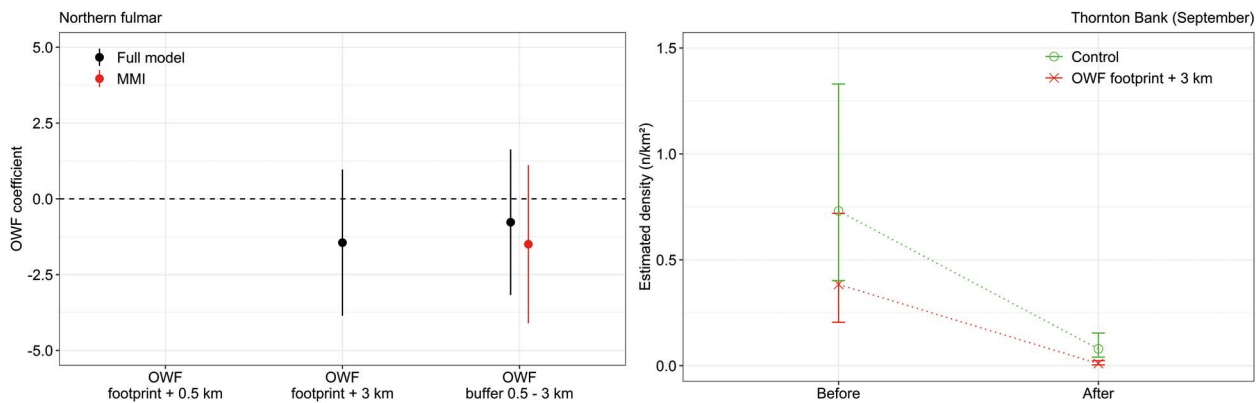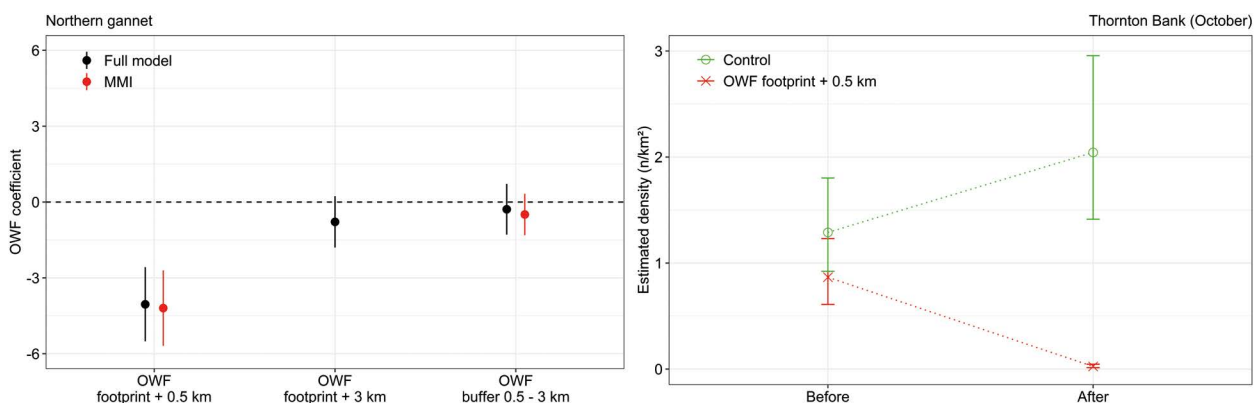

Figure 4. Modelling results for northern fulmar in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (±SE) for the month with maximum numbers on the right.

3.3.2. Northern gannet

Results for northern gannets showed strongly negative and significant MMI and full model OWF coefficients of -4.20 and -4.05 respectively (both implying 98% lower numbers than expected) for the "OWF footprint + 0.5 km" area (fig. 5). OWF coefficients for the buffer area were much less

pronounced and not significant, with values of -0.49 and -0.29 estimated through the MMI and full model respectively. Concluding, northern gannets showed strong displacement of the Thornton Bank OWF footprint area, yet this effect was no longer apparent in the surrounding buffer zone.
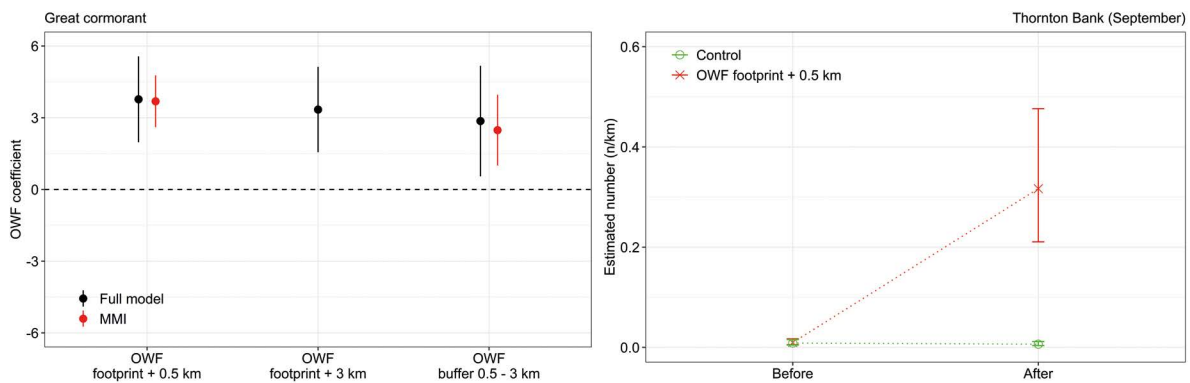

Figure 5. Modelling results for northern gannets in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the month with maximum numbers on the right.

3.3.3. Great cormorant

With only 9 observations of 1 to 4 birds prior to April 2008, great cormorants were virtually absent in the study area before wind farm construction. Post-construction observations were much more numerous but largely limited to turbine-associated birds in the impact area (143 individuals, see table 2) and to only 6 observations of 1 to 5 birds in the control area. Not unexpectedly, this species showed strongly positive and significant displacement coefficients. For the footprint area, MMI and full model OWF coefficients equalled 3.69 and 3.77, representing an increase in numbers with factors 40 and 43 respectively. For the buffer zone, a 17-fold increase in numbers was predicted by the full model (OWF coefficient = 2.86). Great cormorants were thus found to be strongly attracted to the Thornton Bank wind farm area and its immediate surroundings (fig. 6).
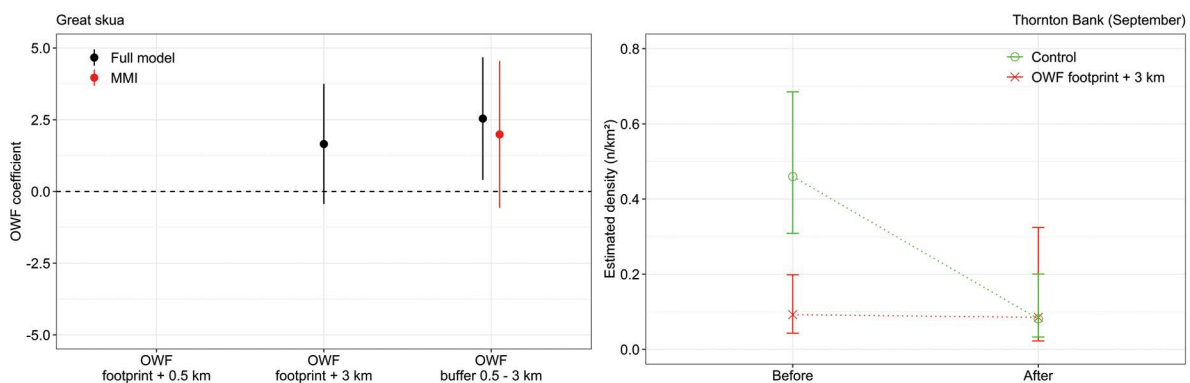

Figure 6. Modelling results for great cormorants in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI estimates of the number observed per km (± SE) for the month with maximum occurrence on the right.

3.3.4. Great skua

As for northern fulmar, no great skuas were observed inside the wind farm, hampering meaningful statistics and explaining the empty space in the left panel of fig. 7. All other coefficients were found positive, but only the full model OWF coefficient of 2.54 for the buffer area appeared to be statistically significant. The results thus indicate avoidance of the footprint area (no post-construction observations!) combined with attraction towards the immediate surroundings of the wind farm. But considering the generally very low encounter rate in the study area (0.014 birds per km²), combined with a lack of consistency between MMI and full model coefficient significance, the results for great skua should be interpreted with care and firm conclusions are difficult to draw.
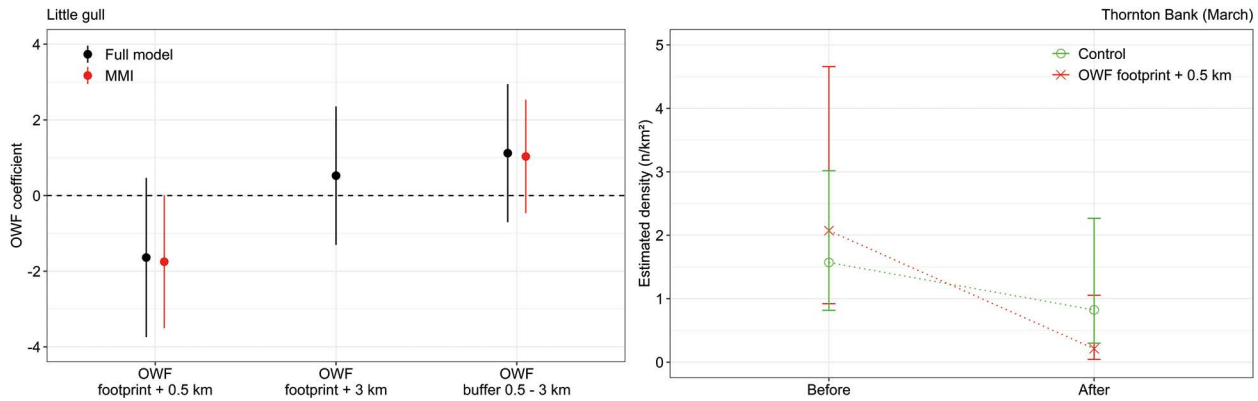

Figure 7. Modelling results for great skua in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the month with maximum numbers on the right (but note that the zero inflation of 71% is not accounted for in these estimates).

3.3.5. Little gull

As in previous reports, the results for little gull showed an interesting pattern of avoidance of the OWF footprint area (negative coefficients of -1.64 and -1.75 for the full model and MMI alternative), opposed to attraction to the surrounding buffer zone (positive coefficients of 1.12 and 1.03) (fig. 8). Neither full model nor MMI OWF coefficients, however, proved statistically significant, leaving the results for little gull inconclusive.
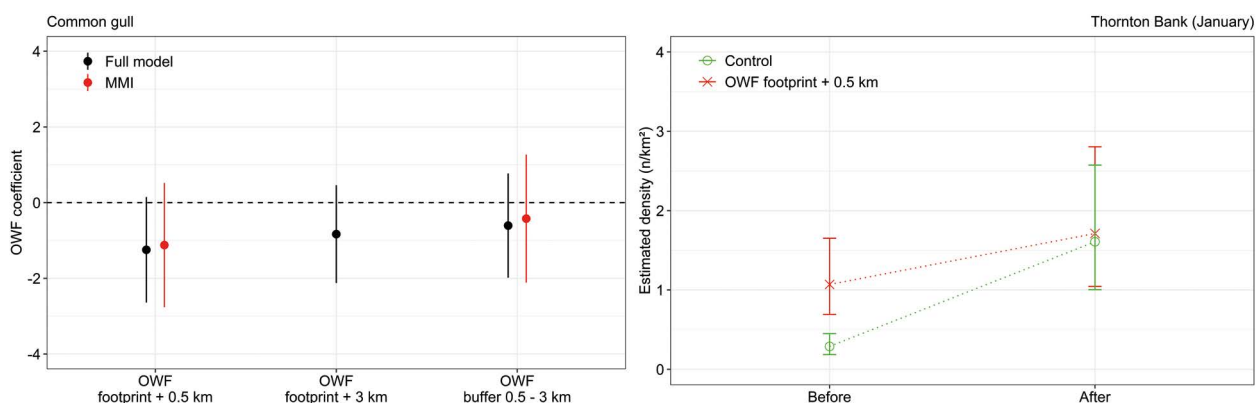

Figure 8. Modelling results for little gull in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (±SE) for the month with maximum numbers on the right.

3.3.6. Common gull

Between the reference and impact period, numbers of common gull in the study area increased (fig. 9, right panel). This increase, however, was less prominent in the wind farm area and its immediate surroundings, resulting in negative full model OWF coefficients ranging between -1.25 and -0.61 for all three data selections. None of these, however, nor the highly similar MMI coefficients, significantly differed from zero and no firm conclusions on the effect of the Thornton Bank OWF on the presence of common gulls can therefore be drawn.
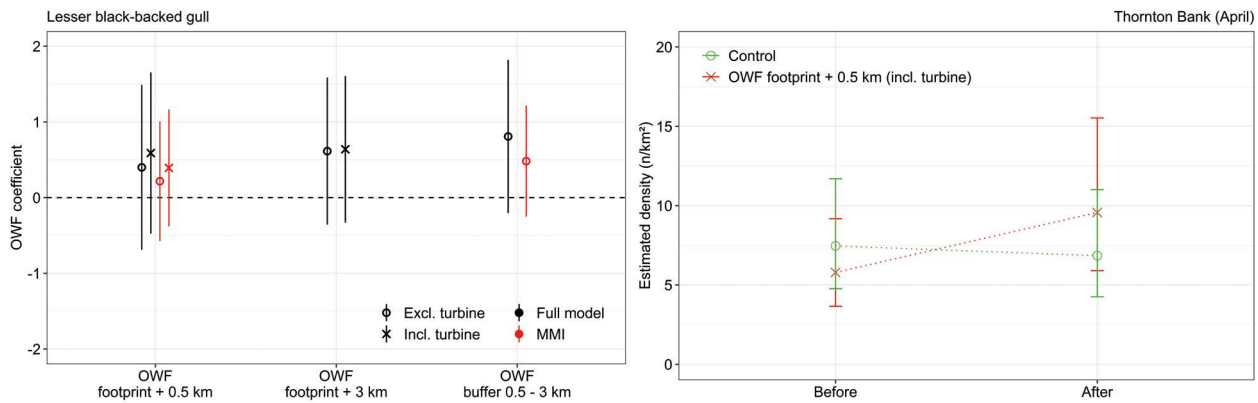

Figure 9. Modelling results for common gull in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the month with maximum numbers on the right.

3.3.7. Lesser black-backed gull

The full model OWF coefficients for lesser black-backed gull were all found to be above zero, ranging between 0.40 and 0.81, with little variation caused by including or excluding turbine-associated birds. The MMI coefficients were slightly lower with values between 0.22 and 0.48 (fig. 10). None of these coefficients, however, significantly differed from zero, leaving the results for lesser black-backed to be inconclusive.
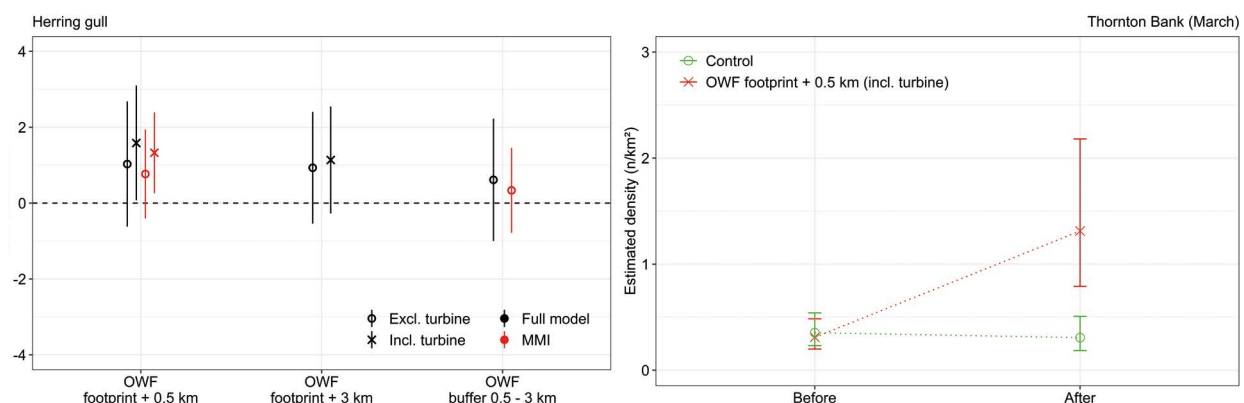

Figure 10. Modelling results for lesser black-backed gull in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) including turbine-associated birds for the month with maximum numbers on the right.

3.3.8. Herring gull

Positive MMI and full model OWF coefficients were found for all data selections, suggesting an overall attraction of herring gulls to the wind farm area (fig. 11). These effects were generally not statistically significant, except when modelling the adjusted response (including turbine-associated birds) for the "OWF footprint + 0.5 km" area. For the latter, the full model predicts a density 4.9 times higher compared to the value expected based on the trend in the control area (OWF coefficient = 1.58). The MMI alternative prediction also differed significantly from zero, representing a factorial effect of 3.8 (MMI OWF coefficient = 1.33). In conclusion, a significant attraction effect of herring gulls to the footprint area of the Thornton Bank wind farm could be demonstrated, provided birds associated with the turbines are included in the analysis.
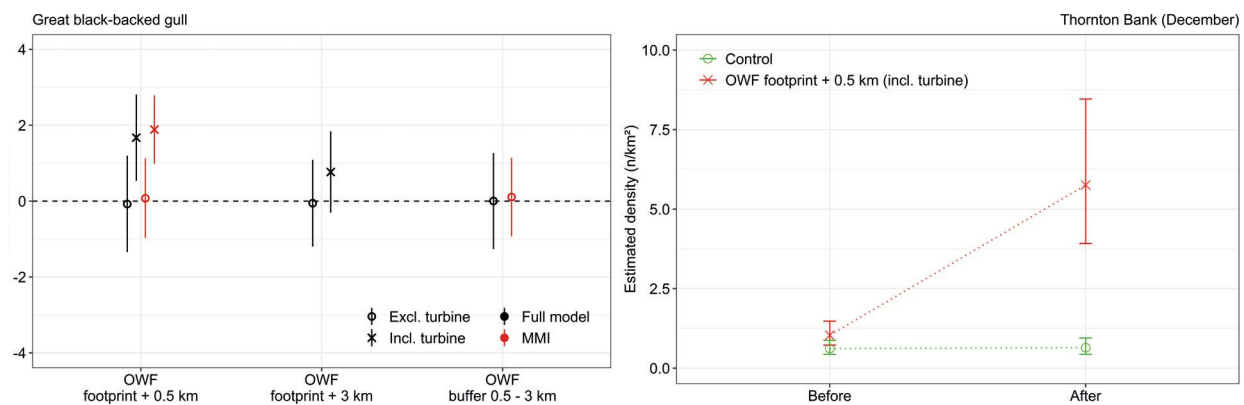

Figure 11. Modelling results for herring gull in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) including turbine-associated birds for the month with maximum numbers on the right.

3.3.9. Great black-backed gull

Highly positive and significant OWF coefficients were found for great black-backed gull occurrence inside the wind farm footprint, provided birds roosting on the turbines were included in the analysis (*i.e.* applying the adjusted response variable). For the footprint area, the full model OWF coefficient equalled 1.67, implying a 5.3 times higher density than expected, with the MMI coefficient

being even more pronounced with a value of 1.89 and a factorial effect of 6.6 (fig. 12). Interestingly, model coefficients when not including turbine-associated birds as well as those obtained for the buffer area all approached zero, emphasising the important role of turbine-association in the observed attraction effect of great black-backed gulls towards the Thornton Bank wind farm.
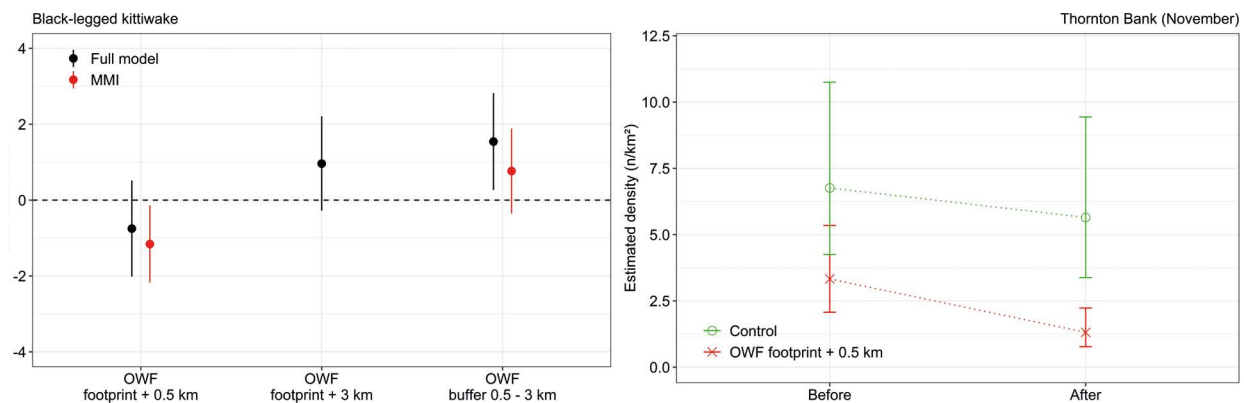

Figure 12. Modelling results for great black-backed gull in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) including turbine-associated birds for the month with maximum numbers on the right.

3.3.10. Black-legged kittiwake

Results for this species showed comparison with those for little gull, indicating avoidance of the footprint area opposed to attraction to the buffer zone (fig. 13). For the footprint area, only the MMI effect appeared significant (MMI coefficient = -1.16, *i.e.* a 69% decrease), while for the buffer area

this was exclusively so for the full model OWF coefficient (= 1.54, *i.e.* a 4.7 factorial increase). Due to inconsistency between the significance levels of the MMI and full model OWF coefficients, the results for black-legged kittiwake should yet be regarded as inconclusive.
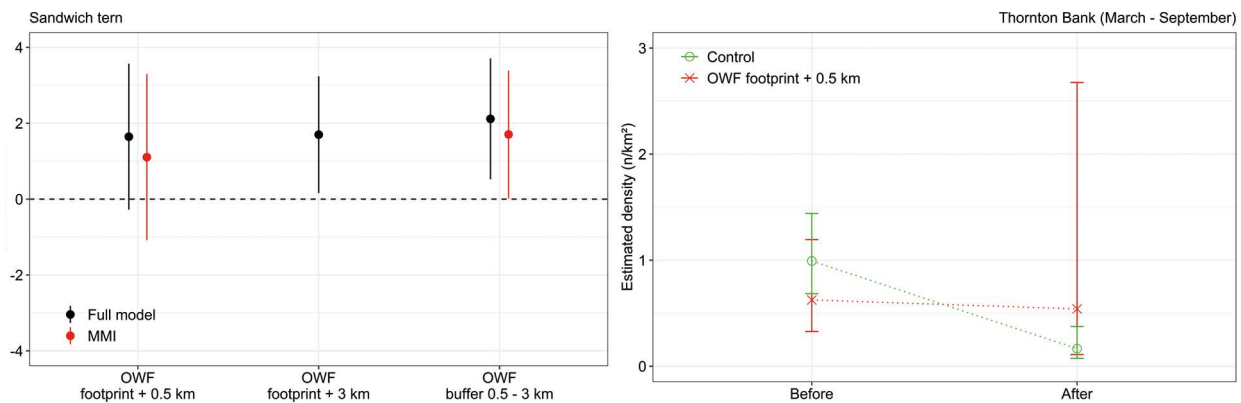

Figure 13. Modelling results for black-legged kittiwake in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the month with maximum numbers on the right.

3.3.11. Sandwich tern

Due to fitting problems, we only used Sandwich tern data collected from March till September, and no longer considered seasonal variation in the model. Sandwich terns showed a steady trend in the impact area compared to a strong decrease in the control area, resulting in positive OWF coefficients for all three data selections, varying between 1.10 and 2.11 (fig. 14). Only for the buffer zone, the full model OWF coefficient of 2.11 and the MMI coefficient of 1.70 were significantly different from zero, corresponding to factorial changes of 8.2 and 5.5 respectively. The results for Sandwich tern thus suggest attraction at least to the wind farm surroundings, yet should be interpreted with care considering the low number of positive counts after impact (only 10 post-construction observations of 1 up to 17 birds per observation).

Figure 14. Modelling results for Sandwich tern in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the period March to September on the right (but note that the zero inflation of 77% is not accounted for in these estimates).

3.3.12. Common guillemot

For the "OWF footprint + 0.5 km" area, significantly negative values of -0.92 and -1.00 (corresponding to 60 and 63% decreases) were found for the full model and MMI OWF coefficient respectively (fig. 15). In the buffer zone, these coefficients approached zero. The results for common guillemot thus demonstrate clear displacement of the Thornton Bank wind farm itself, but no such effect in the surrounding buffer zone.
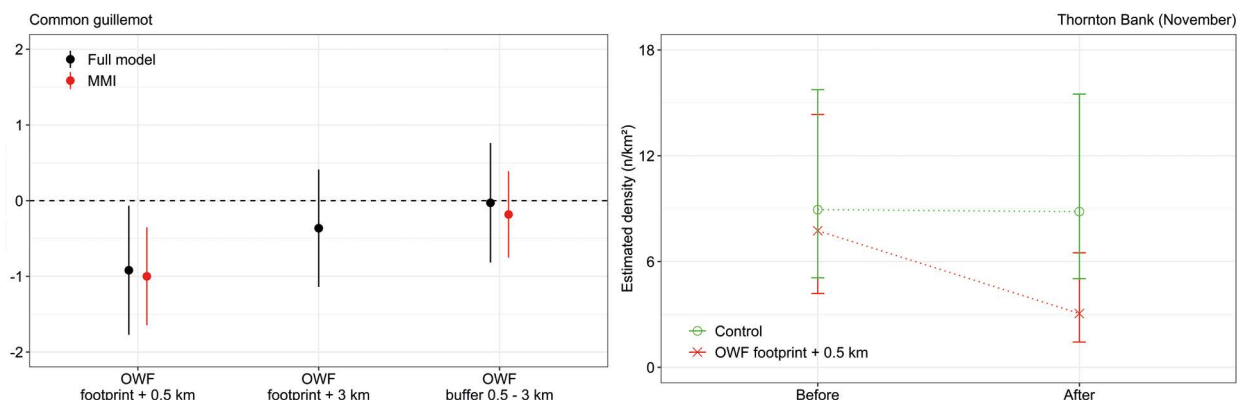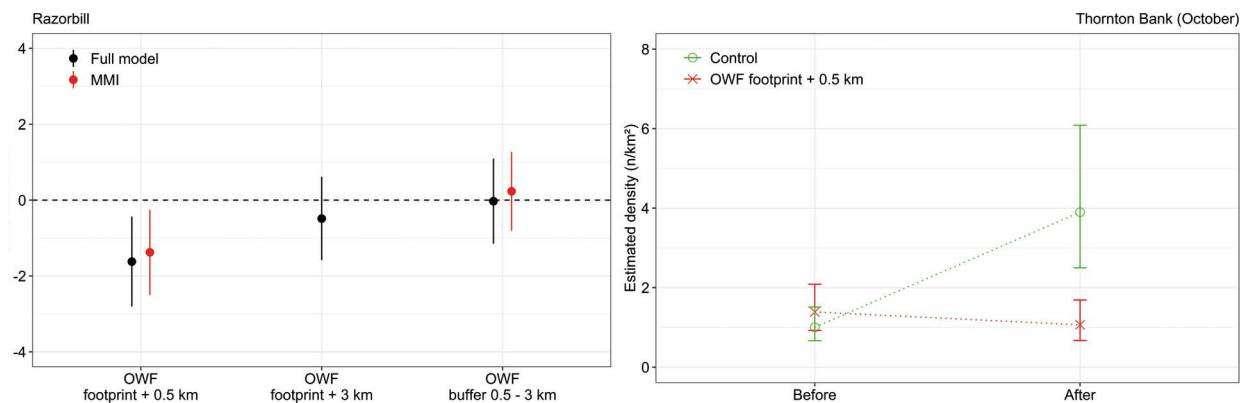

Figure 15. Modelling results for common guillemot in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the month with maximum numbers on the right (but note that the zero inflation of 10% is not accounted for in these estimates).

3.3.13. Razorbill

For razorbill, a highly comparable pattern was found as for common guillemot, with significantly negative footprint coefficients opposed to buffer coefficients approaching zero (fig. 16). The coefficients for the "OWF footprint + 0.5 km" area were estimated at -1.38 and -1.62 through MMI and the full model respectively, corresponding to 75% and 80% lower numbers than expected. As for common guillemot, a clear displacement of razorbills occurred from the Thornton Bank wind farm itself, with no observable effect in the 0.5-3 km buffer zone.

Figure 16. Modelling results for razorbill in the Thornton Bank study area with OWF coefficients and their 95% confidence intervals on the left and BACI density estimates (± SE) for the month with maximum numbers on the right.

3.3.14. Summarising tables

Our BACI monitoring results are summarised in table 4, listing the OWF coefficients and corresponding P-values as estimated throughout the modelling process. The full range of impact model coefficient estimates is listed in table 6 in the appendix.

After six years of post-impact monitoring at the Thornton Bank OWF, the impact area appeared to be avoided by three species, *i.e.* northern gannet, common guillemot and razorbill. In the "OWF footprint + 0.5 km" area, these species dropped in numbers by 98%, 60-63% and 75-80% respectively (the ranges originating from variation between the full model and MMI coefficient estimates). Strikingly, none of these three species appeared to be displaced from the wind farm buffer zone 0.5-3 km away from the wind farm edge. Black-legged kittiwakes too showed avoidance of the wind farm footprint area, yet only the MMI coefficient proved statistically significant.

Attraction to the wind farm footprint area could be demonstrated for herring and great black-backed gulls, for which the BACI models predicted a factorial change in densities of 3.8-4.9 and 5.3-6.6 respectively. These results only account for models including turbine-associated birds by applying the adjusted response variable. Both species were indeed often observed associated with the turbines (table 2). Occurrence of great cormorants too was concentrated on or near the jacket-turbine foundations, and despite their rather low numbers, the species showed major attraction effects and an increase with a factor 40-43 (amplified by the fact that great cormorants were quasi-absent in the study area before wind farm construction). Except for great cormorants aforementioned attraction effects were limited to the wind farm footprint area.

Table 4. BACI monitoring results for the C-Power wind farm at the Thornton Bank after 6 years of operation, listing full model OWF coefficients (and their P-values) and MMI coefficients; model results based on the adjusted response variable for the three large gull species are indicated by "(T)" in the species column; cells in bold indicate a statistically significant model coefficient, while coloured text indicates consistency in significance between the full model and MMI coefficient

	OWF footprint + 0.5 km			OWF footprint + 3 km		OWF buffer 0.5-3 km		
SPECIES	Full model	P-Value	MMI	Full model	P-Value	Full model	P-Value	MMI
Northern fulmar	-28.65	1.000	/	-1.44	0.240	-0.77	0.529	-1.49
Northern gannet	**-4.05**	**0.000**	**-4.20**	-0.78	0.129	-0.29	0.573	-0.49
Great cormorant	**3.77**	**0.000**	**3.69**	**3.34**	**0.000**	**2.86**	**0.015**	**2.48**
Great skua	-114.17	0.998	/	1.66	0.121	**2.54**	**0.020**	1.99
Little gull	-1.64	0.127	-1.75	0.52	0.574	1.12	0.229	1.03
Common gull	-1.25	0.079	-1.12	-0.83	0.207	-0.61	0.387	-0.42
Lesser black-backed gull	0.40	0.471	0.22	0.61	0.215	0.81	0.117	0.48
Lesser black-backed gull (T)	0.59	0.278	0.39	0.64	0.196			
Herring gull	1.03	0.221	0.77	0.93	0.216	0.61	0.455	0.34
Herring gull (T)	**1.58**	**0.040**	**1.33**	1.14	0.114			
Great black-backed gull	-0.08	0.907	0.07	-0.05	0.926	0.00	0.998	0.11
Great black-backed gull (T)	**1.67**	**0.004**	**1.89**	0.77	0.160			
Black-legged kittiwake	-0.75	0.245	**-1.16**	0.96	0.129	**1.54**	**0.018**	0.77
Sandwich tern	1.64	0.094	1.10	**1.70**	**0.031**	**2.11**	**0.009**	**1.70**
Common guillemot	**-0.92**	**0.034**	**-1.00**	-0.36	0.357	-0.03	0.943	-0.18
Razorbill	**-1.62**	**0.007**	**-1.38**	-0.49	0.387	-0.03	0.960	0.23

Interestingly, some species showed (indications of) attraction to the surrounding buffer area, *i.e.* great skua, little gull, black-legged kittiwake and Sandwich tern, which except for the latter coincided with (indications of) avoidance of the footprint area. The OWF coefficients obtained for these species, however, were mostly not significant. Only for Sandwich tern we found consistency in statistical significance between the full model and MMI coefficients estimating the observed attraction to the wind farm buffer zone.
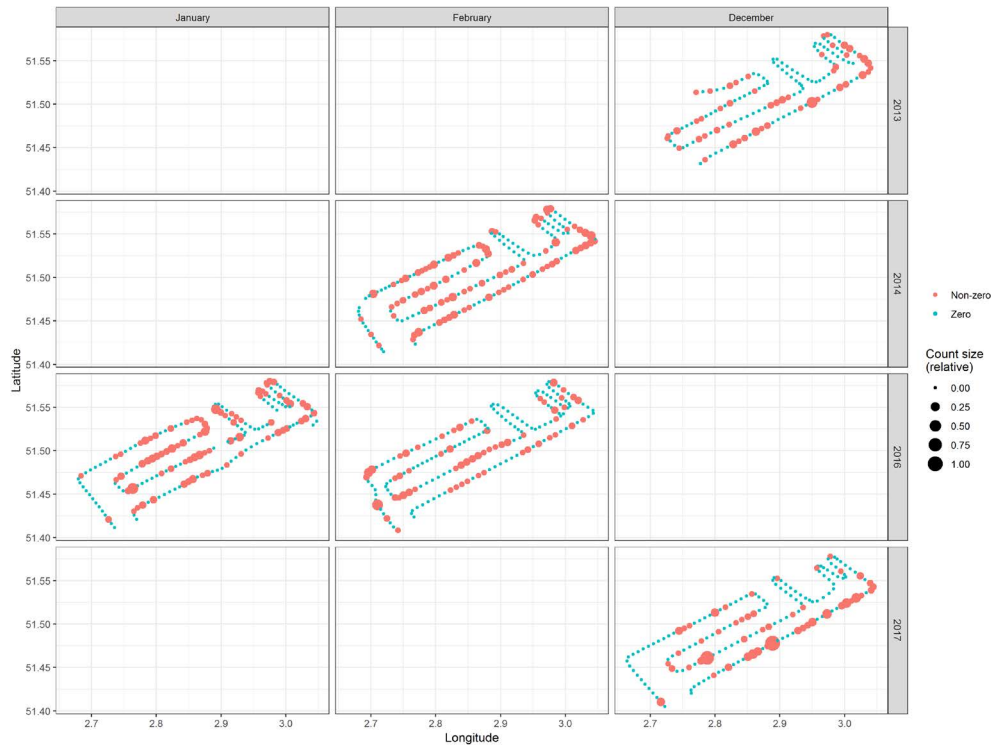
3.4. Explorative INLA analysis

3.4.1. Data selection

Five post-construction surveys were selected based on their coverage of the study area, the total number of guillemots counted and the proportion of non-zero counts. As such, numbers counted inside the transect varied from 93 to 173 and the proportion of non-zero counts between 29 and 46% (fig. 17).

3.4.2. INLA model results

We fitted four different Poisson models using INLA: one without spatial correlation, one with spatial correlation (estimated across surveys), one with replicated spatial correlation (estimated for each survey) and a last one with replicated spatial correlation including an area factor assigning counts to either the control, buffer or impact area. Doing so, we observed strong successive decreases in the wAIC values of the fitted models (table 5) implying that

Figure 17. Count results for common guillemot during the five post-construction surveys used in the explorative INLA analysis (coloured red symbols indicate positive observations and symbol size indicates count size relative to the maximum single count of 24 guillemots in the data subset).
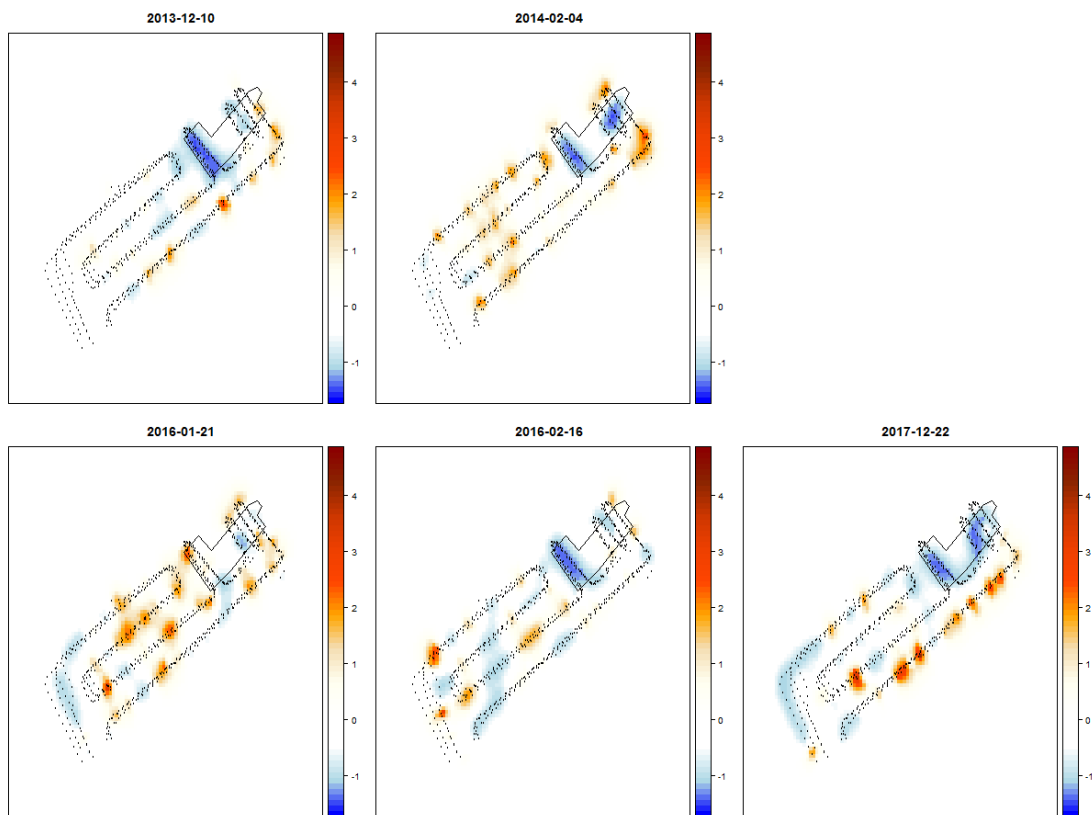
Figure 18. Spatial random field values (projected on a 100 x 100 dimension grid) as obtained through the "Poisson + repl. SRF" model of common guillemot numbers encountered during five surveys in the Thornton Bank study area (with the wind farm border indicated by the black polygon).

Table 5. INLA models results in terms of wAIC values (SRF = spatial random field)

Model type	wAIC
Poisson	2604.2
Poisson + SRF	2205.0
Poisson + replicated SRF	1839.7
Poisson (incl. area factor) + replicated SRF	1822.1

accounting for spatial correlation, especially when replicated for the different surveys, resulted in major model improvement.

A distribution plot of the "Poisson + repl. SRF" spatial random field across the study area (fig. 18) can be regarded as a heat map of unexplained variance, with blue areas (~ negative values of the spatial random field) delineating zones where occurrence appeared to be lower than expected by the (fixed part of the) model, whereas red areas highlight zones with higher occurrence. Looking at fig. 18, interestingly, this pattern of hot and cold spots varies strongly between surveys, except for the clear and recurrent cold spots prevailing inside the wind farm boundaries in 4 out of 5 surveys, which provides strong indication of a wind farm displacement effect.

To quantitatively estimate this wind farm displacement effect, we plotted density plots of the spatial random field values obtained at grid locations falling within the wind farm boundaries (fig. 19). These plots show that most values are indeed below zero, with a mean of -0.60 across surveys, representing 45% lower numbers of common guillemot inside compared to outside the wind farm.

When including an area factor in the model, the recurrent cold spots inside the wind farm are no longer visible when plotting the spatial random field (fig. 20), as the effect is absorbed by the factor variable in the fixed part of the model. Instead, we obtained an "important" coefficient value of -1.50 for the impact area, corresponding to 78% lower numbers. Note that in Bayesian analyses, a coefficient is regarded as "important" when the 95% credible interval of its posterior probability distribution does not contain zero (which would imply no effect) (fig. 21).

As a final step, we performed a simulation exercise to compare the models' data distribution with the original data distribution. For this specific "high count" data subset, a Poisson distribution performed quite well in predicting the number of zeros present in the original data (fig. 22).
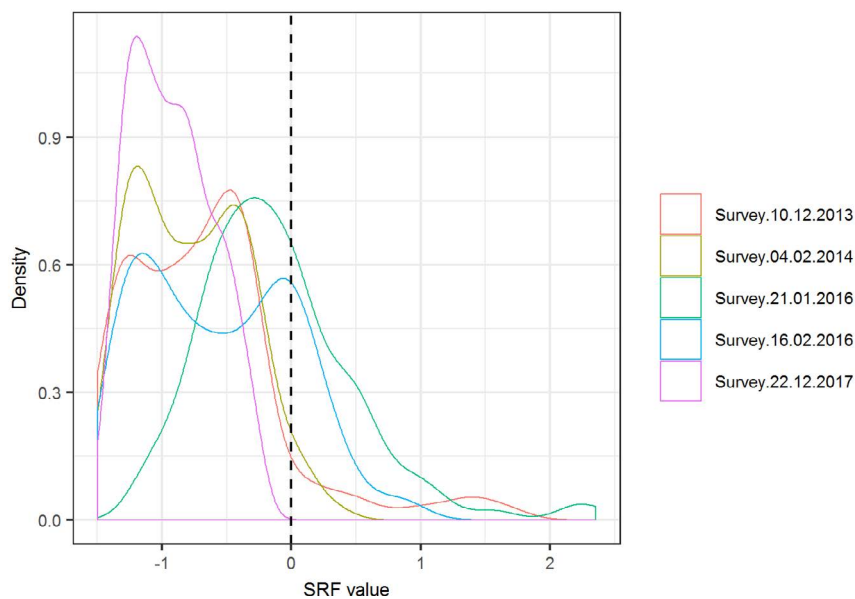
Figure 19. Density-plots of the spatial random field values within the wind farm boundaries for each of the five surveys.
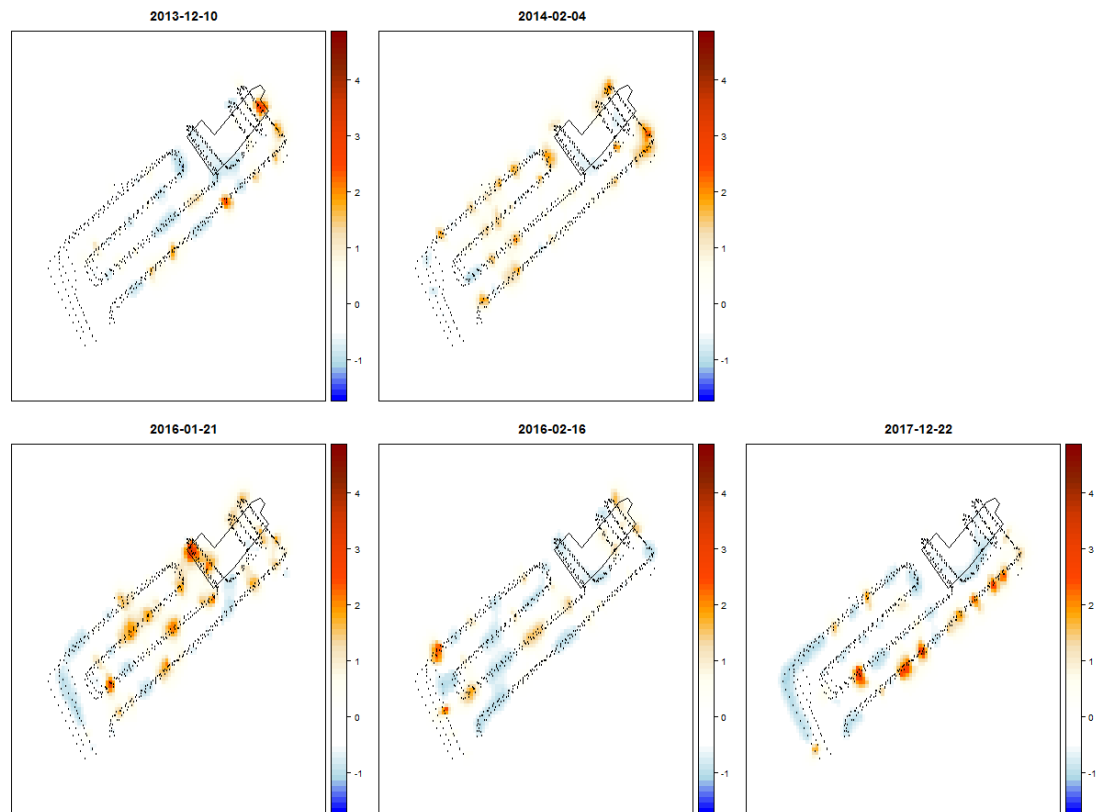
Figure 20. Spatial random field (projected on a 100 x 100 dimension grid) as obtained through the "Poisson (incl. area factor) + repl. SRF" model of common guillemot numbers encountered during five surveys in the Thornton Bank study area (with the wind farm border indicated by the black polygon).
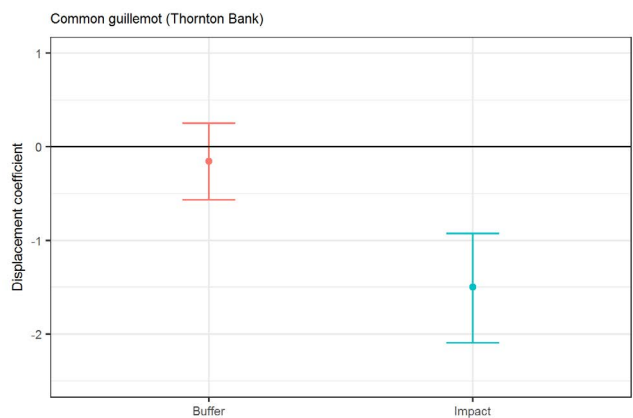
Figure 21. Posterior mean coefficient estimates for the area factor as estimated by the "Poisson (incl. area factor) + repl. SRF" model with indication of the 95% credible interval.
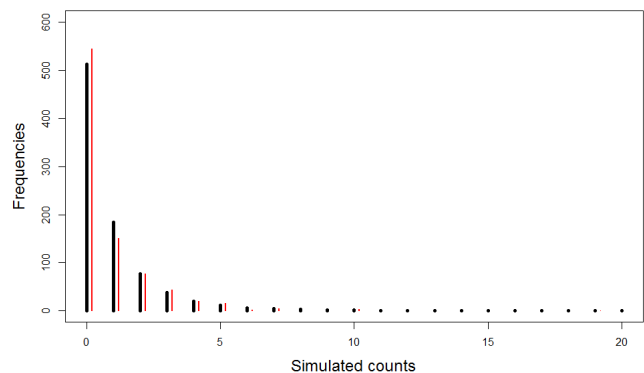
Figure 22. Average frequency table of simulated bird counts based on the "Poisson (incl. area factor) + repl. SRF" model (200 simulations – black bars) opposed to the original data histogram (red bars).

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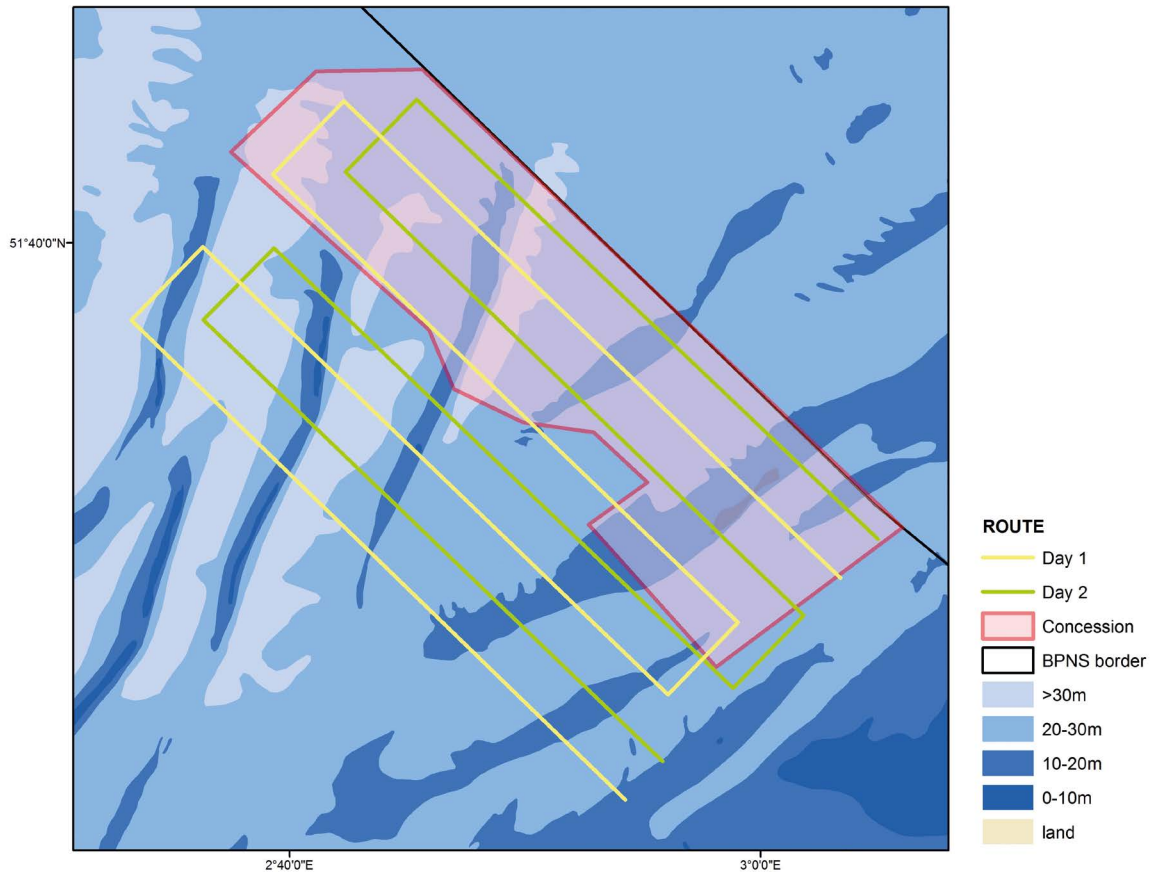
Figure 23. Proposal for a two-day concession-wide seabird displacement monitoring scheme.

3.4.3. Conclusions and future prospects

Depending on the strategy followed, modelling (a subset of) our data in their original resolution and taking into account the spatial correlation predicted common guillemot displacement levels of 45 to 78%, which is comparable to the 63% figure obtained through BACI modelling of day totals per area (yet applying the full dataset including pre-construction data). While the INLA modelling strategy probably needs some fine-tuning (*e.g.* by defining more informative priors), this exercise already illustrates how displacement can successfully be detected using a rather limited set of post-construction data. This offers interesting perspectives towards future monitoring, which will be reoriented towards assessing concession-wide displacement. Up until now our research focused on single isolated wind farm sites surrounded by 3 km buffer zones, but these buffer areas are now increasingly occupied by wind turbines of newly constructed, adjacent wind farms. From a seabird displacement monitoring perspective, the Belgian OWF concession zone should therefore better be considered as one large wind farm cluster from now on. Taking into account the vast area of the concession zone itself, and the need to monitor an equally wide reference area, we propose a two-day monitoring scheme as illustrated in fig. 23. Interestingly, this monitoring scheme can take full account of inshore-offshore gradients in seabird distribution (and possibly in displacement levels), and may also allow to detect other variability in species-specific displacement levels resulting from differences in wind farm configuration.

4. Discussion

4.1. Local displacement results in a wider perspective

We have now conducted 5 and 6 years of post-construction monitoring in and around the Bligh Bank and Thornton Bank offshore wind farms respectively. Compared to the Thornton Bank OWF, the Bligh Bank OWF is located further offshore (46 km) and consists of smaller and more closely spaced 3 MW turbines built on monopile foundations. Yet the results of both displacement studies are highly similar (compare Vanermen *et al.* 2016). In both wind farm areas, northern gannet, common guillemot and razorbill showed a marked and significant decrease in numbers, while the opposite was observed for herring and great black-backed gulls. Only for lesser black-backed gulls, results differed markedly, with a strong attraction effect at the Bligh Bank opposed to non-significant, moderate attraction effects at the Thornton Bank. The estimated effect sizes too are surprisingly alike, *e.g.* avoidance estimates of 82% *vs.* 98% for northern gannet, 75% *vs.* 63% for common guillemot and 67% *vs.* 75% for razorbill, next to factorial increases of 4.3 *vs.* 3.8 for herring gull and 3.6 *vs.* 6.6 for great black-backed gull at the Bligh Bank and Thornton Bank respectively. Avoidance effects at the Thornton Bank were limited to the "footprint + 0.5 km" area, while at the Bligh Bank, common guillemots were also displaced from the "buffer 0.5-3 km" zone.

The wind farm displacement effects found in the BPNS correspond well to other European study results. Based on a comparison of 16 European studies (Vanermen & Stienen 2019), avoidance was reported in 7 out of 9 studies on northern gannet, and wherever detected, displacement seemed to be strong and comparable with Belgian results, *e.g.* reductions of 93% at PAWP, 74% at OWEZ, 95% at Greater Gabbard and 79% at Alpha Ventus (Leopold *et al.* 2013; APEM 2014; Welcker & Nehls 2016). For auks, displacement was confirmed in 5 out of 7 studies on common guillemot, in 6 out of 7 studies on razorbill and in 3 out of 3 studies on auks as a species group. Levels of change were generally less pronounced compared to Belgian results, yet still amounted to 45 and 24% decreases of common guillemots at the Dutch PAWP and OWEZ wind farms, an 80% reduction of razorbills at PAWP and a 75% lower number of auks (common guillemot + razorbill) at the German Alpha Ventus wind farm (Leopold *et al.* 2013; Welcker & Nehls 2016). Great black-backed gulls were found to be attracted to 5 out of 8 study sites, yet the increase in numbers was nowhere as high and pronounced as in the two Belgian wind farms. Lastly, for great cormorants, 4 out of 5 European studies demonstrated strong attraction effects. Results for herring gull, on the other hand, showed inconsistency between European studies, as was the case for quite a number of (mainly gull) species.

In conclusion, offshore displacement studies have thus shown quite good consistency in results for at least some species, opposed to major differences in study outcomes for others. Local parameters such as food abundance in and outside the impacted area, the location of the OWF relative to the colony or feeding grounds and wind farm configuration might all be important in explaining the observed (differences in) responses but are mostly not accounted for in current studies. There is also likely to be a seasonal aspect in a bird's reaction towards wind farms. The Robin Rigg study site for example is the only one included by Vanermen & Stienen (2019) where common guillemots are more abundant during the breeding season than during winter, and one of the few not finding an avoidance response. In winter, common guillemots occur widespread and can be expected to be quite flexible in compensating habitat loss by moving elsewhere to forage (Dierschke *et al.* 2016). Breeding common guillemots, however, become central-place foragers and are more constrained in terms of distribution and habitat choice. As resources become limited, individuals may be

prepared to take more risks and be more tolerant of the presence of wind farms (Hötker 2017). Another temporal aspect regarding displacement effects relates to the fact that OWFs are still a relatively new phenomenon, and in time seabirds might habituate to their presence (*e.g.* Drewitt & Langston 2006; Fox *et al.* 2006; Petersen & Fox 2007). The occurrence of common scoters in and around Horns Rev 2 provides a good example of potential habituation. Although assumed to be displaced based on the first three years after wind farm construction, Petersen & Fox (2007) reported common scoters to occur in numbers higher than ever before in the fourth year post-construction, with a maximum of no fewer than 4,624 birds within the wind farm footprint. It was hypothesised that this strong change in distribution suggesting habituation might as well have been due to an unknown shift in food abundance, again emphasising the importance of accounting for local factors such as food availability to increase the reliability of conclusions drawn on wind farm induced seabird displacement.

4.2. Displacement effect versus impact

Importantly, a displacement effect, *i.e.* a change in distribution or numbers, should not be regarded synonymous with a displacement *impact*, which rather refers to changes in fitness and survival (Masden *et al.* 2010a). Birds subject to OWF displacement might respond by flying or swimming around rather than entering the wind farm and/or by spending time searching for alternative foraging habitat, all implying increased energetic costs. The alternative foraging areas may prove to be of minor quality, or may be subject to increased inter- and intraspecific competition due to the inflow of displaced birds, resulting in decreased food intake. Initial behavioural displacement responses may thus eventually lead to decreased body condition, increased mortality and/or decreased productivity (Fox *et al.* 2006; Petersen *et al.* 2014). Importantly, displacement impact is likely to vary with the different stages in

the species' annual life cycle, *i.e.* breeding, non-breeding and dispersal/migration periods. During the breeding season, birds suffering from poor body condition due to the displacement may abandon ongoing breeding efforts or not even attempting to breed in a particular year or an even longer period (Furness 2013). Displacement from OWFs within the foraging range of a colony could also result in longer foraging flights, leading to decreased chick provisioning and/or increased chick predation both resulting in reduced chick survival (Searle *et al.* 2014). Crucially, seabirds commuting daily between their breeding colony and feeding grounds may interact with OWFs several times a day and are therefore suspected to be particularly vulnerable to displacement impact (Masden *et al.* 2010b), despite the possibility of being less susceptible to avoidance responses as outlined above. During the non-breeding season, seabirds are no longer constrained to feeding grounds near the colony. On the other hand, mortality of seabirds is often highest in winter, and displacement effects will be additive to other factors causing this annual survival bottleneck. Due to carry-over effects, displacement during winter might further lead to a poor body condition at the onset of the breeding season, again affecting productivity (Dierschke & Garthe 2006; Furness 2013).

The quantitative translation of a displacement effect into its ultimate impact in terms of reproductive success or chances of survival, however, is highly challenging (Fox *et al.* 2006). Crucial parameters to feed population models needed to unravel displacement impact on demographic parameters often lack empirical evidence (Green *et al.* 2016). In the case of seabirds, density-dependency, seabird population carrying capacities as well as spatio-temporal variation in habitat quality are all important knowledge gaps. Seabird colony size for example is considered to be limited by intraspecific competition for food within the foraging range (Furness & Birkhead 1984),

highlighting the potential ecological consequences of reducing the foraging habitat available to a specific colony. But while there is (some) evidence of density-dependency affecting seabird populations during the breeding season, there is virtually no mentioning of such mechanisms occurring during the non-breeding season, which is no surprise as seabirds are much more difficult to study in their extensive offshore winter ranges (Busch *et al.* 2015). The assessment of relative habitat quality is also key to reliably assess displacement impacts on seabirds as displacement from poor instead of high-quality habitat is anticipated having less impact (Furness 2013; Warwick-Evans *et al.* 2017). Another knowledge gap highlighted by Searle *et al.* (2014) is the relationship between adult body mass and expected survival over the following year, which would help to quantify aforementioned carry-over effects, and accordingly, there is very few empirical information on the effect of fledging chick mass on post-fledging survival.

While a single OWF generally affects only small numbers of birds relative to their population sizes, the cumulative impact of all current and future developments may be severe and potentially even greater than the sum of single wind farm impacts (Masden *et al.* 2010a). Displacement affecting birds during the non-breeding season at a location several thousands of kilometres away from the colony may have fitness consequences that only become apparent a few months later during the breeding season or even in the years thereafter. Ideally, a cumulative impact assessment therefore includes all existing and new wind farm developments within the year-round distribution range of the species or population under study. Several cumulative impact assessments based on individual-based models have been conducted so far (Topping *et al.* 2011; Searle *et al.* 2014; Warwick-Evans *et al.* 2017), all showing

that displacement effects may indeed have consequences for individual fitness and survival, which may ultimately lead to seabird population declines. The models applied, however, still rely on numerous assumptions on key seabird ecology aspects. Filling in the knowledge gaps highlighted in this discussion would support a more reliable assessment of the actual and cumulative ecological consequences of extensive offshore wind farm installations and should be the primary goal of future research.

Acknowledgements

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Appendix

Table 6. Impact model coefficients for all species studied in the Thornton Bank OWF study area (zero-inflation shown in the response scale, all other coefficients in the link scale)

Species	Impact polygon	Intercept	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	Sin (1/4yr)	Cos (1/4yr)	BA	CI	Fishery	OWF	Theta	Zero-inflation
Northern fulmar (GLM NB)	OWF footprint + 0.5 km	-1.37	-0.83	0.51					-2.39	-0.62	0.99	-28.65	0.08	
	OWF footprint + 3 km	-1.29	-0.98	0.11					-2.22	-0.64	0.76	-1.44	0.07	
	OWF buffer 0.5 - 3 km	-1.30	-0.99	0.12					-2.22	-0.64	0.76	-0.77	0.07	
Northern gannet (GAM NB)	OWF footprint + 0.5 km	-0.47	s(month)						0.46	-0.40	0.34	-4.05	0.32	
	OWF footprint + 3 km	-0.40	s(month)						0.03	-0.48	0.27	-0.78	0.31	
	OWF buffer 0.5 - 3 km	-0.43	s(month)						0.05	-0.46	0.29	-0.29	0.31	
Great cormorant (GLM NB)	OWF footprint + 0.5 km	-6.15	-0.03	0.86	0.11	-1.36			-0.29	0.12		3.77	0.47	
	OWF footprint + 3 km	-6.31	-0.10	1.18	0.09	-1.49			-0.29	0.16		3.34	0.46	
	OWF buffer 0.5 - 3 km	-6.53	0.39	1.85	-0.34	-1.81			-0.42	-0.25		2.86	0.16	
Great skua (ZIP)	OWF footprint + 0.5 km	-2.40	-1.50	-0.02					-1.68	-1.64	-0.26	-114.17		0.66
	OWF footprint + 3 km	-2.28	-1.51	0.16					-1.73	-1.61	-0.50	1.66		0.71
	OWF buffer 0.5 - 3 km	-2.29	-1.59	0.22					-1.67	-1.59	-0.57	2.54		0.73
Little gull (GLM NB)	OWF footprint + 0.5 km	-2.57	2.31	2.29	-1.90	-0.70			-0.65	0.28		-1.64	0.12	
	OWF footprint + 3 km	-2.65	2.30	2.40	-1.95	-0.70			-0.63	0.31		0.52	0.12	
	OWF buffer 0.5 - 3 km	-2.63	2.29	2.39	-1.94	-0.71			-0.63	0.30		1.12	0.12	
Common gull (GLM NB)	OWF footprint + 0.5 km	-4.35	2.53	3.15	-0.86	-0.28			1.72	1.31	0.50	-1.25	0.29	
	OWF footprint + 3 km	-4.38	2.39	3.26	-0.70	-0.40			1.61	1.39	0.20	-0.83	0.30	
	OWF buffer 0.5 - 3 km	-4.35	2.34	3.21	-0.57	-0.37			1.65	1.45	0.02	-0.61	0.26	
Lesser black-backed gull (GAM NB)	OWF footprint + 0.5 km	-0.31	s(month)						-0.08	-0.24	0.69	0.40	0.31	
	OWF footprint + 0.5 km (T)	-0.23	s(month)						-0.14	-0.27	0.65	0.61	0.33	
	OWF footprint + 3 km	-0.20	s(month)						-0.20	-0.36	0.67	0.81	0.30	
	OWF footprint + 3 km (T)	-0.31	s(month)						-0.09	-0.25	0.68	0.59	0.32	
	OWF buffer 0.5 - 3 km	-0.23	s(month)						-0.14	-0.27	0.65	0.64	0.33	

Species	Impact polygon	Intercept	Sin (1yr)	Cos (1yr)	Sin (1/2yr)	Cos (1/2yr)	Sin (1/4yr)	Cos (1/4yr)	BA	CI	Fishery	OWF	Theta	Zero-inflation
Herring gull (GLM NB)	OWF footprint + 0.5 km	-2.27	1.24	0.12					-0.14	-0.12	0.71	1.03	0.15	
	OWF footprint + 0.5 km (T)	-2.28	1.19	0.29					-0.31	-0.18	0.80	0.93	0.17	
	OWF footprint + 3 km	-2.39	1.38	0.46					-0.24	-0.24	0.99	0.61	0.15	
	OWF footprint + 3 km (T)	-2.28	1.24	0.17					-0.14	-0.13	0.72	1.58	0.18	
	OWF buffer 0.5 - 3 km	-2.26	1.17	0.29					-0.33	-0.18	0.79	1.14	0.19	
Great black-backed gull (GAM NB)	OWF footprint + 0.5 km	-1.93			s(month)				-0.11	0.49	1.46	-0.08	0.25	
	OWF footprint + 0.5 km (T)	-2.14			s(month)				-0.15	0.51	1.51	-0.05	0.29	
	OWF footprint + 3 km	-2.19			s(month)				-0.15	0.52	1.56	0.00	0.24	
	OWF footprint + 3 km (T)	-1.84			s(month)				0.04	0.52	1.48	1.67	0.31	
	OWF buffer 0.5 - 3 km	-1.95			s(month)				-0.07	0.53	1.44	0.77	0.32	
Black-legged kittiwake (GAM NB)	OWF footprint + 0.5 km	-0.14			s(month)				-0.18	-0.71	0.72	-0.75	0.24	
	OWF footprint + 3 km	-0.15			s(month)				-0.65	-0.92	0.48	0.96	0.22	
	OWF buffer 0.5 - 3 km	-0.11			s(month)				-0.70	-0.95	0.43	1.54	0.20	
Sandwich tern (ZINB)	OWF footprint + 0.5 km	-0.01	1.67	6.57	-1.27	-1.94			-1.79	-0.46		1.64	2.11	0.77
	OWF footprint + 3 km	-0.05	1.81	6.83	-1.41	-2.11			-1.93	-0.44	/	1.70	1.72	0.75
	OWF buffer 0.5 - 3 km	-0.04	1.89	6.95	-1.49	-2.14			-1.92	-0.45		2.11	1.73	0.77
Common guillemot (ZINB)	OWF footprint + 0.5 km	-2.79			s(month)				-0.01	-0.14		-0.92	0.91	0.10
	OWF footprint + 3 km	-2.97			s(month)				-0.14	-0.14		-0.36	0.91	0.11
	OWF buffer 0.5 - 3 km	-3.05			s(month)				-0.13	-0.15		-0.03	0.89	0.10
Razorbill (GAM NB)	OWF footprint + 0.5 km	-4.13			s(month)				1.36	0.32		-1.62	0.40	
	OWF footprint + 3 km	-4.90			s(month)				1.08	0.30		-0.49	0.41	
	OWF buffer 0.5 - 3 km	-4.82			s(month)				1.05	0.29		-0.03	0.40	

CHAPTER 9

THE INFLUENCE OF METEOROLOGICAL CONDITIONS ON THE PRESENCE OF NATHUSIUS' PIPISTRELLE (*PIPISTRELLUS NATHUSII*) AT SEA

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Abstract

Bats undertaking seasonal migration between summer roosts and wintering areas can cross large areas of open sea. Given the known impact of onshore wind turbines on bats, concerns were raised on whether offshore wind farms pose risks to bats. Better comprehension of the phenology and associated weather conditions of offshore bat migration will provide a science base for mitigating the impact of offshore wind turbines on bats. This study investigated the weather conditions linked to the occurrence of bats in an offshore wind farm in the Belgian part of the North Sea during autumn 2017. We installed seven ultrasonic recorders, registering the echolocation calls of bats, on seven different wind turbines. A total of 142 bat recordings were registered during 23 nights throughout the entire study period. All echolocation calls were identified as calls from the species Nathusius' pipistrelle *Pipistrellus nathusii*. Wind speed seemed to have a large influence on the presence of bats during the study period, with 87% of the detections when the wind speed was maximally 5 m/s. The wind direction is also important for the recorded bat activity at sea, with a clear peak in occurrence when wind originated from the East and the South East. Bat activity was further positively related to temperature and barometric pressure. This study sheds light on the meteorological conditions that favor bat activity at sea. The collection of more data and multivariate analysis would allow to make statistically sound conclusions about the most important variables that explain offshore bat activity in the Southern North Sea.

1. Introduction

Bats undertaking seasonal migration between summer roosts and wintering areas can cross large areas of open sea (Rodrigues *et al.* 2015). This is also the case in the Southern North Sea, where bats have been frequently recorded in the last years (*e.g.* Walter *et al.* 2007; Boshamer & Bekker 2008; Skiba 2009; Leopold *et al.* 2014;

Brabant *et al.* 2016; Lagerveld *et al.* 2017). Nathusius' pipistrelle (*Pipistrellus nathusii*) is the species that is most frequently reported at sea (Hüppop & Hill 2016; Lagerveld *et al.* 2017; Brabant *et al.* 2018), but also common noctules (*Nyctalus noctula*), parti-coloured bats (*Vespertilio murinus*) and Leisler's bats (*Nyctalus leisleri*) have been observed.

Given the known impact of onshore wind farms on bats, offshore wind farms (OWFs) also pose risks to bats (*e.g.* Rodrigues *et al.* 2015; Baerwald & Barclay 2014; Rydell *et al.* 2010; Voigt *et al.* 2012; Lehnert *et al.* 2014; Brabant *et al.* 2018). Because most OWFs are out of the foraging range of local bats, particularly migratory bats may be at risk. Sightings of bats are regularly reported from OWFs (fig. 1; *e.g.* Lagerveld *et al.* 2014; Hüppop & Hill 2016; Brabant *et al.* 2017). Lagerveld *et al.* (2014) reported that these occurrences are generally limited to periods with calm weather suitable for long-distance migration. Also, Hüppop and Hill (2016) hinted towards a weather-dependency of the offshore presence of bats at a research platform 45 km of the German coast. A better comprehension of the phenology and associated weather conditions would provide a science base for mitigating the impact of wind turbines on bats at sea.

This study investigated the weather conditions linked to the occurrence of bats in an OWF in the Belgian part of the North Sea (BPNS) during an autumn migration period. We were particularly interested in shedding a light onto those weather conditions needed for offshore bat migration.

2. Material and methods

The C-Power wind farm is located on the Thornton Bank in the BPNS at approximately 27 km from the nearest point at the Belgian coastline (fig. 2). The wind farm consists of 54 wind turbines and one offshore transformer platform. Six turbines have a capacity of 5 MW; the other 48 are 6.15 MW turbines. The turbines have a cut-in wind speed of 3.5 m/s, a rotor diameter of 126 m, and the hub height is approximately 94 m above sea level.

We installed seven ultrasonic recorders (batcorder 3.0/3.1 EcoObs Ltd., Germany), registering the echolocation calls of bats, on

Figure 1. Bat specimens that were sighted on offshore wind farms (OWFs) in spring 2019. Left picture: bat sp. roosting in the grate floor of a turbine in the Belgian Nobelwind OWF (8 April 2019); right picture: bat sp. roosting on the foundation of a turbine in the Belgian C-Power OWF (30 April 2019).
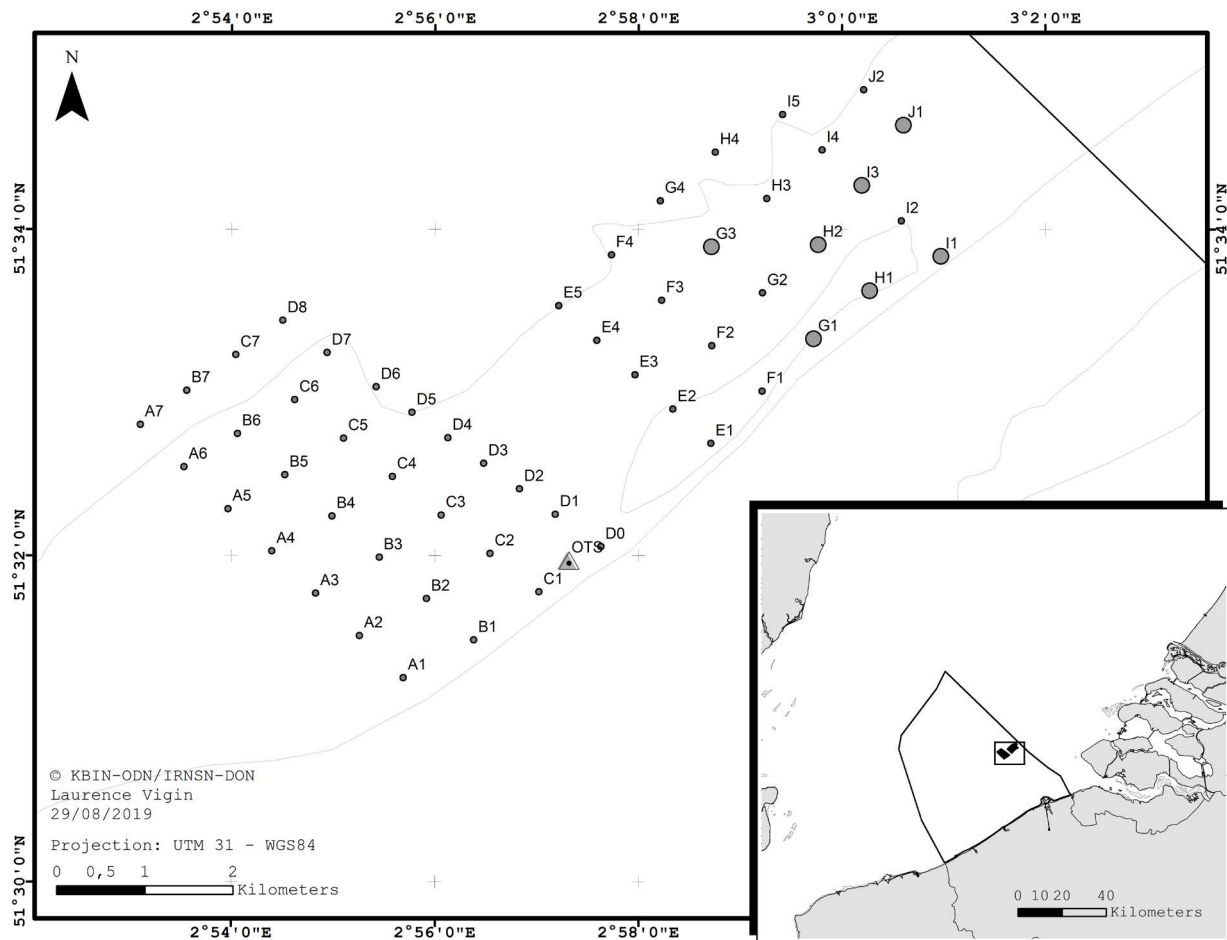
Figure 2. Lay-out of the C-Power wind farm on the Thornton Bank in the Belgian part of the North Sea. Each dot represents a wind turbine. Turbines G1, G3, H1, H2, I1, I3 and J1 (indicated by the large dots), in the North-East of the wind farm, were equipped with a batcorder on the transition piece (16 m amsl). Meteorological data were collected at the offshore transformation station (OTS).

seven different wind turbines in the C-Power wind farm (fig. 2). The batcorders were installed on the service platform of the turbines, at approximately 16 m above MSL. Each recorder was powered by a solar panel. The recorded data were locally stored on SD memory cards. The batcorders were installed on 8 August 2017 and were operational until 30 November 2017. We made full spectrum recordings in .RAW format (sampling rate: 500 kHz; record quality: 20; threshold amplitude (sensitivity): -36 dB; post trigger: 400 ms; threshold frequency (sensitivity): 30 kHz). A threshold frequency of 30 kHz was used to avoid wind turbine generated noise in the dataset. This setting does not allow to reliably sample Nyctaloid bats (*i.e.* a

species group that includes genera *Nyctalus*, *Vespertilio*, *Eptesicus*) that have a frequency of maximum energy (FME) lower than 30 kHz (Barataud 2015). Therefore, this study focused on pipistrelle bats, which are most frequently recorded offshore (see above).

Detections were processed and visualised with the software program Sonochiro 3.3.3 (Biotope, France). Automated species identifications were verified by a bat expert. 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 once are valued in the same way and render one DP10).

The meteorological variables wind speed, wind direction, visibility and atmospheric pressure were collected by C-Power at their offshore transformation station (fig. 2), with a 1- to 10 minute resolution. Air temperature data were retrieved from the Scheur Wielingen measuring pile, which is part of the Flemish banks monitoring network of the Agency for Maritime Services and Coast. For further analysis, the meteorological values nearest to the time of a bat detection, were linked to that detection. The number of bat recordings were normalised by dividing the number of records (DP10) by the frequency of occurrence of that meteorological variable (*e.g.* wind direction) during the study period.

We calculated the tailwind component (TWC) and crosswind component (CWC) to assess the influence of the wind on the recorded bat activity. Calculations were done following the methodology of Hüppop and Hilgerloh (2012):

- TWC = cos (observed wind direction - tailwind direction) x wind speed;

- CWC = sin (observed wind direction - tailwind direction) x wind speed.

The tailwind direction is defined as the direction of migration minus 180°. Positive TWC values mean tailwind, negative values are headwind. Positive CWC are winds from the left of the bat migrating into the presumably preferred direction, winds from the right are negative CWC.

Hüppop and Hill (2016) assumed a WSW direction of bat migration in autumn for similar calculations with data from the FINO1 platform in the German Bight based on recoveries of ringed Nathusius' pipistrelles (Pētersons 2004; Vierhaus 2004). Reproduction areas of Nathusius' pipistrelle are located in North-Eastern Europe, whereas the wintering areas are in the South-West

(Dietz & Kiefer 2016), this implies a SW migration during autumn. Therefore, we calculated TWC and CWC for two scenarios: (1) direction of flights to the WSW (*i.e.* bats crossing the North Sea towards the UK from the Dutch Zeeland coast, hereby passing the study site); and (2) SW direction of flight. Positive CWC mean in both cases offshore crosswinds.

3. Results

Bats were registered throughout the entire study period, from the end of August until the end of November, with a peak between the 23rd and 29th of September. A total of 142 bat recordings, equaling 68 DP10, were made by all seven batcorders. All echolocation calls were identified as calls from the species Nathusius' pipistrelle *Pipistrellus nathusii*. Bat activity was recorded during 22 nights (fig. 3).
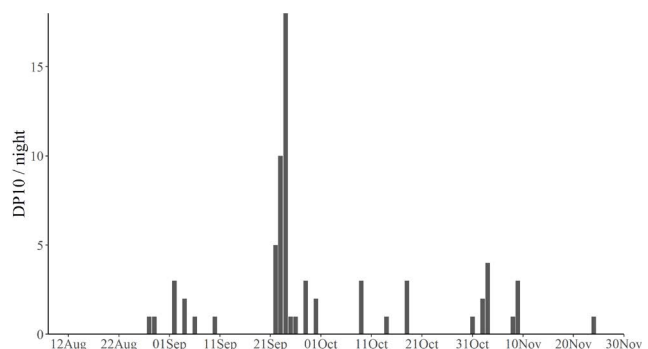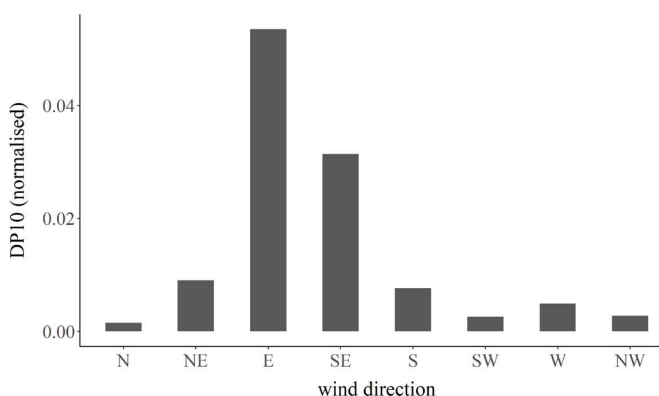

Figure 3. Detection positive 10 minutes (DP10) per night from 8 August until 30 November 2017.

The recorded wind speed at night was on average 7.6 ± 4.5 m/s during the study period, with a median value of 7.0 m/s and a maximum of 27.4 m/s. The mean wind speed at the time bats were recorded was 3.1 ± 1.9 m/s, with a median value of 2.4 m/s. 66% of the DP10 occurred when the wind speed was lower or equal to 3 m/s; 87% when wind speed was maximally 5 m/s (fig. 4). No bats were recorded when wind speed was higher than 11 m/s.
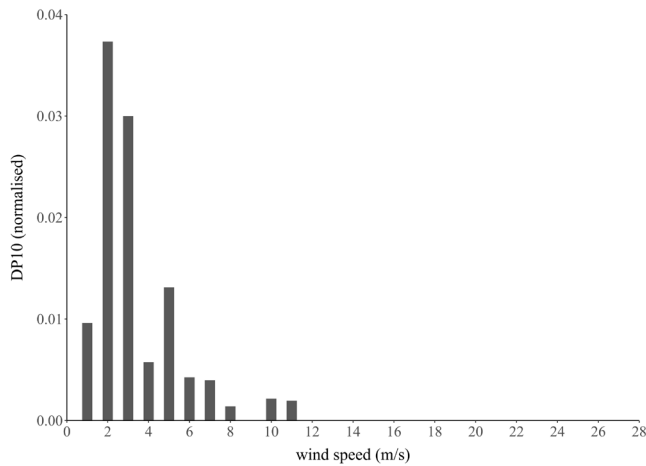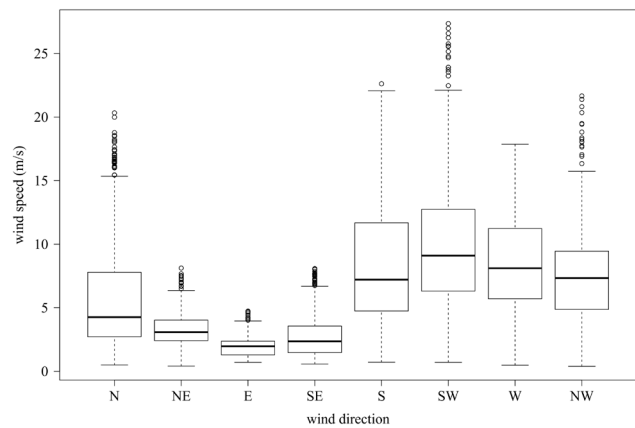
Figure 4. Normalised detection positive 10 minutes (DP10) of bat recordings in relation to wind speed. The range on the X-axis is the range of wind speed measured during the study period.

Most recordings were made during easterly (n = 16 DP10) and southeasterly (n = 18 DP10) wind and to a lesser extent wind from the south (n = 12 DP10). These are the wind directions that were the least frequently measured during the study period. Normalising the number of recordings per wind direction results in a clear peak in bat activity during easterly and southeasterly winds (fig. 5). These are the wind directions were the lowest wind speed was measured (fig. 6).

Figure 6. Boxplot of the wind direction and wind speed during the entire measurement period. Line in the box is the median value. Lower and upper limits of the box represent 25th and 75th percentile of the data, respectively. The upper whisker is defined as a 75th percentile + (1.5 x spread). The lower whisker is 25th percentile − (1.5 * spread), the spread being 75th − 25th percentile.

On average, bats were detected during a slight headwind (mean TWC = -0.53 ± 2.79) and offshore crosswind conditions (mean CWC = 1.19 ± 1.96), when we assume that the direction of migration is WSW (fig. 7). This is also the case when assuming a SW migration direction (mean TWC = -0.95 ± 2.74; mean CWC = 0.89 ± 2.02; fig. 8).
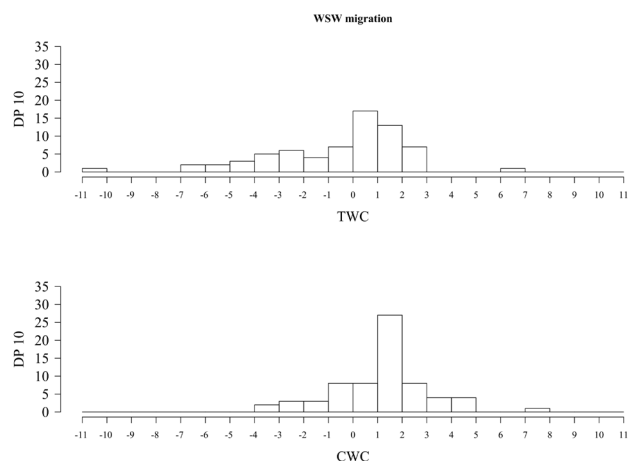

Figure 5. Normalised bat activity in relation to wind direction, *i.e.*, detection positive 10 minutes (DP10) of bat recordings divided by the frequency of occurrence of the wind direction during the study period.
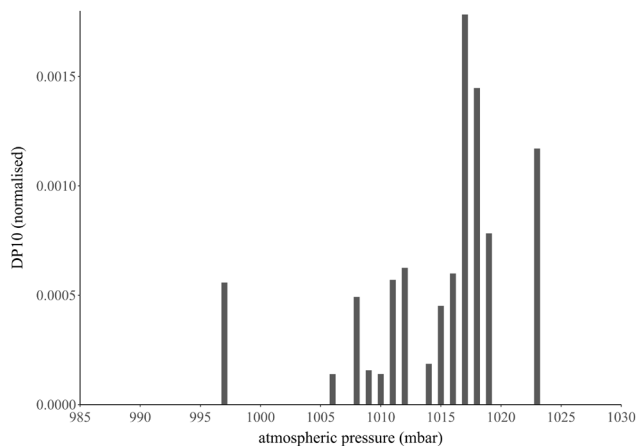

Figure 7. Detection positive 10 minutes (DP10) of bat recordings in relation to the tailwind and cross wind component, assuming a WSW migration direction. Positive TWC = tailwinds; negative TWC = headwinds. Positive CWC = offshore wind; negative CWC = onshore wind.

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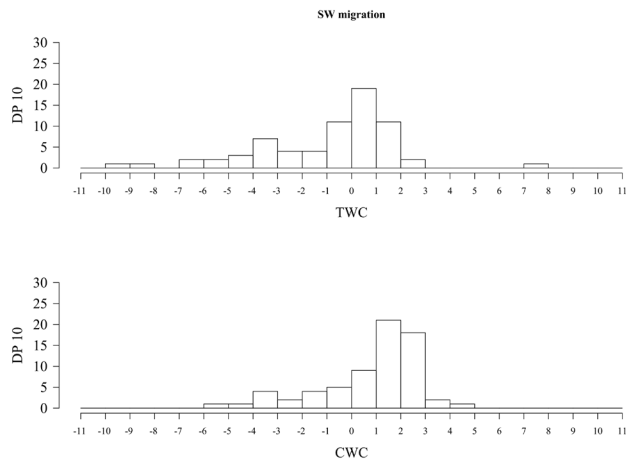
Figure 8. Detection positive 10 minutes (DP10) of bat recordings in relation to the tailwind component and cross wind component, assuming a SW migration direction. Positive TWC = tailwinds; negative TWC = headwinds. Positive CWC = offshore wind; negative CWC = onshore wind.

The mean atmospheric pressure during the study period was 1010.7 ± 7.7 hPa, with a median value of 1012.3 hPa. During bat recordings the atmospheric pressure was 1015.2 ± 4.0 hPa and a mean of 1017.0 hPa (fig. 9). Nighttime temperature was 13.6 ± 3.5°C on average during the study period. The mean temperature when bats were recorded was 13.8 ± 2.4°C (fig. 10).

Figure 9. Normalised detection positive 10 minutes (DP10) of bat recordings in relation to atmospheric pressure (hPa). The range on the X-axis is the range of atmospheric pressure measured during the study period.
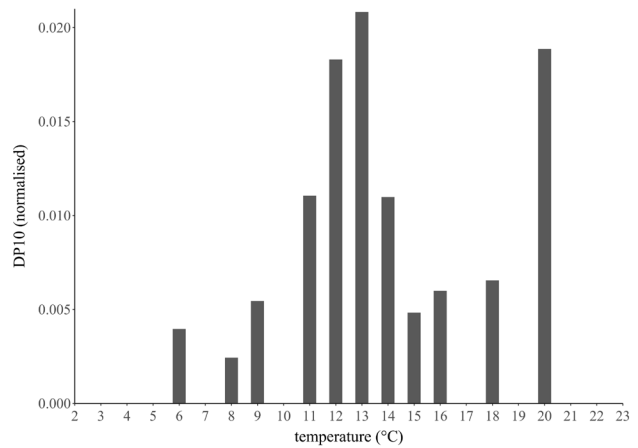
Figure 10. Detection positive 10 minutes (DP10) of bat recordings in relation to temperature (°C). The range on the X-axis is the range of temperature values measured during the study period.

4. Discussion

All recordings that were made were identified as Nathusius' pipistrelles. The threshold frequency used in this study (30 kHz, to avoid turbine generated noise), however, prevented to reliably detect Nyctaloid bats (genera *Eptesicus*, *Vespertilio* and *Nyctalus*). Therefore, our conclusions are only valid for pipistrellus bats. However, other studies showed that this is by far the most common species detected at the North Sea, with occurrence rates between 82 and 100% (Lagerveld *et al.* 2014; Hüppop & Hill 2016).

The highest number of detections were registered in the month of September (DP10 = 47) and, to a lesser extent, October (DP10 = 8) and November (DP10 = 11). This coincides with the known migration peak period for Nathusius' pipistrelle (*i.e.* mid-August to mid-October; Rydell *et al.* 2014; Barataud 2015). The relatively high number of detections in November is remarkable. Less detections than expected were recorded in August (study started August 8th), *i.e.* two DP10. As the weather in August 2017 was colder than average and with a lot of precipitations, we expect that the migration period of bats started later than on average.

Wind speed seemed to have a large influence on the presence of bats during the study period, with 66% of the DP10 recorded when the wind speed was lower or equal to 3 m/s, 72% when wind speed was lower or equal to 4 m/s and 87% when wind speed was not higher than 5 m/s. The cut-in wind speed, *i.e.* wind speed at which the wind turbine rotor starts rotating, of operational and planned wind turbines in the BPNS is between 3.0 and 4.0 m/s. This means that most of the bat activity took place when the turbines were not operational. Increasing the cut-in wind speed to 5 m/s would be an effective mitigation measure and is as such already imposed in the planned Dutch Borssele wind farms between August 15 and October 31 (Lagerveld *et al.* 2017).

The wind direction is also important for the recorded bat activity at sea, with a clear peak in occurrence when wind originated from the E and the SE. This coincides with the findings of Lagerveld *et al.* (2017) who report highest bat activity at sea during wind directions between NE and SE. However, the mean wind speed was lowest when the wind direction was NE, E and SE. Most likely, the low wind speed was a more important driver of the reported bat activity.

By calculating the TWC and CWC, the wind direction and wind speed at the time of bat recordings are combined. This results in, on average, slight headwind and crosswind conditions, both when assuming a WSW and SW migration direction. Hüppop and Hill (2016) reported that the highest bat activity at a research platform in the German Bight coincides with crosswind conditions, which suggests that wind drift is the main driver of bat occurrence at sea. Given the high number of recordings during low wind speed conditions, it seems unlikely that offshore drift explains the occurrence of bats in our study site. It might be the case for some recordings, but intentional migration across the North Sea channel seems more likely for most recordings.

Bender and Hartman (2015) showed that bat activity on land was positively related to average nightly temperature and average nightly barometric pressure. Our results suggest that this is also the case for bat activity across the North Sea.

This study sheds light on the meteorological conditions that favor bat activity at sea. Multivariate analysis would allow to make statistically sound conclusions about the most important variables that explain bat activity. At this point, however, we do not have enough data to perform reliable multivariate statistical analysis. Therefore, it is needed to continue to collect recordings of bats at sea. More data will then result in more robust conclusions about the drivers of the activity of bats at sea in order to support future policy decisions with regards to offshore wind farms.

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CHAPTER 10

AN ANALYSIS OF HARBOUR PORPOISE STRANDINGS AFTER A DECADE OF OFFSHORE WIND FARM CONSTRUCTION IN THE SOUTHERN NORTH SEA

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Abstract

The Southern North Sea is an important hub for offshore wind energy with nearly 40 operational offshore wind farms, and many more planned. In most cases, the construction of offshore wind farms requires the installation of large hollow steel piles using high-energy impact hammers. In addition, geophysical site surveys regularly include seismic airgun surveys which operate over large areas for prolonged periods of time. Both of these processes generate very high sound levels in the surrounding waters, which can be detrimental to marine mammals if these are exposed to them. Increased noise levels over a large area can affect marine mammals in several ways, ranging from behavioural responses, masking of acoustic signal detection and temporary to permanent hearing loss and physical injury. All of these can lead indirectly to an increased mortality rate or, due to stress, to a compromised reproduction. In this study we examined, over a period of fourteen years, whether prolonged periods of intermittent high intensity impulsive sound influenced the temporal pattern of strandings on Belgian beaches of the harbour porpoise (*Phocoena phocoena*), the most common cetacean in the Southern North Sea. Generalized Additive Mixed modelling revealed a strong seasonal pattern in strandings, with a first peak in strandings in spring (March-May) and a second, less pronounced, in September. In addition, our analysis revealed a significantly higher occurrence of stranded harbour porpoise on Belgian beaches in months with prolonged periods of intermittent high intensity impulsive sound which is suggestive of increased mortality. An in-depth analysis of age, sex, cause of death, and overall health (prior to death) of the stranded specimens will help determine what drives this additional mortality and reduce the uncertainty due to the biases associated with the use of this strandings data.

1. Introduction

Harbour porpoises (*Phocoena phocoena*) are the most consistently present cetacean observed in the North Sea (Hammond *et al.* 2002) and the Belgian part of the North Sea (BPNS) (Haelters *et al.* 2011). The species is protected

by both national (Belgian Government 2001) and EU law (EU 1992). In the North Sea, the harbour porpoise is considered vulnerable because of high bycatch levels and its sensitivity to increasing noise pollution. In the last two decades, there has been a strong increase in activities generating prolonged periods with intermittent high intensity impulsive underwater sound (Slabbekoorn *et al.* 2010; Shannon *et al.* 2016). High levels of impulsive underwater sound are generated when large steel turbine foundations are hammered into the seabed as well as during seismic surveys. Concerns over the possible impact of high intensity impulsive sound generated during the construction of offshore wind farms on marine mammals in general, and harbour porpoise in particular, has been a driving force in determining national regulations in North Sea countries *(*Rumes *et al.* 2016). These concerns originate from the fact that exposure to high intensity impulsive underwater sound affects porpoises over large distances (Haelters *et al.* 2013; Brandt *et al.* 2016; Rumes *et al.* 2017). Potential effects include physical injury, physiological dysfunction, behavioural modification and masking of sound used by the animals themselves (see *e.g.* Carstensen *et al.* 2006; Parvin *et al.* 2007; Bailey *et al.* 2010). At the level of individual porpoises these effects and their consequences on vital rates (survival, maturation, reproduction) vary in significance from negligible to fatal (Marine Mammal Commission 2007; Brandt *et al.* 2016). When the vital rates of large numbers of individuals are influenced, this will ultimately result in population level effects in line with the Population Consequences of Acoustic Disturbance (PCAD) conceptual model (NRC 2005). Both the iPCOD (interim Population Consequences of Disturbance; Harwoord *et al.* 2014; Nabe-Nielsen & Harwood 2016) and DEPONS (disturbance effects on the Harbour Porpoise population in the North Sea; Nabe-Nielsen *et al.* 2018) models aim to quantitatively assess population consequences of such sub-lethal

behavioural effects. Resultant predictions for the impact of wind farm construction in the North Sea on harbour porpoise population have ranged from indistinguishable from the baseline scenario (Nabe-Nielsen *et al.* 2018) to a 40% chance of exceeding a 5% overall population reduction (SEANSE-study, TNO). However, due to a lack of data on the behaviour and fitness of those porpoises that are exposed to high levels of underwater sound, we must rely on indirect measures of activity such as changes in click rates and feeding buzzes to determine disturbance and use the aforementioned demography (iPCoD) and process-based (DEPONS) models to translate these to population consequences.

The investigation of strandings may provide additional insights into the state of the porpoise population. Strandings provide a sample of this population and show variability in the incidence of disease and causes of death. Ten Doeschate *et al.* (2017) demonstrated how examination of baseline patterns can facilitate the detection of unusual variability in stranding rates of harbour porpoises. In this report, we determined whether temporal patterns in observed porpoise strandings at the Belgian coastline (as a proxy for local population mortality) are influenced by prolonged (multiple consecutive months) periods of intermittent high intensity impulsive sound.

2. Material and methods

2.1. Data collection

In Belgium, strandings are reported to the national strandings netwerk (Haelters *et al.* 2013b). Given the relatively short length of the coastline (66.6 km) and the dense habitation and intensive recreational use along most of the coast, we assume that the large majority of stranded animals is reported within 24 hours. For this study, we have analysed strandings data from 2005 to 2018. Only those strandings that included data on

Table 1. Overview of the number of harbour porpoise (HP) strandings per month along the Belgian coastline between 2005 and 2018. Underlined numbers indicate months when pile driving was taking place.

Year	January	February	March	April	May	June	July	August	September	October	November	December	Total HP strandings
2005	3	6	4	21	18	7	3	13	6	4	2	2	89
2006	2	2	20	18	17	9	1	13	5	1	3	3	94
2007	1	9	9	13	14	4	4	13	12	4	3	0	86
2008	1	3	11	9	8	7	5	4	3	8	3	0	62
2009	6	3	6	5	5	5	3	10	13	8	1	1	66
2010	2	2	4	6	7	4	2	8	5	3	3	2	48
2011	3	2	6	25	10	4	16	25	7	5	5	8	116
2012	4	4	18	13	11	2	9	6	18	5	1	5	96
2013	3	7	11	38	26	20	13	6	8	13	2	2	149
2014	3	4	23	14	7	18	14	9	24	5	0	6	127
2015	2	2	5	6	7	8	6	3	5	5	3	0	52
2016	1	4	31	23	14	6	14	5	17	12	6	4	137
2017	1	5	8	5	5	9	19	13	4	13	5	6	93
2018	0	1	3	16	12	16	11*	3	7	15	0	2	86

The asterisk (*) indicates the seismic survey in July 2018.

stranding date and location were included in this analysis as animals that were reported floating at sea were excluded. We decided not to use data before 2005, as piling only started in 2009, and in the late 1990s and the early 21st century a major shift occurred in the distribution of porpoises in the North Sea, from the northwest to the Southern North Sea, (Hammond *et al.* 2013), leading to a major increase in numbers of stranded animals in the SSouthern North Sea. For the same time period, information on hydraulic pile driving activity in the Belgian North Sea was derived from the OD Nature pile driving dataset. No pile driving took place in the adjacent Dutch and French parts of the North Sea in this period.

The dataset thus includes strandings on Belgian beaches for fourteen years. Pile driving was ongoing in seven of these years and, in addition, in 2018 an experimental seismic survey took place in the month prior to the start of pile driving (table 1). The purpose of this latter survey was to test the response of tagged cod to the sound levels produced by during oil and gas exploration surveys (PCAD4COD-project).

Exposure to intermittent high intensity impulsive sound has the potential to affect porpoise survival on multiple time scales. A disturbed animal may not be able to feed for a certain period time (different iterations of the iPCoD models assume 6-24 hours – Harwoord *et al.* 2014, Nabe-Nielsen & Harwood 2016; Booth *et al.* 2019) but this alone is unlikely to result in direct mortality in adult porpoises (Kastelein *et al.* 2018). However, repeated exposures may result in starvation or reduced resistance to pathogens, thus resulting in indirect mortality. For our analysis we focused on changes in the numbers of strandings at the Belgian coast in those months with ongoing intermittent high intensity impulsive sound. We acknowledge that, for some individuals exposed to high intensity impulsive sound, mortality may occur only at a later date. However, harbour porpoises are highly mobile and there are major seasonal changes in porpoise distribution in the Southern North Sea (Gilles *et al.* 2016). Thus, those individuals that do die at a later date as an indirect result of cumulative exposure to impulsive sound generated in our waters are less likely to end up stranding at the Belgian coast.

2.2. Data analyses

We explored whether there was a linear relationship between the number of porpoise strandings over a whole year and the number of months of intermittent high
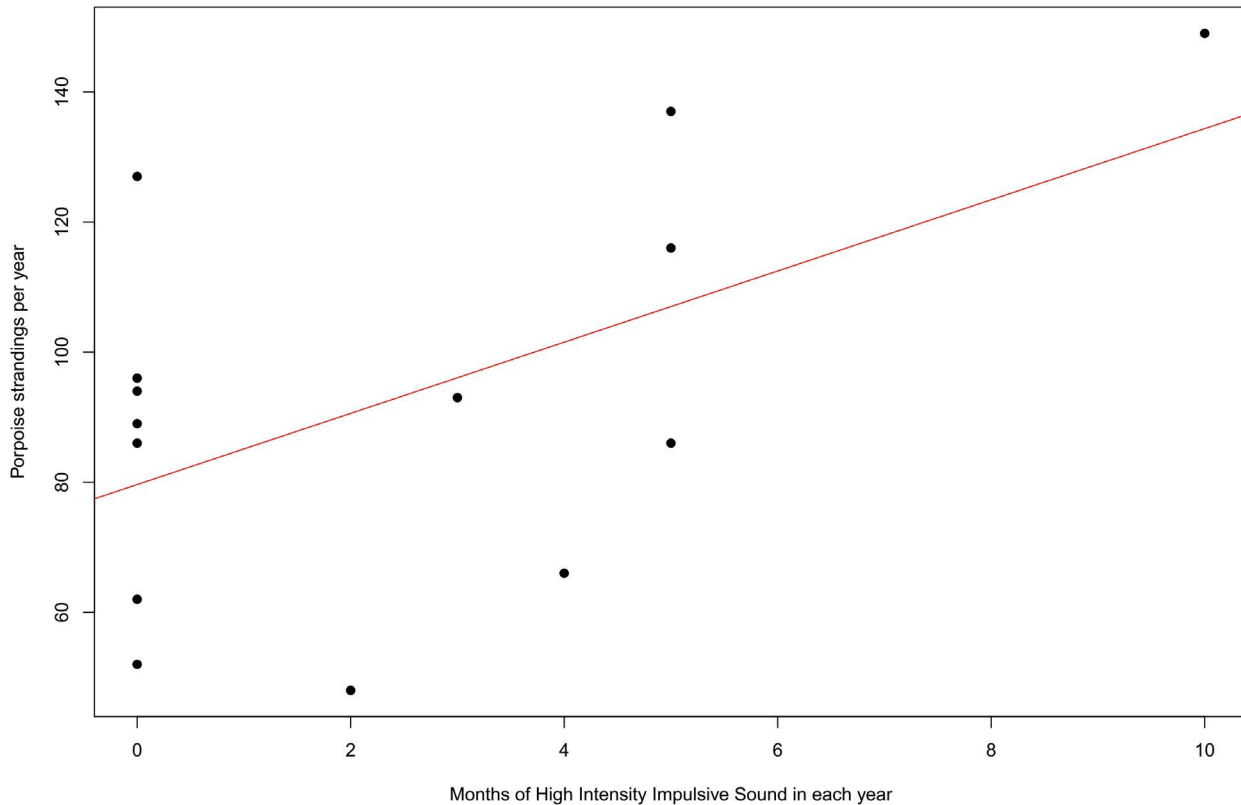
Figure 1. Yearly number of porpoise strandings at the Belgian coast in the period of 2005-2018 in relation to the number of months high intensity impulsive sound in these years.

intensity impulsive sound using the e1071 package (Meyer *et al.* 2019) available in R (R Core Team 2013).

We investigated the effect of high intensity impulsive sound on temporal stranding patterns through General Additive Mixed Modeling, using the *gamm4* package (Wood & Scheipl 2017) available in R (R Core Team 2013). Total number of strandings per month was modelled as function of a circular cubic spline for month, and presence/absence of high intensity impulsive sound generating activities in that month. Year was considered as a random variable (Ten Doeschate *et al.* 2017). A preliminary data-exploration (Zuur & Leno 2016) did not reveal any outliers, hence all data were used in the analyses. As data are counts, and therefore equal or larger than zero, we initially applied the GAMM using a Poisson error distribution, which resulted in an overdispersed model. Therefore, we applied GAMM using a Negative Binomial

error distribution. As *gamm4* does not calculate the minimum value of the dispersion parameter *k*, we ran the model with a range of *k*-values and calculated approximate AICs. The k-value resulting in the lowest AIC was selected to run the final GAMM. Model validation was performed according to Zuur & Leno (2016) and mainly consisted of checking homogeneity of residuals and inspecting graphs of residuals *versus* all covariates in the model.

3. Results

For the period of 2005-2018, there is a weak, but positive correlation between the number of porpoise strandings per year and the number of months of intermittent high intensity impulsive sound (r = 0.54; p < 0.05; fig. 1). Exploratory analysis showed that, especially during late spring and summer, high numbers of stranded porpoises were observed during months with high intensity impulsive sound (fig. 2).

Porpoise strandings per month
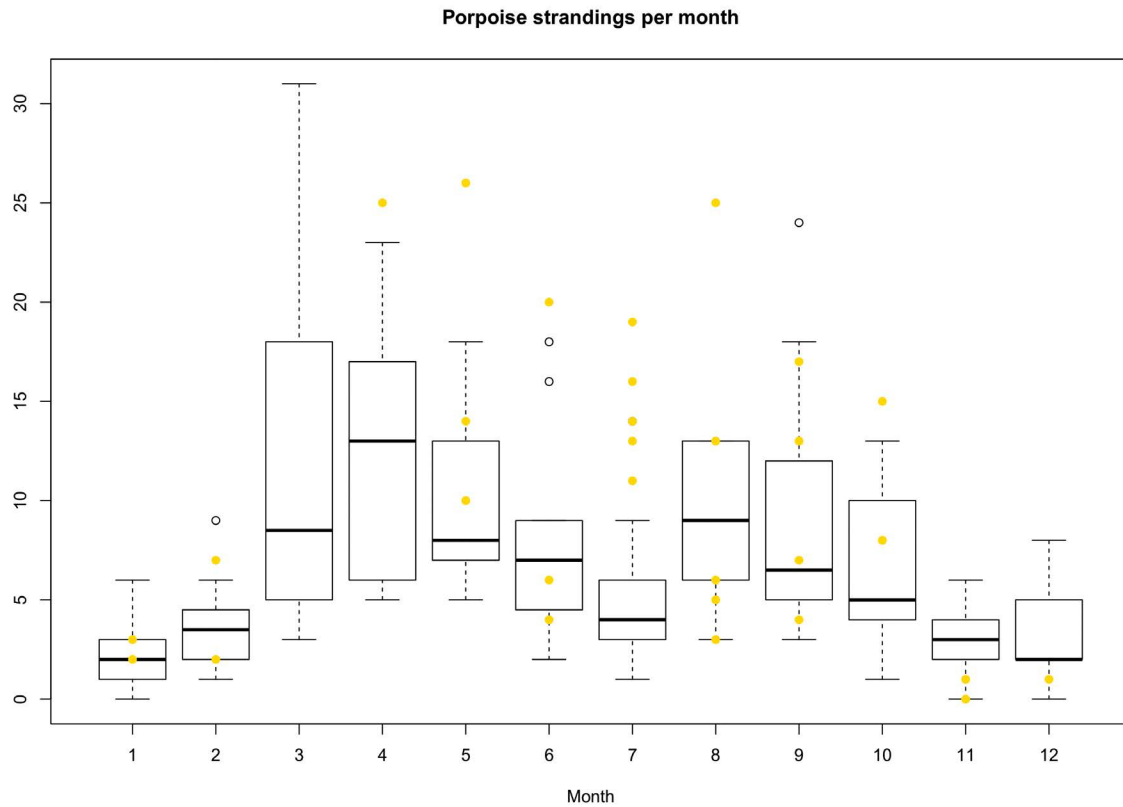
Figure 2. Monthly number of porpoise strandings at the Belgian coast in the period of 2005-2018. Boxplots only include months without high intensity impulsive sound. Yellow dots represent data of months with high intensity impulsive sound.
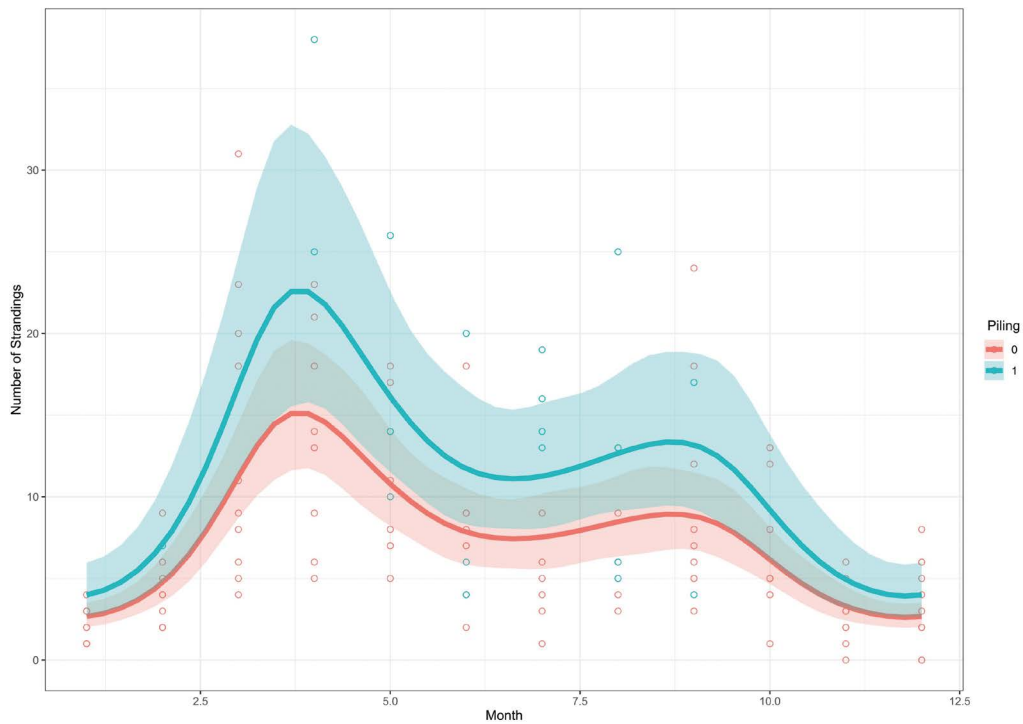
Figure 3. GAMM predicted number of harbour porpoise strandings in months with (solid blue line) and without high intensity impulsive sound (solid red line) for an average year. Shaded areas represent the confidence intervals. Actual data are displayed as dots (data: monthly number of porpoise strandings at the Belgian coast in the period of 2005-2018).

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Generalized Additive Mixed modelling revealed a strong seasonal pattern in porpoise strandings along the Belgian coastline, with a low amount of strandings in winter, a first peak in spring (March-May) and a second, less pronounced, in late summer (September). The selected model suggested number of porpoise strandings was significantly related to month of the year (smoother estimated degrees of freedom = 5.6; p-value < 0.001; fig. 3). If we look at the effect of high intensity impulsive sound on this temporal stranding pattern, then a picture emerges with significantly higher numbers of strandings in those months with intermittent high intensity impulsive underwater sound (p-value < 0.01).

4. Discussion

In previous years, 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) and have used this information to model the consequences of pile-driving at (local) population scale (Rumes *et al.* 2018) using demography-based modelling, such as the interim Population Consequences of Disturbances model (iPCoD). In this report we discuss the seasonal pattern in porpoise strandings along the Belgian coastline and show that there is an increased number of strandings of harbour porpoise during months when high intensity impulsive sound is present in the BPNS.

4.1. Temporal stranding pattern

We observed a strong seasonal pattern in porpoise strandings along the Belgian coastline, with a low amount of strandings in winter, a first peak in spring (March-May) and a second, less pronounced, in late summer (September). This pattern is consistent with the results of aerial surveys (Haelters *et al.* 2015) and passive acoustic monitoring

(Haelters *et al.* 2016; Augustijns 2018), with both also showing a seasonal pattern with higher detection rates in the end of winter to early spring. Gilles *et al.* (2016) found that, in spring, harbour porpoise hotspots were situated mainly inshore in the southern and south-eastern part of the North Sea and shifted offshore and to western areas in summer. Thus, from winter to summer, changes in porpoise distribution are likely to be the main driver for the observed pattern in observations and strandings. The second peak, in late summer, would appear not to reflect porpoise density in Belgian waters as determined by aerial surveys (Haelters *et al.* 2015). Passive acoustic monitoring (Haelters *et al.* 2016; Augustijns 2018) does observe a second and lower peak in detections around September-October. Part of the increase in strandings in late summer may be due to a temporary increase in mortality amongst juvenile porpoise failing to find enough food. Starvation is a common cause of death for porpoises in Belgian and adjacent French waters with necropsies showing that up to 70% of porpoises examined had an empty gastro-intestinal tract, indicating no recent food intake (Jauniaux *et al.* 2002).

Given the strong seasonal pattern in porpoise abundance in Belgian waters and the variability in timing of high intensity impulsive underwater sound generating activities, it is perhaps unsurprising that there is no clear linear relationship between the yearly number of porpoise strandings and the number of months of intermittent high intensity impulsive sound. However, if we look at the number of strandings over multiple months then prolonged periods with intermittent high intensity impulsive underwater sound do coincide with increased numbers of strandings. Further, more detailed analysis is required to determine whether this is due to causation (strandings reflect increased albeit indirect mortality) or correlation (with pile driving coinciding with periods of increased – natural – mortality).

4.2. Further analysis

There were considerable differences both in timing (winter *vs* spring/summer) and intensity of high intensity impulsive sound generating works (ranging from 14 days to 72 days of impulsive sound in 2010 and 2013 respectively). As the impact of disturbance on vital rates of porpoises is assumed to be exponential only once a certain threshold is exceeded (Booth *et al.* 2019) and, is likely subject to seasonal changes, we intend to repeat the analysis using only those years where prolonged high intensity impulsive sound was generated during spring-summer which coincides with the peak calving period (Lockyer 1995).

In addition, it may be advisable to include data on the sex and age of those stranded porpoises as vulnerability to disturbance is likely going to be different for *e.g.* calves and pregnant females (Booth *et al.* 2019). Data on the physical condition of stranded porpoises prior to death (*e.g.* blubber thickness and stomach content) may also reveal possible knock-on effects of disturbance. However, it should be noted that these data are not always available and depend on the state of decomposition at the time of stranding.

5. Conclusion

An exploratory analysis of the effect of high intensity impulsive sound on temporal stranding patterns along the Belgian coast suggests that these periods of high intensity impulsive sound often coincide with significantly increased overall porpoise stranding rates and is suggestive of increased mortality. In order to test this hypothesis, further analyses need to be conducted taking into account those factors that will influence porpoise mortality due to disturbance and thus indirectly stranding rates. These factors include cumulative exposure, age, sex and seasonal variation in vulnerability, and density dependent responses.

Acknowledgements

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