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

EMPIRICAL EVIDENCE INSPIRING PRIORITY MONITORING, RESEARCH AND MANAGEMENT

Edited by
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CONTEXT

The European Directive 2009/28/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. With eight operational wind farms, the first zone is completely filled. A second area for renewable energy of 285 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 2019.

DEGRAER Steven, BRABANT Robin, RUMES Bob & VIGIN Laurence

EXECUTIVE SUMMARY

EMPIRICAL EVIDENCE INSPIRING PRIORITY MONITORING, RESEARCH AND MANAGEMENT

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By the end of 2020, *i.e.* twelve years after the installation of the first wind turbines in the Belgian part of the North Sea, an installed capacity of 2.26 Gigawatts (GW), consisting of 394 offshore wind turbines and covering 238 km², will be operational in the Belgian part of the North Sea (chapter 1). They are expected to produce an average of 8 TWh annually, which is around 10% of the total national electricity demand or nearly 50% of the electricity needs of all Belgian households. Although no new projects are scheduled in the next years, long-term developments include an additional zone of 285 km² for a production capacity of ~2 GW of offshore wind energy which has been delineated in the new marine spatial plan. With 523 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 will undoubtedly continue to be a major concern in the years to come. We hence continue to be ever more faced with the challenge to optimise measures to combat the energy crisis in the light of combatting the biodiversity crisis. Tackling this challenge will necessitate the generation of new knowledge as well as a maximum uptake of existing and new knowledge to facilitate an environment-friendly management of offshore renewable energy developments, hence inspiring priority monitoring, research and management. This knowledge should cover a broad range of ecosystem components from soft sediment and (artificial) hard substrate invertebrates and fish to seabirds and marine mammals, as well as their interactions, all of which are impacted by offshore renewable energy developments in different ways.

Given the potential negative impact on the marine ecosystem, excessive sound levels are considered one of the main pressures during the construction phase of offshore wind farms; this particularly when pile driving is involved. Next to sound production, propagation and mitigation modelling, *in situ* measurements of piling events have received a lot of attention during the last decades, also in the Belgian part of the North Sea. After ten years of pile driving activities in Belgian waters, the last turbine foundation was hammered in the seafloor of the first offshore renewable energy zone on 2 January 2020. Until construction activities start in the second Belgian renewable energy zone, this was the last event of pile driving in Belgian waters. The last turbine foundations installed, consisting of extra-large steel monopiles with a diameter of 7.4 and 8 m, were piled using large hydraulic hammers of 3000 and 4000 kJ, respectively (chapter 2). The associated excessive sound levels were reduced using a double big bubble curtain (DBBC) mitigation technique for the first time in Belgian waters. A double bubble curtain is formed around a pile by freely rising bubbles created by compressed air injected into the water through two rings of perforated pipes encircling the pile. *In situ* measurements of underwater sound generated during 14 full pile driving events showed zero to peak sound levels ranging from 183 to 193 dB re 1 μ Pa when normalised to a distance of 750 m from the source. This represented an estimated zero to peak sound level reduction of 12-20 dB re 1 μPa. The variability in efficiency of sound level reduction may be explained by the technical set-up of the DBBC but also by environmental conditions like tidal currents which should receive extra attention while designing the DBBC as to optimise sound reduction. We further detected the efficiency of sound level reduction to be higher for frequencies above 300 Hz while mainly lower frequency sounds are emitted during pile driving.

Marine mammals are particularly at risk of being negatively impacted by the excessive sound levels produced during pile driving. Sound mitigation measures to reduce that impact are therefore outlined in the environmental license conditions. Initial license conditions were aimed at preventing near-field injury to individual animals and included the use of an acoustic deterrent device as well as a prohibition on starting pile driving if a marine mammal was observed in the vicinity of the construction zone (chapter 3). Progressive insight in the potential population consequences of far-field behavioural disturbance led to the formulation of further permit conditions. These included a seasonal pile driving ban from 1 January to 30 April, and an obligation to use noise mitigation measures that limit the transmission of noise pollution to the marine environment. The interim Population Consequences of Disturbance model (iPCOD), simulating how different approaches to sound mitigation during pile driving can impact a harbour porpoise *Phocoena phocoena* population over a period of 25 years, showed that the applied mitigation measures reduced the average porpoise population decline at the end of the offshore wind farm construction period by 50%, while the application of currently available measures would have reduced the population decline by 97%. Possible improvements to the environmental license conditions include optimising the use of acoustic deterrent devices, formalising obligatory marine mammal surveys, and requiring developers to comply with the national threshold for impulsive underwater sound. The associated direct cost (~0.5% of the construction cost of an average-sized offshore wind farm) and indirect costs (cf. no indication of an increased installation time) of the application of sound mitigation measures will not affect the overall economic viability of future projects.

Seabird collision risks need our continued attention, because the resulting additional mortality may have a substantial impact at the population level for long-lived seabird species with a delayed maturity and small clutch size. Despite the considerable uncertainty about the absolute number of seabird collisions, collision models do allow identifying which species face the highest collision risk and show great value in the collision risk analysis of different scenarios for offshore wind farm development. They hence offer a promising tool for strategic marine planning at both national and regional scale. We assessed the number of possible seabird collision victims for the fully developed first offshore renewable energy zone in the Belgian part of the North Sea based on the latest available knowledge on collision risk modelling (chapter 4). This was done for the six most abundant species, *i.e.* black-legged kittiwake *Rissa tridactyla*, lesser black-backed gull *Larus fuscus*, great black-backed gull *Larus marinus*, herring gull *Larus argentatus*, common gull *Larus canus* and northern gannet *Morus bassanus*. In total, 70 casualties per year (standard deviation: 53) are expected. This number may arise to 290 casualties (standard deviation: 205) depending on the source of the avoidance rates in the model, which still are heavily debated in the scientific community. With respectively 54% and 27% of the total number of collisions, the highest number of collisions are expected for greater and lesser black-backed gull, *Larus fuscus* and *Larus marinus*. With an increasing number of offshore wind farms built and planned in the North Sea, population level effects caused by additional mortality through collisions cannot be excluded and may conflict with seabird conservation goals. Our results demonstrate turbines with a larger distance between the sea surface and the lower tip of the rotor to result in lower collision risk and a high turbine density to result in a higher collision risk. This knowledge is of direct use for a seabird-friendly siting and design of future offshore wind farms.

Not all seabird species are equally at risk of colliding with the turbine blades. Twelve years of baseline monitoring of seabird displacement has revealed distinct patterns in the tendency of seabird species to either avoid or to be attracted to these OWFs. Striking parallels among wind farms were detected for some seabird species, while other species showed a substantial inconsistency in displacement among different wind farms. Because of limited insight in what is driving the variation in observed patterns, impact study results so far have had limited value in predicting expected displacement rates elsewhere. Increased knowledge on cause-effect relationships would strongly benefit future planning and impact assessments. The future monitoring focus should therefore be oriented towards more targeted research, aiming to an understanding of the actual impact of offshore wind farms on individual birds or bird populations, next to aspects supporting mitigation. The first Belgian offshore renewable energy zone been fully developed has now created a momentum to revisit the research strategy (chapter 5). Three future focus research themes were identified. First, we aim to perform more targeted analyses to look for a correlation between wind farm characteristics and locally observed displacement rates within the Belgian wind farm concession zone. Secondly, we will develop empirically informed species distribution models for northern gannet *Morus bassanus*, common guillemot *Uria aalge* and razorbill *Alca torda*. Together with prospects on wind energy developments and empirically-assessed displacement rates, the species-specific number of birds affected can be estimated, allowing for recommendations for mitigating and compensating measures in future marine spatial planning. Thirdly, we advise for tracking studies of lesser black-backed gull *Larus fuscus* to generate unprecedented knowledge on behavioural and foraging-related activities inside offshore wind farms. This will shed a light onto additional or decreased collision risk as a consequence of gull behaviour inside wind farms.

Impacts on the benthic communities are more subtle and cannot easily be classified as positive or negative. As part of the artificial reef effect, offshore wind turbines may for example impact the seafloor beyond their actual footprint. Locally modified water currents around turbine foundations, as well as the depositional flow of faecal pellets and other detrital material produced by suspension-feeding fouling organisms on the foundations, have earlier been suggested to contribute to a process of sediment fining and organic matter enrichment close to and in the wake of wind turbines; this with consequent shifts in macrobenthos community composition, diversity and abundance. An analysis of three years of soft sediment macrobenthos data in the vicinity of jacket foundations provided equivocal support for this (chapter 6). Seven to nine years after installation, sediments at 37.5 m from the jacket foundations indeed had a significantly higher proportion of fine sand compared to samples collected at further distance (350-500 m from the foundations). This was, however, only occasionally accompanied by a higher organic matter content. Although, average macrobenthos abundance and diversity were higher and higher nearby the turbines, these differences could not always be backed up statistically. Still, the macrobenthos community structure did consistently differ between nearby and distant locations. At nearby locations, macrobenthos significantly changed with time which was largely attributable to a decline in the abundance of the amphipod *Urothoe brevicornis* and the polychaetes *Spiophanes bombyx* and *Nephtys cirrosa*, typical for natural surroundings permeable sediments, and to an increase of several other species, in particular the sand mason *Lanice conchilega*, formerly unknown for this environment. This evolution is likely to increase the small-scale benthic heterogeneity because of positive feedback loops from the bio-engineering activity of *Lanice conchilega* aggregations on sediment fining and enrichment. Ideally, future basic monitoring need to better incorporate small-scale variability in its sampling design, whereas targeted monitoring efforts should be directed at determining the spatial extent over which fining and enrichment is manifesting.

Changes to the larger soft sediment epibenthos and demersal fish communities that reside on the soft sediments in between the turbines were equally expected. However, nine years after construction no drastic changes could be detected for two offshore wind farms (chapter 7). The epibenthos and demersal fish assemblages remained structured mainly by temporal variability due to local and large-scale changes in temperature and climate (*in casu* the North Atlantic Oscillation and Atlantic Multidecadal Oscillation indices), rather than by the potential small-scale effect of the offshore wind farms. Still, some secondary effects are seen, which may be interpreted as the first signs of a refugium effect and an expansion of the artificial reef effect, hinting towards a positive effect of offshore wind farms on the epibenthos and demersal fish community. The refugium effect is suggested by the increased fish densities of some common soft sediment-associated fish species like common dragonet *Callionymus lyra*, solenette *Buglossidium luteum*, lesser weever *Echiichthys vipera* and plaice *Pleuronectes platessa* in one of two studied wind farms. This effect may result from fisheries exclusion combined with increased food availability. An expansion of the artificial reef effect is suggested through the appearance of an increased number of hard substrate-associated species, *e.g.* long-clawed porcelain crab *Pisidia longicornis*, edible crab *Cancer pagurus* and seabass *Dicentrarchus labrax* on the soft sediments. Increased densities of common squid *Loligo vulgaris* in one wind farm consisting of jacket foundation wind turbines could be an indication that cephalopods use the jacket foundations as substrate for egg deposition. However, the clearest indication for the artificial reef effect expansion was the increased abundances of blue mussel *Mytilus edulis* and anemones on the soft sediments in between the piles (> 200 m), two taxa dominating the epifouling communities on the turbines. Although mussel densities were still low (max. ~15 ind./1000 m^2), they may contribute to a future increased soft bottom habitat heterogeneity. Future monitoring is expected to demonstrate a continued heterogenisation of the soft sediment habitat with local biodiversity hotspots linked to patchy mussel drop offs.

Offshore wind turbine foundations indeed are heavily colonised by fouling organisms that mainly consist of suspension feeding invertebrates influencing local food web properties with possible knock-on effects at the wider spatial scale (chapter 8). A higher food web complexity was observed for the communities occurring in deeper parts of the turbines and in zones where organic matter accumulates (surrounding soft sediments and scour protection layer) compared to the upper parts of the foundation. The high trophic diversity and low redundancy in the soft sediment, scour protection layers and the *Metridium* zone on the foundation further suggest resources partitioning among and within the communities inhabiting these zones. The hard substrate species had larger trophic niches, exploiting a wider range of food sources, which allows for the co-existence of a multitude of species and a wide distribution of fouling organisms within and across depth zones. Two of the most common species, *i.e.* the amphipod *Jassa herdmani* and the mussel *Mytilus edulis* play a key role in the carbon assimilation process: we estimated that these two species' populations in the whole offshore renewable energy zone graze ~1.3% of the total annual local primary production of the Belgian part of the North Sea. While negligible relative to the primary production quantity, the consequent local deposition of organic matter likely has a considerable local effect on the sedimentary habitat and may indirectly influence the wider marine food web. The locally increased carbon assimilation (linked to secondary production) on its turn locally was shown to support higher trophic levels, demonstrated by benthic and bentho-pelagic species like sculpin *Myoxocephalus scorpioides*, cod *Gadus morhua* and pouting *Trisopterus luscus*, utilising the offshore wind farms as feeding grounds for longer periods. Even though these findings support the production hypothesis, further research is needed to prove that the locally increased secondary production of fish also has positive knock-on effects at the fish population level.

CHAPTER 1

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

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Abstract

By the end of 2020, with the completion of the Northwester 2 and Seamade projects, an installed capacity of 2.26 Gigawatt (GW), consisting of 394 offshore wind turbines, will be operational in the Belgian part of the North Sea (BPNS). They are expected to produce an average of 8 TWh annually, which is around 10% of total national electricity demand. Although no new projects are scheduled in the next few years, long term developments include an additional zone for 2 GW of offshore wind energy which has been identified in the new marine spatial plan.

With 523 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, 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

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 285 km² is reserved in the marine spatial plan that came in force on 20 March 2020.

The European Directive 2009/28/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 the end of 2020. Offshore wind farms in the BPNS will make an important contribution to that goal.

On 31 December 2019, Belgium submitted a National Energy and Climate Plan to

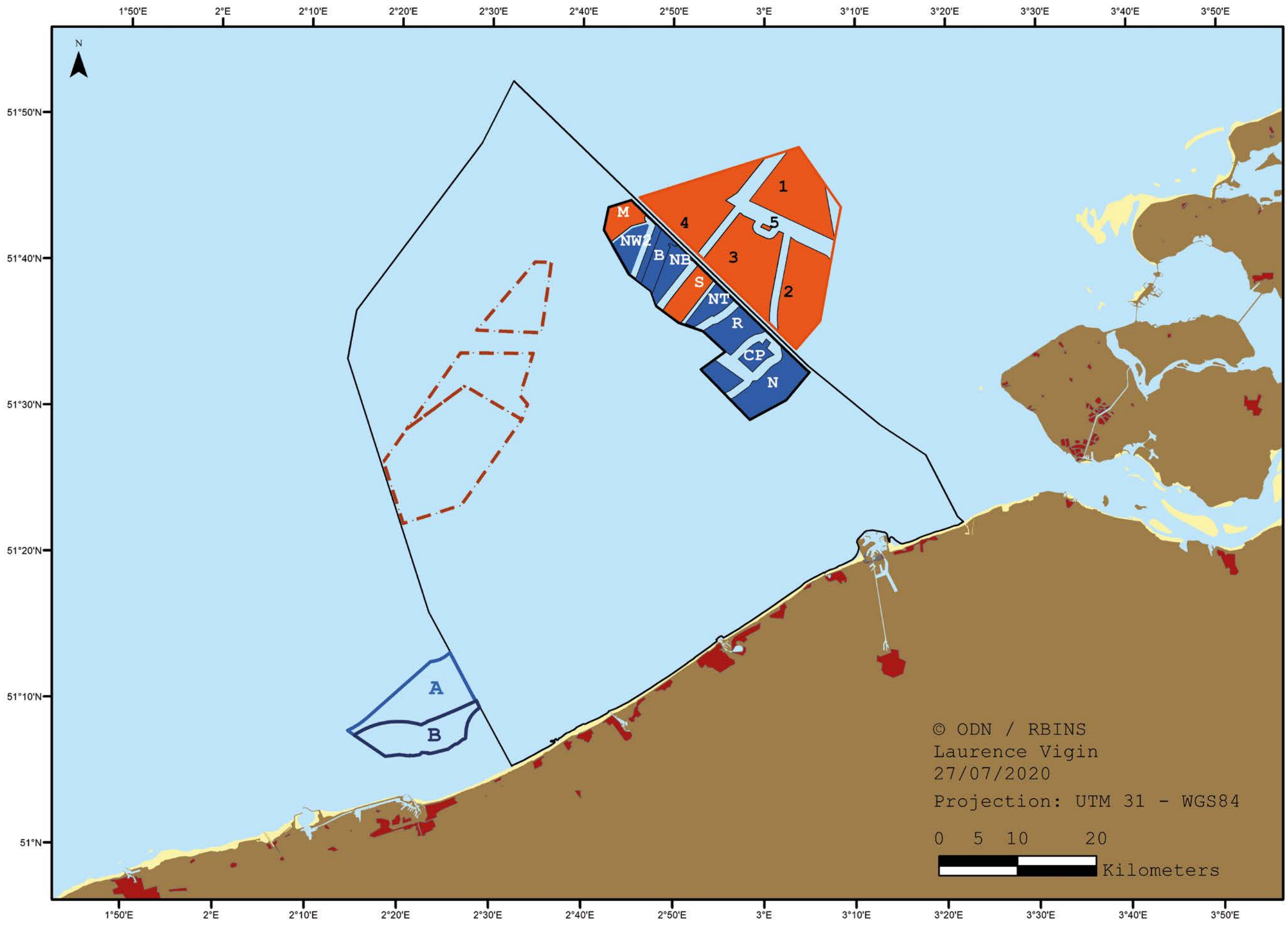

Figure 1. Current and planned zones for renewable energy in and around the Belgian Part of the North Sea with indications of wind farms that are operational (blue) or currently under construction (orange). N: Norther, CP: C-Power, R: Rentel, NT: Northwind, S: Seamade - former Seastar zone, NB: Nobelwind, B: Belwind, NW2: Northwester 2, M: Seamade - former Mermaid zone, 1-5: Borssele wind farm zones 1-5 (Netherlands), A-B: sites of proposed Dunkerque offshore wind farm (France). Dashed lines: locations of the new renewable energy zone as delimited in the marine spatial plan 2020-2026.

the European Commission which envisions a target figure of 17.5% for the contribution of the production of electricity from renewable energy sources by 2030. This plan anticipates 4 GW of operational offshore wind.

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.

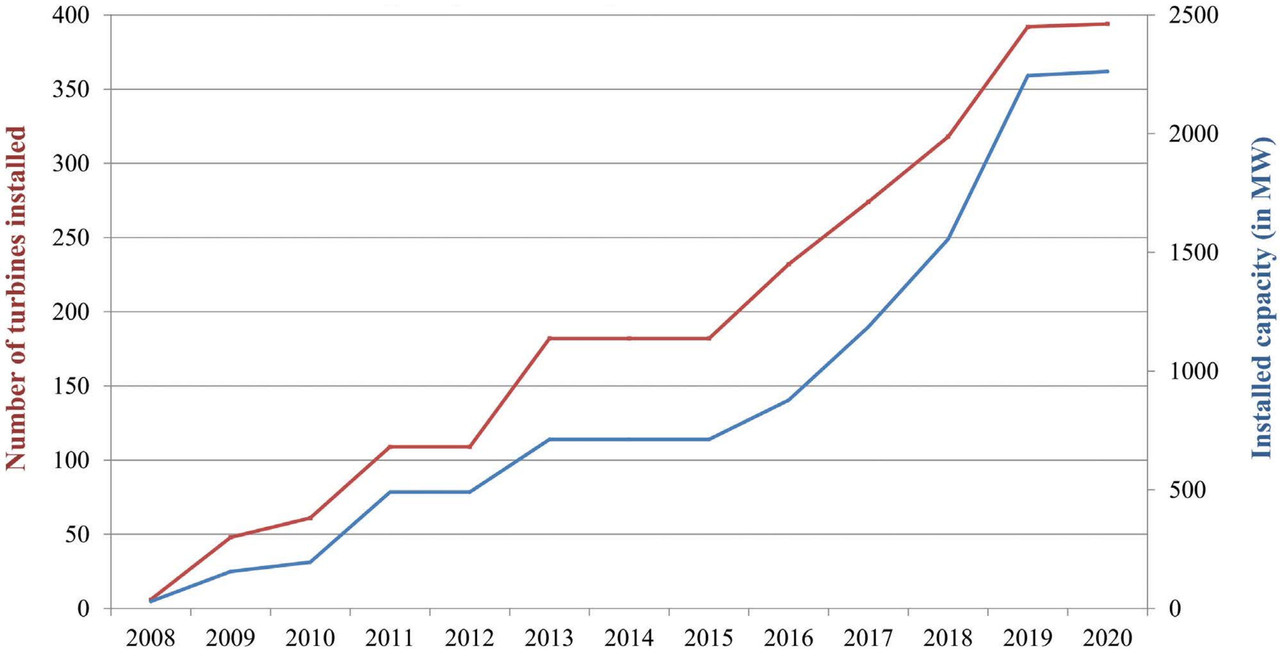

Figure 2. Number of offshore wind turbines installed and installed capacity in the Belgian part of the North Sea since 2008.

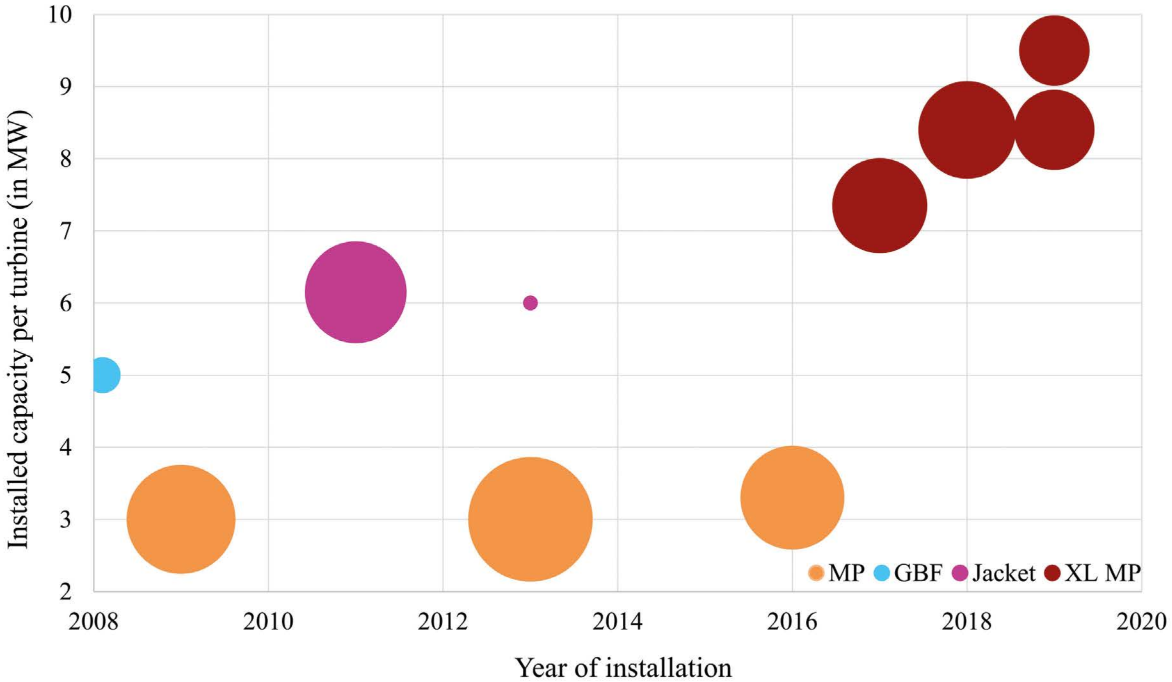

Figure 3. Overview of the timing, individual capacity and foundation type of offshore wind turbines installed in the Belgian part of the North Sea since 2008. The size of the bubbles is proportional to the number of turbines installed per project or phase (see table 1). MP: monopile foundation; GBF: gravity based foundation; Jacket: jacket foundation; XL MP: monopile foundations exceeding approx. 7 m in diameter.

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 2 & Mermaid (table 1; fig. 1). On 20 July 2018, the merger between the Seastar and Mermaid projects was finalized and the resulting merged project was named Seamade NV. By the end of 2020, when all Belgian wind farms are built, there will be a little less than 400 operational wind turbines in the Belgian part of the North Sea (fig. 2). The entire first 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 capacity density of the first wind energy zone, defined as the ratio of the wind energy zone rated capacity to its ground area, is at 9.5 MW/km² among the highest in Europe which results in a higher levelized cost of electricity then other North Sea countries. 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/km² for future developments in order to be able to realize maximum energy yields, and thereby reduce production costs. Over the last decade, installed capacity per turbine has gradually increased with extra-large monopiles (*i.e.* with a diameter larger than 7 m) becoming the dominant foundation type in our (shallow) waters (fig. 3).

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 an environmental 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 2020, the second marine spatial plan for the BPNS (Royal Decree of 22 May 2019 establishing the marine spatial planning for the period 2020 to 2026 in the Belgian sea-areas) came into force. This 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) for the period 2020-2026. Management actions,

Table 1. Overview of wind farms in the Belgian part of the North Sea

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 & 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 2019
Northwester 2		23	9.5	219	Operational since May 2020
SeaMade		58	8.4	487	Operational by the end of 2020

Figure 4. The NEMOS Wave Energy Converter prototype at the site of the Blue Accelerator (photo NEMOS).

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. In this revision of the marine spatial plan, the Belgian federal government has delineated three new zones for renewable energy which cover a total area of 285 km^2 and are located at least 32 km from the coast (fig. 1). These new zones were named the Princess Elisabeth Zone and would be suitable for an additional 2 GW of installed capacity. 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.

This second Belgian zone for marine renewable energy is partly located inside a designated Natura 2000 area. A targeted research programme was designed in order to determine whether and how renewable energy development is compatible with the conservation objectives for this Natura 2000 area. This programme commenced in 2019 and is expected to last four years. The first results will become available for the 2021 monitoring report.

3. 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 realized. 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 (now Seamade) project allows for an installed capacity of 20 MW of WECs no actual WEC deployment is foreseen in the immediate future.

Test sites are an essential element of any emerging technology developments including wave energy extraction. One such test site, the maritime innovation and development platform Blue Accelerator, was constructed of the coast of Ostend in April 2019. Since October 2019, NEMOS GmbH has been testing and evaluating the performance and survivability of its complete NEMOS Wave Energy Convertor prototype at this site (fig. 4).

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CHAPTER 2

AN EVALUATION OF THE NOISE MITIGATION ACHIEVED BY USING DOUBLE BIG BUBBLE CURTAINS IN OFFSHORE PILE DRIVING IN THE SOUTHERN NORTH SEA

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Abstract

In Belgian waters, in 2019, two wind farm construction projects were ongoing, the Northwester 2 (NW2) and the Seamade (SEA) projects. For both projects, turbine foundations were installed using hydraulic pile driving technique with a double big bubble curtain (DBBC) deployed to minimise the underwater sound levels emitted. For the installation of steel monopiles of 7.4 and 8 m diameter, large hydraulic hammers were used of 3000 and 4000 kJ respectively. These projects were the first to use the DBBC mitigation system in Belgian waters. In this study, the underwater sound generated during 14 full-pile driving events, seven per project, was analysed *in situ*. Measured zero to peak sound levels (L_{z-p}) showed values ranging from 183 to 193 dB re 1 µPa when normalised to a distance of 750 m from the source. This represented an estimated sound reduction of 20 dB re 1 µPa for NW2 and 12 to 20 dB re 1 µPa for Seastar. This made NW2 the first offshore wind farm whose pile driving was in compliance with the current Belgian Marine Strategy Framework Directive (MSFD) threshold for impulsive underwater sound.

1. Introduction

In Belgian waters, a zone was reserved to develop energy production at sea (fig. 1). By the end of 2020, the last two wind farms of this zone, Northwester 2 (NW2) and Seamade (comprised of the Mermaid & Seastar zones), will have completed construction activities and together with the other six parks, represent nearly 400 operational wind turbines with a combined installed capacity of 2200 MW. Together with the adjacent Dutch Borssele wind energy zone (1500 MW), this area is rapidly becoming the world's largest operational offshore wind energy area.

Steel monopile foundations are by far the most widely adopted substructure foundation system in the shallow Southern North Sea (see chapter 1 in this report).

Construction of NW2 and Seamade required the installation of 24 and 28 large steel monopiles of 7.4 to 8 m diameter, respectively. A large hydraulic hammer was required to drive these steel piles ~50 m into the seafloor. As a consequence, a large quantity of energy was introduced underwater in the form of sound.

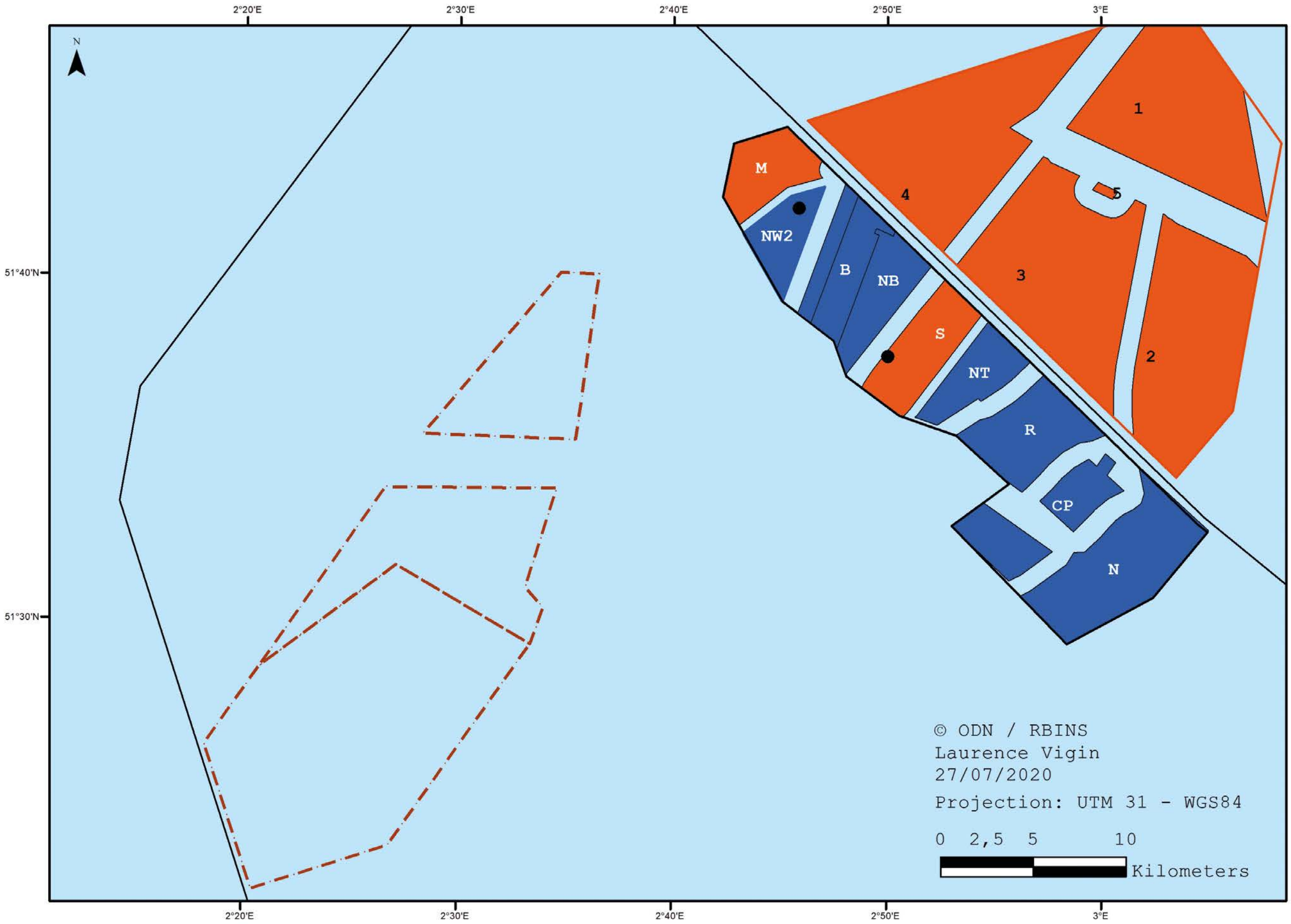

Figure 1. Belgian and adjacent Dutch zone of offshore energy production. Belgian zone (North to South). M: Mermaid; MW2: Northwester 2; B: Belwind; NB: Nobelwind; S: Seastar; NT: NorthWind; R: Rentel; CP: C-Power; N: Norther. The hydrophone mooring sites used in this study are represented by the black dot inside S and NW2. Dashed lines: locations of the new Belgian renewable energy zone as delimited in the marine spatial plan 2020-2026.

The piling of such a large steel monopile of 8 m diameter can produce zero to peak sound levels ($L_{z\text{-}p}$) of more than 200 dB re 1 µPa at 750 m from the source (Bellmann *et al.* 2017). In Belgium, the impulsive underwater sound should not exceed an $L_{z\text{-}p}$ of 185 dB re 1 µPa at 750 m distance from the source (Belgische Staat 2018). The environmental license conditions of NW2 and Seastar requires the developers to comply with this national threshold for impulsive underwater sound.

Both concessioners proposed to use a sound mitigation system consisting of a double big bubble curtain (DBBC). A bubble curtain is formed around a pile by freely rising bubbles created by compressed air injected into the water through a ring of perforated pipes encircling the pile. In a DBBC, two rings of perforated pipes are positioned on the seafloor around the foundation to be piled. Compressors located on the construction vessel feed air into the pipe. The air passes into the water column by regularly arranged holes. Freely rising bubbles form a large curtain around the entire structure, even during tidal conditions, thus shielding the environment from the noise source (OSPAR 2014).

At the end of July 2019, the project NW2 (219 MW installed power) started the construction with the monopile A02 on 29 July and completed the piling work with the installation of the monopile D01 on the

14 November 2019. NW2 is located at about 50 km off the Belgian North Sea coast and is one of the wind farms further away from the coast together with Mermaid.

The Seamade project is comprised of two zones: Mermaid (252 MW installed power) and Seastar (235.2 MW installed power). Seamade construction started on 8 September at Mermaid and on 21 September for Seastar with monopile SE1 at some 40 km from shore. The Seastar zone is located between Nobelwind and Northwind. The Seamade piling work was concluded on 2 January 2020 with the last Seastar monopile SF2.

The purpose of this report is to quantify the emitted underwater sound measured *in situ* at sea and to discuss whether the use of the double big bubble curtain (DBBC) noise mitigation system, as selected by both projects, is appropriate to comply with Belgian legislation on emitted impulsive underwater noise.

2. Material and methods

2.1. Construction activities and local conditions

The large steel monopiles at NW2 were drilled in the seafloor using a hydraulic hammer IHC Hydrohammer S-3000 capable of producing a maximum percussive energy of 3000 kJ while the installation at Seastar used an IHC Hydrohammer S-4000 capable of a maximum of 4000 kJ. NW2 deployed the hydraulic hammer from the jacking-up platform ‘Vole au vent’ while the DBBC was operated by the support vessel Thor Express using 21 compressors of a total Free Air Delivery (FAD) of 0,45 m^3 min^{-1} m^{-1}. The nozzle-hoses’ length was 720 m for the inner BBC and 900 m for the second BBC layer. At Seastar the IHC S-4000 was operated from the jack-up platform ‘Innovation’ for every piling while the DBBC was operated by the support vessel Master express equipped with 24 SCANTECH ST 1600 HAT RS compressors capable of a FAD of 40,5 m^3 min^{-1} each into a double bubble curtain HY100. The nozzle-hoses’ length was 750 m (inner hose) and 990 m (second layer).

The BPNS is the seat of strong semi-diurnal tides. At both construction sites, the tidal current can be more than 1.5 m/s at a given time during the moon/tidal cycle (Belgian Nautical chart D11). In this North Sea zone, the semi-diurnal tidal current is changing speed and direction all along the 12 h 25 tidal cycle.

2.2. Research strategy

The underwater sound generated by driving of 7.4 and 8 m diameter steel monopiles into the seabed was measured *in situ* during construction. Fourteen full pile driving events were recorded from 29 July 2019 to 21 December 2019 (tables 1 & 3). Various metrics, including L_{z-p}, sound pressure level (SPL), and the sound exposure levels of a single stroke 95 percentile (SEL_{95}) were considered.

2.3. Underwater sound measurement equipment

The underwater sound was recorded from two moored stations (fig. 2). 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 to recover the mooring block after deployment. The pop-up buoy was attached rigidly to the concrete block to avoid perturbing sound. The sound recorder manufacturer RTsys calibrated the complete measurement chain before shipping from the factory. The calibration was verified before and after deployment in the laboratory. This calibration was made using a calibrator

Figure 2. Illustration of the mooring design of the underwater sound measurement equipment (here Norther deployment).

Table 1. Position of the monopiles and instruments in UTM31, distance from the monopile to the measuring equipment NW2 for Northwester 2, and SEA for Seastar mooring.

Names	Center Point Position (Coordinates)		Distance to NW2	Distance to SEA
	UTM 31			
	Easting	Northing	in m	in m
A02	484308	5727043	912	
A03	484641	5728036	947	
B01	483747	5727286	468	
D02	482093	5725927	2458	
D04	480887	5726411	3151	
F02	483509	5724926	2837	
F03	483242	5724201	3587	
NW2 instrument	483737	5727754		
SEA instrument	488500	5719748		
SS-OSS	489230	5719261		877
SA02	489882	5718628		1779
SA03	490550	5718361		2475
SF01	490488	5719306		2037
SB03	487278	5718257		1927
SA04	489842	5717880		2300
SA05	489013	5717467		2337

Table 2. Specificities of the double big bubble curtain, nozzle hose (as provided by concessioners Northwester 2 – NW2 and Seamade – SEA).

FAD available NW2	0.45 $m^3 m^{-1} min^{-1}$
Diameter holes NW2	1.5 mm every 200 mm
Length inner & outer NW2	720 m & 900 m
Distance from monopile & DBBC	55 m & 40 m

FAD available SEA	> 0.5 $m^3 m^{-1} min^{-1}$
Diameter holes SEA	2 mm every 200 mm to 300 mm
Length inner & outer SEA	750 m & 990 m
Distance from monopile & DBBC	100 m & 40 m

B&K 4229 (piston-phone) equipped with a ½ inch precision microphone B&K 4191-l.

The mooring for NW2 was deployed on 26 July 2019 from the research Vessel Simon Stevin at position (WGS84) N 51° 42,03; E 002° 45,88 (fig.1) and for Seastar by the BNS Belgica on 6 November 2019 at the position (WGS84) N 51° 37,72; E 2° 50,03 (fig.1). The distance between the measuring equipment and the piling locations ranged from 468 m to 3587 m. No surface marker was left on-site to reduce risks to navigation inside the construction zone and avoid any perturbing sound originating from a line linking a surface buoy to the mooring.

2.4. Underwater sound measurements and post-treatment

Sound pressure was recorded continuously at a sampling rate of 78125 Hz and stored on hard drives as WAV files.

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 750 m, were computed following Norro *et al.* (2013).

2.5. Double big bubble curtain (DBBC) specificities

Mitigation measures introduce an ‘insertion loss’ between the sound source and the surrounding environment. Both projects used a double big bubble curtain to reduce the emitted underwater sound produced by pile driving. However, the detailed technical specificities are different for each project, and they are summarised in table 2.

3. Results

During the pile driving at NW2, $L_{z\text{-}p}$ at 750 m from the source ranged between 183 to 185 dB re 1 µPa (table 3).

While during the piling of Seastar the level zero to peak ranged from 183 to 193 dB re 1 µPa.

The insertion loss efficiency was increasing at higher frequencies while one may remember that pile driving is producing its highest energy at low frequencies (below 300 Hz; fig. 3).

4. Discussion

4.1. Noise reduction achieved

During the hydraulic pile driving operations at Northwester 2, the *in situ* measured $L_{z\text{-}p}$ values remained below the threshold imposed by the Marine Strategy Framework Directive (MSFD) in Belgian waters. This was almost the case for the piling operated at Seamade even if on some occasions the limit of 185 dB re 1 µPa at 750 m from the source was exceeded (table 3).

In general, a level reduction of about 25 dB (peak level $L_{z\text{-}p}$) is assumed for a DBBC (OSPAR 2014; Bellmann *et al.* 2017). As no pile driving events have been realised without mitigation for these projects, it is difficult to accurately assess the achieved

Table 3. Name, distance from the recorder, date of operation, pile diameter, zero to peak sound levels in dB re 1 µPa as measured *in situ*, SEL_{95} computed, maximal energy deployed by the hammer and $L_{z\text{-}p}$ normalised at 750 m distance from the pile driving location for seven piling events measured at the Northwester 2 (NW2) and Seastar (SEA).

Names	dist.	dist.	Date of piling	Pile diameter	L_{ZP} measured	SEL_{95} measured	Max. energy	L_{ZP} at 750 m
	m	m		M	dB re 1 µPa	dB re 1µPa² S	kJ	
NW2	Hammer S-3000 & Mitigation = DBBC							
A02	912		29/07/2019	7.4	181	156	1600	183
A03	947		15/08/2019	7.4	180	159	1480	183
B01	468		04/08/2019	7.8	188	162	1500	183
D02	2458		12/08/2019	7.6	174	157	1600	185
D04	3151		27/08/2019	7.4	172	154	2100	185
F02	2837		07/08/2019	7.8	173	154	2200	185
F03	3587		25/08/2019	7.8	173	152	2200	185
SEA	Hammer S-4000 & Mitigation = DBBC							
SS-OSS		877	20/11/2019	8	192	170	3500	193
SA02		1779	07/12/2019	8	182	163	2600	188
SA03		2475	05/12/2019	8	177	155	1600	185
SF01		2037	11/12/2019	8	178	158	2500	185
SB03		1927	21/12/2019	8	181	155	1700	187
SA04		2300	04/12/2019	8	176	156	2300	183
SA05		2337	03/12/2019	8	179	158	2000	186

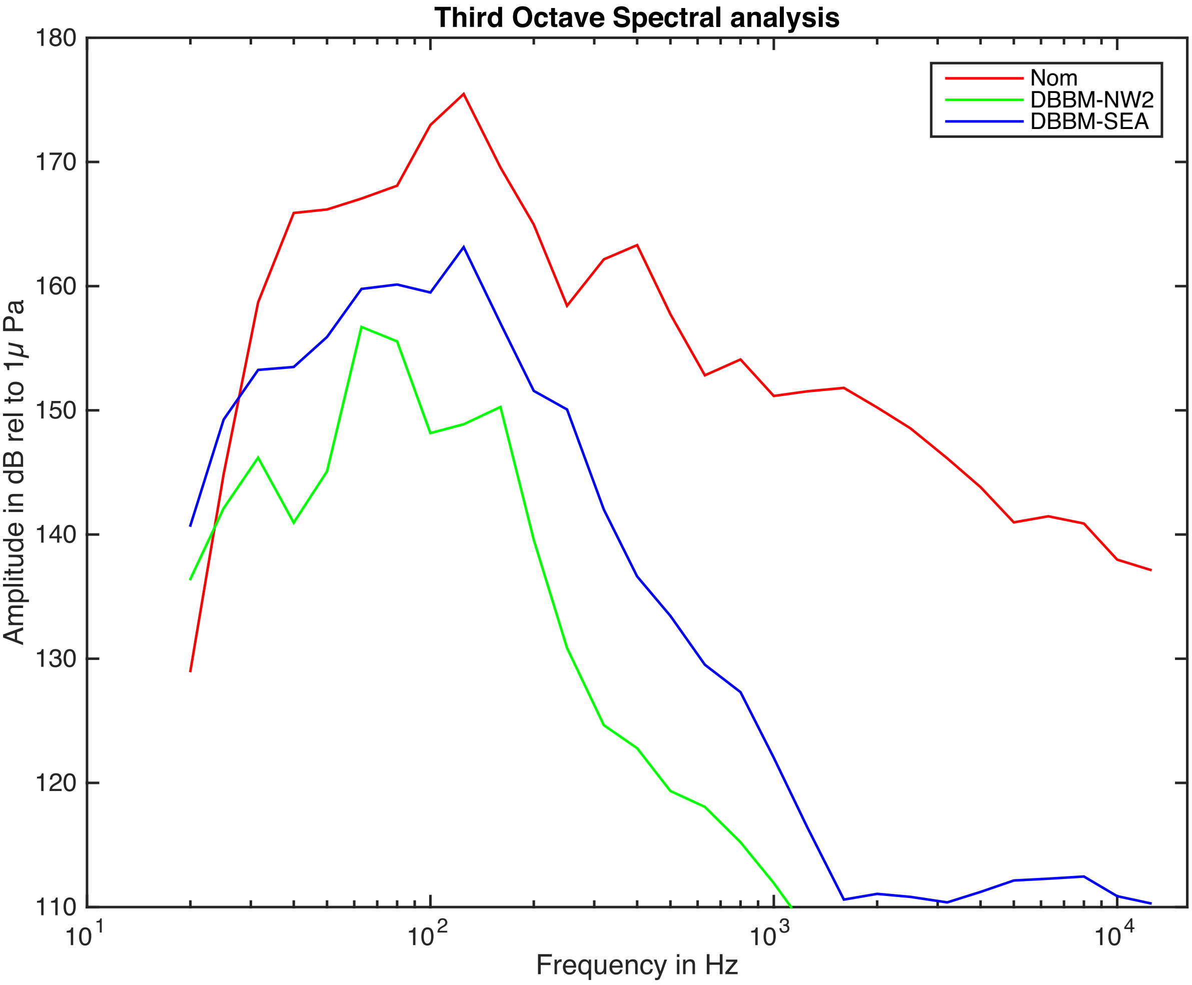

Figure 3. Spectral analysis of the underwater sound pressure level measured *in situ* during piling operated without mitigation measures (Norro 2018) and using the double big bubble curtain at Northwester 2 and Seastar. Respectively Nom, DBBM-NW2, and DBBM-SEA.

reduction of the DBBC in these cases. For a rough assessment, theoretical values given by Bellmann *et al.* 2017 can be used. For an 8 m diameter steel monopile zero to peak level of 205 dB re 1 µPa is expected when no mitigation is used. Using that figure for the piling operated at NW2, an efficient 20 dB re 1 µPa is observed while it is lower for Seastar with a range of 12 to 20 dB as observed in table 3. Norro (2019) showed the possible influence of high tidal current featuring the Belgian continental plate on bubble curtains' efficiency, which may explain this difference (see below).

4.2. Technical aspects affecting noise mitigation

Both projects used different hydraulic hammers, with 3000 kJ for NW2 and 4000 kJ for Seastar but similar mitigation measures with DBBC. However, the setup of the DBBC was different for the two projects (table 2).

Another consideration is the optimisation of the DBBC. One important aspect is the size and distance between the successive holes on the nozzle hose (table 2). Based on Bellmann (2017), the optimal sound reduction is achieved for a hole of 1 to 2 mm diameter and spaced by 20 to 60 cm. Both projects presented here fit the ‘optimal’ requirements. The same applies to the Free Air Delivery (FAD) even if that parameter is more difficult to assess in the field. It is based on a theoretical value computed from the number and the given specification of the used compressor. On some occasions, the complete number of compressors could not be used. On one occasion during our survey of Seastar and for the installation of SB03, 22 compressors out the 23 foreseen were used.

One may remark that in case of damage to the nozzle hoses, some leakage may appear that will reduce the system’s efficiency. No such damage was reported.

Table 3 showed that both projects’ maximum energy were different with a higher value for Seastar. It is generally accepted that an increase of 500 kJ for the hammer energy corresponds to an increase of 2-3 dB re 1 µPa in $L_{z\text{-}p}$ measured at 750 m distance from the source (Müller *et al.* 2019).

4.3. Operational aspects affecting noise mitigation

The tidal current may affect the efficiency of bubble curtains. When the current is high, the bubble curtain may be shifted by the current and may be displaced beyond the monopile’s footprint, reducing the mitigation effect. NW2 considered a maximum tidal current of about 1.2 m/s, while Seastar considered a mean current of 0.6 m/s. It is the local instantaneous tidal current that affects the bubble curtain. Suppose the mean current is used for the setup of the DBBC. In that case, it is advisable to concentrate the piling operation on the tidal windows characterised by tidal current speeds lower or equal to the mean current speed value. The shape of the DBBC is better designed, taking into account the maximum tidal current’s local direction and speed.

Another point to discuss is the difference in the sound mitigation system’s efficiency with the frequency of the sound. Figure 3 confirms that the efficiency of the mitigation measure is not equal at every frequency. The efficiency is higher for frequencies above 300 Hz while the energy emitted underwater is mainly below that frequency during piling operated installation.

To comply with Belgian MSFD regulation Belgische Staat (2018) it is advisable to, at least, test on-site the combination of sound mitigation measures as they will be deployed and used before the construction work starts and not to rely only on theoretically predicted efficiency. From the experiment conducted during the Norther wind farm construction in 2017 (Norro 2018), it was shown (Bellmann personal communication) that sound reduction figures obtained from the German waters cannot be transferred as such in Belgian water. This is at least due to the difference in tidal current between the North Sea regions.

5. Conclusion

Compared to previous projects (Norro 2018, 2019), the use of DBBC enhanced compliance with the national MSFD limit with $L_{z\text{-}p}$ ranging from 183 to 193 dB re 1 µPa at 750 m from the source. The sound emitted during pile driving for the Seamade project regularly exceeded the Belgian MSFD threshold for impulsive sound of 185 dB re 1 µPa at 750 m despite using a DBBC noise mitigation system. However, using a similar DBBC, the Northwester 2 project managed to comply with regulations. Little differences between both project setups existed except for the hydraulic hammer used that was more powerful for the Seamade project with a 4000 kJ hydraulic hammer deployed to install a little larger monopile. Further improvements on noise reduction could be obtained by an obligatory fine-tuning of noise mitigation measures at the start of new projects *e.g.* the influence of tidal currents must be taken into account with pile driving restrictions the windows of maximal tidal current if necessary.

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CHAPTER 3

FIT FOR PORPOISE? ASSESSING THE EFFECTIVENESS OF UNDERWATER SOUND MITIGATION MEASURES

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Abstract

In this chapter, we review how developers complied with the environmental license conditions formulated to mitigate the potential negative impacts of pile driving on marine mammals in the Belgian part of the North Sea (BPNS), whether this impacted the timing of development and what the likely consequences were for marine mammals. Between 2009 and 2020, offshore wind farm developers in the BPNS complied to a large extent with those environmental license conditions formulated to mitigate the potential negative impacts of pile driving on marine mammals. However, we did identify several possible improvements to these environmental license conditions, including changes in the use of acoustic deterrent devices, formalising obligatory mammal surveys, and requiring developers to comply with the national threshold for impulsive underwater sound. The reduction in the costs of applying noise mitigation measures ensures that these suggested improvements should not affect the economic viability of future projects.

1. Introduction

In December 2000, the first offshore wind farm (OWF) in the North Sea became operational. It was located 1.6 km off the coast of Blyth, England and comprised two 2 MW turbines. Since then, offshore wind in the North Sea has grown with leaps and bounds, and 20 years later the North Sea has a total installed capacity of 22 GW. This growth is expected to accelerate as, in order to meet the EU objective of reaching net-zero greenhouse gas emissions by 2050, offshore wind capacity in the North Sea should increase to a total installed capacity of at least 150 GW in the next thirty years (North Seas Energy Cooperation 2020). The installation of wind turbines, electric transformer stations and power cables in the marine environment has a range of environmental effects which depend among others on the location, timing and methods of installation (Lindeboom *et al.* 2011; Degraer *et al.* 2013). Prior to installing a renewable energy project in the North Sea, a developer must obtain an environmental permit (see chapter 1

Table 1. Overview of environmental permit conditions for hydraulic pile driving in the Belgian part of the North Sea (2004-2015)

Project name	Environmental permit granted	Use of Acoustic Deterrent Device	Marine mammal survey prior to pile driving	Seasonal pile driving ban from January 1st to April 30th	Noise mitigation measures for impact pile driving*	Start pile driving	Pile driving events*
C-Power	14-04-2004 Modified 25-04-2008	“to be used at least half an hour prior to the start of pile driving” “starting one hour before up to the start of pile driving”	No Yes, “half an hour prior to the start of activities”	No No, but an additional cost applies when pile driving in this period	No	07-04-2011	54
Belwind	20-02-2008	“starting one hour before up to the start of pile driving”	Yes, “half an hour prior to the start of activities”	No, but an additional cost applies	No	07-09-2009	67
Northwind	19-11-2009					07-04-2013	73
Nobelwind	13-05-2015					16-05-2016	51
Norther	18-01-2012		Yes, “before and during pile driving activities”	Yes	SBBC	06-08-2018	45
Rentel	08-02-2013	“starting half an hour before up to start of pile driving”				21-07-2017	43
SeaMade	07-02-2014 (Seastar) 13-04-2015 (Mermaid)				DBBC	08-09-2019	60
Northwester 2	18-12-2015					29-07-2019	24
Elia MOG	07-07-2014				GABC	04-11-2018	3

* A pile driving event refers to any instance where hydraulic pile driving takes place after a period of at least three hours of no pile driving. Therefore, the installation of a single turbine foundation can comprise multiple pile driving events. A short description of the noise mitigation measures is provided in the results section.

** All environmental permits also require the developer to use a ‘ramp-up’ or ‘soft start’ procedure at the start of pile driving. In this case, lower hammer energy levels are used to start the pile driving process, and then the force of pile driving is gradually increased.

of this report), which includes terms and conditions intended to minimise and/or mitigate the impact of the project on the marine ecosystem. Some of the mitigation measures formulated to reduce the impact of OWF construction on marine mammals are considered onerous by developers as they increase project cost both directly (*i.e.* the cost of the mitigation measures) and indirectly (by increasing construction time; Koschinski & Lüdemann 2013). In addition, there is a discussion as to whether the benefits of using of an Acoustic Deterrent Device (ADD) to scare away marine mammals prior to pile driving outweigh the negative consequences of thereby prolonging the introduction of high levels of underwater sound into the marine environment (Graham *et al.* 2019; Rose et *al.* 2019). Given that construction activities in the first Belgian offshore energy zone have come to an end, and that new mitigation measures need to be prepared in anticipation of the licensing of the second Belgian offshore energy zone (see chapter 1 in this report), now is a good time to evaluate those mitigation measures formulated to reduce the impact of pile driving on marine mammals.

In this paper we combined data from developers and regulators to analyse whether developers complied with the environmental license conditions formulated to mitigate the potential negative impacts of pile driving on marine mammals in the Belgian part of the North Sea (BPNS), how this impacted the timing of development and what the likely consequences were for marine mammals.

2. Material and methods

2.1. Data acquisition and analysis

Using the website of the Scientific Service Management Unit of the Mathematical Model of the North Sea (MUMM: https://odnature.naturalsciences.be/mumm/en/windfarms/), we accessed the environmental permits of the nine wind farms and the offshore switchyard platform that engaged in pile driving activities in the BPNS in the period 2009-2020 (permits granted 2004-2015). Prior to 2009, no pile driving occurred in the BPNS. Those terms and conditions in the environmental permits that were intended to minimise and/or mitigate the impact of OWF construction on marine mammals are listed below (table 1). Initial permit conditions were aimed at preventing near-field injury to

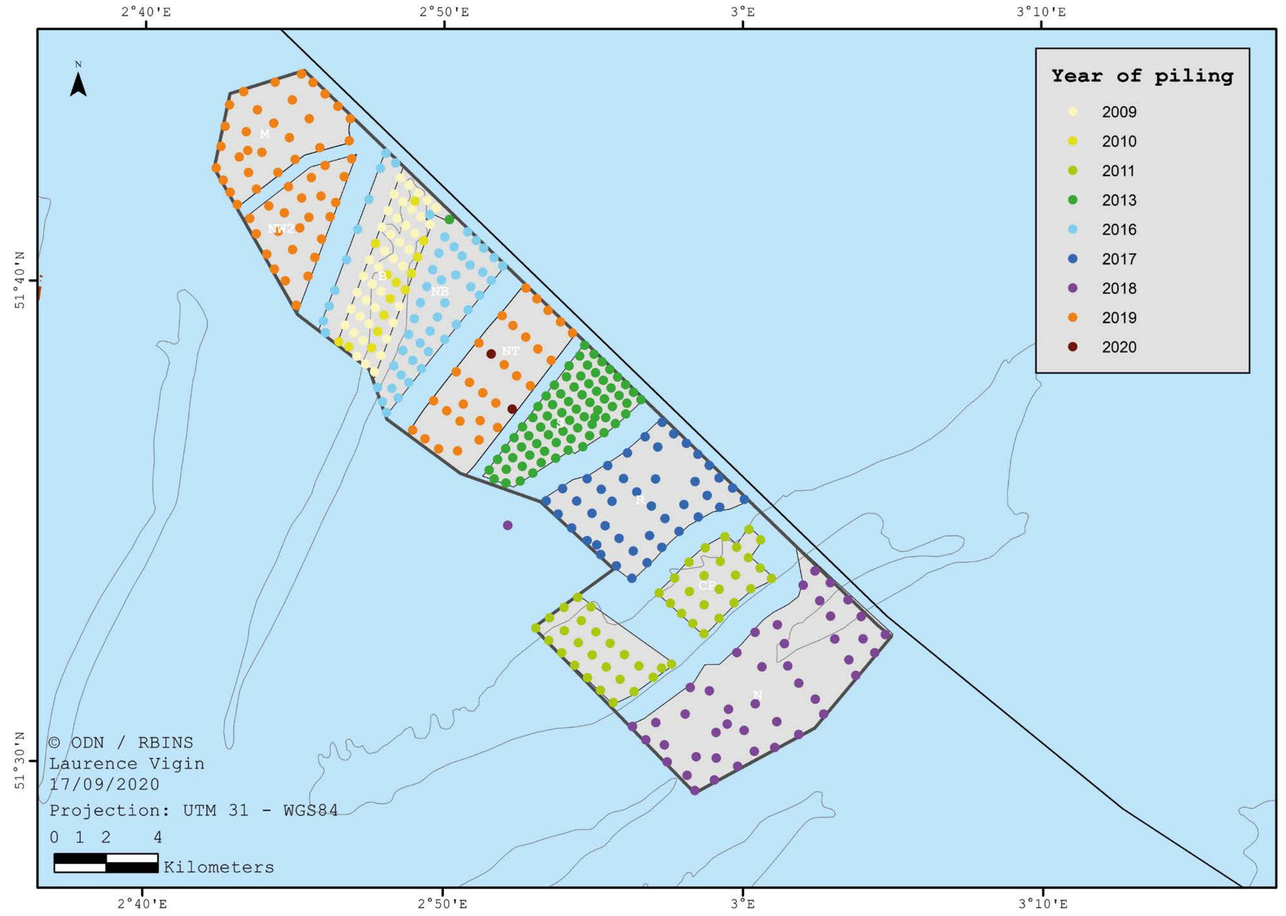

Figure 1. Timing and location of pile driving events in the Belgian part of the North Sea (period 2009-2020, data RBINS).

individual animals, and included the use of an acoustic deterrent device (ADD) as well as a prohibition on starting pile driving if a marine mammal was observed in the vicinity of the construction zone. Progressive insight in the potential population consequences of far-field behavioural disturbance resulting from exposure to excessive levels of impulsive underwater sound led to the formulation of further permit conditions. These included a seasonal pile driving ban from January 1st to April 30th, and an obligation to use noise mitigation measures that limit the transmission of noise pollution to the marine environment. All permits also included an obligation to use a soft start or ramp up method, whereby a pile is initially driven with low hammer energy which is gradually increased with increasing soil penetration. Since a soft start is an operational necessity for pile driving, we don't consider it as a real mitigation measure for the purposes of this paper.

Compliance of the developer with these conditions was checked based on the information provided by the developer in the daily reports on piling activities and confirmed by the aerial surveillance program as well as the environmental monitoring conducted, both coordinated by the Operational Directorate Natural Environment of the Royal Belgian Institute of Natural Sciences. The timing and location of pile driving events in the BPNS are shown in fig. 1.

The interim Population Consequences of Disturbance (iPCOD) model was developed to assess the potential effects of anthropogenic noise, associated with offshore renewable energy developments, on harbour porpoise populations (Harwood *et al.* 2013; Nabe-Nielsen & Harwood 2016). In this model population dynamics are simulated based on the birth and average survival rates of harbour porpoises in the North Sea, derived from expert elicitation (Booth *et al.* 2019; Sinclair *et al.* 2019). In this report, we used the latest update of the iPCOD model (version 5.2 – released on 2 October 2019) to compare the consequences of the realised pile driving activities in the BPNS (2009-2019) on a local population of harbour porpoise under three scenarios': without any mitigation measures (no mitigation), a second scenario with the applied mitigation measures (as-it-happened) accounting for the applied mitigation measures, and accounting for the observed reductions in underwater sound (as reported in Norro 2018, 2019 and this report), and a final scenario (optimal configuration) in which the most successful combination of mitigation measures (as applied for NW2 in 2019; Norro, this volume) is assumed for all OWF construction in the BPNS. For an overview of the input parameters, and underlying assumptions used in these scenarios': see Rumes & Debusschere (2018). In theory each pile driving event could lead to two residual days of disturbance as the reduction of detection rates has been observed starting one day before and up to two days after pile driving (Brandt *et al.* 2016; Rumes *et al.* 2017). However, given the nature of the actual piling calendars – with piling activities often taking place less than 24 h after the previous event – it was decided that each day with pile driving could lead to only one residual day of disturbance. Every scenario was simulated 1000 times.

Statistical analyses and coding were performed in the Rstudio environment under R version 4.0.0. Plots were generated in Rstudio using both the lattice and ggplot2 packages.

3. Results

3.1. Compliance

3.1.1. Use of Acoustic Deterrent Device

Seal scarers were originally designed to deter seals from fishing gear and aquaculture installations to avoid depredation on fish. In the context of OWF construction, they can be referred to as both acoustic deterrent devices (ADDs) and acoustic harassment devices (AHDs). ADDs transmit short sounds in a frequency range of 10-40 kHz (most often

Figure 2. Lofitech Acoustic Deterrent Device of the type used during the construction of the Belwind wind farm. The figure shows the control unit (left) and transducer (on the right). The control unit contains a pulse generator and an amplifier and transmits random bursts of audio frequency signals to the transducer, where they are converted into intense sound (from www.seiche.com).

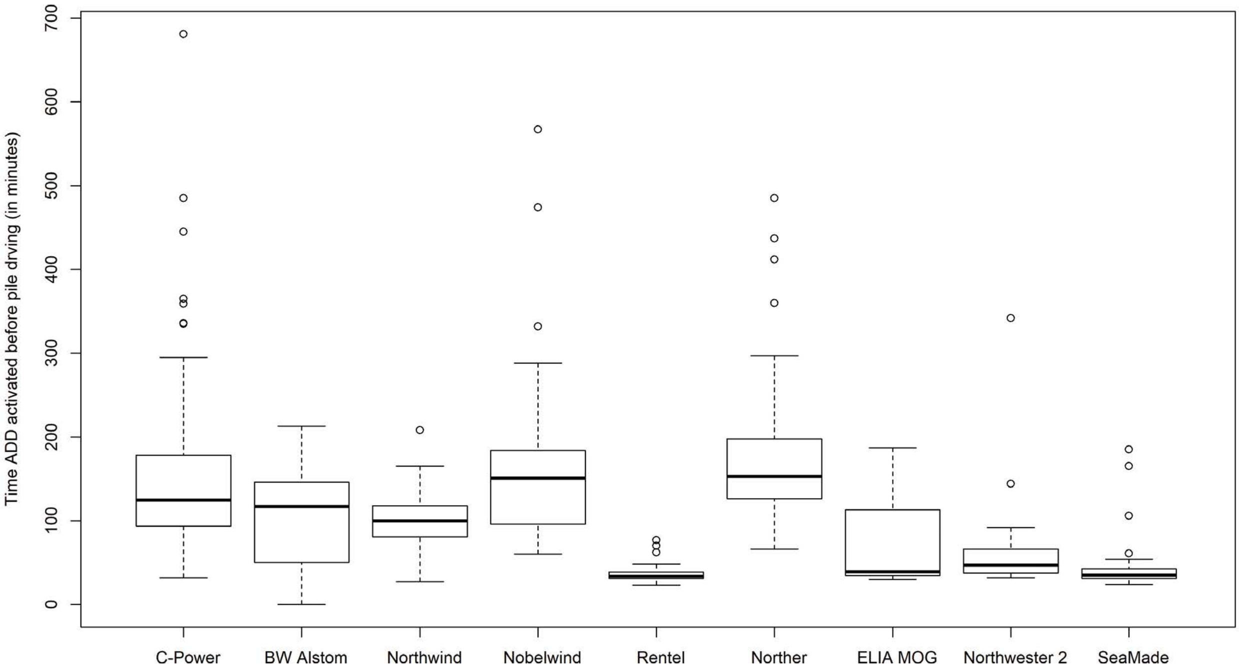

Figure 3. Timing of ADD activation relative to the start of hydraulic pile driving.

with main energy at 10-14 kHz). The length of pulses and intervals between them are often randomised to decrease the potential for animals habituating to the sounds, so that aversion effects can be maintained over time (Hermannsen *et al*. 2015). In compliance with their environmental permit, all developers reported on the use of ADDs prior to the start of pile driving. The devices used were approved by MUMM prior to deployment. Most projects used a LofiTech Seal scarer which operates at a frequency of 14 kHz (fig. 2). For the Norther project a FaunaGuard was used which operates at frequencies of 60-150 kHz and is designed to deter harbour porpoises. The different sounds are based on the hearing range and sensitivity of this species (frequency spectrum) and the reaction threshold levels, based on known literature and extensive behavioural response experiments (Van der Meij *et al.* 2015). As stipulated in the permit conditions, the ADDs had a source level of 170 to 195 $\text{dB}_{\text{p-p}}$ re 1 μPa. In practice, ADDs were often activated well before pile driving started and they were used for a much longer period than anticipated (fig. 3, data on ADD activation was not available for the Belwind project). In some extreme cases, this was due to technical difficulties preventing the start of hydraulic pile driving. On average ADD deployment lasted for 134 minutes (SD: 109 minutes). In only two instances the ADD was not used. In a few instances, the ADD was only turned off hours after pile driving was ended resulting in a significant prolongation of the period of acoustic disturbance.

3.1.2. Marine mammal survey prior to pile driving

Based on the daily reports provided by the developers, a watchman conducted a marine mammal survey prior to pile driving whenever this was possible (daylight hours). No marine mammals were observed by the watchmen nor was any mortality of animals (fish, seabirds, squids, or marine mammals) reported. As a consequence, at no point was pile driving delayed due to the presence of marine mammals in the vicinity of the pile driving platform.

3.1.3. Seasonal pile driving ban from January 1st to April 30th

As of 2013, the environmental permit of new projects included a seasonal ban on hydraulic pile driving in Belgian waters from January 1st to April 30th in order to avoid the period with consistently highest local harbour porpoise densities. This was respected by all projects, except for the Seamade project that installed its last two turbine foundations on January 1st and 2nd 2020 after obtaining a derogation from the Minister.

3.1.4. Noise mitigation measures for impact pile driving

Concern over the high levels of underwater noise being generated during pile driving operations for the building of the first OWFs (Norro *et al.* 2010, 2013) and the observed large scale avoidance of the construction zone by porpoises (Haelters *et al.* 2010) has led to the formulation of a threshold for impulsive underwater sound in the BNS at 185 dB re 1 μPa (Sound Pressure Level, zero to peak) at 750 m from the source (Anonymous 2012). In compliance with their environmental permits, since 2017, all subsequent projects have used various types of noise mitigation measures (see table 1) with varying levels of success (Norro 2018, 2019; chapter 2 in this report). These included single big (SBBC), double big (DBBC) and grout annulus bubble curtains (GABC). A bubble curtain is formed around a pile by freely rising bubbles created by compressed air injected into the water through a ring of perforated pipes encircling the pile. A SBBC is a ring of perforated pipes positioned on the sea floor around the foundation to be piled. Compressors located on the construction vessel or on a platform feed air into the pipe. The air passes into the water column by regularly arranged holes. Freely rising bubbles form a large curtain around the entire structure, even in tidal conditions, thus shielding the environment from the noise source (Koschinski & Lüdemann 2013). DBBCs add a second of ring of perforated pipes around a BBC. The GABC is generated by blowing air into the annulus between the skirt sleeve and pin pile of the jacket foundation. Within the annulus, the air bubbles are protected from the current resulting in a stable air-water mixture that acts as an impedance barrier to the pressure waves generated by the pile during each hammer strike. At the top of the annulus, about 10 m above seabed, the bubbles drift out and are subjected to the current, carrying them away from the pile (Lippert *et al.* 2017).

3.2. Effects of mitigation measures on offshore wind farm construction

Since all projects are different in ways that influence the speed of construction (foundation type, soil conditions, weather conditions in the year of installation…) we limited ourselves here to a rough comparison of the average installation time per foundation (installation period divided by the number of installed foundations) for projects with and without noise mitigation measures (table 2). The ELIA MOG project and Belwind Alstom demo turbine could not be included in this comparison as these required the installation of only a single foundation eliminating the need to move the construction vessel and set up noise mitigation measures. Based on the available data, there is no indication that the need for noise mitigation increased installation time for offshore wind projects in the BPNS. In recent years, the overall time needed to install a project's foundations has decreased as fewer, larger turbines are being installed (table 2).

3.3. Population consequences of mitigation for marine mammals

The mean porpoise population decline at the end of the OWF construction period exceeded 1.5% for both the scenario without any mitigation measures as well as the scenario accounting for the applied mitigation measures (fig. 4). For the scenario in which the most successful combination of mitigation measures is assumed for all OWF construction in the BPNS, mean porpoise population decline at the end of the construction period was only 0.1% (table 3). More importantly, relative differences between the scenarios indicate that the applied mitigation measures reduced mean porpoise population decline at the end of the OWF construction period by 50%, and that currently available mitigation measures would have reduced porpoise population decline by 97%.

4. Discussion

4.1. Compliance

4.1.1. Optimising the use of acoustic deterrent devices

ADDs have been widely used during the construction of OWFs in the North Sea in order to make sure that marine mammals vacate the immediate vicinity of construction sites prior to the start of pile driving and thereby avoid near-field injury (JNCC 2010; Hermannsen *et al.* 2015; Brandt *et al.* 2016). Yet, there is increasing evidence that the use of ADDs can cause a prolonged

Table 2. Installation time for offshore wind projects in the Belgian part of the North Sea

Project	Foundations	Start installation	End installation	Noise mitigation	Construction period (days)	Days needed per foundation
Belwind	56	07/09/2009	04/02/2010	None	150	2.7
C-Power	49	04/04/2011	21/08/2011	None	139	2.8
Northwind	73	07/04/2013	09/09/2013	None	155	2.1
Nobelwind	51	16/05/2016	22/09/2016	None	129	2.5
Average	**57.3**				**143.3**	**2.54**
Rentel	43	21/07/2017	23/09/2017	BBC	64	1.5
Norther	45	06/08/2018	12/11/2018	BBC	98	2.2
Northwester 2	24	29/07/2019	13/11/2019	DBBC	107	4.5
Seamade	60	08/09/2019	02/01/2020	DBBC	116	1.9
Average	**43.0**				**96.3**	**2.52**

BBC: Big Bubble Curtain. DBBC: Double Big Bubble Curtain.

Table 3. Overview of the outcome of the three scenarios' simulated with the iPCOD model, showing the mean decrease (%) in porpoise population size at six (end of the first phase of constructions), twelve (end of the second phase of constructions), and twenty-five (end of the simulation) years after the start of the piling calendar

	Mean decline in porpoise population* after			Reduction compared
Scenario	6 years	12 years	25 years	to 'No mitigation'
1. No mitigation	1.81	3.75	3.42	–
2. As-it-happened	0.92	1.88	1.73	50%
3. Optimal configuration	0.04	0.11	0.09	97%

* as compared to an unimpacted population

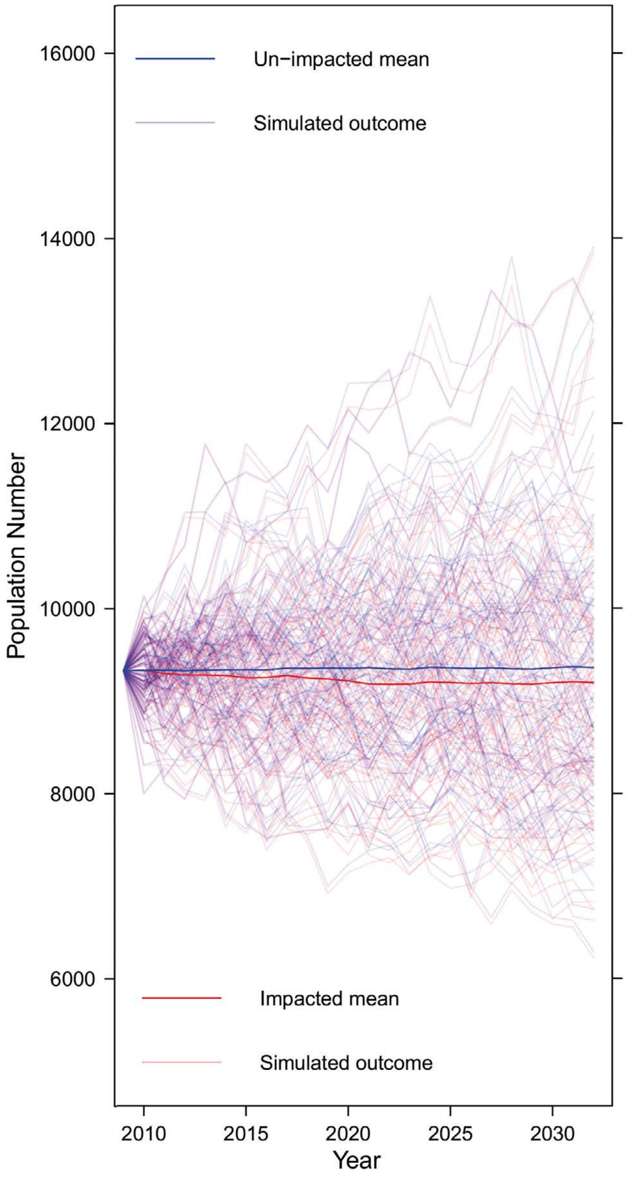

Figure 4. Line plots showing the mean population trajectories and the 1000 simulated populations for the un-impacted population (blue), the impacted population (red), both overlain for scenario 2 with the actually applied mitigation measures showing a decrease in mean porpoise population size of 1.73% after 25 years.

introduction of high levels of underwater sound into the wider marine environment which may have far-field disturbance effects (Brandt *et al.* 2013; Graham *et al.* 2019; Rose *et al.* 2019). The environmental permits include clear guidance on ADD use in order to limit prolonged introduction of high levels of underwater sound: ADDs are to start an hour (permits up to 2012) or half an hour (permits since 2013) before the start of pile driving and are to be shut down when pile driving starts. Even accounting for occasional technical difficulties, for the projects in the BPNS, ADDs were too often turned on too soon and left on longer than needed. It is recommended that for future projects, the timing of any use of ADDs is monitored closely during the construction period allowing for immediate remedial actions. It has also been suggested to use a substantially less powerful pinger as an initial deterrent prior to the use of the ADD (Skjellerup *et al.* 2014). Such a pinger could potentially replace the ADD altogether, as both increased vessel noise (Dyndo *et al.* 2015) and the constant use of sonar by large vessels in the construction zone are likely to have already deterred most nearby porpoises (Rose *et al.* 2019).

4.1.2. Formalising obligatory marine mammal surveys prior to pile driving

Passive acoustic monitoring has shown that there are still porpoises present in the construction zone during foundation installation (Rumes *et al.* 2017). To avoid injuring these animals, pile driving activities cannot commence and must be stopped when a marine mammal is observed at less than 500 m from the construction vessel. The environmental permit requirement to conduct a marine mammal survey prior to pile driving was inspired by the statutory nature conservation agency protocol for minimising the risk of injury to marine mammals from piling noise (JNCC 2010). However, in contrast to the aforementioned protocol, it was not required that this survey was conducted by an appropriately trained marine mammal observer (MMO), which may explain why not a single marine mammal was observed during these surveys in the past ten years. For the next project, it is worth having a trained MMO on board conducting regular visual marine mammal surveys and evaluating how this influences detections. In addition to observer training, there are several factors that likely contributed to the fact that no marine mammals were observed, including those that influence both visibility (pile driving at night and during low visibility) and availability (avoidance of work zone due to high vessel activity, low seasonal porpoise densities).

4.1.3. Seasonal pile driving ban from January 1st to April 30th

There is a long history of regulatory agencies using seasonal restrictions on activities to avoid harming marine mammal populations (Richardson 1995). In the North Sea, the Dutch authorities initially banned pile driving for offshore wind farms from January 1st to July 1st (Arends *et al.* 2009), and later moved to a dynamic underwater sound threshold (MEZ 2015). In the BPNS, a seasonal ban on hydraulic pile driving in Belgian waters from January 1st to April 30th has been enforced since 2016. Overall compliance with the seasonal pile driving ban has been very good with (almost) no pile driving taking place in this period of highest local porpoise densities.

4.1.4. Efficient noise mitigation measures are likely to reduce the impact of pile driving

From 2017 onwards, underwater noise mitigation measures were used during pile driving. Unfortunately, there was a lower than expected performance of the SBBC, which is likely due to local hydrodynamic conditions and/or sub-optimal use of the equipment (Norro 2018, 2019). As a result, most projects routinely exceed the national threshold on impulsive underwater sound set at 185 dB re 1 µPa (sound pressure level, zero to peak; SPL_{z-p}) at 750 m from the source (Anonymous 2012). For the last two projects, which used DBBC, the *in situ* measured SPL_{z-p} mostly remained below the national threshold (Norro, chapter 2 in this volume) showing that it is possible to adequately reduce the amount of underwater noise being introduced into the marine environment by using a combination of noise mitigation measures. It is recommended that future projects only be allowed to continue construction after they have demonstrated their ability to comply with national underwater sound regulations using data from the first few pile driving events. Such a regulation is already enforced in two other North Sea countries: Germany (Anonymous 2017) and the Netherlands (MEZ 2015). If these noise mitigation measures result in a meaningful reduction of porpoise displacement range and duration, then it could be considered eliminating the need for a seasonal pile driving ban.

4.2. Effects of mitigation measures on offshore wind farm construction

Noise mitigation measures during pile driving are intended to benefit the marine environment but should not threaten the renewable energy goals by impacting the project's viability. The direct costs of applying noise mitigation measures during pile driving currently are less than € 5m (~0.5% of the construction cost) for an 80 turbine OWF-project (Verfuss *et al.* 2019). This is much lower than the costs in 2011 to 2014, which ranged from € 15m to € 36m for an equally sized OWF (Philipp 2018). Indirect costs associated with the use of noise mitigation measures are assumed to result from prolonged installation schedules and an overall increase in complexity and risk (Verfuss *et al.* 2019). Experiences in the BPNS suggest that the use of noise mitigation did not increase installation time for offshore wind projects. This is likely since SBBCs and DBBCs were used, which can be deployed independently from the installation vessel (Koschinski & Lüdemann 2013). This is in line with results from Germany, where despite a strict noise threshold, wind farm construction has proceeded at pace, even with declining government subsidies (Andresen 2017). This indicates that although the economic cost of compliance with the underwater noise regulations may affect the profitability of offshore wind farms, they have not affected their economic viability (Merchant 2019).

4.3. Population consequences of mitigation for marine mammals

We used the interim Population Consequences of Disturbance model (iPCOD) to simulate how different approaches to noise mitigation during pile driving for offshore construction can impact a harbour porpoise population over a period of 25 years and found that currently available mitigation technologies can avoid 97% of the porpoise population decline anticipated under a 'no mitigation' scenario. Applied mitigation measures are assumed to work in two ways: by reducing the animals experiencing permanent threshold shift through the use of acoustic deterrent devices, and by reducing the number of animals disturbed by impulsive sound by limiting the transmission of sound waves. For similar projects, differences in the configuration and nature of noise mitigation measures resulted in major changes in the size of the area impacted by high levels of impulsive sound (see Norro, chapter 2 of this volume). An optimal configuration of mitigation measures will thus reduce the number of porpoises that are being disturbed by an order of magnitude. However, the assumptions made about the effect of noise mitigation on the (spatial and temporal) magnitude of porpoise disturbance are possibly overly optimistic. In their study of German OWF construction, Rose *et al.* (2019) could not demonstrate a further reduction in displacement of porpoises during construction, despite a considerable improvement in noise-mitigation systems used.

5. Conclusion

Between 2009 and 2020, OWF developers in the BPNS largely complied with the national environmental license conditions formulated to mitigate the potential negative impacts of pile driving on marine mammals. We have identified possible improvements to these environmental license conditions, including optimisation in the use of acoustic deterrent devices, formalising obligatory marine mammal surveys, and requiring developers to comply with the national threshold for impulsive underwater sound. A reduction in the costs of applying noise mitigation measures ensures that the suggested improvements should not affect the economic viability of future projects.

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CHAPTER 4

COLLISION RISK FOR SIX SEABIRD SPECIES IN THE FIRST BELGIAN OFFSHORE WIND FARM ZONE

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Abstract

Collision of seabirds with turbines is a direct impact of offshore wind farms (OWFs) resulting in additional mortality. The numerous operational and planned offshore wind farms in the North Sea, an area of great importance for millions of seabirds during their different life stages, raise concern about the possible impact they might have on seabird populations.

Now that a first wind farm zone in the Belgian part of the North Sea, comprising nine OWFs, is (nearly) completed, we assessed the number of possible seabird collision victims based on the latest available knowledge on collision risk modelling.

A total of 69.5 ± 53.0 casualties per year for six selected seabird species, which are the most abundant inside the Belgian OWFs, are estimated. This total figure arises to 290.3 ± 205.4 depending on the source of the avoidance rates in the model. Of the six species included in the study, the highest number of collisions are expected for greater and lesser black-backed gull. Despite considerable uncertainty about the absolute number of collisions, the model identifies which species face the highest risk and shows great value in the comparison of different scenarios for wind farm developments and should be used as a tool for strategic marine planning at a national or regional scale. With an increasing number of OWFs built and planned in the North Sea, population level effects caused by additional mortality through collisions cannot be excluded and developments could conflict with seabird conservation goals.

1. Introduction

The collision of seabirds with the rotor blades of turbines is a direct impact of offshore wind farms (*e.g.* Fox et al. 2006; Drewitt & Langston 2006; Furness *et al.* 2013). The resulting additional mortality may have a substantial impact at a population level because seabirds are long-lived species with a delayed maturity and small clutch size (Croxall & Rothery 1991; Sæther & Bakke 2000; Stienen *et al.* 2007).

Internationally highly important numbers of seabirds breed along the North Sea coasts, totalling more than 4 million individuals. These birds make intensive use of the North Sea for feeding during at least part of the year (Tasker *et al.* 1987; Mitchell *et al.* 2000). During autumn and spring, an estimated number of 1.0-1.3 million seabirds annually migrate through the 'migration bottleneck' of the Southern North Sea, including the Belgian part of the North Sea (Seys 2002; Stienen *et al.* 2007). The large number of operational and planned OWFs in this area therefore raised concern about the impact on seabird communities. In the first Belgian zone for renewable energy, oriented perpendicular to the main seabird migration route, nine wind farms are operational (see chapter 1). Prior to developments in a second area, we intend to assess the number of likely seabird collision victims based on the latest available knowledge. Searching for carcasses, as it is done in wind farms on land, is not an option offshore, so the only possible way to assess this impact is by modelling the risk of collision for birds. These collision risk models (CRM) are based on input data related to wind farm configuration and turbine dimensions, as well as species-specific parameters such as bird dimensions, flight activity and local bird density.

2. Material and methods

2.1. Research strategy

Accurate information on turbine dimensions is available for all nine OWFs in Belgian waters. Also, post-construction seabird surveys have been conducted for over five years in two of these wind farms (Vanermen *et al.* 2016, 2019). We used the resulting post-construction seabird density data to estimate the total number of collision victims within all Belgian OWFs for the six most abundant seabird species occurring inside the wind farms. Post-construction data are not yet available for more recently built wind farms, but the above-mentioned density data were used as a proxy for the other wind farms.

2.2. Collision risk modelling

Estimating bird collisions at sea can be done using a collision risk model (CRM) that calculates the risk per species based on technical wind farm and turbine specifications, bird-related parameters and bird densities. The CRM most frequently used is the one developed by Band (2012). Masden (2015) developed a CRM, based on the Band model, that includes uncertainty and variability of the input variables. The Masden (2015) model was further improved by McGregor *et al.* (2018) to develop a stochastic version of the Band (2012) collision risk model, providing a more robust and transparent method of accounting for uncertainty in the estimation of seabird collision rates.

The Band model (Band 2012) has undergone several iterations over the years and now provides four different options for calculating collision risk. Option 3 of the extended model uses species-specific flight height distributions from Johnston *et al.* (2014), in contrast to the basic model that assumes a uniform distribution of the flight height between the lowest and the highest level of the rotor swept area. As option 3 is considered the most realistic calculation (McGregor *et al.* 2018), this is what we used.

The stochastic CRM (sCRM) is available in two forms: a Shiny app based on the R-code, available as an online tool (https://dmpstats.shinyapps.io/avian_stochcrm/) and as a package that can be downloaded and run locally (https://github.com/dmpstats/stochCRM). We used the online application. The input variables needed for the sCRM are further described in the paragraphs below.

2.3. Species selection

The focus of this study was on the six most abundant seabird species inside the Belgian offshore wind farms: black-legged kittiwake *Rissa tridactyla*, lesser black-backed gull *Larus fuscus*, great black-backed gull *Larus marinus*, herring gull *Larus argentatus*, common gull *Larus canus* and northern gannet *Morus bassanus*. Other species were not selected because of insignificant post-construction densities inside the wind farms or because they are at low risk

Table 1. Bird related input data for the stochastic collision risk model

Species	Northern gannet	Common gull	Lesser black-backed gull	Herring gull	Great black-backed gull	Black-legged kittiwake
Avoidance rate (%)[1]	99.9	99.8	99.8	99.9	99.6	99.8
SD Avoidance rate (%)[1]	0.03	0.07	0.06	0.05	0.11	0.06
Body_Length (m)[2]	0.94	0.41	0.58	0.6	0.71	0.39
SD Body_Length (m)[2]	/	/	0.03	/	/	0.005
Wingspan (m)[2]	1.725	1.11	1.43	1.44	1.58	1.08
SD Wingspan (m)[2]	/	/	0.0375	/	/	0.0625
Flight_Speed (m/s)[1]	13.33	9.8	10.13	9.68	9.78	8.71
SD Flight_Speed (m/s)[1]	4.24	3.63	3.93	3.47	3.65	3.16
Nocturnal_Activity (% of diurnal activity)	0.25[3]	0.5[3]	0.43[4]	0.01[4]	0.5[3*]	0.5[3]
Flight	Flapping	Flapping	Flapping	Flapping	Flapping	Flapping
Proportion Flight	1	1	1	1	1	1

[1] Skov *et al.* (2018), [2] Snow & Perrins (1998), [3] Garthe & Hüppop (2004; *common gull not mentioned, therefore we took the same value as for other gull species mentioned in this study), [4] Gyimesi *et al.* (2017).

of collision due to their low-flying height (*e.g.* razorbill *Alca torda*, common guillemot *Uria aalge*). Great cormorant *Phalacrocorax carbo* was not considered either, despite the fact that this species is frequently observed perching on the jacket turbine foundations in the C-Power wind farm on the Thornton Bank (Vanermen *et al.* 2019). This species, however, was rarely observed flying inside the wind farm, resulting in negligible densities of flying birds.

2.4. Bird related input data

Avoidance rates are taken from Skov *et al.* (2018), who determined these in an empirical study. Body length and wingspan are taken from Snow and Perrins (1998). Flight type for seabirds is regarded as flapping, not gliding. Proportion in flight is set at 1, as the density data are based on flying birds only.

2.5. Bird density data

Monthly post-construction bird surveys started in 2010 in the Belwind OWF on the Bligh Bank and in 2013 in the C-Power OWF on the Thornton Bank and were continued for five years. Details on these surveys can be consulted in Vanermen *et al.* (2016, 2019). During these surveys flying birds and birds on the water were counted separately. We selected only the flying birds to calculate seasonal densities as input for the sCRM.

Post-construction data are not yet available for the other wind farms, but the post-construction density data of the Bligh Bank and Thornton Bank offshore wind farms were used as a proxy for the other wind farms. The Thorton Bank data were used for the Southern parks (Norther, C-Power and Rentel), the Bligh Bank data for the northern wind farms (Northwind, Seastar, Nobelwind, Belwind, Northwester 2 and Mermaid; fig. 1).

2.6. Turbine related input data

The variables of the wind farms and wind turbines are given in table 2. Wind farm and turbine specific input data were collected with the help of the wind farm operators. Rotor speed and pitch were taken from Gyimesi *et al.* (2018). Informations on turbine activity per month were taken from Masden *et al.* (2015).

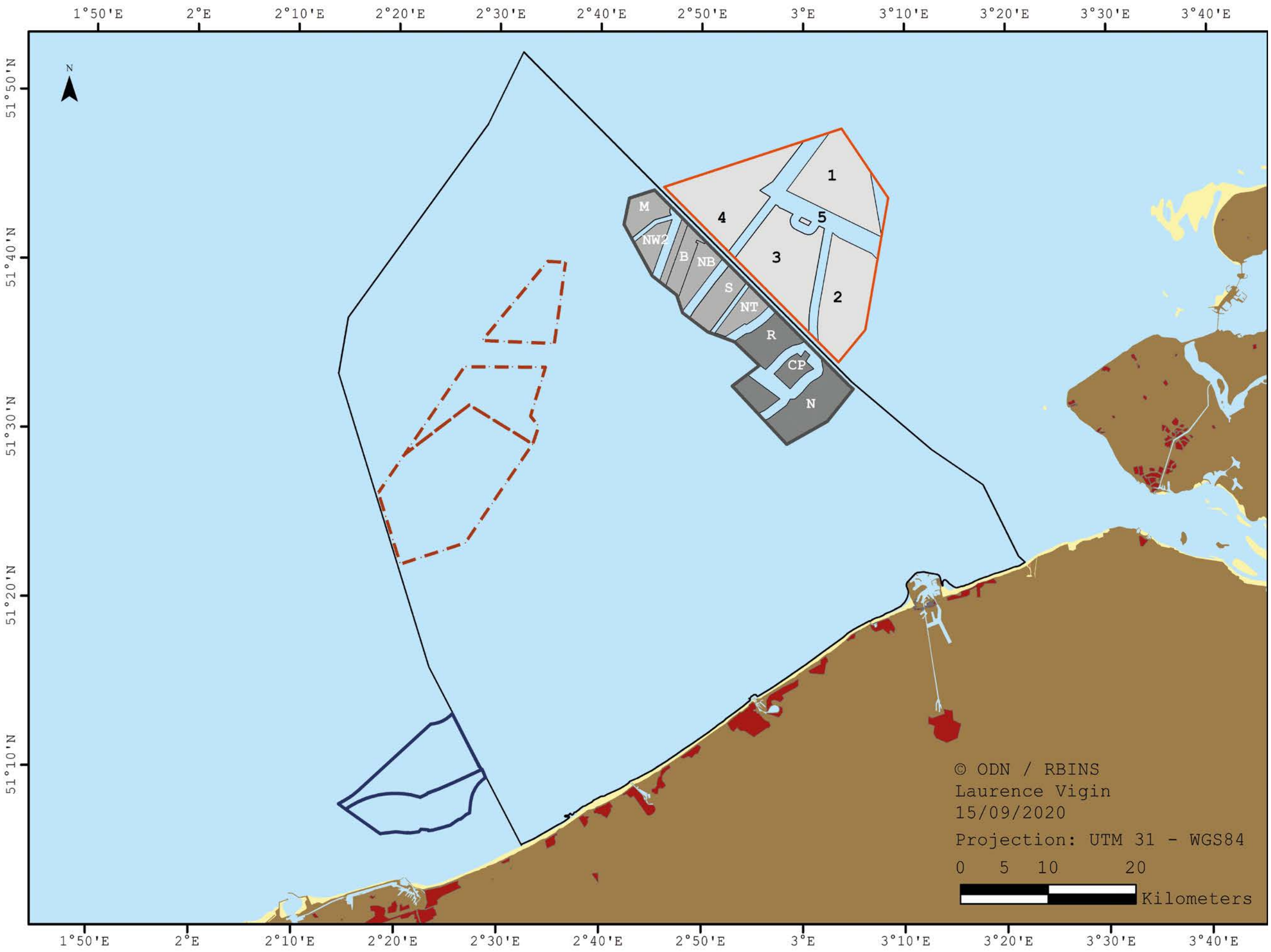

Figure 1. Map of Belgian part of the North Sea with indication of the nine offshore wind farms (OWFs) that are operational or being finalized. For the darker grey OWFs, the Thornton Bank bird density data were used, for the lighter grey, the Bligh Bank density data were used. The second zone for wind energy is indicated by the dashed polygon, the Borssele wind farm zone (in the adjacent Dutch waters by the red polygon and the French wind farm zone near Dunkerque by the blue polygon.

Table 2. Wind farm and turbine related input data for the stochastic collision risk model

	N of turbines	Width (km)	Latitude (°)	Tidal offset (m)	Turbine model (MW)	N of blades	Rotor radius (m)	Air gap (m)	Max blade width (m)	Rotor speed (rpm)	Pitch (°)
Norther	44	4.3	51.52	4.3	8.4	3	82	25	5.4	10.95	5.2
C-Power	54	4.4	51.55	4.3	6.15	3	63	32	5	12.22	5.6
Rentel	42	4.7	51.59	4.3	7.35	3	77	28.5	5	11.62	5.4
Northwind	72	3.1	51.62	4.3	3	3	56	27	4	14.85	6
Seastar	30	2.8	51.64	4.3	8.4	3	83.5	25.5	5.4	10.95	5.2
(No)Belwind*	106	5.1	51.67	4.3	3.3	3	56	27	4	14.85	6
Northwester 2	23	4.2	51.69	4.3	9.5	3	82	24.5	5.4	10.52	5.1
Mermaid	28	3.6	51.71	4.3	8.4	3	83.5	25.5	5.4	10.95	5.2

* the Nobelwind OWF is built around the Belwind OWF and therefore Belwind and Nobelwind are considered as one project. Belwind and Nobelwind have different turbines (Vestas V90 and Vestas V112 respectively). We used the Nobelwind turbine dimensions as a worst-case scenario.

Table 3. Post-construction density data (mean (n/km²) ± SD) of flying individuals of six seabird species inside the wind farms on the Bligh Bank and the Thornton Bank in winter (December, January, February), spring (March, April, May), summer (June, July, August) and autumn (September, October, November)

Thornton Bank (9/2013-12/2018)						
Season	Northern gannet	Common gull	Lesser black-backed gull	Herring gull	Great black-backed gull	Black-legged kittiwake
Winter	0.00 ± 0.00	0.43 ± 0.76	0.01 ± 0.04	0.01 ± 0.04	0.09 ± 0.17	0.49 ± 0.58
Spring	0.00 ± 0.00	0.00 ± 0.00	0.22 ± 0.34	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Summer	0.02 ± 0.06	0.00 ± 0.00	0.20 ± 0.14	0.00 ± 0.00	0.04 ± 0.06	0.00 ± 0.00
Autumn	0.00 ± 0.00	0.04 ± 0.11	0.04 ± 0.11	0.01 ± 0.04	0.14 ± 0.22	0.09 ± 0.25
Bligh Bank (10/2010-4/2015)						
Season	Northern gannet	Common gull	Lesser black-backed gull	Herring gull	Great black-backed gull	Black-legged kittiwake
Winter	0.00 ± 0.00	0.14 ± 0.24	0.04 ± 0.07	0.05 ± 0.17	0.04 ± 0.09	0.64 ± 0.64
Spring	0.03 ± 0.07	0.02 ± 0.04	0.27 ± 0.45	0.03 ± 0.08	0.02 ± 0.08	0.14 ± 0.34
Summer	0.00 ± 0.00	0.00 ± 0.00	0.16 ± 0.19	0.00 ± 0.00	0.02 ± 0.05	0.00 ± 0.00
Autumn	0.03 ± 0.09	0.01 ± 0.03	0.04 ± 0.10	0.00 ± 0.00	0.20 ± 0.27	1.14 ± 0.19

3. Results

3.1. Post-construction bird densities

The resulting density data of flying individuals of the six-target species (table 3) were used to calculate the annual number of collision victims.

3.2. sCRM results

The sCRM was run for 1000 iterations of the input variables, resulting in an overall number of collision victims ± standard deviation. This was done for each wind farm and then the model outputs were summed to get an overall number of collisions per species for the entire Belgian wind farm zone (table 4). As such, a total of 69.5 ± 53.0 casualties per year for the six selected seabird species are expected. The highest numbers are expected for great and lesser black-backed gulls, with respectively 54.3% and 27.1% of the total number of collisions. Only 0.7% of the collisions are expected to be Northern gannets.

4. Discussion

The resulting collision estimates are significantly lower than the outcome of an earlier study on collision risk in the BPNS. Brabant and Vanermen *et al.* (2015) estimated a yearly 102 [22; 704] seabird collisions for a single wind farm (Belwind) for the same six

Table 4. sCRM option 3 output resulting in a total estimated number of collisions per species per year (± SD) for the eight Belgian offshore wind farms in the first zone for renewable energy

	Winter	Spring	Summer	Autumn	N collisions/year (± SD)
Black-legged kittiwake	3.2 ± 2.9	1.1 ± 1.3	0.0 ± 0.0	1.0 ± 0.9	5.3 ± 7.4
Common gull	4.6 ± 5.0	0.4 ± 0.6	0.0 ± 0.0	0.6 ± 0.7	5.5 ± 9.1
Great black-backed gull	8.2 ± 6.7	3.9 ± 4.5	4.4 ± 3.6	21.3 ± 18.3	37.7 ± 45.8
Herring gull	0.9 ± 1.6	0.7 ± 1.2	0.0 ± 0.0	0.1 ± 0.2	1.7 ± 4.2
Lesser black-backed gull	1.2 ± 1.1	10.3 ± 9.1	5.3 ± 4.3	2.1 ± 1.9	18.8 ± 23.6
Northern gannet	0.0 ± 0.0	0.2 ± 0.2	0.1 ± 0.1	0.2 ± 0.3	0.5 ± 0.8
Total	18.1 ± 9.0	16.5 ± 10.3	9.7 ± 5.6	25.2 ± 18.4	69.5 ± 53.0

species. This exceeds by far the results of this updated calculation where we expect a total of 69.5 ± 53.0 collisions per year for nine wind farms. The main reason for this strong decrease is the use of the empirical avoidance rates from Skov *et al.* (2018). These vary between 0.996 and 0.999 for the selected species (see table 1). In the 2015 study we applied an avoidance rate of 0.976 for all species, a figure taken from Krijgsveld *et al.* (2011). This implies that the number of collisions decreases with a factor 6 to a factor 24 only by updating the avoidance rate. The discussion on the avoidance rates is still ongoing within the scientific community. Bowgen and Cook (2018) state that the empirical avoidance rates of Skov *et al.* (2018) cannot be used directly in the sCRM as they do not incorporate model error or how birds respond in relation to other factors, for example weather conditions. Using the avoidance rates recommended by Bowgen and Cook (2018) increases the number of estimated collisions with a factor ranging from 2.5 to 15 for the species included in this study. The overall number of collisions by the nine OWFs would then be 290.3 ± 205.4 instead of 69.5 ± 53.0.

Leemans *et al.* (2019) also used the sCRM to estimate collisions of lesser black-backed gull and black-legged kittiwake for different development scenarios of offshore wind farms in the North Sea. For the first Belgian wind farm zone, *i.e.* the nine wind farms we included, they estimate that 41 lesser black-backed gulls and 3 black-legged kittiwakes would collide per year. Our calculations result in 18.8 ± 23.6 annual collisions for lesser black-back gull and 5.3 ± 7.4 black-legged kittiwakes. The difference for lesser black-backed gull can be explained by the input data for flying altitude. While we used the species-specific flight height distributions as modelled by Johnston *et al.* (2014), Leemans *et al.* (2019) used GPS logger data of lesser black-backed gulls from the Netherlands, Belgium and England (Gyimesi *et al.* 2017). These GPS logger data showed that approximately 34% lesser black-backed gulls fly at the collision risk height between 25 and 150 m (Gyimesi *et al.* 2017), while for the modelled distributions of Johnston *et al.* (2014) this is only 22%. Another explanation for the difference can be found in the seabird density data being used. Leemans *et al.* (2019) made use of data presented by van der Wal *et al.* (2018), which are higher than the post-construction density data used in this study. The other input variables Leemans *et al.* (2019) used were identical to this study.

The results also nicely reflect the dimensions and density of the turbines in different wind farm: turbines with a larger area between the sea surface and the lower tip of the rotor (*i.e.* air gap, table 2) will result in lower number of collision victims (*e.g.* C-Power) and a high turbine density will result in higher number of collisions (*e.g.* (No) Belwind). These conclusions need to be taken into account in the planning and design of future developments in the North Sea (*e.g.* the second wind farm zone in the BPNS) *e.g.* by requiring developers to install fewer, larger turbines.

There is large uncertainty about the absolute number of collisions, and that outcome largely differs depending on the input variables of which the avoidance rates and the flight speed of birds have the largest impact. The approach is, however, very useful for use in a relative manner to compare different scenarios for wind farm development which is also recommended by Cuttat and Skov (2020) and to identify which species face the highest risk of collision. Furthermore, these collision risk assessments become increasingly relevant when they are conducted at a national or regional scale as a means of strategic marine planning, opposed to being applied during the licensing or consenting procedure of a single wind farm. Nevertheless, these results indicate the order of magnitude of the number of collisions. In our study, the highest number of collisions are to be expected for greater and lesser black-backed gull. These species were also identified by Furness *et al.* (2013) as being most vulnerable to collision mortality. Large gull species have the highest risk of collision because they fly at rotor height more frequently compared to the other species in this study (*e.g.* northern gannet) and their relatively high density inside the OWFs. With an increasing number of OWFs built and planned in the North Sea, population level effects caused by additional mortality through collisions cannot be excluded and developments could conflict with seabird conservation goals.

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Annex: collision estimates per wind farm

Norther	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.3 ± 0.6	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.3	0.4 ± 0.7
Common gull	0.7 ± 2.0	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.3	0.8 ± 2.0
Great black-backed gull	1.7 ± 3.6	0.0 ± 0.0	0.7 ± 1.6	2.5 ± 7.0	4.9 ± 8.1
Herring gull	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1	0.1 ± 0.2
Lesser black-backed gull	0.1 ± 0.2	1.1 ± 2.4	0.7 ± 1.3	0.3 ± 0.6	2.1 ± 2.8
Northern gannet	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1

C-Power	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.2 ± 0.5	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.2	0.3 ± 0.5
Common gull	0.9 ± 2.4	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.3	1.0 ± 2.4
Great black-backed gull	1.3 ± 2.8	0.0 ± 0.0	0.6 ± 1.1	1.9 ± 3.9	3.7 ± 5.0
Herring gull	0.0 ± 0.2	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.2	0.1 ± 0.2
Lesser black-backed gull	0.1 ± 0.2	1.1 ± 3.0	0.6 ± 1.5	0.3 ± 0.7	2.0 ± 3.5
Northern gannet	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0

Rentel	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.2 ± 0.5	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.2	0.3 ± 0.5
Common gull	0.8 ± 1.8	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.2	0.9 ± 1.8
Great black-backed gull	1.2 ± 2.4	0.0 ± 0.0	0.5 ± 1.2	1.8 ± 3.9	3.5 ± 4.7
Herring gull	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1	0.1 ± 0.2
Lesser black-backed gull	0.1 ± 0.2	0.9 ± 2.2	0.6 ± 1.1	0.3 ± 0.6	1.8 ± 2.5
Northern gannet	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1

Northwind	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.6 ± 1.3	0.3 ± 0.6	0.0 ± 0.0	0.2 ± 0.4	1.0 ± 1.5
Common gull	0.6 ± 1.7	0.1 ± 0.3	0.0 ± 0.0	0.1 ± 0.2	0.8 ± 1.7
Great black-backed gull	1.1 ± 2.1	1.1 ± 2.1	0.8 ± 1.5	4.3 ± 8.7	7.3 ± 9.3
Herring gull	0.3 ± 1.0	0.2 ± 0.8	0.0 ± 0.0	0.0 ± 0.0	0.5 ± 1.3
Lesser black-backed gull	0.3 ± 0.5	2.0 ± 3.9	1.0 ± 2.2	0.4 ± 0.7	3.7 ± 4.6
Northern gannet	0.0 ± 0.0	0.1 ± 0.2	0.0 ± 0.0	0.1 ± 0.2	0.1 ± 0.2

Seastar	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.3 ± 0.7	0.1 ± 0.3	0.0 ± 0.0	0.1 ± 0.2	0.5 ± 0.8
Common gull	0.2 ± 0.4	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1	0.2 ± 0.4
Great black-backed gull	0.5 ± 1.0	0.4 ± 1.0	0.3 ± 0.6	1.6 ± 3.2	2.7 ± 3.5
Herring gull	0.1 ± 0.5	0.1 ± 0.3	0.0 ± 0.0	0.0 ± 0.0	0.2 ± 0.5
Lesser black-backed gull	0.1 ± 0.3	1.0 ± 2.3	0.5 ± 1.1	0.2 ± 0.4	1.7 ± 2.6
Northern gannet	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1	0.0 ± 0.1

Nobelwind	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	1.1 ± 2.2	0.5 ± 1.0	0.0 ± 0.0	0.3 ± 0.6	1.9 ± 2.5
Common gull	1.0 ± 3.0	0.2 ± 0.5	0.0 ± 0.0	0.1 ± 0.3	1.3 ± 3.0
Great black-backed gull	1.5 ± 3.3	1.5 ± 3.7	0.9 ± 2.0	5.7 ± 12.0	9.6 ± 13.2
Herring gull	0.3 ± 1.1	0.3 ± 0.8	0.0 ± 0.0	0.0 ± 0.0	0.6 ± 1.4
Lesser black-backed gull	0.3 ± 0.7	2.7 ± 6.2	1.2 ± 2.4	0.5 ± 1.2	4.9 ± 6.8
Northern gannet	0.0 ± 0.0	0.1 ± 0.2	0.0 ± 0.0	0.1 ± 0.2	0.2 ± 0.2

Northwester 2	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.2 ± 0.4	0.1 ± 0.2	0.0 ± 0.0	0.1 ± 0.1	0.4 ± 0.5
Common gull	0.2 ± 0.4	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1	0.2 ± 0.4
Great black-backed gull	0.5 ± 0.9	0.4 ± 0.8	0.3 ± 0.6	1.6 ± 2.8	2.8 ± 3.1
Herring gull	0.1 ± 0.2	0.1 ± 0.2	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.3
Lesser black-backed gull	0.1 ± 0.2	0.7 ± 1.5	0.3 ± 0.8	0.1 ± 0.3	1.2 ± 1.7
Northern gannet	0.0 ± 0.0	0.0 ± 0.4	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.1

Mermaid	Winter	Spring	Summer	Autumn	Total/year
Black-legged kittiwake	0.2 ± 0.5	0.1 ± 0.3	0.0 ± 0.0	0.1 ± 0.2	0.4 ± 0.6
Common gull	0.2 ± 0.7	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1	0.3 ± 0.7
Great black-backed gull	0.5 ± 1.0	0.5 ± 0.9	0.3 ± 0.7	2.0 ± 4.1	3.3 ± 4.4
Herring gull	0.1 ± 0.2	0.1 ± 0.2	0.0 ± 0.0	0.0 ± 0.0	0.1 ± 0.2
Lesser black-backed gull	0.1 ± 0.2	0.7 ± 1.6	0.4 ± 0.9	0.1 ± 0.3	1.4 ± 1.9
Northern gannet	0.0 ± 0.0	0.0 ± 0.1	0.0 ± 0.0	0.0 ± 0.1	0.0 ± 0.1

CHAPTER 5

BELGIAN SEABIRD DISPLACEMENT MONITORING PROGRAMME: A FEASIBILITY STUDY ON FUTURE RESEARCH POSSIBILITIES

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Abstract

As a first step towards a meaningful continuation of the Belgian seabird displacement monitoring programme, this chapter aims to identify relevant knowledge gaps and feasible research possibilities. After a decade of baseline displacement monitoring in and around single offshore wind farms, future focus should be oriented towards more targeted research, aiming to address specific issues on the actual impact of offshore wind farms on individual birds or bird populations, next to aspects supporting mitigation. As such, we identified three major future research themes: the correlation between displacement and wind farm characteristics, large gull movements in and around offshore wind farms and an empirically informed species-distribution model to support marine spatial planning.

1. Introduction

From 2008 until 2019, our research programme has focused on seabird displacement in two individual offshore wind farms (OWFs) located at the Bligh Bank and Thornton Bank. This has revealed distinct patterns in the tendency of seabird species to either avoid or to be attracted to these OWFs (*e.g.* Vanermen *et al.* 2019a). For a certain range of species we found striking parallels in displacement results between both Belgian wind farms as well as between Belgian and foreign North Sea studies. Not unexpectedly, for other species there was substantial inconsistency between results. Local seabird distribution, densities, seasonality and wind farm characteristics (overall size of the development, turbine size and density) are all hypothesised to potentially affect seabird displacement rates. Yet because of limited insight in what is driving the variation in observed patterns, impact study results so far have had limited value in predicting expected displacement rates elsewhere. Clearly, increased knowledge on cause-effect relationships would strongly benefit future planning and impact assessments. This chapter will therefore look for additional, yet feasible research possibilities to continue to provide valuable input in the ongoing scientific discussions on seabird displacement caused by offshore wind farms.

2. Feasibility study on cause-effect relationships

Degraer *et al.* (2013) strongly promoted the continuation of a 'basic monitoring programme' studying the impact of OWFs on all ecosystem components. Such 'basic monitoring' should be designed in a manner that it allows to keep track of major and unforeseen impacts, thus functioning as a finger on the pulse of environmental impact development. Additionally, the Belgian wind farm research programme aims at conducting 'targeted monitoring', in search for cause-effect relationships. In the end, the results of such monitoring research may allow to extrapolate observed impacts and to provide valuable input for future wind farm planning and design regarding mitigation strategies.

Displacement research at the Bligh Bank and Thornton Bank was continued until five and six years after construction respectively. In order to assess possible habituation effects, we originally planned to repeat the displacement research from 10 years after construction on. However, this is now considered unfeasible as the overall setting has changed tremendously, with new wind turbines now present in areas that were used for monitoring the reference situation before. It might therefore be difficult to disentangle habituation effects from effects due to an increased number of turbines in the direct vicinity of the wind farm under study. Meanwhile, the near-future situation also offers new research opportunities. The concession zone will soon become one large wind

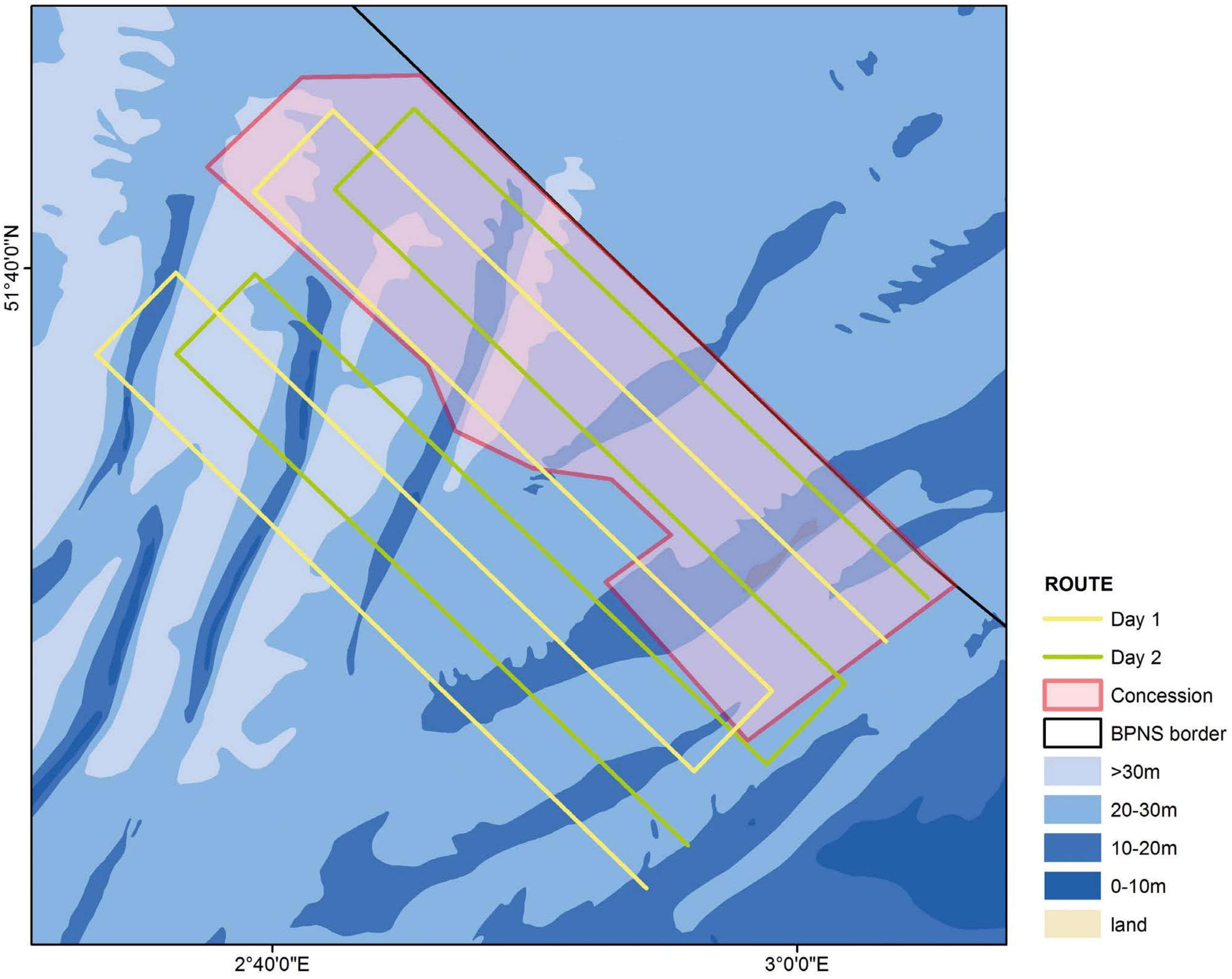

Figure 1. Proposal for a concession-wide seabird displacement monitoring scheme.

farm area of nearly 500 turbines, consisting of nine adjacent wind farms, each with their own specific features (see chapter 1). Next to wide-scale displacement patterns, monitoring the full concession zone and an adjacent control area would allow us to look for a **correlation between wind farm characteristics and locally observed displacement rates**, thus offering a perfect integration between basic and targeted monitoring. In Vanermen *et al.* (2019a), we already proposed a ship-based seabird monitoring scheme covering the area in two days (see fig. 1). Actually performing this scheme will not be planned before all wind farms have become operational, to avoid local access limitation due to construction activities, and to assure a stable situation throughout the programme.

In the near past, the Dutch government (Rijkswaterstaat) already commissioned a study to investigate whether the varying response of common guillemots *Uria aalge* observed at different sites could be related to wind farm configuration, by bringing together data from eight European OWFs. Unfortunately, Zuur (2018) could not find convincing displacement responses in any of the wind farms, let alone a correlation between displacement rate and OWF configuration. But despite applying state of the art Bayesian statistics (INLA), the analysis was performed in a way that makes it very hard to reliably detect OWF-induced guillemot displacement. The authors looked for displacement by modelling the spatial distribution of common guillemots and considering the percentages of 'importantly' negative and positive spatial random field (SRF) values inside distance bands of 5, 10, 15, 20 and 25 km around the wind farm. It was further hypothesised that OWF disturbance would result in a general increase in the percentage of importantly positive SRF values with increasing (incremental) distance from the OWF, as opposed to a decrease in the percentage of importantly negative SRF values. However, in this set-up the wind farm itself covers only a minor part of the inner 5 km circle. Clearly, wind farm disturbance could simply involve the redistribution of birds within this first 5 km distance band (*i.e.* from inside the wind farm boundaries to its near vicinities), in which case this particular displacement effect could never be detected applying this strategy. And even if birds would be displaced outside the first 5 km distance band and numbers would spread over a wide area up to 25 km, it still seems unlikely that this would generate a substantial increase in the percentage of positive SRF values. Therefore, we aim to perform more targeted analyses to look for response differences between sites, either at the scale of the wider North Sea region (as performed by Zuur 2018) or more locally within the Belgian wind farm concession zone.

Another (hypothesised) cause-effect relationship is the **attraction of large gulls to OWFs** as a result of increased food availability. Vanermen *et al.* (2013) therefore recommended conducting research on behavioural and foraging-related actions of large gulls inside OWFs. Tracking studies may generate valuable and detailed information on the movements and behaviour of individual birds inside OWFs. At the same time, tracking data may help to fill in notable knowledge gaps on meso- and micro-scale avoidance, nocturnal activity and whether or not a bird's response varies according to meteorological circumstances, all of which would provide valuable input for collision risk assessments. In the framework of the Belgian OWF research programme, efforts have been made in describing turbine-associated foraging behaviour of large gulls and analysing GPS data of lesser black-backed gulls *Larus fuscus* to assess their movements inside OWFs (Vanermen *et al.* 2018, 2019b). Also note that the tagging of lesser black-backed gulls in Belgian colonies only goes back to 2013, the year in which the Thornton Bank OWF became fully operational, and that the current data therefore do not allow a before-after comparison of bird movements. Furthermore, during the breeding season, the Thornton Bank is just outside the gulls' main distribution range, resulting in a relatively low number of records inside and near the wind farm. Interestingly, the current installation of the Norther wind farm just southeast of the Thornton Bank (and closer to the shore) does offer the opportunity to compare the distribution of tracked lesser black-backed gulls in and around an OWF site before and after construction, provided a comparable tagging effort of lesser black-backed gulls in the colonies of Zeebrugge and Ostend is ensured (Vanermen *et al.* 2019b). Up until now, most gull tracking studies in relation to OWFs focused on aspects regarding collision risk, for example on flight height distribution (Corman & Garthe 2014; Ross-smith *et al.* 2016; Borkenhagen *et al.* 2017) and on potential overlap with OWFs in terms of foraging range (Wade *et al.* 2014; Thaxter *et al.* 2015) or year-round movements (Thaxter *et al.* 2019). Surprisingly, few studies have thus reported on within-OWF movements and behaviour (but see Thaxter *et al.* 2018; Vanermen *et al.* 2019b), aiming to unravel why large gulls visit wind farms and to reveal whether gull behaviour inside wind farms may lead to additional or decreased collision risk. Increased knowledge on the matter may also be accomplished by analysing accelerometer data incorporated in the GPS tags, through which Bouten *et al.* (2013) could easily distinguish between standing, soaring, floating and flapping behaviour in lesser black-backed gulls.

A third major knowledge gap which we would like to highlight here is the impact of displacement on the survival and reproduction rate of individual birds. Investigating this particular impact implies assessing the effect of habitat loss on a bird's energy balance and studying the correlation between bird condition and demographic parameters. Being central-place foragers, GPS tagging of breeding birds offers opportunities to link foraging habitat and at-sea behaviour with their reproductive success. This kind of research, however, grows increasingly difficult when targeting wintering seabirds that often do not come to land for several months in a row, such as wintering divers, gannets and auks in Belgian waters. Despite these difficulties, this is the exact study aim in the 'red-throated diver energetics' JNCC project (https://jncc.gov.uk/our-work/rtde-project/). In this project, red-throated divers Gavia stellata are tagged with geo-locators and time-depth recorders (TDRs) to reveal where and for how long divers forage during the non-breeding season. During the 2018 breeding season, 74 adults breeding in Scotland, Finland and Iceland have been tagged, with tags retrieved during 2019 and further retrievals planned for 2020. Data analysis will provide an indication of where each individual wintered next to detailed information on dive depth, duration and frequency. If divers would appear to forage for only a small part of each day, it could be inferred they are easily capable of meeting their energetic requirements in the non-breeding season and so may have the capacity to accommodate the additional energetic costs of displacement (O'Brien *et al.* 2018, 2020). In Belgian waters, the main species displaced by OWFs do not breed anywhere close, and birds would need to be captured at sea. Divers have been caught at sea and subsequently tagged by German researchers (see for example, www.divertracking.com), and this has been demonstrated to work for auks and gannets too (Bugoni *et al.* 2008; Ronconi et al. 2009; Chimienti *et al.* 2017). The main problem, however, lies in the need to recapture birds in case one wants detailed information of diving behaviour through the use of a TDR. Note that a raw classification in bird behaviour with no necessity of recapturing is possible by applying a tri-axial accelerometer (often incorporated in GPS trackers), designed to monitor body movement (*e.g.* Bouten e*t al.* 2013; Patterson *et al.* 2019). Accelerometer data can further be used to calculate the 'overall dynamic body acceleration', which in turn is a proxy for energy expenditure (Sotillo *et al.* 2019).

Lately, individual-based models (IBMs) are considered to be a most promising tool to determine the (cumulative) effect of displacement on demographic parameters (*e.g.* Topping & Petersen 2011; Searle *et al.* 2014; Warwick-Evans *et al.* 2018), while not necessarily relying on GPS tracking data. Topping & Petersen (2011) defined an IBM as 'a computational model for simulating the actions and interactions of autonomous individuals in a defined virtual world, to assess their effects on the system as a whole'. In a recent study by Kooten *et al.* (2019), a method was developed to estimate the effect of habitat loss on five seabird species (red-throated diver, northern gannet *Morus bassanus*, sandwich tern *Thalasseus sandvicensis*, razorbill *Alca torda* and common guillemot), over their full life cycle and across the larger North Sea area. First, the authors constructed habitat maps linking distributional seabird at sea data with abiotic variables. Next, they determined the cost of habitat loss using an individual-based energy-budget model, by combining this with the habitat model predictions and the expected degree of displacement. Eventually, the 'cost' of habitat loss is expressed in terms of reduced survival rates following a change in the availability of foraging area in several OWF scenarios. The authors highlight that there are large sources of uncertainty that may influence the outcome, for example the unbalanced coverage of seabird at sea data across the North Sea and a lack of insight in density-dependent mechanisms.

3. Seabird sensitivity map of the Belgian part of the North Sea

Regarding the potential cumulative effect of the current and new wind farm concession zone (delineated at the Hinder Banks), and the potential need for mitigating measures by means of a marine protected area, we also aim to perform an analysis on the number of seabirds expected to be impacted by displacement at the Belgian part of the North Sea. This would further allow to identify possible bottlenecks for the objectives set for the Marine Strategy Framework and Bird Directives. We will therefore develop species distribution models, linking observational seabird at sea data with a range of explanatory environmental variables (Waggit *et al.* 2019; Kooten *et al.* 2019). Overlaying this with current prospects of wind energy developments, next to empirically assessed displacement rates would result in species-specific estimations of the number of birds affected. Species distribution models at the scale of the Belgian part of the North Sea could meanwhile serve as an instrument to delineate areas which are particularly valuable to seabirds. In the first place we should focus on those species known to be vulnerable to displacement in Belgian waters: northern gannet, common guillemot and razorbill. Next we could extend to those species for which results are yet unclear due to low detection rates or statistically insignificant avoidance effects, *i.e.* northern fulmar *Fulmarus glacialis*, great skua *Stercorarius skua*, little gull *Hydrocoloeus minutus* and black-legged kittiwake *Rissa tridactyla*.

4. Conclusion

Seabird displacement research in the coming years should evolve around the following three major themes. First of all, the new basic monitoring programme, as proposed in Vanermen *et al.* (2019a), will continue assessing species-specific displacement rates, meanwhile looking for correlations with wind farm configuration characteristics. Secondly, a continuation of the tracking network of large gulls in Belgian colonies with suitable GPS trackers will be assured. Focus will be on lesser black-backed gull, a species which ranges far more offshore compared to herring gull *Larus argentatus*, and that has been demonstrated to interact with OWFs much more frequently, at least in Belgian waters. Finally, we will perform a detailed study on the potential effect of OWF-related habitat loss on the scale of the Belgian part of the North Sea, based on a species distribution model (making use of environmental explanatory variables) and empirically observed seabird displacement rates. Such should allow to do profound recommendations for mitigating and compensating measures in future marine spatial planning. The focus here should be on species known to be sensitive to displacement such as divers, gannets and auks. It should further be noted that these species do not occur as breeding birds anywhere near Belgian waters. Extending the monitoring programme with research on the impact of displacement on survival and reproduction rates is therefore much less feasible, considering the logistic and budget-related bottlenecks.

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CHAPTER 6

TURBINE-RELATED IMPACTS ON MACROBENTHIC COMMUNITIES: AN ANALYSIS OF SPATIAL AND TEMPORAL VARIABILITY

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Abstract

We investigated spatiotemporal variability in sediment properties and in macrobenthos communities ‘very close’ (*i.e.* at a distance of 37.5 m) to, and ‘far’ (*i.e.* at a distance of 350-500 m) from jacket foundations in the C-Power offshore wind farm on the Thornton Bank over a time span of three consecutive years. We anticipated that the locally modified water currents around wind turbines, as well as the depositional flow of faecal pellets and other detrital material produced by filter-feeding epifauna living on the foundations, could both contribute to a process of sediment fining and organic matter enrichment close to, and in the wake of, wind turbines. Such effects would in turn be expected to cause shifts in macrobenthos community composition, diversity and abundance. Our three-year analysis provided equivocal support for this hypothesis. Sediments ‘very close’ to jacket foundations had a significantly higher proportion of fine sand compared to samples collected at further distance, but this was not accompanied by a higher organic matter content in 2018 and 2019, rather the contrary. Average macrobenthos abundance and diversity were always higher ‘very close’ to the turbines, but these differences were not consistently statistically significant. Macrobenthos species composition, however, did consistently differ between ‘very close’ and ‘far’ stations. It also changed significantly with time in the ‘very close’ stations, a difference which was largely attributable to a pronounced decline in the abundances of three otherwise dominant species of permeable sediments on the BPNS, and to an increase of several other species, in particular of the bio-engineering, small reef-building polychaete *Lanice conchilega*. This evolution should be carefully monitored in following years, the more so since feedback loops can be expected from the activity of *L. conchilega* on sediment fining and enrichment, because *L. conchilega* enhances the deposition and retention of fine particulate material from the benthic boundary layer. Overall, there appears to be a clear trend for sediments to become finer and organically enriched ‘very close’ to jacket foundations, with concomitant effects on the abundance, diversity and species composition of macrofauna. However, there is a large variability between turbines, and an

increased small-scale heterogeneity around individual turbines as a result of the emergence of small biogenic reefs further adds to an overall prominent variability, which renders statistically robust conclusions on the exact evolutions of sediments and macrobenthos under the influence of turbines difficult to draw. Future monitoring should therefore try to better incorporate small-scale variability in its sampling design, whereas targeted monitoring efforts should be directed at a better elucidation of the spatial scale over which fining and enrichment effects are being manifested.

1. Introduction

The 2020 Belgian targets for renewable energy depend in part on an expansion of offshore wind farms (OWFs) in the Belgian part of the North Sea (BPNS; Rumes *et al.* 2017). Within the eastern part of the BPNS, three parks are already fully operational (C-Power, Northwind and Belwind), a fourth one (Norther) having recently been constructed and having entered the operational phase as of 27 may 2020 (hence after the 2019 autumn monitoring campaign). On top of the 2020 targets, the Belgian government has decided to double the capacity of electricity outputs from wind energy, with a major contribution coming from offshore installations. In order to reach this target, a new concession area in the western part of the BPNS, near the border with France, has been designated for possible exploitation after 2020 (Rumes & Brabant 2018). Because wind farms introduce hard substrate in otherwise soft sediment environments, and because every stage (pre-construction, construction, operational and decommissioning phase) of the development of an OWF can potentially impact these soft sediments and their communities of living organisms (*i.e.*, benthos), consistent monitoring of impacts, as well as of (sometimes subtle) changes in environmental factors which could lead to impacts, remains of utmost importance (Gill *et al.* 2018).

The current OWFs are situated in generally medium to coarse sandy sediments with a low organic matter content (Van Hoey *et al.* 2004; Byers *et al.* 2013). Such sediments tend to be characterised by relatively poor macrobenthic communities in terms of both density and species diversity (Van Hoey *et al.* 2004; Reubens *et al.* 2009; Coates *et al.* 2014). However, the operational phase of an OWF may result in a modification of the habitat as a result of at least two possible mechanisms. First, wind turbines may affect local hydrological conditions (in terms of both currents and water column stratification; hydrology, sediment type, water column stratification) and infaunal community structures. At the same time, wind turbines rapidly become colonized by a dense community of epifouling fauna, which to a large extent obtains its food from the water column while depositing both suspended sediment and organic matter particles to the seafloor surrounding the turbines. Both processes can affect sediment granulometry and organic matter content, both of which are extremely important environmental factors that structure the abundance, diversity and species composition of benthic communities (De Backer *et al.* 2014; Maar *et al.* 2009; Dannheim *et al.* 2019; Gill *et al.* 2018; Coates *et al.* 2014).

The prediction that the sediments surrounding wind turbines may progressively become enriched in fine sediment particles as well as organic matter were supported by the results of a targeted study that focused on a single gravity-based foundation in the C-Power OWF (Coates *et al.* 2014). The researchers hypothesised that such sediment fining and organic matter enrichment would likely result in changes in macrobenthic assemblages surrounding wind turbines, which would be expected to become more abundant and diverse, as is usually the case in similar finer sediments offshore (Wilding *et al.* 2012). However, tests of this hypothesis in the framework of the yearly large-scale monitoring from 2015 onwards failed to discover any significant local enrichment effects when looking at multiple turbines (Colson *et al.* 2017). Because sampling in those campaigns focused at two specific distances from turbines, *i.e.* 50 m ('close') and 350-500 m ('far'), we revised the sampling strategy based on the consideration that turbine effects might be spatially more limited and therefore not generally observable at a distance of 50 m. Hence, from 2017 onwards, we shifted the 'close' samples as closely as possible (in terms of sampling) to the turbines, *i.e.* at 37.5 m from the center of a turbine (Lefaible *et al.* 2018). We also considered the possibility of differences in turbine effects depending on turbine type. Indeed, the study by Coates *et al.* (2014) was done at a gravity-based turbine, but the majority of turbines at the C-Power OWF have jacket foundations, whereas those at Belwind, and indeed most turbines that have recently been deployed or are being planned, are monopiles.

Results of the 2017 monitoring campaign revealed significant sediment fining, organic enrichment and changes in macrobenthic communities (higher densities, diversity and different compositions) at 37.5 m from jacket-based foundations in C-Power (Lefaible *et al.* 2018). Impacts around the monopiles in Belwind were less pronounced, with differences in sediment composition, macrobenthos abundance and diversity not being significant between the 'very close' and 'far' stations, whereas species composition did exhibit significant differences between both distances from the turbines (Lefaible *et al.* 2018). Still, a recurrent observation is that inter-turbine variability within an OWF is high, both in terms of epifouling and macrobenthic communities (Jak & Glorius 2017). Hence, generalisations about turbine-related impacts should not be based on the results of a single monitoring campaign, but should ideally be firmly rooted in patterns that are robust over space and time.

When the same sampling strategy was used in 2018, the higher fine sediment fraction 'very close' to jacket foundations was again observed, yet the organic enrichment was not (Lefaible *et al.* 2019). Such absence of an organic enrichment could obviously imply that there is no clear turbine effect. Alternatively, since the presence or absence of an effect is assessed from a comparison between 'very close' and 'far' stations, and since turbine effects may, with time, expand over a larger area, we cannot automatically rule out that an increase in sediment organic content occurred at locations well beyond the 'very close' stations, which could also erase significant differences between 'very close' and 'far' samples (Lefaible *et al.* 2019). This aspect is currently under investigation in the framework of a targeted monitoring that has been performed in August 2020 (Lefaible *et al.* in prep.), but would also be expected to show up in a multi-year comparison of sediment characteristics of 'far' stations. At the same time, to ascertain that any organic matter enrichment effects are local effects that are probably caused by the presence of turbines, it remains important to also monitor the evolution of the nearby reference area, which is characterised by a very similar sediment type and macrobenthos community, yet is not influenced by wind turbines. Due to the work restrictions imposed by UGent in response to the SARS-CoV-2 epidemic, these samples of the 2019 campaign have not yet been processed and analysed, but they will be done as soon as possible, enabling a comparative analysis of spatiotemporal trends inside and outside an offshore wind farm.

In this report, we focus on macrobenthic communities and their sedimentary habitat around selected jacket foundations in the C-Power OWF. More specifically, we investigate patterns of sediment fining, organic matter content and macrobenthic community abundance, species richness

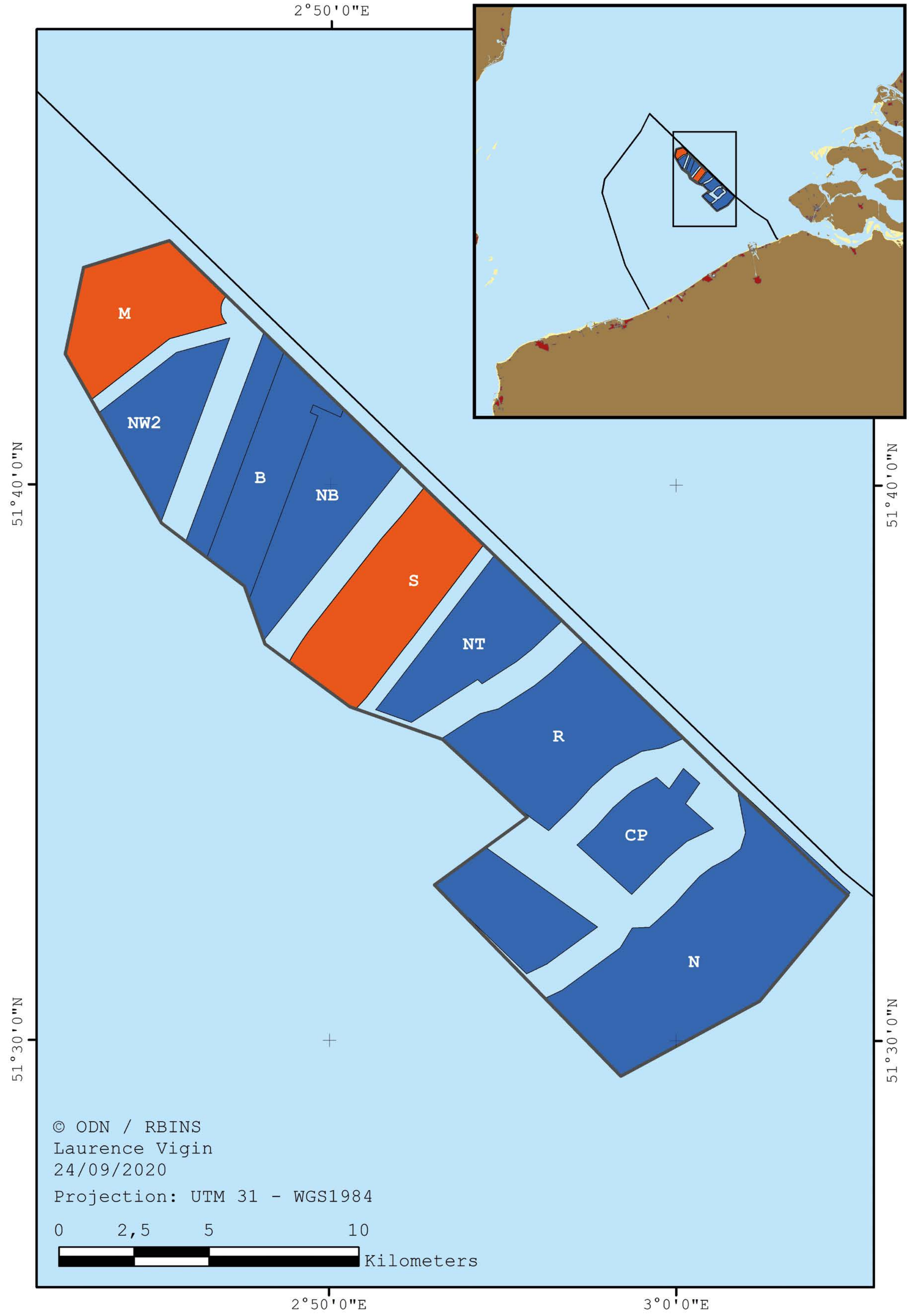

Figure 1. Wind farm concession area in the Belgian part of the North Sea. Blue areas represent the currently operational offshore wind farms (Norther, C-Power, Rentel, Northwind, Nobelwind, Belwind and Northwester 2), while orange areas are wind farms that are under construction (Seastar and Mermaid).

and taxonomic composition around selected turbines based over a 3-year series of consecutive monitoring campaigns. We also zoom in on the temporal patterns in abundance of fine sand-associated macrobenthic species that proliferate close to the turbines and compare them to the temporal patterns in species that naturally occur in the permeable sediments of the Thornton Bank.

2. Materials and methods

2.1. Study area

Sampling in 2019 was conducted in the concession area of the C-Power and Belwind OWFs (fig. 1), but in this report, we will focus on results of the C-Power monitoring only. This OWF is located on the Thornton Bank (TB), approximately 30 km offshore from the Belgian coast. C-Power became fully operational in 2013, and this OWF is composed of 6 gravity-based foundations and 48 jacket foundations (Rumes *et al.* 2017).

Because of unfavourable weather conditions, which precluded the use of the lighter vessel Aquatrot which is needed to approach the turbines until a distance of 37.5 m, the autumn 2019 campaign had to be split up over several subcampaigns. This is not unusual and has happened before, but the rough weather conditions caused a larger-than-usual time gap of nearly two months between the collection of the 'far' and 'very close' samples on the Thornton Bank.

On top of the later-than-planned collection of the 'very close' samples came the SARS-CoV-2 epidemic which imposed strict measures, including the complete closure of our laboratory facilities at UGent for a period of more than two months. As a consequence, not all samples of the 2019 monitoring campaign have been processed and analysed (see table 1 for overview). However, a representative set of 'very close' and 'far' samples from the C-Power OWF (jacket foundations) had been analysed in time, and since they matched sampling stations from the previous years, we were able to use a set of 'very close' and 'far' samples from 12 turbines (see table 1, Thornton Bank samples, and fig. 2: 12 'very close' stations and their corresponding 'far' stations were fully processed and included in this analysis) in a spatiotemporal data analysis on both

Table 1. Overview of the sampling moments, stations, vessels, sample numbers and already processed samples (in light blue) collected during the autumn 2019 monitoring. All collected samples are available at the Marine Biology Research Group, UGent

	Date of sampling	Vessel	Station	Number of samples taken	Number of processed samples
Bligh Bank	October 2019	RV Belgica	BB_FAR	36	7
	January 2020	Aquatrot	BB_VC	12	6
			BB_FAR_EXTRA	6	2
Thornton Bank	October 2019	RV Belgica	TB_FAR	32	16
	December 2019	Aquatrot	TB_VC	16	12
Bligh Bank Reference	October 2019	RV Belgica	BB_REF	16	0
Thornton Bank Reference (Goote Bank)	October 2019	RV Belgica	BGR	25	0

BB = Bligh Bank (Belwind OWF), TB = Thornton Bank (C-Power OWF). VC = 'very close' samples, taken at a distance of 37.5 m from the centre of a turbine; FAR = samples taken further away from the turbines, at distances between 350 and 500 m.

sediment characteristics and macrobenthos communities.

2.2. Sample design, collection and treatment

The potential effects of turbine presence (C-Power) on macrobenthos communities were tested by conducting spatial and temporal comparisons. For this purpose, samples were collected at two distances from the turbines during autumn 2017 through 2019 on board the vessels Aquatrot (for 'very close' samples) and RV Simon Stevin and RV Belgica. 'Very close' samples (TB: 16) were taken at approximately 37.5 m from the centre of the turbine, which is the closest distance from the turbine which can be relatively easily sampled without the help of divers, whereas 'far' samples (TB: 32) were collected in the middle between four surrounding wind turbines (*i.e.*, at the farthest possible distance from the nearest turbines), *i.e.* at distances in between 350 and 500 m from any wind turbine (fig. 2).

The samples were collected from the vessels by means of a 0.1 m^2 Van Veen grab. A plexiglass core (Ø 3.6 cm) was taken from each Van Veen grab sample to collect the environmental data, which included sediment grain size distribution and total sediment organic matter content (TOM). After drying at 60°C, the grain size distribution was measured using laser diffraction on a malvern mastersizer 2000G, hydro version 5.40. Grain size distributions were used to determine the fine sand fraction (125-250 µm) in each sample. Total organic matter (TOM) content was calculated per sample from the difference between dry weight (determined after drying for 48 h at 60°C) and ash-free dry weight (2 h at 500°C).

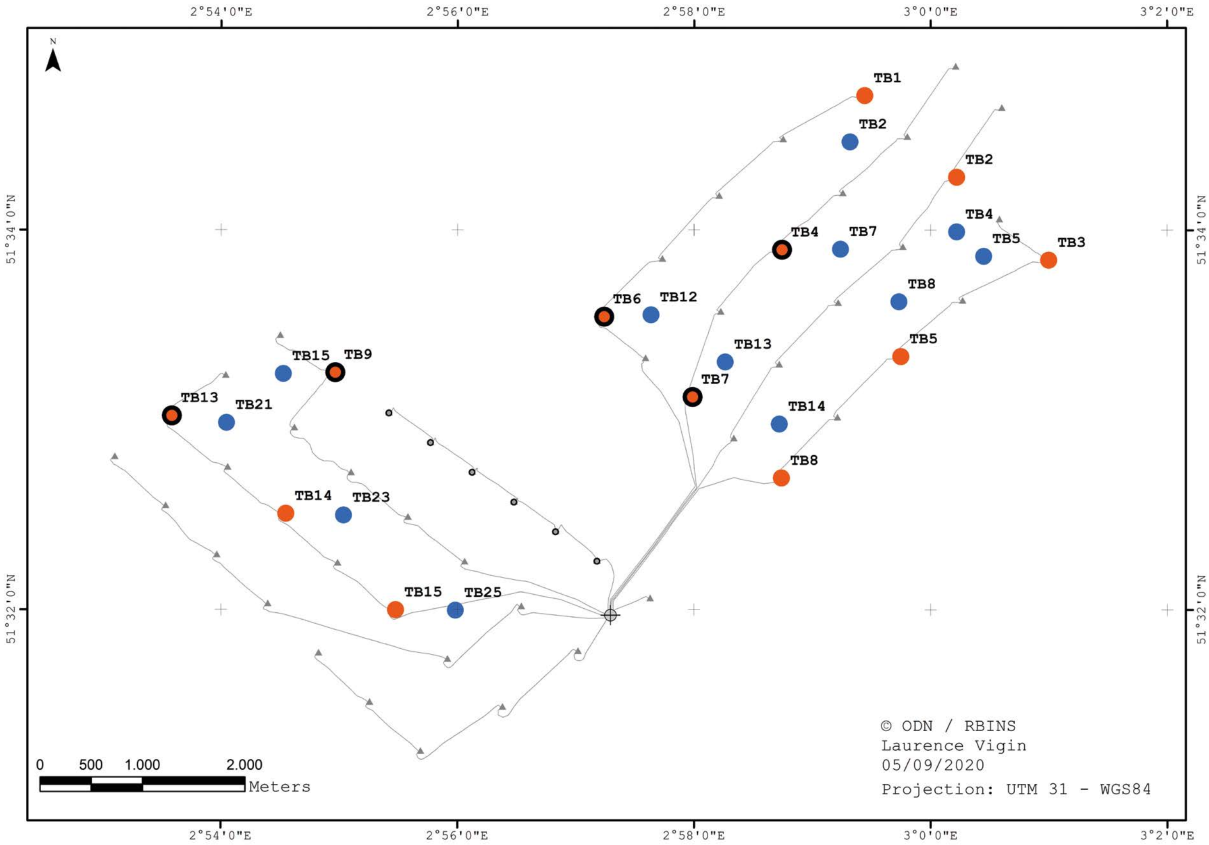

Figure 2. Overview of the selected 'far' (350-500 m; blue dots) and 'very close' (37.5 m; orange dots) locations at the Thornton Bank used in the present study of spatial and temporal variability. Very close locations highlighted in figs 3 and 4 are presented in this figure as grey dots with orange fill.

The rest of the sample was sieved on board the ship over a 1 mm mesh-sized sieve, and the macrofauna was sorted and 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 as blotted wet weight (mg). In this report, these taxa are further referred to as species.

2.3. Data analysis

Prior to statistical analysis, the total abundance (ind. m^{-2}) and number of species (species richness, S) were calculated from the dataset. Univariate analysis (two-way ANOVA) was performed to assess differences between distances from the turbines ('far' *vs* 'very close') and years in terms of the above-mentioned biological variables and of the sediment fine sand fraction and TOM content. We used the paired 'very close' and 'far' samples of twelve selected turbines for which the data of three consecutive years were available. Assumptions of normality and homogeneity of variances were tested with a Shapiro-Wilk – and a Levene test, respectively, and log transformations were performed if these assumptions were not met. Tukey HSD post-hoc tests were performed to investigate significant differences in the interaction term Position x Year and/or in the factor Year.

Potential effects of distance and year on macrobenthic community structure were investigated with a Permutational multivariate Analysis of Variance (Permanova), based on a Bray-Curtis resemblance matrix of 4th root transformed data, with a fully crossed fixed two-factor design with factors distance (two levels: 'far' *vs* 'very close') and year (three levels: 2017 – 2018 – 2019). Because a significant result of a Permanova can be caused by truly significant differences between levels of a factor, but also by a strong heterogeneity among replicate data, homogeneity of multivariate dispersions was verified using the permutest routine (distances among centroids; Anderson 2006; Anderson *et al.* 2006; in case this test indicated that variances were significantly heterogeneous, we present cautious interpretations of the Permanova results. Similarity Percentages (SIMPER) routine analysis was subsequently performed to identify the contributions of individual species to the dissimilarity between groups of samples (Clarke & Gorley 2006).

Additionally, linear mixed models were used to predict patterns in the densities of selected macrobenthic species in the 'very close' samples based on year, fine sand fraction and TOM content. Potential multicollinearity was verified using a Variance Inflation Factor (VIF). The final model was obtained by stepwise removal of non-significant parameters until all remaining partial regression coefficients were significant. The residuals were then inspected to detect outliers, which were subsequently removed from the models. A Shapiro-Wilk test was used to verify whether residuals were normally distributed.

All analyses were performed in R (version 3.5.2) with the packages car (Fox & Weisberg 2019), vegan (Oksanen *et al.* 2013) and RVAideMemoire (Hervé 2020).

3. Results

The fine fraction of the sediment (125-250 µm) did not vary significantly over the years, but was consistently higher in 'very close' sediments compared to sediments 'far' from the turbine (two-way ANOVA, factor position: $F_{1, 66} = 21.383$, $p < 0.001$). While not statistically significant, fig. 3 suggests a trend of increasing fine sand fraction and organic matter content with time at the 'far' stations. The fraction of organic matter in the sediment (TOM) varied significantly among years and distances from turbines (two-way ANOVA, interaction Year x Position: $F_{2,66} = 3.54$, $p = 0.03$), but when investigated in more

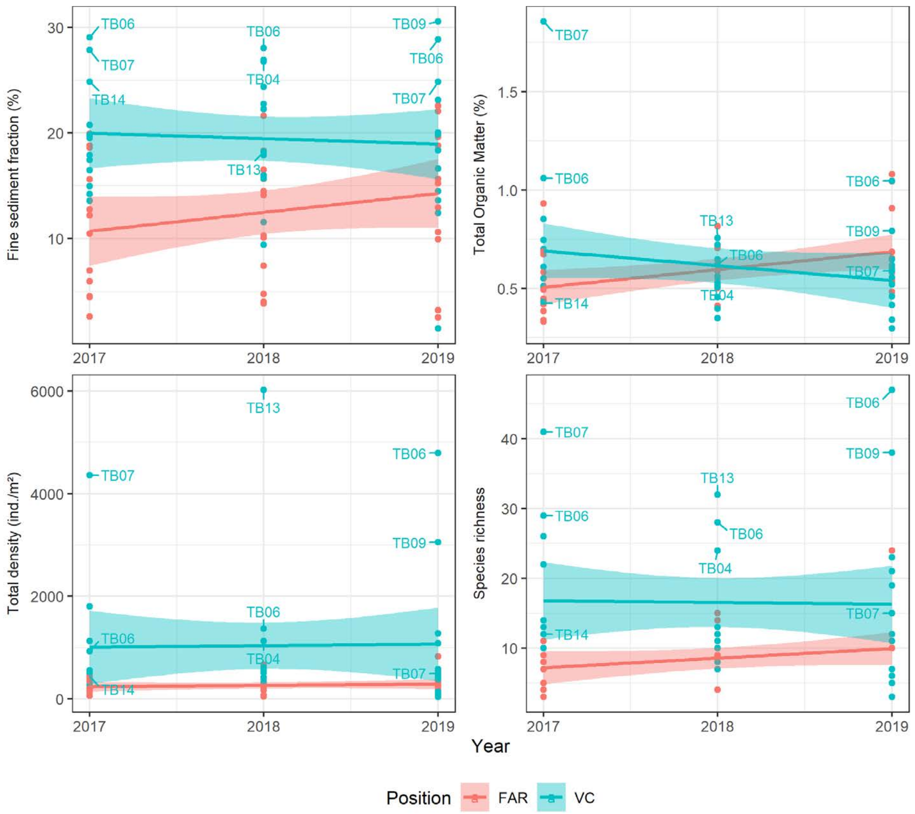

Figure 3. Temporal patterns in the fine sand fraction (125-250 µm; top-left panel) and the total organic matter content (top-right panel) of the sediment, macrobenthic total densities (bottom-left) and species richness (bottom-right) in stations 'very close' to (VC, *i.e.* at 37.5 m) and 'far' from (FAR, *i.e.* at 350-500 m) wind turbines. Positive outliers were visually identified and related to the corresponding turbine at which they were recorded.

detail, no clear differences between pairs of data were apparent (all pairwise comparisons within this interaction factor had $p > 0.05$; fig. 3).

Irrespective of the year in which they were sampled, macrobenthic densities and species richness were significantly higher 'very close' to the turbines compared to at further distances (two-way ANOVA, factor position: $F_{1, 66} = 25.22$, $p < 0.001$ for density and $F_{1, 66} = 20.28$, $p < 0.001$ for species richness; fig. 3).

A large scatter was apparent from the patterns in these univariate variables (fig. 3). Throughout the three years of this investigation, consistently higher fractions of fine sediment, higher macrobenthic densities and species richness were recorded 'very close' to six out of twelve turbines (see fig. 3).

Whereas macrobenthos density and species richness remained stable over the years, significant temporal patterns were apparent at the level of community structure: the communities 'very close' to the jacket foundations were significantly different from those 'far' from the turbines, but this difference was dependent on the year (two-way Permanova, interaction Position x Year: $F_{1,71} = 2.03$ $p = 0.03$). The communities 'very close' to the turbines were significantly different from their 'far' counterparts in 2017 and 2018 (pairwise comparisons: $p < 0.05$), but not in 2019 (pairwise comparison: $p = 0.08$). In contrast to communities 'far' from the turbines, those 'very close' to the turbines evolved over time: those of 2019 were different from those of 2017 and 2018 (pairwise comparisons: $p < 0.05$). SIMPER analysis

Table 2. Results of SIMPER analysis, showing the species that contributed most (up to a cumulative value of approximately 70%) to the difference in community composition in the sediments ‘very close’ to the turbines on the Thornton Bank between 2017 and 2019. Species highlighted in bold were selected for further temporal analysis; species in black are typically associated with natural permeable sediments in BPNS sandbanks, while those in blue tend to be associated with fine sandy sediments in the BPNS. Untransformed abundance data are shown and cumulative contributions are reported as proportions of 1

	Av. abundance 2017 (ind.m^{-2})	Av. abundance 2018 (ind.m^{-2})	Av. abundance 2019 (ind.m^{-2})	Cumulative sum 2017-2019
Urothoe brevicornis	**164.17**	**182.50**	**67.50**	**0.21**
Nephtys juv.	22.50	45.00	81.67	0.30
Nephtys cirrosa	**65.83**	**78.33**	**48.33**	**0.35**
Actinaria sp.	32.50	20.00	26.67	0.39
Terebellidae juv.	**12.50**	**38.33**	**73.33**	**0.42**
Nototropis swammerdamei	10.83	0.00	47.50	0.46
Bathyporeia elegans	16.67	27.50	20.83	0.49
Spiophanes bombyx	**27.50**	**51.67**	**12.50**	**0.52**
Nototropis falcatus	0.00	0.00	38.33	0.55
Echinocardium cordatum	10.00	16.67	20.00	0.58
Lanice conchilega	**0.00**	**17.50**	**75.83**	**0.60**
Abludomelita obtusata	23.33	2.50	32.50	0.62
Eumida sanguinea	**6.67**	**2.50**	**60.00**	**0.65**
Edwardsia sp.	1.67	0.83	47.50	0.67
Spio sp.	13.33	1.67	18.33	0.69

showed that the abundances of most species encountered in the 2019 samples collected ‘very close’ to the turbines were higher than in 2017 and 2018 (table 2). The exceptions to this rule were the burrowing amphipod species *Urothoe brevicornis*, the interstitial polychaete *Spiophanes bombyx*, and the errant polychaete *Nephtys cirrosa*, all three of which had declined in abundance. The abundance of unidentified *Actiniaria* sp. was variable between 2017 to 2019, but this could have resulted from an identifier effect (cf. abundance of the actiniarian *Edwardsia* sp. increased in 2019). Also the abundance of the amphipod *Bathyporeia elegans* was variable between the years.

We further investigated patterns in the abundance of the above three species that are typically associated with permeable sandbank sediments on the BPNS, and that declined between 2017 and 2019. We then compared these with patterns in the abundances of the tube-building polychaete species *Lanice conchilega* and of juvenile Terebellidae, which are probably (mainly) juveniles of *L. conchilega*, as well as of the associated polychaete species *Eumida sanguinea*. *Lanice conchilega* and *E. sanguinea* are typical representatives of low-dynamic habitat and increased markedly in abundance in the sediments close to the turbines between 2017 and 2019 (fig. 4). This increase in Terebellidae and in *E. sanguinea* was mainly driven by local effects at four specific turbines; at these turbines, the increase was statistically significant for *L. conchilega* (linear regression, log[*Lanice conchilega* density + 1] = -4188 + 2 * Year; n = 12; R^2_{adj} = 0.6; p = 0.002), but not for *Eumida sanguinea* nor for the juvenile Terebellidae. At the same time, it is important to note the extremely low abundances of these three species in the ‘far’ samples away from the turbines. Except for one turbine, lower abundances of the

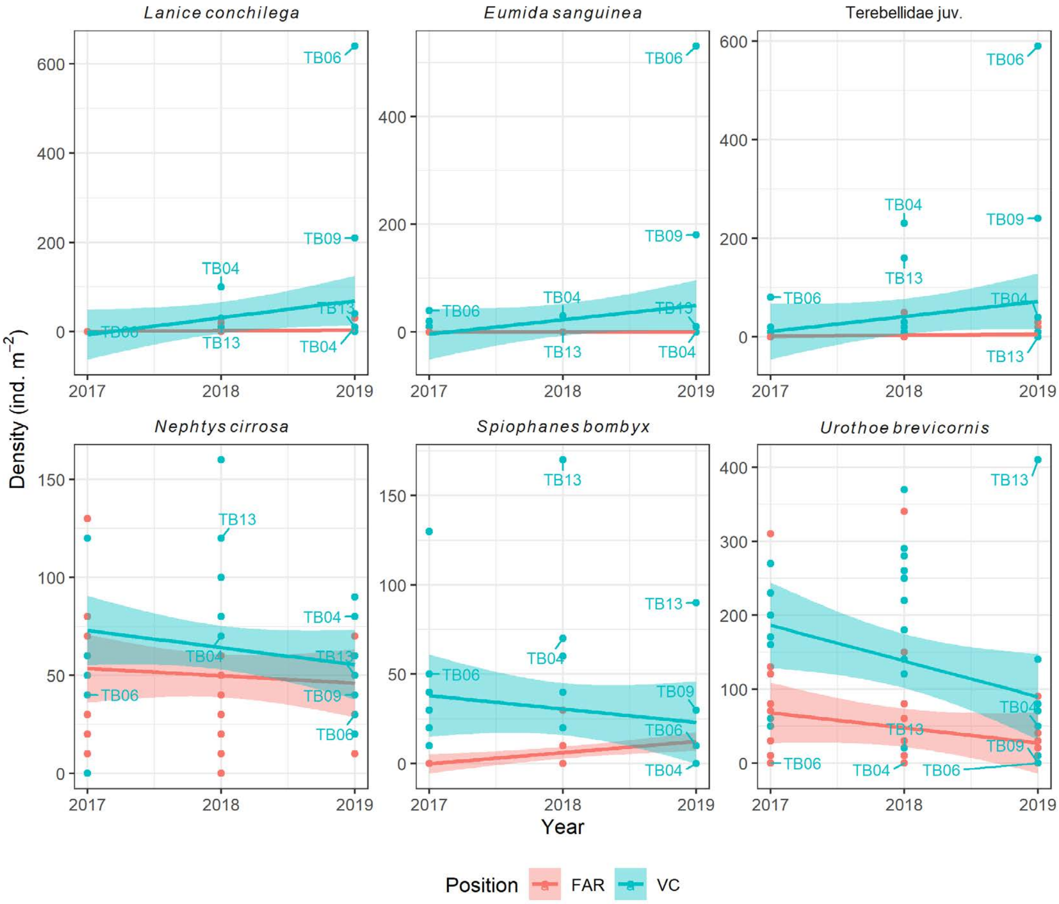

Figure 4. Temporal patterns in the abundances of *Lanice conchilega*, *Eumida sanguinea* and juve-nile Terebellidae, and of *Nephtys cirrosa*, *Spiophanes bombyx* and *Urothoe brevicornis* in stations 'very close' to (VC, *i.e.* at 37.5 m) and 'far' from (FAR, i.e. at 350 – 500 m) wind turbines. Positive outliers have been visually identified and related to the corresponding turbine at which they were recorded.

species associated with permeable sands were observed at the very same turbines where the species from low-dynamic habitats became more abundant.

The temporal patterns in the abundances of the selected species 'very close' to the turbines were primarily dependent on the fraction of fine sediment. The abundances of polychaete species associated with low-dynamic habitats were positively correlated with the fraction of fine sediment. In addition, *L. conchilega* abundances increased over time, whereas abundances of *E. sanguinea* were positively correlated with higher TOM content of the sediment. These models were highly significant and nearly 40% of the variance in the models was explained by the predictor variables (table 3).

In contrast, only very little variability in the abundances of the species associated with permeable sandbank sediments was explained by the predictor variables used in these models. The densities of the polychaete *N. cirrosa* did not show any dependence on year, fine sediment fraction or TOM content, whereas abundances of the amphipod *U. brevicornis* decreased significantly over the years. The densities of the interstitial polychaete *S. bombyx* slightly increased with the fine sand fraction: 10% of the variance in its abundance was explained by this variable, but the model was only just significant ($p = 0.04$; table 3).

Table 3. Results of linear mixed models on log-transformed abundances of selected macrobenthic species in the sediments very close to the turbines. The partial regression coefficients of the intercept and different predictor variables are shown when significantly different from zero, as well as the model adjusted R^2 and global p-value

Log-transformed species abundances	Intercept	Year	Fine sediment fraction	TOM	R^2_{adj}	Global p-value
Lanice conchilega	-0.002**	0.9**	0.2**	0	0.38	< 0.001
Eumida sanguinea	0	0	0.09*	2.7**	0.37	< 0.001
Terebellidae juv.	0	0	0.2***	0	0.38	< 0.001
Urothoe brevicornis	2177**	-1*	0	0	0.16	0.01
Spiophanes bombyx	0	0	0.1*	0	0.10	0.04
Nephtys cirrosa	0	0	0	0	0	> 0.05

*** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$

4. Discussion

Artificial hard substrates such as wind turbines can alter local hydrodynamics by modifying local currents: they may increase the current speed on one side of the turbine while reducing it on the other, creating a mixture of more dynamic and more sheltered areas (Coates *et al.* 2014; Rivier *et al.* 2016) and hence a higher spatial heterogeneity. Erosion and resuspension of sediment may occur on that side of the turbine foundation where the current speed increases (Rivier *et al.* 2016), resulting in an increased bed shear stress, in mobilized fine sediment, and hence in an enhanced turbidity in the water column (Baeye & Fettweis 2015). In the wake of the turbine, reduced currents allow the deposition of this fine sediment and a lower seabed shear (Rivier *et al.* 2016), resulting in a sediment matrix that contains a higher proportion of fine particles, a concomitantly lower permeability, and therefore a higher retention of organic matter (Coates *et al.* 2014; De Backer *et al.* 2014). This phenomenon is referred to as the sediment fining hypothesis. The tidal dynamics add a further level of complexity, in that the described pattern is not 'stationary' but travels around a turbine with the movement of the tidal ellipses.

In addition to a higher retention of deposited organic matter by less permeable sediments, sedimentary organic matter content can be further enhanced by the depositional flow of the faecal pellets and organic detritus shed by the abundant epifouling filter feeders that rapidly and massively colonise the turbines (Coates *et al.* 2014; van Deurs 2012). This in turn causes an increased food availability for the benthos near the foundations, and is expected to enhance local macrobenthos biodiversity. This suite of events is collectively referred to as the enrichment hypothesis (Coates *et al.* 2014; De Backer *et al.* 2014; Inger *et al.* 2009).

4.1. Sediment granulometry and organic matter content provide only partial support for the sediment fining and enrichment hypotheses

The 2019 results show partial evidence of the 'sediment fining theory', with a significantly higher fine sand fraction at the 'very close' stations on the Thornton Bank compared to the 'far' stations (Bruns 2020). This corroborates results from 2017 and 2018 (Lefaible *et al.* 2018, 2019). By contrast, sediment TOM – while indeed being significantly higher in 'very close' compared to 'far' samples in 2017 – exhibited a different pattern in 2018 and

2019, being even a little higher in the 'far' samples. While not shown in this report, this trend was found on both the Thornton and Bligh Bank (Bruns 2020). This trend does not follow from a decreased TOM content in 'very close' samples in the two most recent monitoring years, but rather from a tendency towards an increasing TOM content – albeit as yet not statistically significant – in the 'far' stations. This trend definitely requires a close follow-up in the coming years; if sustained or even strengthened, it might indicate a spatial expansion of what has hitherto generally been considered to be a very localised impact of turbines on the seafloor.

The operational phase of an OWF entails a complex interplay between abiotic and biotic components, which may also act at partly different spatial scales, rendering a correct interpretation of observed patterns, and hence an accurate prediction of future ones, particularly challenging (Dannheim *et al.* 2020). In addition to changes in hydrodynamics, the epifouling (mainly filter-feeding) fauna that colonises the turbine foundations (De mesel *et al.* 2015) is likely a key factor in any observed shifts in sediment organic matter content in the immediate vicinity of the turbines, and as such likely contributes to shifts in abundance, diversity and species composition of macrobenthic communities. In turn, shifts in the functional composition of the macrobenthos may further stimulate sediment fining and organic matter retention if species are favoured that enhance the deposition and retention of fine particles from the benthic boundary layer (De Backer *et al.* 2014; Dannheim *et al.* 2020).

It is therefore paramount that the current monitoring is continued in the next years, so that trends in sediment fining and enrichment over time and space can be properly documented. It would be advisable, though, to better characterise the organic matter in the sediments at different distances from turbines in order to have a clearer picture of its main origin(s), and to trace the fate of particulate organic matter produced by fouling fauna: is it deposited onto the sediment in the wake of the turbines, and if so, within what distance range? To what extent is it consumed by benthic macrofauna and/or decomposed by microbenthos, and to what extent does it eventually end up being buried into these sediments? In August 2020, a targeted monitoring was performed around a single turbine on the Bligh Bank and, in 2021, a similar monitoring is planned around a jacket foundation on the Thornton Bank to sample for sediment granulometry, organic matter and macrobenthos along a more detailed distance gradient from the turbine (*i.e.*, including additional distances on top of the 'very close' and 'far' from the present monitoring design), in order to obtain a sharper picture of the extent of any sediment fining or enrichment effects. The results of that targeted monitoring may affect the design of future monitoring campaigns. In the meantime, initiatives are taken by Royal Belgian Institute of Natural Sciences and Ghent University to obtain funding to trace the fate of faecal pellets of the dominant epifouling species into the water column, sediment and fauna around wind turbines.

4.2. Macrobenthos communities around wind turbines exhibit a shift in species composition and diversity

While the data from the past two years hence do not clearly corroborate the enrichment hypothesis in terms of sediment TOM content, enrichment of macrobenthos communities in the 'very close' sediments in terms of a higher species richness and overall abundance appear to be two fairly consistent trends over the past three years, although the values fluctuate substantially. In 2017 and 2018, significantly higher macrobenthic densities and species richness were observed in the 'very close' compared to the 'far' stations (Lefaible *et al.* 2018, 2019). While average species richness in the 'very close' stations was even slightly higher in 2019 than in 2018, the high variability among replicate samples rendered the difference with the 'far' stations statistically not significant in 2019 (Bruns 2020). Likewise, while very similar macrofaunal abundances in 'very close' samples were obtained in 2017 and 2019, being roughly threefold higher than in the 'far' samples, this difference was not statistically significant in 2019 as a consequence of high variability among turbines (Bruns 2020).

Similar to abundance and diversity, macrobenthos community composition differed substantially between 'very close' and 'far' stations, these differences being statistically significant in 2017 and 2018, but not in 2019. Unlike the results on abundance and diversity, macrobenthos species composition in the 'very close' samples also differed significantly between years, more specifically between 2019 on the one hand and 2017 and 2018 on the other. SIMPER analysis demonstrated the contributions of the different taxa to these among-year differences in community composition. A closer inspection of table 2 highlights that most of the taxa from the 'very close' samples were equally or more abundant in 2019 than in the two previous years. Notable exceptions to this pattern were *Urothoe brevicornis* (Amphipoda), *Spiophanes bombyx* and *Nephtys cirrosa* (both Polychaeta), all three of which declined in 2019. These three species are not just any species; they were the dominant taxa in sediments of the Thornton Bank during a baseline study in 2005, prior to any wind farm construction activity in the area (De Maersschalck *et al.* 2006). *Urothoe brevicornis* consistently remained the most abundant species in 2017 and 2018, whereas *N. cirrosa* ranked among the top-three of most abundant taxa until 2018. In 2019, however, neither of these species still ranked among the three most common taxa in 'very close' stations, although some caution is due for *N. cirrosa*, since a relatively high abundance of unidentified juveniles of the genus Nephtys was observed in 2019. All three species, *N. cirrosa*, *U. brevicornis* and *S. bombyx*, are key species of a *Nephtys cirrosa* community (Van Hoey *et al.* 2004; De Maersschalck *et al.* 2006; Breine *et al.* 2018), which is typical of well-sorted, medium to coarse, permeable sands in the BPNS. Such community is characterised by a moderate to low species richness (3-12 species per sample) and a rather low abundance (Van Hoey *et al.* 2004; Breine *et al.* 2018). Abundance-wise, the values we observed for total macrofauna were near the upper limit of what has been described for the *Nephtys cirrosa* community, whereas diversity-wise, the number of taxa consistently exceeded typical values for this community over the past three years (Lefaible *et al.* 2018, 2019; this report). It is tempting to suggest that the lower abundances of these three typical inhabitants of permeable sediments in 2019 could be a logical consequence of a sediment fining process. However, the measured environmental variables (*i.e.* fine sand fraction and total organic matter content) explained very little (up to 10% in *S. bombyx*) to no (*N. cirrosa* and *U. brevicornis*) spatiotemporal variation in the abundances of these key species. Hence, other drivers of the population abundances of these species are likely to be at play, perhaps including interactions with other macrobenthos species.

Indeed, the majority of the other macrobenthos species had very similar or (sometimes much) higher abundances in 2019 compared to the previous two years, which, along with the decline in two out of the three previously most common species, resulted in a significantly higher evenness in the 2019 macrobenthos communities 'very close' to the turbines (Bruns 2020). Among the taxa that exhibited a marked increase in abundance in 2019 were

Lanice conchilega, juvenile Terebellidae and *Eumida sanguinea* (Polychaeta). Given that juvenile Terebellidae are particularly difficult to identify to species level, and that *L. conchilega* is a terebellid polychaete, it is plausible that part or most of the juvenile Terebellidae belonged to this species. *Lanice conchilega* is a bio-irrigator which contributes to sediment bioturbation and a bio-engineer which, when present in sufficiently high densities, can alter the water flow in the benthic boundary layer, thus affecting bottom shear stress and the deposition and retention of fine particulates (De Backer *et al.* 2014; Alves *et al.* 2017), thereby contributing to both a fining of the sediment texture and an increasing organic matter content in surficial sediment. Indeed, sufficiently dense *L. conchilega* patches can function as small reefs which offer both food and shelter to a variety of benthic fauna (Callaway 2006; Rabaut *et al.* 2009). Their tubes also create habitat which provides a living space for other fauna. *Eumida sanguinea*, for instance, often hides inside *L. conchilega* tubes, provided these are of sufficient size. Hence, its increase in 2019 probably reflects its close association with *L. conchilega* (Callaway 2006; Rabaut *et al.* 2008). Both *L. conchilega* and *E. sanguinea* being typical representatives of fairly fine-grained, low-dynamic habitats, it is not surprising that variability in their abundances could be well explained by the fraction of fine sediment and – for *E. sanguinea* – TOM content. Indeed, models with these two predictor variables significantly explained nearly 40% of the variance in the population abundances of these two species. Any relationship between TOM content and *L. conchilega* abundance could be difficult to interpret, since both options – not mutually exclusive – exist: the species may respond positively to finer sediments with higher TOM content, but may in turn contribute to an increase in TOM in the sediments. The high patchiness – see further – of sediments with *L. conchilega* reefs further adds to the difficulty in finding statistically significant relationships at the scale and design of our sampling.

The development of *L. conchilega* aggregations represents a strong deviation from the baseline assessment in 2005 (De Maersschalck *et al.* 2006)– together with the declining abundance of previously dominant species typical of permeable sediments, and with the increased macrobenthos diversity – suggests that the macrobenthos communities 'very close' to wind turbines exhibit intermediate characteristics between a *Nephtys cirrosa* and an *Abra alba* community, the latter being characteristic for fine to medium sandy sediments in the BPNS (Breine *et al.* 2018). Given that an *Abra alba* community is further characterised by high overall macrofauna abundances and a prominent presence of bivalves (Breine *et al.* 2018; Van Hoey *et al.* 2004), two features which are not (yet) met in the 'very close' stations, we can at present not claim that an overall shift is taking place from a *Nephtys cirrosa* to an *Abra alba* community. The development of *Lanice* reefs may in turn negatively affect species like *Urothoe brevicornis* and *Spiophanes bombyx*, two interstitial species which may suffer from space and/or interference competition with tube-building Terebellidae.

However, as for the sediment granulometry and organic matter results, the spatial and temporal variability in macrobenthos communities in terms of abundance, richness and species composition calls for a cautionary interpretation at this stage. Indeed, the macrobenthos of coastal waters is typically characterised by a large year-to-year variability as a consequence of, among other things, climatic conditions (particularly peak winter temperatures and storms). Hence, the current trends need to be closely monitored in future years in order for real patterns to be more conclusively discriminated from largely stochastic variability. In that respect, we cannot exclude that the late sampling of the 'very close' stations due to harsh (stormy) weather conditions in October-November could have affected the 2019 results. Also, while the establishment of a population of a bio-engineering species like *L. conchilega* seems to fit a broader picture in which fining of the sediment facilitates settlement of *L. conchilega*, which in turn contributes to further (fine) sediment accumulation and habitat diversification, an unequivocal elucidation of cause-and-effect relationships and of feedback loops (here for instance between sedimentation and tube building; Alves *et al.* 2017b) in this chain of processes will require dedicated research efforts, as well as a more detailed investigation of the spatial patterns around wind turbines. Moreover, since ecological engineering effects of *L. conchilega* can vary strongly with population dynamics (Alves *et al.* 2017a), evaluating the importance of such feedback loops may prove yet more difficult.

4.3. Macrobenthic communities and sediment granulometry + organic matter content 'very close' to wind turbines exhibit a high degree of patchiness

The above-described development of high abundances of Terebellidae and of *E. sanguinea* 'very close' to wind turbines in 2019 was not a general phenomenon, but essentially occurred near roughly half of the investigated turbines, with particularly high abundances near turbines TB04, TB06, TB09 and TB13. The sediments 'very close' to these turbines were also characterised by some of the highest fine sand fractions, along with those of turbines TB07 and TB14, where *L. conchilega* was not abundantly present. When we focus on the first four turbines, there was a statistically significant increase in *L. conchilega* abundance with time (linear regression, log[*Lanice conchilega* density +1] = -4188 + 2 * Year; n = 12; $R^2_{adj} = 0.6$; p = 0.002). Sediments near three out of these four turbines were also characterised by low abundances of the species associated with permeable sands, supporting the idea that a shift is taking place from a community typical of permeable, well-sorted sands to one of fine to medium sediments. At the 'far' stations away from the turbines, these same three species were extremely rare or absent, suggesting that if any expansion of the sediment fining and enrichment effects is taking place, it has not yet reached 'far' stations and/or has not attained a level where it attracts species that are characteristic of more sheltered sediments and which may enhance community shifts through their bio-engineering capacity.

The high abundances of *E. sanguinea*, a species which tends to associate with *L. conchilega*, indicate that *L. conchilega* were already forming small reefs near some of the turbines. Such reefs affect the deposition and retention of fine particulate matter from the benthic boundary layer and can therefore have reverberating effects on the shifts in sediment properties and associated macrobenthic communities. At the same time, given their typically small size (≤ 10 m^2; Rabaut *et al.* 2008), *Lanice* reefs increase the small-scale patchiness of benthic communities and of sediment characteristics. Indeed, the reefs typically occupy ≤ 18 % of a suitable habitat/sediment (Rabaut *et al.* 2009). This may in part explain why during the last three years, *L. conchilega* and *E. sanguinea* seem not to consistently occur at the same wind turbines: given their small size and relatively restricted space occupation, *Lanice* reefs may be easily missed by Van Veen grab sampling. The high heterogeneity of sediments around turbines as caused by hydrodynamic properties (Dannheim *et al.* 2020; see section 4.1) can be strongly enhanced at an even smaller spatial scale by the development of bio-engineering fauna.

An additional factor which may contribute to the high variability among 'very close' stations of replicate turbines are mussel clumps which get detached from the foundations and settle onto the sediment, where they can create patches of locally enriched (with organic matter) sediments that can be attractive to macrobenthic deposit feeders and scavengers, as well as create small-scale habitat heterogeneity (Lefaible *et al.* 2019; Bruns 2020). This phenomenon has repeatedly been observed over the past three years of monitoring.

Given that these sources of small-scale heterogeneity cause large variability in sedimentary and biological properties around different turbines, they also hamper the detection of robust, statistically underpinned trends. It may therefore be necessary to reconsider the current monitoring design, where a reduction of the number of turbines included could compensate for a higher effort in sampling the local spatial variability at the turbine level, explicitly encompassing *Lanice* patches, *Mytilus* clumps and 'bare' sediments.

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CHAPTER 7

A DECADE OF SOFT SEDIMENT EPIBENTHOS AND FISH MONITORING AT THE BELGIAN OFFSHORE WIND FARM AREA

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Abstract

Since 2005, an environmental beam trawl survey has been set-up in the Belgian part of the North Sea, designed to investigate the effect of offshore wind farms (OWFs) on epibenthos and demersal fish that reside on the soft sediments in between the turbines. The survey follows a Before/After – Control/Impact design and focusses on the two oldest concession areas: C-Power located on the Thornton Bank (54 turbines, 325 MW) and Belwind (55 turbines, 165 MW) located on the Bligh Bank. This contribution presents the results of resp. 8 and 9-year post-construction beam trawl monitoring. We specifically aimed at (1) investigating more in depth long-lasting changes both at the assemblage and the species level using a combination of multivariate and univariate techniques, and (2) exploring which environmental variables affect species assemblages associated with wind farms at a larger scale to better include natural variability.

Results showed that soft sediment epibenthos and fish assemblages in between the turbines underwent no drastic changes due to the presence of the OWFs at mid/longer term. The assemblages were mainly structured by temporal variability due to changes in temperature and climate indices NAO and AMO, and this degree of change is proportionally much larger than the local effect of the present OWF areas. Nevertheless, some significant secondary effects could be clearly related to the presence of the OWFs pointing to an expansion of the OWF effect beyond the immediate vicinity of the turbine on two fronts: (1) an expansion of the reef effect, and (2) signals of a refugium effect.

Expansion of the reef effect is suggested through the appearance of an increased number of hard substrate-associated species like long-clawed porcelain crab *Pisidia longicornis*, edible crab *Cancer pagurus* and seabass *Dicentrarchus labrax* in the soft sediment trawls. Also increased densities of common squid *Loligo vulgaris* were observed in C-Power, probably an indication for the use of the jacket foundations as substrates for egg deposition. However, the clearest indication for reef expansion was the significantly increased abundances of blue mussel *Mytilus edulis* and anemones *Anthozoa* sp., two species dominating the epifouling communities on the turbines. Although, densities were still low, they could increase heterogeneity in the soft-bottom sediments in between foundations in the future. Slightly, but significantly, increased fish densities of

some common soft sediment-associated fish species (common dragonet *Callionymus lyra*, solenette *Buglossidium luteum*, lesser weever *Echiichthys vipera* and plaice *Pleuronectes platessa*) inside the C-Power wind farm compared to the outside reference area seem to be the first signs of a refugium effect, probably related to a combination of fisheries exclusion and increased food availability. More pronounced effects were found for C-Power than for the more offshore Belwind OWF, stressing that effects might be site-specific and that extrapolation of these findings to other OWFs should be done with care.

1. Introduction

To meet with renewable energy targets, the European Commission advises that Europe needs to generate between 230 and 450 GW of offshore wind energy by 2050, with the centre of gravity for production located in the North Sea region. It is estimated that, by that time, 450 GW could provide almost a third of Europe's entire electricity demand (EWEA 2019). Since 2009, Belgium has been producing offshore wind energy and by the end of 2020, the totally installed capacity, organised within 9 different offshore wind farms (OWFs), will climb to 2262 MW produced by almost 400 turbines, which addresses 10% of the total Belgian need for electricity and meets half of its renewable energy targets (Rumes & Brabant 2019). Due to an additional area of 284 km^2 for the development of wind energy that is foreseen in the new marine spatial plan of 2020-2026, this capacity can even further increase up to 4000 MW (Rumes & Brabant 2019).

This rapid development of renewable wind energy at sea raises questions about its effects on the marine environment, and more importantly on the biota that live in this environment. The most prominent effect of the construction of OWFs is the introduction of artificial hard substrates into typical soft bottom environments, which will inevitably lead to alterations of soft sediment habitats and communities at a variety of spatial scales (Ashley *et al.* 2014 and references therein). The introduction of hard substrates generates a new 'rocky' habitat which attracts hard substrate species (Lindeboom *et al.* 2011; Kerkhof *et al.* 2012; De Mesel *et al.* 2015; Coolen 2017), and creates a reef effect for epibenthic fauna and demersal and bentho-pelagic fish (Reubens *et al.* 2011, 2013; Stenberg *et al.* 2015). These are changes that particularly occur at smaller spatial scales (turbine scale), but may, of course, affect the broader spatial scale through ecological interactions such as trophic linkages and energy flow (Gill & Wilhelmsson 2019). Other effects are underwater sound originating from the operational turbines or electromagnetic radiation created by the infield cables that can affect organisms as well and might disturb communication between individuals (Gill & Wilhelmsson 2019). Additionally, fisheries are excluded from the OWF area, which is another potential effect at play to induce changes on the soft-bottom assemblages (Handley *et al.* 2014).

Within this chapter, the objective is to investigate the potential long-term effect of OWFs in the Belgian part of the North Sea (BPNS) on soft-bottom epibenthos and fish species and assemblages that reside on the sandy sediments in between the turbines. Therefore, ILVO performs beam trawl monitoring surveys since 2005 following a Before/After Control/Impact (BACI) design within two OWFs: C-Power located on the Thornton Bank (54 turbines, 325 MW) and Belwind (55 turbines, 165 MW) located on the Bligh Bank, the first OWFs in Belgian waters. Results presented here are resp. 8 and 9-year post-construction. In previous reports, a post-construction 'overshoot' of epibenthos density and biomass was observed caused by an increase in opportunistic, scavenging species (Derweduwen *et al.* 2016a; De Backer & Hostens 2017). This was, however, a temporary phenomenon lasting only two-year post-construction. After this 'overshoot', 6 to 7-year post-construction, no real significant long-term changes have been observed for the soft-bottom epibenthic and fish assemblages in between the turbines (at distances > 200 m) (De Backer and Hostens 2018). This could be because time after construction was probably still too short and the whole operational OWF area not yet large enough to signal effects of fisheries exclusion beyond the immediate vicinity of the turbine. Another reason can be that natural variation is too large to detect effects at the assemblage level (Dannheim *et al.* 2019) or that effects are restricted to certain species, and as such not easily picked up at the community level.

Therefore, in the current report, we specifically aimed at (1) investigating more in depth long-lasting changes both at the assemblage and the species level using a combination of multivariate and univariate techniques, and (2) exploring which environmental variables affect species assemblages associated with wind farms at a larger scale to better include natural variability. We were mainly interested in 'stable' communities and long-term effects, and therefore focused on the years after the post-construction 'overshoot', so excluding the 2 years after construction from most analyses. These 2 post-construction years also coincide with the pioneering stage of the fouling assemblage on the turbines (Kerkhof *et al.* 2019), indicating as well that these first post-construction years were rather 'unstable'.

2. Material and methods

2.1. Sampling

Since the previous report of De Backer and Hostens (2018), two extra sampling campaigns were performed in autumn 2018 and 2019 with RV Belgica. Beam trawl samples were taken in between the wind farms (4 within C-Power and 3 within Belwind), and at several reference locations away of the concessions (fig. 1). On these track locations, fish fauna and epibenthos were sampled with an 8-meter shrimp beam trawl (22 mm mesh in the cod end) equipped with a bolder-chain. The net was towed for 15 minutes at an average speed of 4 knots over approximately 1 nautical mile. Data on time, start and stop coordinates, trajectory and sampling depth were noted to enable a correct conversion towards sampled surface units. The fish tracks are more or less positioned following depth contours that run parallel to the coastline, thereby minimising the depth variation within a single track, except for tracks 2 and 3 within the C-power concession which are perpendicular to the coastline due to the positioning of the infield electricity cables. Epibenthos and fish were identified, counted, measured (all fish, crabs and shrimps) and wet weighted (all epibenthos) onboard. The samples that could not be fully processed onboard were frozen and further processed in the lab.

2.2. Data used

2.2.1. Biological data

The time series of trawl samples in both C-Power and Belwind dates back to respectively 2005 (some other samples were available for 2004) and 2008 (2004 for 1 reference sample). However, within the sampling period 2004-2019, the sampling design had to be adapted based on previous monitoring results, wind farm accessibility, weather conditions and research vessel availability. An overview on sampled locations in autumn during the entire time period is given in table 1. For an overview map of all track locations, the reader is referred to Vandendriessche *et al.* (2015).

For C-Power, the construction of the second and third phases (48 jacket foundation turbines) took place in 2011. As such, trawls from 2004 up to 2010 were considered to represent baseline conditions (Baseline period) for both impact (Impact) and reference areas (Reference), while trawls after 2013 reflect conditions after which the first effects of the impact had already diminished (operational period; table 1). This way, the post-construction 'overshoot' effect (Derweduwen *et al.* 2016a; De Backer & Hostens 2017) is excluded as much as possible, and the long-term effects of the wind farm can be studied more reliably. Although wind farm construction of the first phase of C-Power (6 gravity-based turbines) started in 2008 and finished in 2009, it was decided to consider these years as baseline conditions, since our sampling locations were located away from these turbines and were as such not (are at least little) impacted by the first construction phase.

For Belwind, construction of the OWF took place in 2010 (55 monopile foundation turbines). The baseline for both impact and reference area was established in 2008 and 2009 (baseline period). Similarly, as for C-Power, the two post-construction years were excluded from statistical analyses to reliably study long-term effects. As such samples from 2013-2019 were used to represent the 'settled' operational wind farm conditions (operational period; table 1).

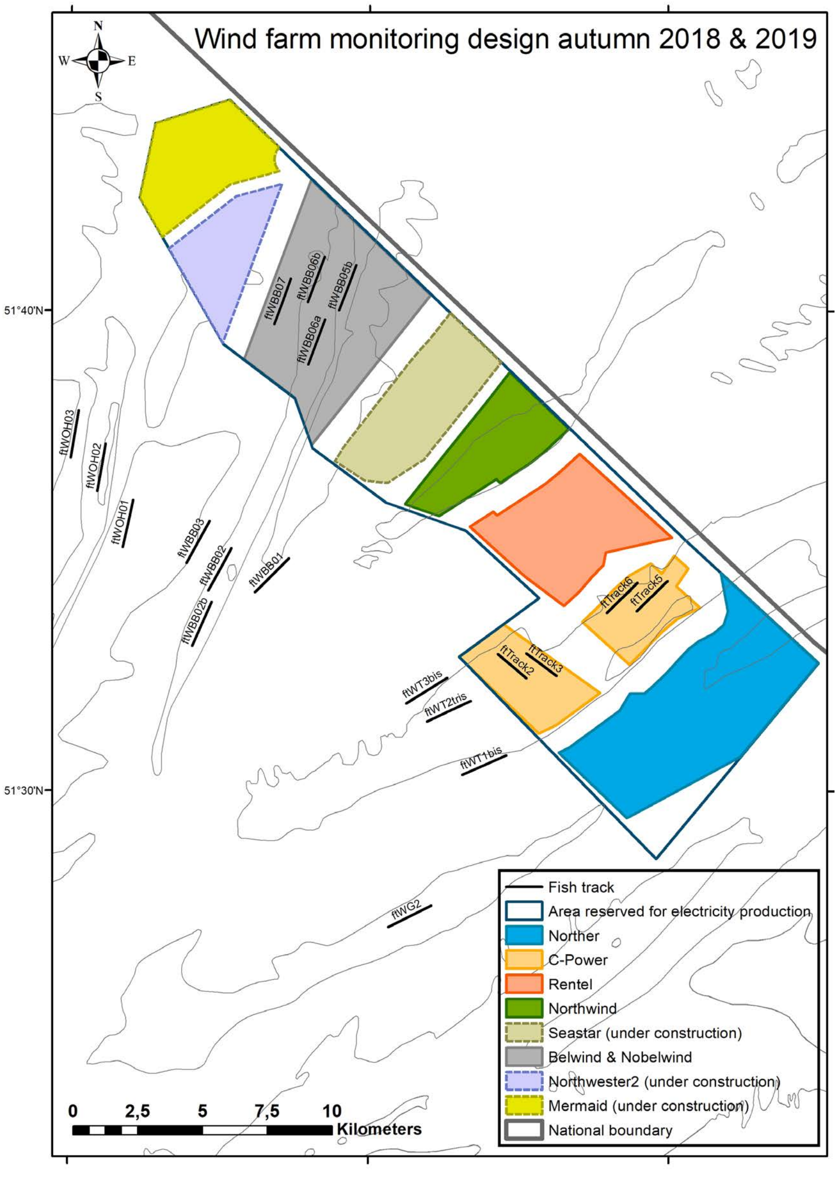

Figure 1: Overview map showing the 2018 and 2019 trawl locations at the C-Power and Belwind concession area and the respective reference locations.

Table 1. Overview table of the sampling design for the C-Power and Belwind wind farm within the time period 2004-2019 with indication of the different periods as used in the analyses. Construction periods are marked in light blue.

C-Power sampling design		Baseline								'Overshoot'		Operational					
Station	**Ref/Imp**	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015**	2016	2017	2018	2019
ftTB1	Ref	l															
ftTB2	Ref	l															
ftTB5	Imp	l															
ftTB8	Imp	l															
ftTrack1	Imp									s							
ftTrack2	Imp									s	s	s		s	s	s	s
ftTrack3	Imp									s	s			s	s	s	s
ftTrack4	Imp									s							
ftTrack5	Imp									s	s	s		s	s	s	s
ftTrack6	Imp									s	s	s		s	s	s	s
ftWG2	Ref		l	l	l	l		s		s	s	s		s	s	s	s
ftWT1	Ref		l			l		s (*)		s	s	s (*)		s (*)	s (*)	s (*)	s (*)
ftWT2	Ref		l			l	l & s	s (*)	s	s	s	s (*)		s (*)	s (*)	s (*)	s (*)
ftWT3	Ref		l			l	l & s	s		s	s	s		s (*)	s (*)	s (*)	s (*)
ftWT4	Imp		l			l		s									
ftWT5	Imp		l			l	l & s	s									
ftWT6	Imp		l			l		s									
ftWT8	Imp		l	l	l	l	l	s									

Belwind sampling design		Baseline							'Overshoot'		Operational						
Station	**Ref/Imp**	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015**	2016	2017	2018	2019
ftWBB01 (G)	Ref	l	l	l	l	l	l	l & s	s	s	s	s		s	s	s	s
ftWBB02 (T)	Ref					l	l	s	s	s	s			s	s	s	s
ftWBB02b (T)	Ref								s	s						s	s
ftWBB03 (G)	Ref					l	l	s	s	s	s	s		s	s	s	s
ftWBB05 (G)	Imp					l			s	s	s	s				s(*)	s(*)
ftWBB06 (T)	Imp					l	l										
ftWBB06a (T)	Imp								s	s				s	s	s	s
ftWBB06b (T)	Imp								s	s	s			s	s	s	s
ftWBB07 (G)	Imp					l	l		s	s	s				s	s	s
ftWOH01 (G)	Ref					l		s	s	s	s	s			s	s	s
ftWOH02 (T)	Ref					l	l	s	s	s	s				s	s	s
ftWOH03 (G)	Ref					l	l	s	s	S	s				s	s	s

Long (l) and short (s) refer to the duration of the trawl (long = 30 minutes, short = 15 minutes). Asterisks (*) refer to stations that were relocated due to logistic considerations. For Belwind, a distinction is made between top samples (T) and gully samples (G). Ref: reference tracks outside concession; Imp: impact tracks in specific wind farm. **: no sampling due to unavailability of sampling vessel.

For all analyses, count data were converted to densities based on the trawled surface area for standardisation to individuals/1000m². Pelagic species (based on www.fishbase.org) such as *Sprattus sprattus*, *Trachurus trachurus*, *Scomber scombrus*, next to jellyfish, bivalves (such as *Abra alba*) and polychaetes were excluded from the analyses, since these are not quantitatively sampled with a beam trawl. So, for the ecosystem component fish, both demersal and bentho-pelagic fish were retained and these are throughout the chapter referred to as fish.

2.2.2. Environmental variables

A time series of the North-Atlantic Oscillation index (NAO), a measure of the pressure difference between Iceland and the Azores was downloaded from the NOAA Climate Prediction Centre website (https://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao.shtml). A similar data set containing values of the Atlantic Multidecadal Oscillation (AMO) was used from the NOAA Earth System Research Laboratory's Physical Sciences Division website (https://www.esrl.noaa.gov/psd/data/timeseries/AMO/). These two climate indices have already been used in several ecological studies, because it has been widely recognised that they can influence the distribution and abundance of marine species. Next to yearly values, winter indices were also calculated by averaging the values of the months December-March. Satellite-originating sea surface temperature (SST) values were obtained through the EU Copernicus Marine Environment Monitoring Service (http://marine.copernicus.eu/) as well as measurements of chlorofyll (chl), phucoxanthin silicon, nitrate (NO_3), ammonium (NH_4) and oxygen (O_2) values. These monthly datasets were averaged over 0.25° x 0.25° grid cells to obtain a standardised resolution and this was done for each year. Trawl locations were assigned to a grid cell and given the environmental values of this grid cell. Further, yearly SST winter values were calculated in the same way as for the climatic indices. Moreover, to consider lagged responses to changes in SST, lagged SST 1-3 years were incorporated as well in the analyses. Lastly, daily discharge values from the adjacent Scheldt estuary were obtained through the Flemish Hydrological Information Centre and mean values for each year were calculated (www.waterinfo.be).

2.3. Data analysis

2.3.1. Multivariate analyses

For this study, we tested multivariate wind farm effects on densities for two ecosystem components (epibenthos and fish) in a two-factorial PERMANOVA (type III sums of squares because of unbalanced design) with factors 'period' and 'impact' (with 'period' being a time factor being either baseline or operational, see section 2.2.1). This was done for both the C-Power and Belwind concession separately. The primary aim was to analyse interaction effects between 'period' and 'impact', since these would reveal whether the changes that occurred could be attributed to the construction of the OWF. When a significant effect for the 'impact x period' interaction term was found, pairwise tests were conducted to test for differences between impact groups (*i.e.* reference versus impact samples) within each period or between periods within each impact group. Similarity percentages (SIMPER) analysis was done to detect the species responsible for the observed changes between groups of samples. Principal coordinates analysis (PCO) was used to visualise the data with additional vector overlay that was based on Spearman correlations ($R > 0.75$).

Distance-based linear models (DistLM) based on BEST selection and AIC_c criterion were used to relate patterns in community structure to the environmental variables, and one extra categorical variable related to the impact with four binary categories: 'reference samples before construction', 'impact samples before construction', 'reference samples after construction' and 'impact samples after construction'. Environmental data were normalised and collinearity among variables was examined using Spearman rank correlation coefficients prior to the DistLM analyses. For linearly dependent variables ($|r| > 0.7$) only one variable was retained in the analysis. As such NO_3, NH_4, phucoxanthin and salinity were excluded from the analyses due to collinearity with chlorophyll. Si, O_2 and winter SST were excluded because of collinearity with average SST. For Belwind, SSTlag1 was also correlated with average SST and excluded from the models. We specified indicators to combine sets of predictor variables to be able to put related variables together in the model (Anderson *et al.* 2008). In total, two individual variables chlorophyll and discharge and four indicators (sets of variables) were included in the DistLM model: temperature including SST, SSTlag1 (not for Belwind), SSTlag2, SSTlag3; NAO including avg. NAO and winter NAO; AMO including avg. AMO and winter AMO; and impact including the four binary categories as mentioned above. We ran both BEST model and marginal tests showing how much each variable explains when taken alone, ignoring all other variables (Anderson *et al.* 2008).

For all these tests, rare species (occurring in < 5% of the samples over the entire time series regardless of their densities but these were usually very low as well) were excluded, density data were square root transformed and similarity among samples was quantified using Bray-Curtis similarity index.

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

2.3.2. Univariate and single species models

A wind farm effect was tested for univariate measures of epibenthic organisms and demersal fish: species number, total density and total biomass (only available for epibenthos). The OWF effect was also studied at species level, using the densities of the ten most abundant species for both epibenthos and fish of both OWFs. Lastly, two epibenthic species *Anthozoa sp*. and *Mytilus edulis*, known to be dominating the fouling community on the turbines (Kerkhof *et al.* 2019), were also included in the analysis since previous research suggested a wind farm effect for both species (De Backer & Hostens 2018).

For both the univariate measures and the single species densities, regression modelling in combination with a BACI (Before-After/Control-Impact)-approach was applied. Adding an impact factor (Reference/Impact; RI), a time factor (Baseline/Operational; BO) and their interaction to each model allowed for testing a wind farm effect (significant interaction), whilst also taking into account any natural variation in time. These analyses were conducted for both OWFs separately, as they differ in depth, foundation type, sediment characteristics and other environmental conditions.

Based on previous analyses of Belwind (*e.g.* Vandendriessche *et al.* 2015), it was decided to include a random effect of depth within the models as the random factor top/gully, resulting in the use of linear mixed modelling for Belwind. As the differences in depth between samples are much less pronounced in C-Power, it was deemed not necessary to include such random factor for the models of C-Power. The univariate measures and the densities of the different species of fish and epibenthos were modelled using generalised least squares (GLS) models for C-Power. A variance structure was included for both model groups. The use of such a variance structure allows for differences in variance between different sample groups and is widely used to control the heterogeneity within the data (Zuur *et al.* 2007). The

final models for both wind farms are given by:
For C-Power: gls(Y ~ RI + BO + RI:BO, weights = var. structure)
For Belwind: lme(Y ~ RI + BO + RI:BO + random [top/gully], weights = var. structure)
The different variance structures that were used are:

- vf = varIdent(form = ~ 1 | BO)
- vf1 = varIdent(form = ~ 1 | RI)
- vf2 = varComb(varIdent[form = ~ 1 | RI], varIdent[form = ~ 1 | BO])
- vf3 = varIdent(form = ~ 1 | IR*BO)

They were included in the weights argument of the model. Choosing the best model was done by testing four different variation structures against the simpler linear model, and selection was based on the Akaike Information Criterion (AIC) with a preference for the simplest model having the lowest AIC-value. The model selection for all models was performed using maximum likelihood (ML) estimation, while the final models were fit using restricted maximum likelihood (REML) estimation. This is mainly important for mixed models, because ML estimation can in this case produce biased estimates for variance and covariance parameters (Zuur *et al.* 2009). Significance tests were conducted only on the final model fitted with REML using a two-way ANOVA (type III, Chisq). The presence of outliers was assessed by using Cleveland dotplots and boxplots; and if present were removed to avoid any misfit of the model. Only data points that would possibly change the outcome of the model were considered as outliers, because a high variation is typical for ecological density data and has to be taken into account when estimating confidence intervals and standard errors. To deal with this high amount of variation, the removal of outliers was chosen over a transformation of the response variable, because the latter changes the entire relationship between the explanatory variables and the response variable for univariate techniques (Keele 2008).

Next to these statistical model results, visualisation of the time series was done in time series graphs based on average values to enable putting the model results in a wider perspective. All analyses, data exploration, data frame manipulations and visualisation were conducted in R using following packages: dplyr, tidyr, ggplot2, ggpubr, nlme and car (R Core Team 2020; version 3.6.1).

3. Results

3.1. Epibenthos

3.1.1. Species number, density and biomass

For C-Power, the time series plot for species number (S) showed a slight general increase over the years for the impact samples (fig. 2). When comparing baseline with operational period, a significant OWF effect for species number (interaction term p = 0.0016) was observed. Average species number increased within the OWF area (avg. S: 13 spp. vs 18 spp.), while average species number was stable for the reference area (avg. S: 16 spp.). Taking a closer look at the species list made clear that the increase in species number is mainly due to the appearance of species associated with hard substrates in the impact samples that were absent in these samples before the construction of the OWF. It concerns *Cancer pagurus*, *Mytilus edulis*, *Ophiothrix fragilis*, *Pilumnus hirtellus* and *Pisidia longicornis*. Epibenthos density and biomass within C-Power showed a very similar pattern over time, with immediately after construction an overshoot in density and biomass within the impact samples (fig. 2). However, no significant OWF effect was observed on the longer term, when modelling baseline versus operational period (resp. p = 0.44 and p = 0.21). For both impact and reference samples, there is a significant increase in both average density (resp. 22 *vs* 48 ind./1000 m^2 and 37 *vs* 52 ind./1000 m^2) and biomass (resp. 59 *vs* 182 g WW/1000m^2 and 94 *vs* 166 GWW/1000 m^2) towards the

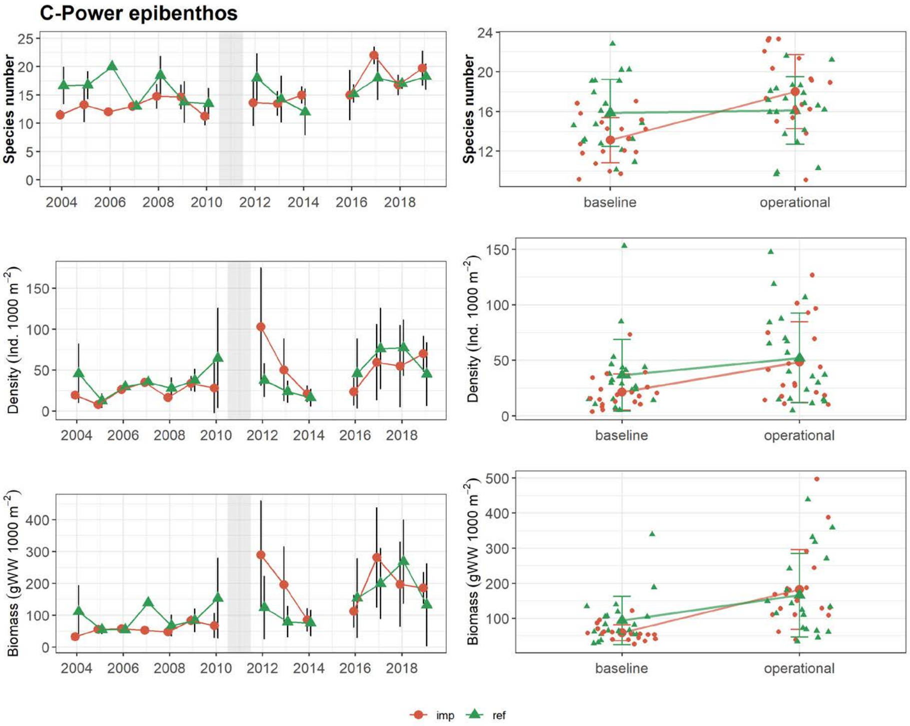

Figure 2. (Left) Time series plots of the univariate variables species number (S), density (N) and biomass for epibenthos at C-Power wind farm. (Right) Mean values (± SD) for baseline (2004-2010) and operational (2014-2019) period allowing to identify offshore wind farm effect. Construction of the second phase of C-Power was in 2011 indicated with a grey rectangle.

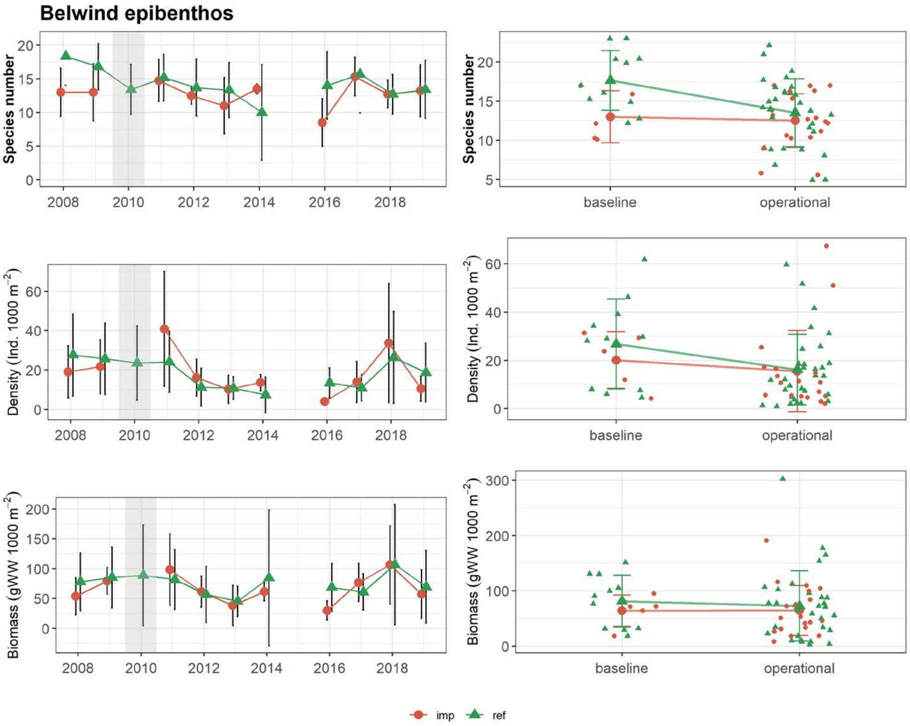

Figure 3. (Left) Time series plots of the univariate variables species number (S), density (N) and biomass for epibenthos at Belwind wind farm. (Right) Mean values (± SD) for baseline (2008-2009) and operational (2012-2019) period allowing to identify offshore wind farm effect. Construction of Belwind was in 2010 indicated with a grey rectangle.

operational period (resp. p = 0.012 and p < 0.001) (fig. 2).

For Belwind, a small significant interaction effect (p = 0.033) was detected for epibenthic species number. Average species number declined within the reference samples from 17 to 13 spp., while for the impact samples it remained stable (around 13 species; fig. 3). Investigating the species list, we as well observed the appearance of some hard substrate-associated species in the impact samples after construction of the OWF namely *M. edulis* and *Necora puber*. Also *Inachus dorsettensis* and *C. pagurus* appeared in the impact samples after construction but this applied as well to the reference samples. For density and biomass, no OWF effect was detected (resp. p = 0.23 and p = 0.5), average density and biomass were very similar in impact and reference samples and in both periods. Nevertheless, similarly as for C-Power, a small overshoot in density was observed right after construction in the impact samples (fig. 3).

3.1.2. Community structure linked to predictor variables

For C-Power, a significant interaction effect (impact x period, $p_{perm} = 0.01$) was observed. Pairwise tests showed that both in the baseline and in the operational period, impact and reference samples significantly differed from each other (resp. p_{perm} = 0.001 and p_{perm} = 0.04), and even higher significant differences occurred between both periods within each impact group (p < 0.001). SIMPER analyses showed that much lower average densities of brown shrimp *Crangon crangon* (REF: 9 *vs* 0.2 ind./1000m²; IMP: 6 *vs* 0.2 ind./1000 m²) and much higher average densities of *Ophiura ophiura* (REF: 2.5 *vs* 14.5 ind./1000m²; IMP: 12.2 *vs* 1.4 ind./1000m²) occurred in the operational period for both impact groups. Difference between reference and impact group within the baseline period was mainly because of higher average densities of *C. crangon* and much higher average densities of *Ophiura albida* (resp. 8.5 *vs* 2.6 ind./1000m²) in reference samples compared to impact samples but overall species composition was very similar. In the operational period, differences in average densities of dominant species also occurred but less pronounced, hence the smaller significant difference and more overlap in the PCO plot (fig. 4). More remarkable was the increased average densities of hard substrate-associated species

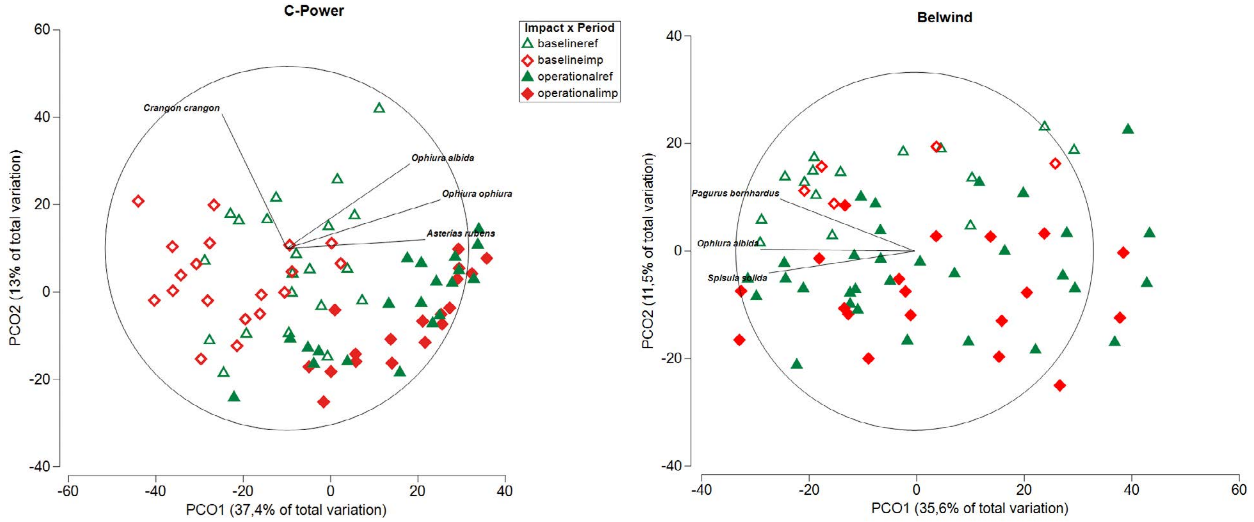

Figure 4. PCO plots based on Bray-Curtis resemblance matrix of square root transformed epibenthic density data for C-Power and Belwind with indication of impact group and sampling period. Vector overlay shows species best correlated with the observed multivariate pattern and is based on Pearson correlation (> 0.75).

Table 2. Proportion of epibenthic community variation that is explained by each individual predictor variable based on marginal DistLM tests for C-Power and Belwind

Variable	Pseudo-F	p-value	Proportion explained (%)
C-Power			
Chlorophyll	1.891	NS (0.07)	2.4
Temperature	7.601	0.0001	29.4
AMO	2.464	0.0097	6.2
NAO	8.721	0.0001	18.9
Discharge	6.636	0.0001	8
Impact	10.070	0.0001	29
Belwind			
Chlorophyll	2.860	0.0158	4
Temperature	2.377	0.0022	9.8
AMO	1.888	0.0372	5.3
NAO	2.641	0.0045	7.3
Discharge	3.241	0.008	4.5
Impact	2.539	0.0013	10.3

M. edulis (resp. 1 *vs* 0.002 ind./1000 m^2) and *Anthozoa* sp. (resp. 1.1 *vs* 0.1 ind./1000m^2) in impact samples compared to reference samples. A DISTLM analysis investigated the relationship between predictor variables (both climate, environmental and impact) and the observed multivariate pattern. Marginal tests showed that all predictor variables, except for chl, had individually a significant relationship with the multivariate data cloud (table 2). Temperature and impact individually explained most of the variation, both 29% (table 2). Together, these five predictor variables explained 55% of the total variation in the epibenthic community structure of C-Power based on BEST model with AIC$_c$ criterion.

For Belwind, no significant interaction effect (impact x period, $p_{perm} = 0.7$) was observed, indicating that the epibenthic community structure on the soft sediments between the turbines was not affected by the presence of the OWF (fig. 4). The only significant difference was found between baseline and operational period ($p_{perm} < 0.001$) and SIMPER indicated that this was mainly because of a decrease in average densities of *O. albida* (5.1 *vs* 2.4 ind./1000 m^2) and *Pagurus bernhardus* (7 *vs* 3.3 ind./1000 m^2). DistLM marginal tests showed that all predictor variables individually explained a smaller or larger part of the observed multivariate variation. Temperature and impact individually explained the highest proportion of the variation, resp. 9.8 and 10.3% (table 2). The BEST model based on AIC$_c$ criterion selected a combination of chl, NAO, discharge and impact which explained 23.4% of the total variation in the epibenthic community structure.

3.1.3. *Single species models*

Top 10 most common species

In C-Power, *P. bernhardus*, *O. albida* and *Asterias rubens* were the top 3 most abundant epibenthic species. For none of them a wind farm effect was observed, nor for the rest of the top 10 except for *Loligo vulgaris* (p = 0.03) (table 3 and figures in annex 1). This species showed a doubling in density for impacted samples in the operational years compared to the baseline (0.23 *vs* 0.46 ind./1000 m^2), while an

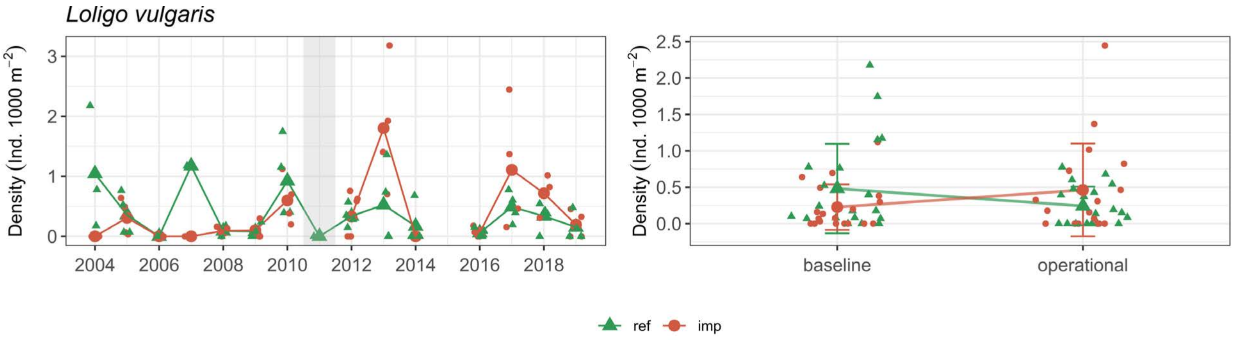

Figure 5. (Left) Time series plots of *Loligo vulgaris* densities in C-Power. (Right) Mean density values (± SD) for *L. vulgaris* in the baseline (2004-2010) and operational years (2014-2019) in C-Power. Construction of the second phase of C-Power was in 2011 indicated with a grey rectangle.

opposite effect could be discerned for the reference samples (0.48 *vs* 0.24 ind./1000 m^2), with low densities overall (fig. 5). When inspecting the time series plot for *L. vulgaris*, very variable densities can be observed with peaks in some years and near-zero density values in other years. However, peaks are mostly occurring simultaneously in both impact and reference areas, except in 2004 and 2007 (only 1 sample) where densities in reference samples were much higher (fig. 5). In operational phase years, a higher density during peaks can be observed for impacted samples compared to reference samples, while the opposite is visible in the baseline years which thus explains why a significant interaction effect was found (fig. 5). A significant time effect was also found for 6 out of the 10 most abundant species, while the impact effect was only significant for *O. albida* (table 3 and figures in annex 1). This impact effect can be explained by a difference in density between reference and impact samples during the baseline years (9.9 *vs* 3.1 ind./1000 m^2), with higher densities found in reference areas in general.

Pagurus bernhardus, *A. rubens* and *O. ophiura* were the three most abundant epibenthic species that were found in Belwind. None of the top 10 epibenthic species showed a significant interaction effect, but for *Sepiola atlantica* a near significant p-value was obtained for the interaction term ($p = 0.05$). This effect is probably detected because of a large peak in density in 2014 in the impact samples, while densities in other years are similar in both impact and reference areas (table 4 and figures in annex 2). For half of the most abundant epibenthic species, a significant time effect was found, while only one species *S. atlantica* showed a significant impact effect.

Focus on two dominant fouling species

For both hard substrate species *Anthozoa* sp. ($p = 0.01$) and *Mytilus edulis* ($p < 0.001$), a wind farm effect was obtained for C-Power (table 3). No or very low densities of these species were found in both reference and impact trawls in the baseline period, while a clear increase in density was observed only in impact samples in the operational period (*Anthozoa* sp.: 0.02 *vs* 4.48 ind./1000 m^2; *M. edulis*: 0.00 *vs* 1.70 ind./1000m^2; fig. 6). In 2017, both species reached a peak in density with a lot of variation between trawls, after which densities declined again. After this decline, *M. edulis* densities in impact samples were only found to be marginally higher than in reference samples. For *Anthozoa* sp., densities in impact samples remained higher than for reference samples during the last two years.

For Belwind, a significant interaction effect could only be detected for *M. edulis* ($p = 0.04$; table 4). For *Anthozoa* sp., the interaction was not significant ($p = 0.19$) as densities also increased in reference samples and are only higher in impact samples in 2018. For both species, the observed wind farm effect seems to appear only after a couple of years after construction (fig. 6).

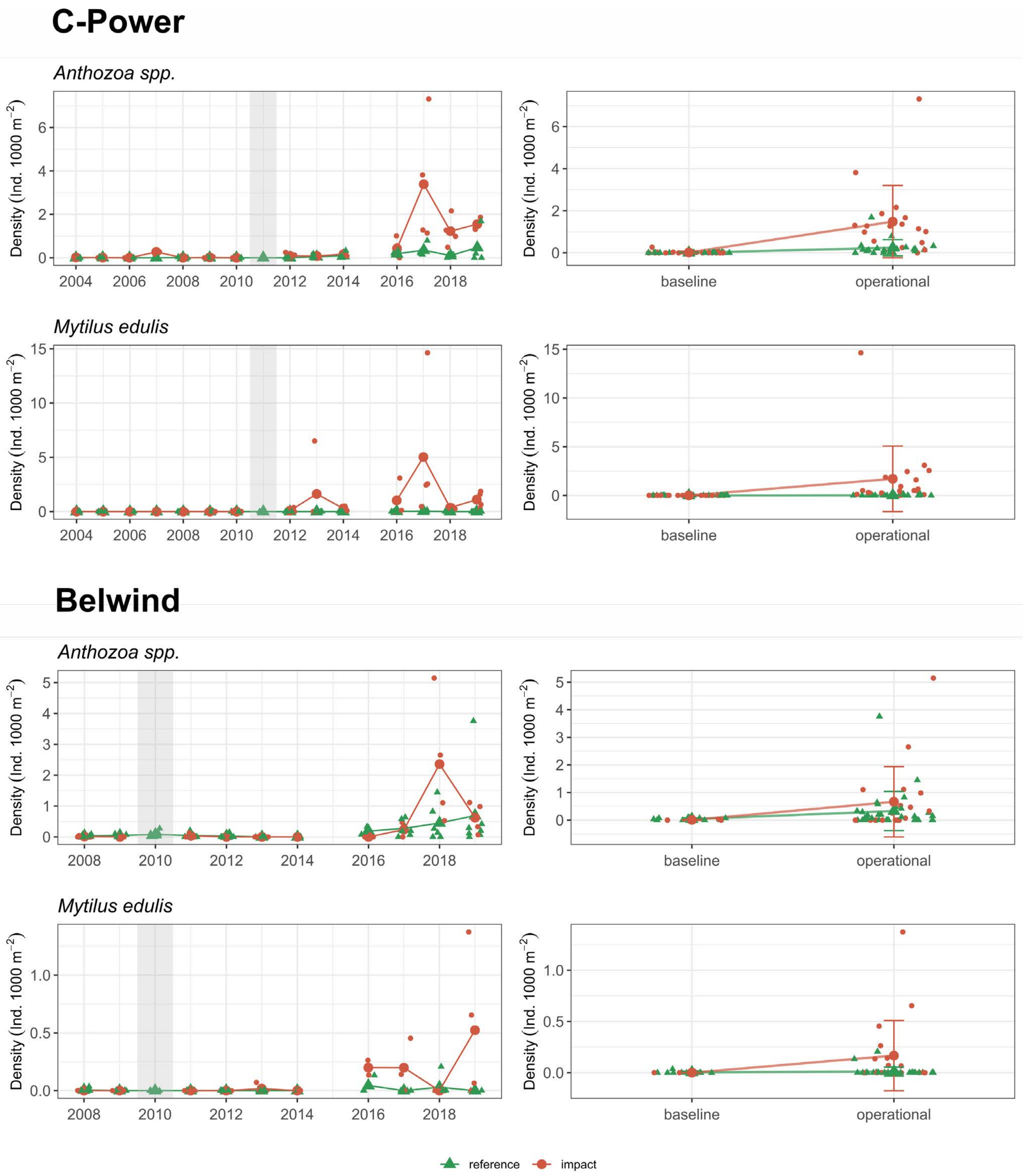

Figure 6. (Left) Time series plots for *Anthozoa* sp. and *M. edulis* densities in both C-Power and Belwind. (Right) Mean density values (± SD) for *Anthozoa* sp. and *Mytilus edulis* in the baseline (C-Power: 2004-2010; Belwind: 2008-2009) and operational years (C-Power: 2014-2019; Belwind: 2013-2019). Grey rectangles indicate construction periods.

Table 3. Overview and results of the GLS models for the ten most abundant epibenthos and fish species together with the two included hard-substrate epibenthic species (light blue) within the C-Power wind farm. The percentage of zero's indicates in how many % of the trawls the species was absent. The different variance structures referred to in the table can be found in the material and methods section. The fourth column indicates if any outliers were removed before fitting the model and how many. The last three columns indicate p-values for all fixed terms of the model obtained from a type III ANOVA test (Chisq). Significant values are indicated in bold ($p < 0.05$).

C-Power epibenthos						
Species	% zero's	Variance structure	Outliers removed	P(Baseline/ Operational)	P(Reference/ Impact)	P(Interaction)
Pagurus bernhardus	0.00	/	yes (1)	**0.009**	0.330	0.320
Ophiura albida	2.56	vf3	no	**0.007**	**0.001**	0.309
Asterias rubens	2.56	vf3	no	**0.002**	0.845	0.614
Ophiura ophiura	1.28	vf3	no	**0.001**	0.378	0.944
Liocarcinus holsatus	6.58	vf1	yes (2)	0.169	0.159	0.566
Sepiola atlantica	19.23	vf3	no	0.432	0.093	0.161
Crangon crangon	21.79	vf3	no	**0.010**	0.262	0.260
Macropodia spp.	32.05	vf3	no	**0.000**	0.407	0.620
Loligo vulgaris	32.05	vf3	no	0.156	0.094	**0.032**
Loligo juv.	43.59	vf3	no	0.124	0.360	0.315
Anthozoa sp.	51.282	vf2	no	**0.003**	0.210	**0.006**
Mytilus edulis	75.641	vf2	no	0.137	**0.033**	**0.028**
C-Power fish						
Species	% zero's	Variance structure	Outliers removed	P(Baseline/ Operational)	P(Reference/ Impact)	P(Interaction)
Arnoglossus laterna	1.30	vf3	yes (1)	**0.007**	0.097	0.395
Limanda limanda	1.32	vf3	yes (2)	**0.034**	**0.015**	0.051
Pleuronectes platessa	1.28	vf3	no	0.376	0.262	**0.010**
Echiichthys vipera	0.00	vf3	no	0.106	0.078	**0.027**
Pomatoschistus	3.85	vf3	no	0.118	0.193	0.183
Callionymus lyra	6.41	vf3	no	**0.005**	**0.003**	**0.002**
Buglossidium luteum	11.54	vf3	no	**0.004**	**0.001**	**0.010**
Mullus surmuletus	14.10	vf3	no	**0.039**	0.192	0.163
Callionymus reticulatus	20.51	vf3	no	**0.003**	0.077	0.072
Hyperoplus lanceolatus	28.21	vf3	no	0.148	0.153	0.146

Table 4. Overview and results of the LMER models for the ten most abundant epibenthos and fish species together with the two included hard substrate epibenthic species (light blue) within the Belwind wind farm. The percentage of zero's indicates in how many % of the trawls the species was absent. The different variance structures referred to in the table can be found in the material and methods section. The fourth column indicates if any outliers were removed before fitting the model and how many. The last three columns indicate p-values for all fixed terms of the model obtained from a type III ANOVA test (Chisq). Significant values are indicated in bold (p < 0.05).

Belwind epibenthos						
Species	% zero's	Variance structure	Outliers removed	P(Baseline/ Operational)	P(Reference/ Impact)	P(Interaction)
Pagurus bernhardus	0.00	/	no	**0.000**	0.612	0.901
Asterias rubens	4.55	vf3	no	**0.004**	0.439	0.419
Ophiura ophiura	7.58	vf2	no	0.296	0.387	0.236
Ophiura albida	10.61	vf3	no	**0.005**	0.191	0.105
Liocarcinus marmoreus	24.62	vf3	yes (1)	0.453	0.961	0.576
Liocarcinus holsatus	27.27	/	no	**0.000**	0.247	0.515
Macropodia	13.64	vf3	no	0.864	0.384	0.724
Spisula solida	15.15	vf3	no	0.833	0.251	0.213
Sepiola atlantica	34.85	vf3	no	0.975	**0.013**	0.053
Loligo juv.	34.85	/	no	**0.001**	0.390	0.330
Anthozoa	45.455	vf	no	0.093	0.101	0.189
Mytilus edulis	83.333	vf	no	0.378	0.898	**0.037**
Belwind fish						
Species	% zero's	Variance structure	Outliers removed	P(Baseline/ Operational)	P(Reference/ Impact)	P(Interaction)
Echiichthys vipera	0.00	vf	no	0.086	0.522	0.252
Pleuronectes platessa	3.03	vf2	no	0.072	0.203	0.170
Arnoglossus laterna	4.55	/	no	0.369	0.525	0.440
Hyperoplus lanceolatus	7.58	vf	no	**0.016**	0.269	0.239
Limanda limanda	16.67	vf2	no	**0.041**	0.965	0.870
Pomatoschistus	16.67	vf3	no	**0.034**	0.994	0.794
Callionymus reticulatus	34.85	vf3	no	**0.000**	0.119	0.089
Ammodytes tobianus	30.77	vf3	yes (1)	**0.006**	0.280	0.677
Mullus surmuletus	27.27	vf2	no	**0.025**	0.061	0.170
Buglossidium luteum	46.97	vf2	no	**0.033**	0.415	0.571

3.2. Fish

3.2.1. Species number, density and biomass

Species number and fish density are very similar over time for impact and reference samples in C-Power (fig. 7). This evidently resulted in an insignificant wind farm effect for both species number ($p = 0.35$) and density ($p = 0.11$; fig. 7).

For Belwind, just as for C-Power, no significant interaction effect was found for species number ($p = 0.14$), nor for fish density ($p = 0.06$). For both variables, a significant time effect was observed with a decrease in both average species number and fish density towards the operational period (fig. 8). When looking at the species list, there was one species for which the appearance seemed an indication of an OWF effect *i.e. Dicentrarchus labrax*, which was absent before construction and now occurred in 20% of the impact samples.

3.2.2. Community structure linked to predictor variables

For C-Power, a significant interaction effect (impact x period, $p_{perm} = 0.004$) was detected. Pairwise tests showed that in the baseline period impact and reference samples differed significantly ($p_{perm} = 0.006$). SIMPER analyses indicated that this was mainly because of density differences in the most common species. For most species, lower average densities were found in the impact samples compared to the reference samples *e.g.* for top 3 contributors to dissimilarity *Callionymus reticulatus* (5.1 *vs* 2.2 ind./1000 m^2), *Callionymus lyra* (3.5 *vs* 0.6 ind./1000 m^2) and *Limanda limanda* (5.5 *vs* 1.3 ind./1000 m^2). In the operational period, reference and impact samples no longer differed significantly ($p_{perm}=0.2$), which is also clear in the PCO plot where they cluster closer together (fig. 9). Pairwise tests also showed significant differences between periods within impact groups ($p_{perm} < 0.0001$ for both). For the reference group, mainly

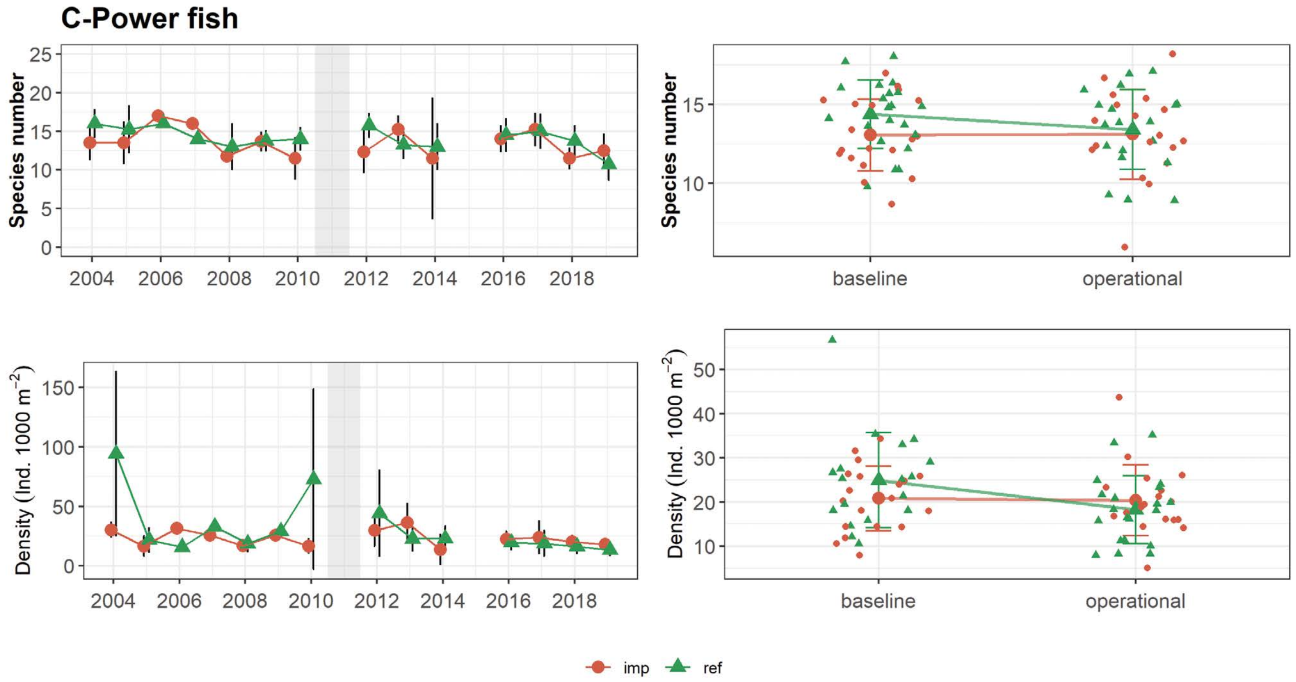

Figure 7. (Left) Time series plots of the univariate variables species number (S), density (N) and biomass for fish at C-Power wind farm. (Right) Mean values (± SD) for baseline (2004-2010) and operational (2014-2019) period allowing to identify offshore wind farm effect. Construction of the second phase of C-Power was in 2011 indicated with a grey rectangle.

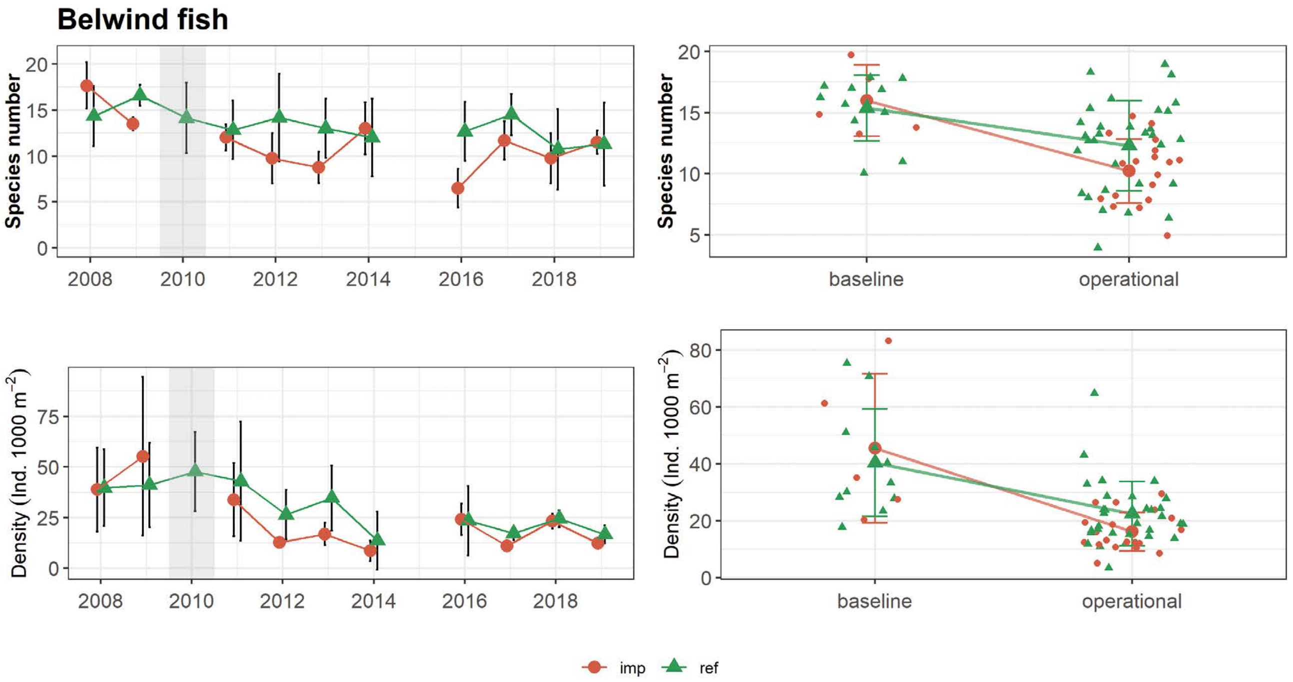

Figure 8. (Left) Time series plots of the univariate variables species number (S), density (N) and biomass for fish at Belwind wind farm. (Right) Mean values (± SD) for baseline (2008-2009) and operational (2012-2019) period allowing to identify offshore wind farm effect. Construction of Belwind was in 2010 indicated with a grey rectangle.

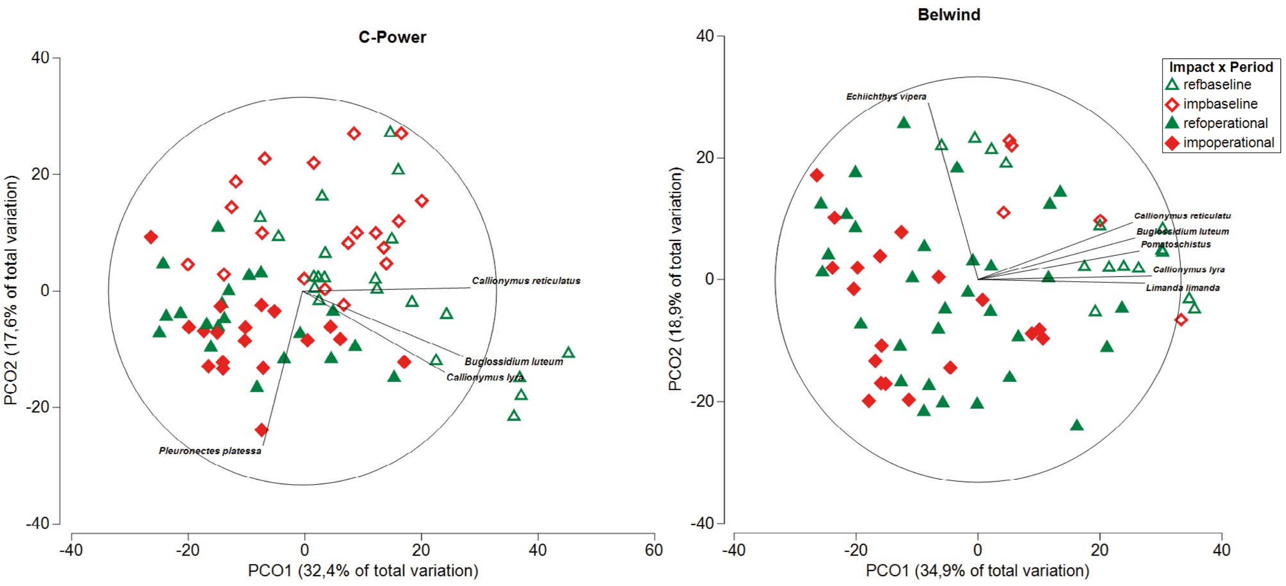

Figure 9. PCO plots based on Bray-Curtis resemblance matrix of square root transformed fish density data for C-Power and Belwind with indication of impact group and sampling period. Vector overlay shows species best correlated with the observed multivariate pattern and is based on Pearson correlation (> 0.75).

Table 5. Proportion of demersal fish community variation that is explained by each individual predictor variable based on marginal DistLM tests for C-Power and Belwind.

Variable	Pseudo-F	p-value	Proportion explained (%)
C-Power			
Chlorophyll	1.121	NS (0.32)	1.5
Temperature	4.855	0.0001	21
AMO	1.459	NS (0.1)	3.7
NAO	3.427	0.0003	8.4
Discharge	3.998	0.0018	5
Impact	7.210	0.0001	22.6
Belwind			
Chlorophyll	6.739	0.0001	9
Temperature	5.570	0.0001	20.2
AMO	2.331	0.0144	6.5
NAO	5.529	0.0001	14.2
Discharge	9.092	0.0001	11.8
Impact	7.289	0.0001	24.9

decreases in common species were observed towards the operational period *e.g. C. reticulatus* (5.1 *vs* 0.1 ind./1000 m²), *C. lyra* (3.5 *vs* 0.6 ind./1000m²) and *L. limanda* (5.5 *vs* 1 ind./1000 m²). While for the impact group, a different pattern is observed with as well a decrease for *C. reticulatus* towards the operational period (2.2 *vs* 0.2 ind./1000 m²), but for most other common species an increase or *status quo* was detected for the operational period with the clearest increase for *Pleuronectes platessa* (0.6 *vs* 2.7 ind./1000 m²). A DistLM analysis investigated the relationship between predictor variables and the observed multivariate pattern. Marginal tests showed that temperature (21%), NAO (8.4%), discharge (5%) and the categorical impact variable (22.6%) individually explained a significant proportion of the total variation in the multivariate data cloud (table 5). The BEST model based on AIC_c criterion included all predictor variables and explained 47.5% of the total variation in the demersal fish community structure.

For Belwind, no significant interaction effect (impact x period, $p_{perm} = 0.7$) was observed, indicating that an effect on the fish community structure on the soft sediments between the turbines by the presence of the OWF could not be demonstrated (fig. 9). The only significant effect was found between baseline and operational period ($p_{perm} < 0.001$). SIMPER indicated this was due to a decrease in densities of the common species in the operational period compared to the baseline period *e.g.* for top 3 contributors to dissimilarity between periods *Echiichthys vipera* (22.1 *vs* 13.7 ind./1000m²), *C. lyra* (1.8 *vs* 0.06 ind./1000 m²) and *Buglossidium luteum* (1.8 *vs* 0.04 ind./1000 m²). Marginal DistLM tests showed that all individual predictor variables explained a significant part of the variation of the observed multivariate pattern (table 5). The individual variables explaining most of the individual variation were temperature (20.2%) and impact (24.9%; table 5). The BEST model based on AIC_c criterion selected temperature, NAO and impact to be included and explained 46.1% of the total variation observed in the fish community structure.

3.2.3. Single species models

For C-Power, the three fish species with the highest presence in the samples were *Arnoglossus laterna, L. limanda* and *P. platessa*. For *B. luteum* ($p = 0.01$), *P. platessa* ($p = 0.01$), *E. vipera* ($p = 0.03$) and *C. lyra* ($p = 0.002$), a significant interaction could be found between the impact factor (RI) and the time factor (BO). Density models of *L. limanda* ($p = 0.05$) and *C. reticulatus* ($p = 0.07$) also showed a near-significant interaction term. For solenette *B. luteum*, a wind farm effect was detected, but this was mainly due to a decrease in density in reference samples over time (3.3 *vs* 0.99 ind./1000 m^2), while the densities in impacted samples remained stable (fig. 10). For common dragonet *C. lyra*, of

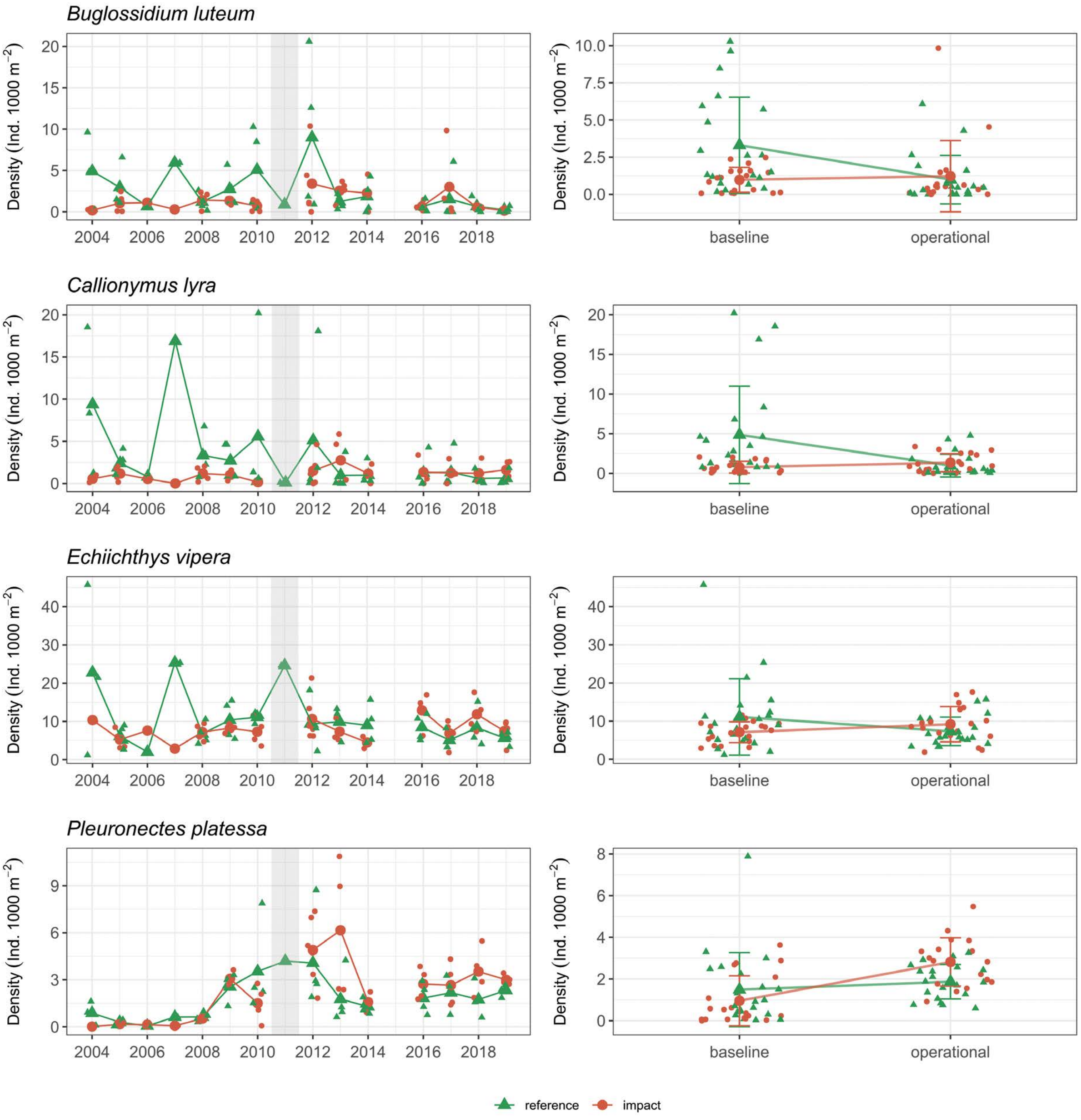

Figure 10. (Left) Time series plots of densities (individuals per 1000 m^2). (Right) Mean density values (± SD) for fish species in C-Power for which a wind farm effect was detected in the baseline (2004-2010) and operational years (2014-2019). Construction of the second phase of C-Power was in 2011 indicated with a grey rectangle.

which its densities also showed a wind farm effect, a similar pattern over time could be discerned. Peaks in densities were similar for both species in 2004, 2007, 2010 and 2012 in reference samples, after which densities stabilised and equalled densities in the impacted areas (fig. 10). A small significant interaction effect ($p = 0.03$) was found for lesser weever *E. vipera* densities in the C-Power wind farm. Densities of this species declined slightly in reference samples, while the opposite trend was observed for impacted samples (fig. 10). The time series plot showed very variable density values from year to year for reference samples, mainly during baseline years and construction, while impact samples showed a steadier course over time. During the last four years of the sampling, density of lesser weever in the impacted areas remained consistent, but marginally, higher than in reference areas (fig. 10). A wind farm effect was also found for plaice *P. platessa* ($p = 0.01$), with higher average densities during operational years in impacted sites compared to the baseline conditions (0.96 *vs* 2.82 ind./1000 m^2). Average densities in reference areas maximised during the construction years of the wind farm, but stabilised again from 2013 onwards. This trend is also visible for impacted areas, but is less clear due to the data gap in 2010. During operational years, densities found in trawls of impacted areas are consistently higher than those found in reference areas, while this was not the case in the baseline period (fig. 10).

Belwind samples had the highest abundances of *E. vipera*, *P. platessa* and *A. laterna*. None of the fish species in Belwind showed a significant wind farm effect. A near significant interaction, however, was obtained for *C. reticulatus* ($p = 0.09$), which can be explained by a steeper decline of densities over time in reference samples compared to impact samples (table 4 and figures in annex 4). For all but three of the 10 most abundant species, a significant time effect was found, which was negative for six out of the seven species (positive for *Mullus surmuletus*). Overall, fish densities seem to have declined over time within this wind farm, but this both for reference and impact area simultaneously (table 4 and figures in annex 4).

4. Discussion

This chapter presents the results of 8 and 9-year post-construction beam trawl monitoring at C-Power (jacket foundations, located 30 km offshore) and Belwind (monopile foundations located 46 km offshore). We specifically focused on the longer-term effects of an established operational OWF on the soft sediment epibenthos and demersal and bentho-pelagic fish in between the turbines (at distance > 200 m), both at assemblage and species level. Therefore, we excluded the first two post-construction years from our analyses in order to exclude the previously observed post-construction ‘overshoot’ effect for some epibenthic species in both wind farms (more pronounced in C-Power), especially visible for *Asterias rubens* and *Pagurus bernhardus* (Derweduwen *et al.* 2016a; De Backer & Hostens 2017; time series graphs in this report).

4.1. Temporal variability, dominant structuring effect for epibenthos and demersal fish

The soft sediment epibenthos and fish assemblages in between the turbines underwent no drastic changes due to the presence of the OWFs at mid/longer term. The species originally inhabiting the sandy bottom are still present and remain dominant in both OWFs. This is in line with other studies *(e.g.* Bergström *et al.* 2013; Stenberg *et al.* 2015), which was to be expected, since changes may take place at different spatial scales and effects are diminishing with distance from the turbine (Dannheim *et al.* 2019, 2020). As such, changes further away from the turbines (> 200 m from the piles) can be presumed to be more subtle. The main observed temporal differences, both at the assemblage and the species level, were often observed in both the impact and reference zones. This showed that the epibenthos and fish assemblages of soft sediments are in first place structured by temporal variability at larger spatial scales, and that this degree of variation is proportionally much larger than the local effect of the presence of hard sub foundations in the OWF areas. For instance, the brown shrimp *C. crangon* was a dominant species in autumn samples at the Thornton Bank before 2011 in both impact and reference locations, but it almost disappeared afterwards. This may be linked to a change in the migration pattern or reproduction cycle in relation to temperature changes (Boddeke 1975; Beukema 1992). Another example is given by reticulated dragonet, *C. reticulatus*, which showed a remarkable decrease over time both at the Bligh Bank and the Thornthon Bank, *i.e.* at a larger spatial scale.

We have introduced environmental predictor variables in our multivariate analyses to explain the observed variation. Generally, around 50% of the variation (except for epibenthos at Bligh Bank) could be explained, which is surprisingly high. Temperature (combination of SST and 1-3 year lagged SST) was one of the important individual variables explaining a large part of the variation, but also the climate indices NAO or AMO were often selected by the models to explain part of the variation. Many studies have shown these variables to be important in structuring biological communities (Harris *et al.* 2014; Ottersen *et al.* 2001). Chlorophyll and daily discharge from the Scheldt contributed much less to the explained variation. These results indicate that temporal variation based on SST and other climate variables are important drivers of the temporal structure in soft sediment epibenthos and fish assemblages.

We also included the categorical variable 'impact' in our analyses, which appeared to be an important explanatory variable as it was selected by all our models. This suggests that the OWF effect might actually be as important as the time aspect. However, due to the binary nature of this categorical variable, also a time factor and a spatial factor (related to the specific location of the beam trawl samples) are encapsulated within this predictor variable, which means that it is actually a combination of time, space and OWF effect. Hence, our primary conclusion remains that the epibenthos and demersal fish assemblages were mainly structured by temporal variability (comparable over larger spatial scales), and that this degree of change is proportionally much larger than the local effect of the present OWFs. Nevertheless, some significant secondary effects could be clearly related to the presence of the OWFs.

4.2. Further expansion of the reef effect?

For C-Power, some obvious OWF effects were discerned in the epibenthic species assemblage, suggesting a further expansion of the reef effect. The number of epibenthic species significantly increased in the impact area (*i.e.* the soft sediments in between the OWF-foundations), mainly due to the appearance of hard-bottom associated species like *Ophiothrix fragilis*, *Pilumnus hirtellus*, *Pisidia longicornis*, *Mytilus edulis* and *Cancer pagurus*. These are all species that were not present in the area before construction of the OWF and now known to occur on the scour protection layer of the turbines (De Mesel *et al.* 2015; Krone *et al.* 2016; Kerkhof *et al.* 2019). For Belwind, some of these hard substrate epibenthic species popped-up as well on the soft sediments, although this was less pronounced. Here, one fish species seabass, *Dicentrarchus labrax*, appeared in the impact samples after construction, while it was absent in the baseline period. This species is known to be attracted to hard substrates (Fabi *et al.* 2004) and has been spotted in schools around the turbines of Belwind (pers. comm. Jan Reubens and own observation).

We found significant density increases of *Mytilus edulis* (in both OWFs) and *Anthozoa* sp. (in C-Power only) in the soft sediments, both species that are fouling on the turbines (De Mesel *et al.* 2013; Kerkhof *et al.* 2019). This was observed for the first time in 2017, where living mussel clumps were found in the beam trawl samples (De Backer & Hostens 2018), and it seems that this pattern continues. Increased densities of both anemones and blue mussels started to appear 5 to 6-year post-construction, coinciding with the *M. edulis-Metridium senile* 'climax' succession stage 6-year post-construction, as described for the hard substrate assemblage on the monopiles in Belwind (Kerkhof *et al.* 2019). For C-Power, only scrape samples from the gravity-based foundations were studied and a climax *Metridium senile* stage was found (Kerkhof *et al.* 2019). No study on the epifouling of the jacket foundations in C-Power is done, but based on the increased *M. edulis* densities in the soft-sediment in this OWF, we can expect that this will be similar to what is described for Belwind. Also other studies showed that jacket-like foundations (oil rigs, gas platforms) exhibit a favourable substrate for blue mussels (Maar *et al.* 2009; Krone *et al.* 2013). These mussel clumps and Anthozoa found in the soft sediments are probably 'knocked off' from the turbines and transported with the currents. Survival chances of *M. edulis* on mobile soft-bottoms at depths of 20 m, with high risk of burial, are probably low (Hutchison *et al.* 2016). Nevertheless, in some macrobenthic soft sediment samples close to the turbines in C-Power (< 50 m), mussel-bed associated communities have been described (Lefaible *et al.* 2019), showing that these species are able to survive (at least for some time) in or on soft sediments. Further, this observation is in line with the so-called mytilusation hypothesis (Krone *et al.* 2013), which predicted that increased mussel biomass at wind farm foundations can produce secondary hard substrate, which may alter the soft-bottom ecosystem. Up until now, densities of mussels at 200 m distance from the turbine are still low and soft-bottom epibenthic species remain dominant, but it is a clear indication that the reef effect is expanding beyond the turbine scale and could thus increase heterogeneity in the soft-bottom sediments at wind farm scale in the (near) future.

Another significant reef effect detected at the species level is the increased density of common squid *Loligo vulgaris* within the C-Power wind farm. *Loligo vulgaris* is a benthic spawner and attaches egg clusters to hard substrata (Hastie *et al.* 2009). Although, densities of *L. vulgaris* greatly varied from year-to-year in the OWF operational period, patterns between reference and impact samples were quite similar over time, but higher densities were observed within the C-Power OWF area. This might be a first indication that *L. vulgaris* uses the jacket foundations for egg deposition. Egg deposition in the North Sea of *L. vulgaris* mainly peaks in late spring/summer depending on water temperature (FAO 2010). Our sampling campaign is late summer/early autumn and mainly juvenile squid are caught, which could originate from the spawning in summer. In an earlier study on squid larvae in C-Power, we did not find an effect (yet) of the OWF (Vandendriessche *et al.* 2016), but as that study was performed in the first years after construction it was probably too early to detect any effects. Partial attraction to gas platforms has been observed previously (Fabi *et al.* 2004). A visual census for egg clusters during the spawning season (*i.e.* late spring/summer) on the jacket foundations would be the best follow-up to confirm or refute this hypothesis.

The above results clearly suggest that with longer time after construction, the reef effect is expanding further into the soft sediments between the turbines, not (yet) replacing the original soft-sediment assemblages but adding slight changes to these communities.

4.3. First signs of fisheries exclusion or increased food availability?

We observed a few significant effects within the soft sediment demersal fish assemblages. These results are in line with Methratta & Dardick (2019), who in a recent review on finfish, observed limited significantly positive effects mainly for species associated with hard bottoms, rather than for soft-bottom-associated species, and larger effects in direct vicinity of the turbines (< 40 m). As our beam trawl samples are located > 200 m from the turbines, with focus on soft-bottom-associated species, it is not surprising that effects were rather limited.

Nonetheless, for C-Power we observed significant effects for four common soft sediment fish species: common dragonet *C. lyra*, lesser weever *E. vipera*, solenette *B. luteum* and plaice *P. platessa*. The first three are small, non-commercial, benthivore species, showing a similar trend: decreasing abundance in the reference area and a *status quo*/slight increase in abundance in the impact area, especially in the later years. This may be a first hint of a 'refugium' effect in between the turbines related to a positive effect of fisheries exclusion on bycatch species. A second explanation for the higher densities of these three fish species may be food availability. In an earlier study, right after construction (Derweduwen *et al.* 2012), fuller stomachs were discerned for these three species in the OWF, indicating that they benefit from the epifouling on the scour protection layer. In a follow-up study, an increased occurrence of *Pisidia longicornis*, a dominant species on the scour protection, was found in the stomachs of lesser weever in the wind farm, indicating a diet change in the OWF (Derweduwen *et al.* 2016b). Most probably, a combination of both increased food availability and fisheries exclusion explains the observed patterns.

The fourth species, for which a significant positive effect with increased abundance in C-Power was observed, is plaice *P. platessa*, a commercial flatfish species. This is a confirmation of earlier indications that plaice was present in higher abundance within the OWF (Vandendriessche *et al.* 2015; De Backer & Hostens 2017). In another study, looking at fishing activities around OWFs in the Belgian North Sea, we found an indication of increased catch rates of plaice around the C-Power wind farm (De Backer & Hostens 2019). All these results are consistent over the years, signalling a 'refugium' effect for plaice between the turbines in the C-Power OWF, that might be an effect of fisheries exclusion. Whether or not in combination with increased food availability requires further research.

For Belwind, no such refugium effect on the demersal fish assemblage has been observed. In general, effects were more pronounced for C-Power than for Belwind. The unbalanced design (with only two years of baseline samples in Belwind) might mask some effects. On the other hand, the long-time series post-construction give a better estimate of the mean population, and time series graphs from both impact and reference areas normally allow to signal potential effects, which was not the case for Belwind. Moreover, the difference for both OWFs is not only found for epibenthos and demersal fish assemblages; also for the soft sediment macrobenthos assemblage the effects are more pronounced in C-Power (Lefaible *et al.* 2019). It remains unclear why this difference between both OWFs exist. It may be related to differences in foundation type, site-specific differences such as distance to shore, depth, hydrographic conditions, sediment or community type. However, differences in surrounding fishing pressure have been noted as well (De Backer & Hostens 2019). The fishing exclusion zone surrounding C-Power has been larger for a longer time period both northwest and southeast, related to the construction of Rentel and Norther. Belwind is surrounded by the Nobelwind concession, but the construction of other neighbouring OWFs only started in 2019. This might lead to a higher fishing effort surrounding Belwind, partially nullifying the refugium effect. The differences do highlight the fact that extrapolation from site-specific OWF effects should be done with care. It also highlights the importance of performing OWF monitoring in different types of wind farms, each with their own specificity.

5. Conclusion

Temporal variation, related to changes in temperature and climate indices, is the main driver structuring epibenthos and demersal fish assemblages, partially masking the potential effect of the presence of OWFs. Nonetheless, we found some clear effects in the soft sediment epibenthic and demersal fish assemblages, which point to an expansion of the OWF effect beyond the immediate vicinity of the turbine, around 8-year post-construction. Effects were subtle but apparent at two levels: (1) an expansion of the reef effect in the soft sediment assemblages, through an increased number of hard substrate-associated species like *Pisidia longicornis*, *Cancer pagurus* and *Dicentrarchus labrax*, and significantly increased abundances of *Mytilus edulis* and *Anthozoa* sp., two species dominating the epifouling communities on the turbines; (2) signals of a refugium effect in C-Power for some common soft sediment-associated fish species (common dragonet *C. lyra*, solenette *B. luteum*, lesser weever *E. vipera* and plaice *P. platessa*) showing higher densities inside the wind farm compared to the reference area, probably related to a combination of fisheries exclusion and increased food availability. More pronounced effects were found for C-Power than for the more offshore Belwind OWF, stressing that effects might be site-specific and that extrapolation of these findings to other OWFs should be done with care.

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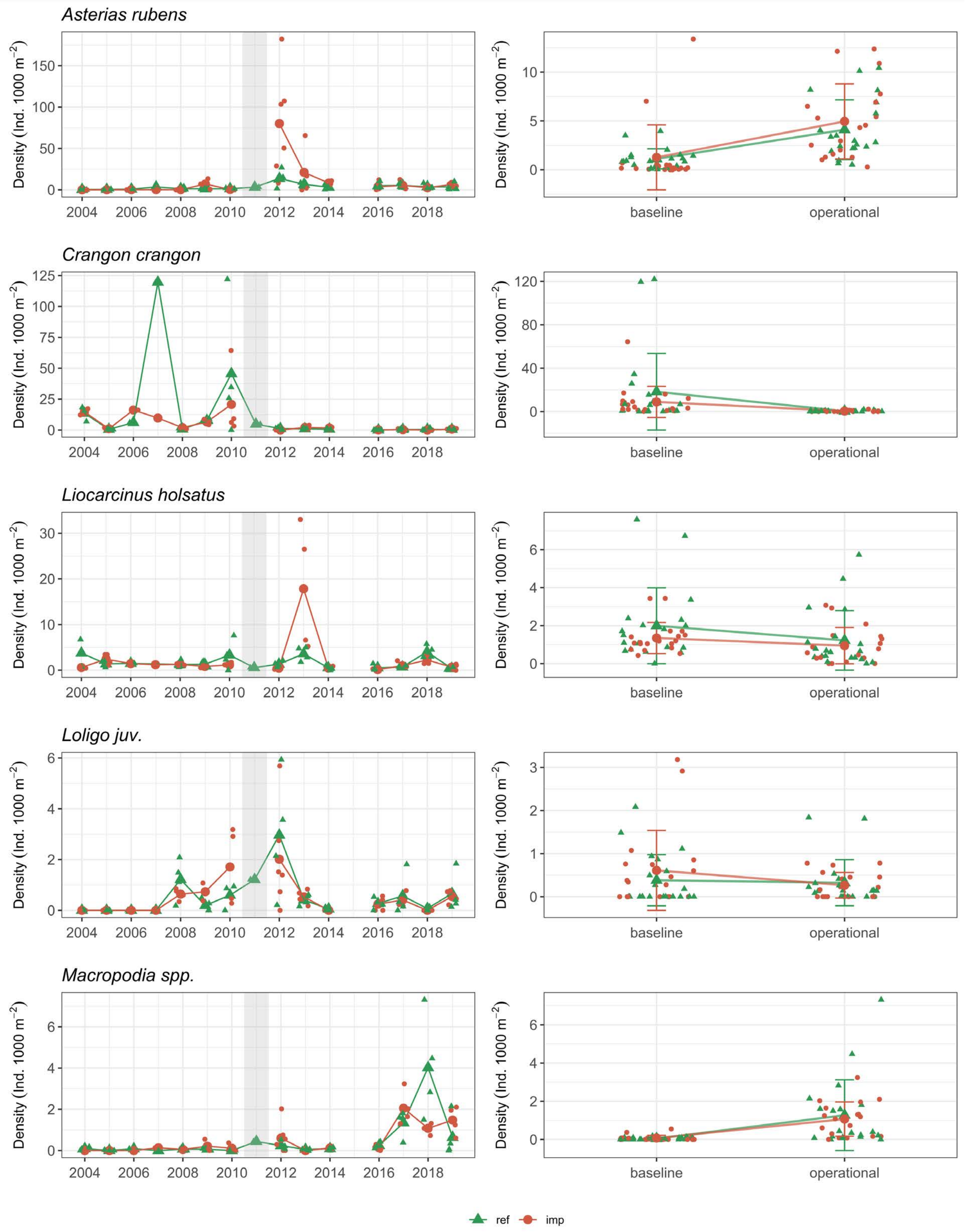
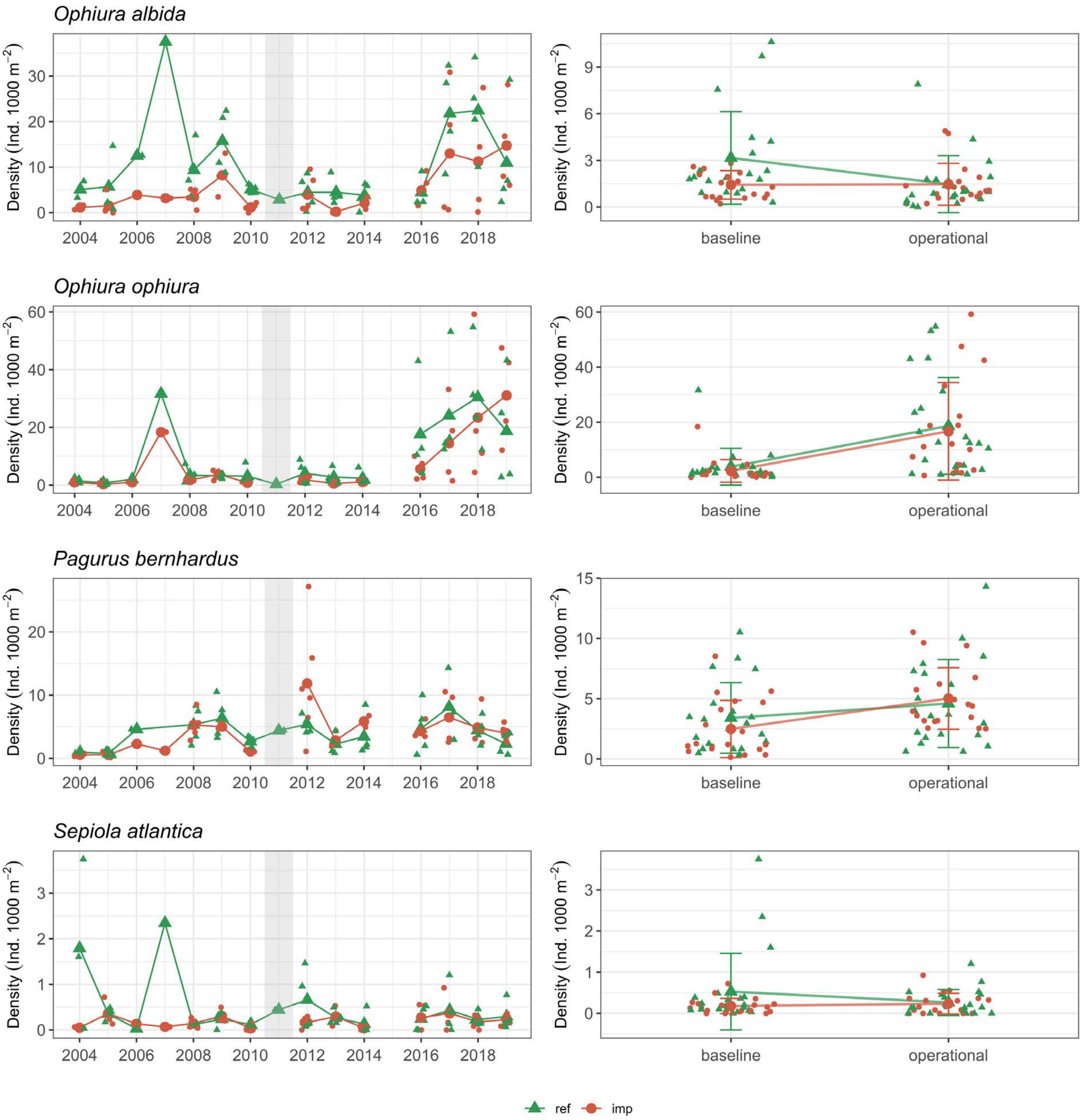
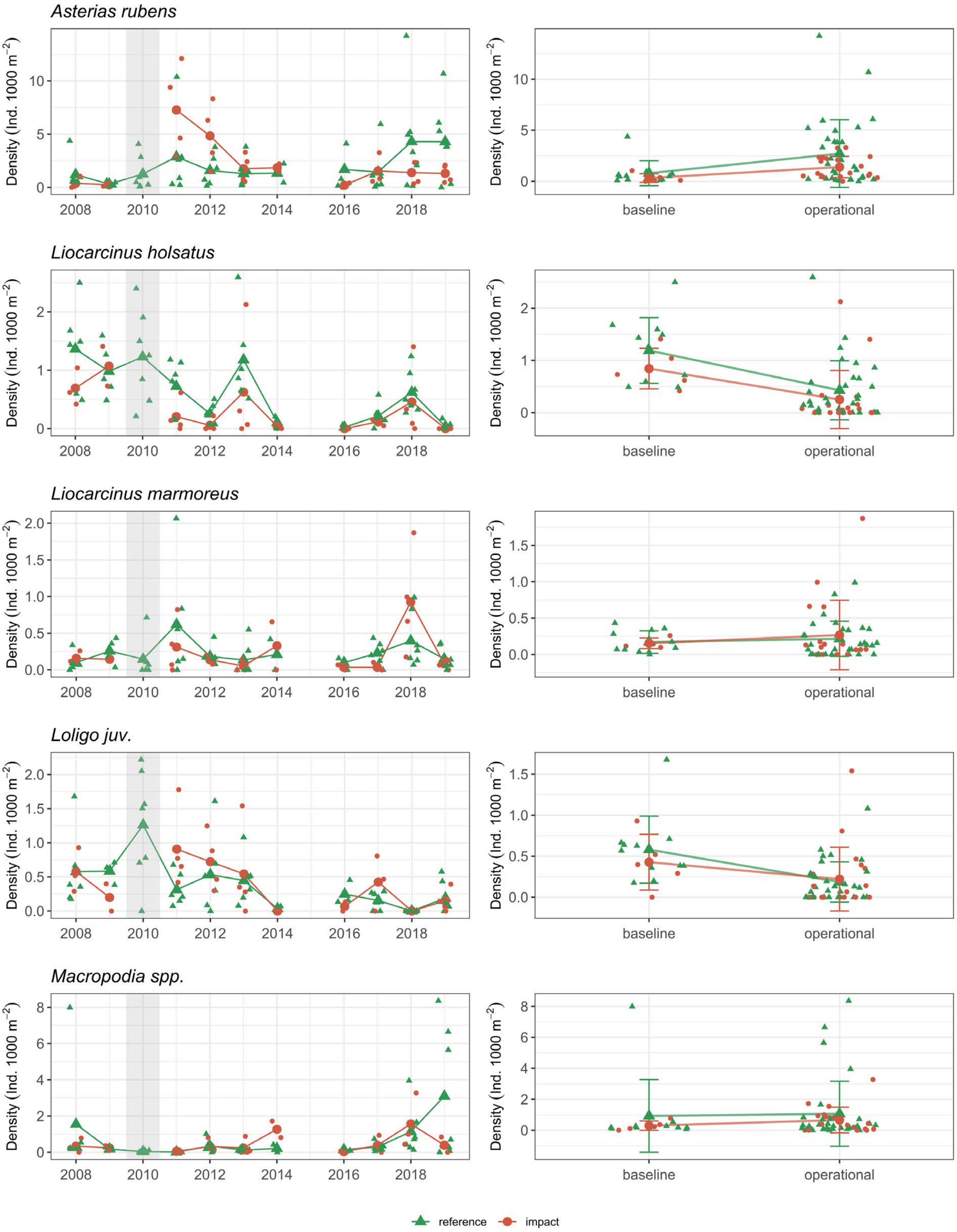
Annex

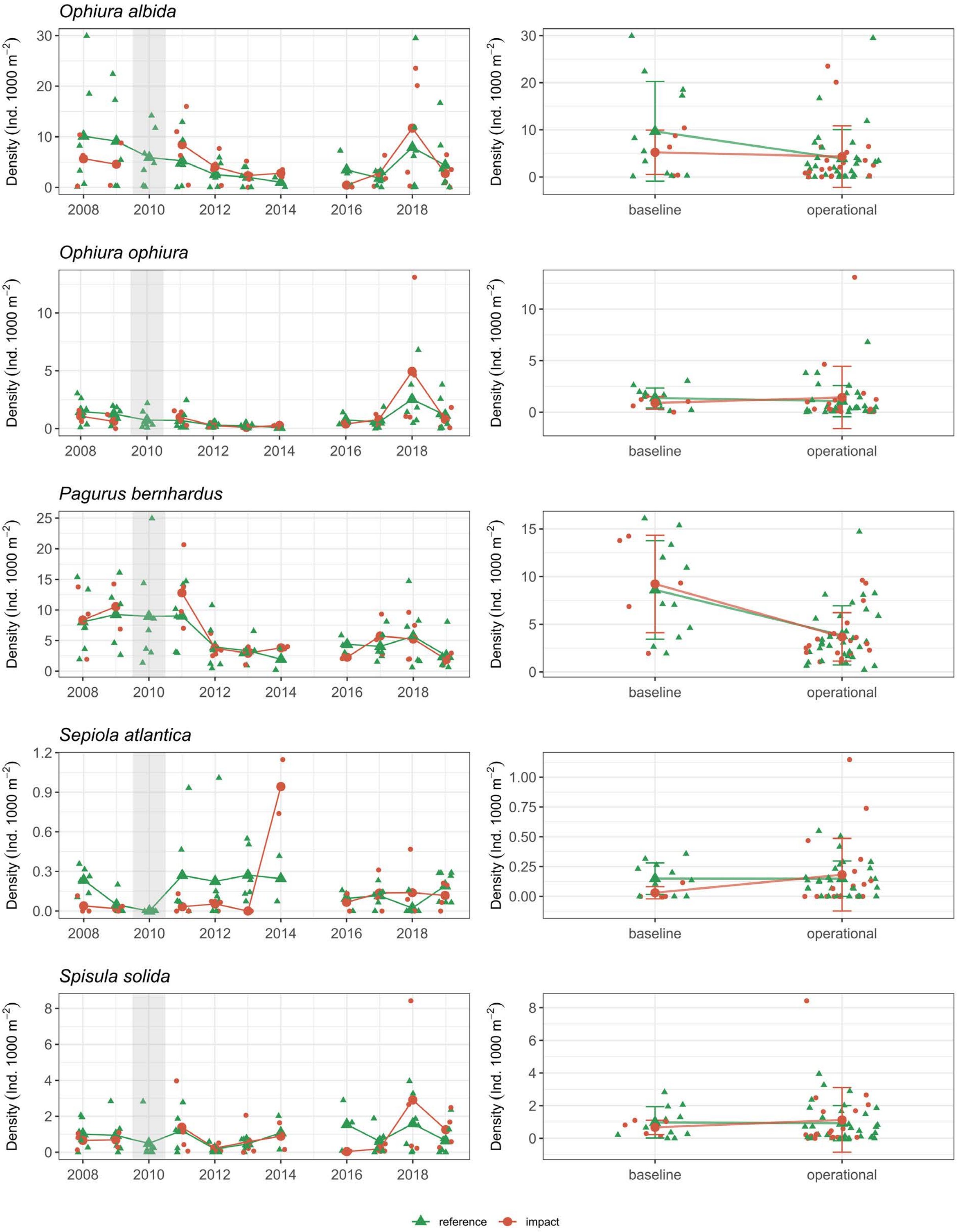
1. Epibenthos C-Power




Ophiura albida
Ophiura ophiura
Pagurus bernhardus
Sepiola atlantica
Density (Ind. 1000 m^{-2})
2004
2006
2008
2010
2012
2014
2016
2018
baseline
operational
ref
imp

2. Epibenthos Belwind

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Ophiura ophiura
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2016
2018
baseline
operational
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10
20
30
5
15
25
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0.3
0.6
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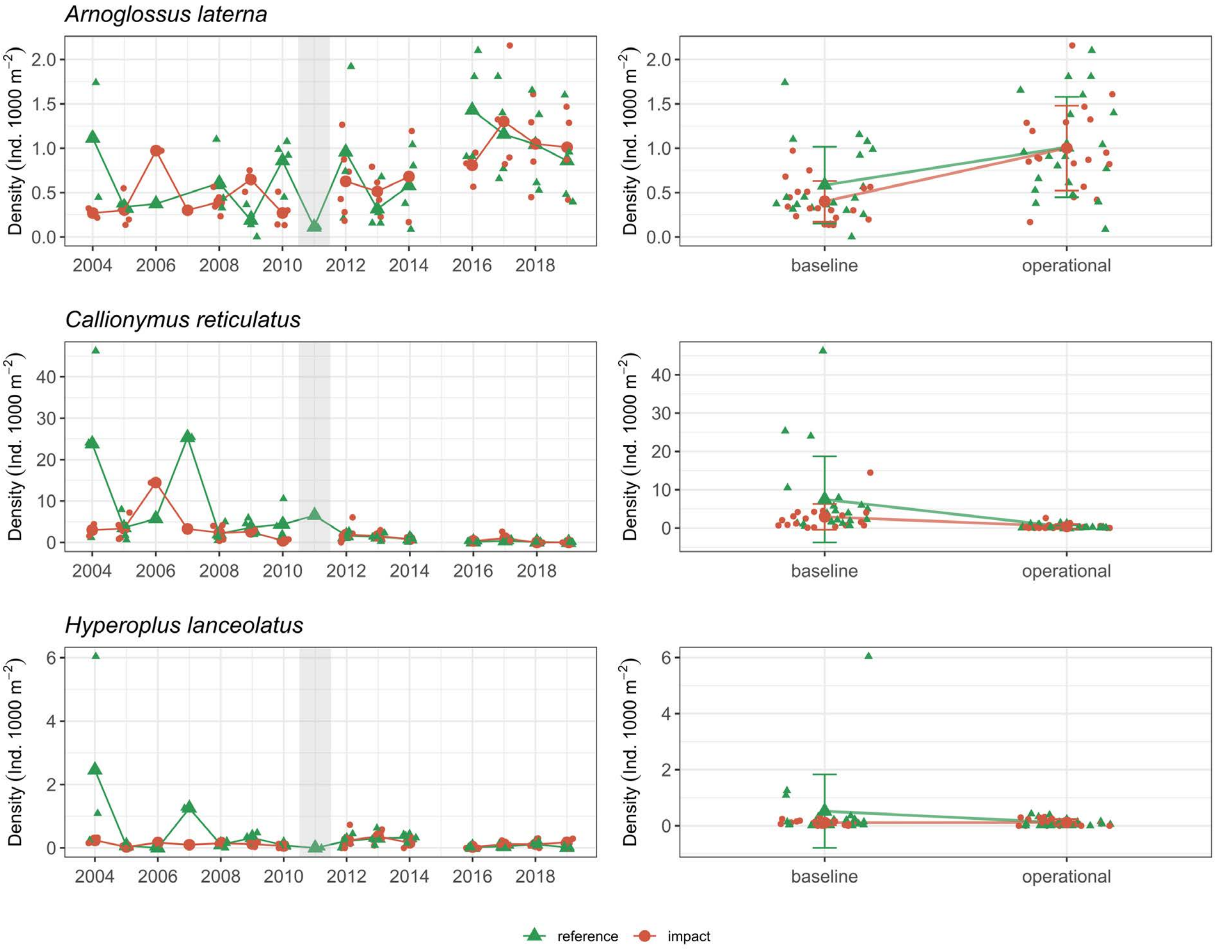
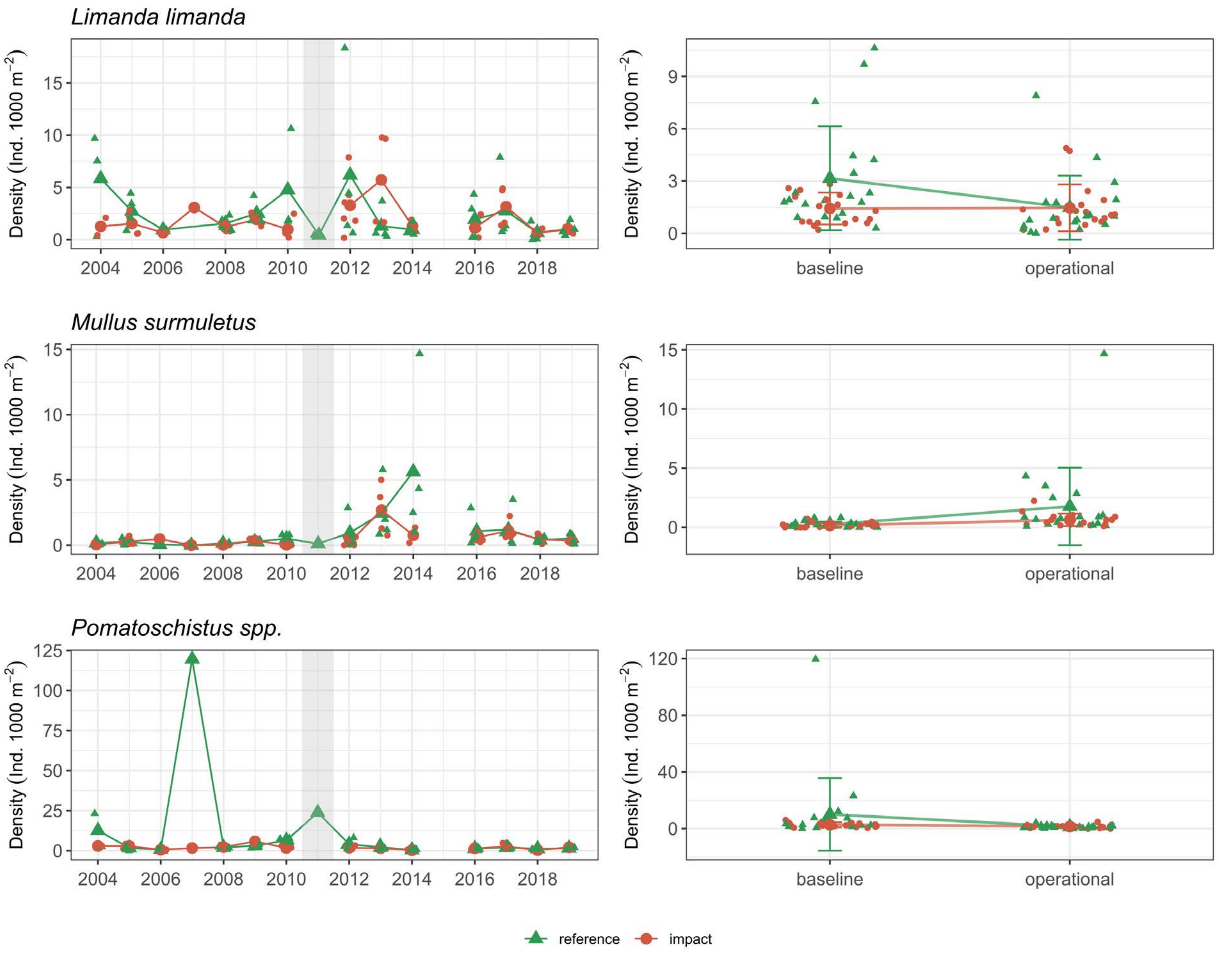
1.2
0.00
0.25
0.50
0.75
1.00
2
4
6
8
reference
impact

3. Fish C-Power




Limanda limanda
Density (Ind. 1000 m^{-2})
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10
15
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2006
2008
2010
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2014
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2018
0
3
6
9
baseline
operational
Mullus surmuletus
Density (Ind. 1000 m^{-2})
0
5
10
15
2004
2006
2008
2010
2012
2014
2016
2018
baseline
operational
Pomatoschistus spp.
Density (Ind. 1000 m^{-2})
0
25
50
75
100
125
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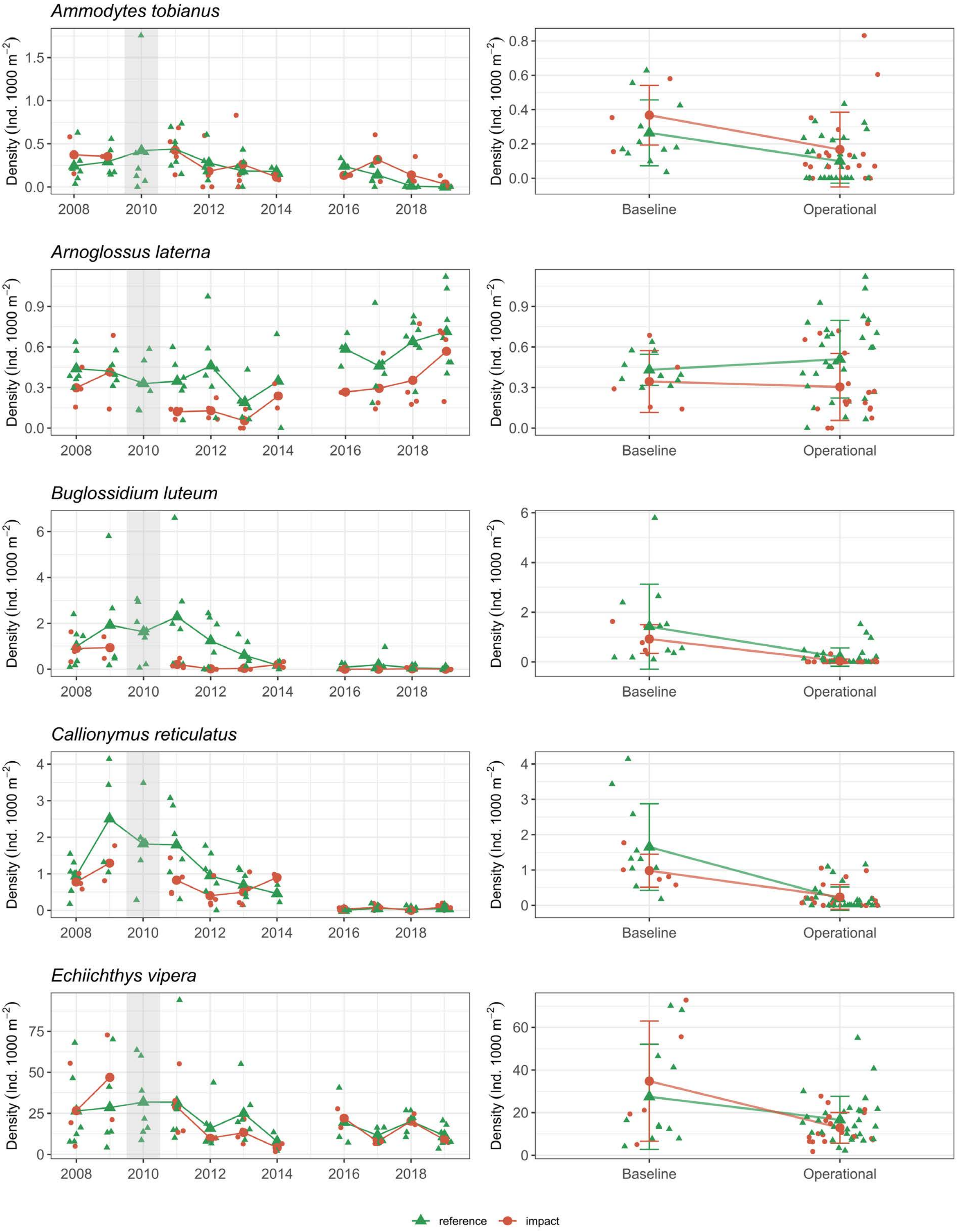
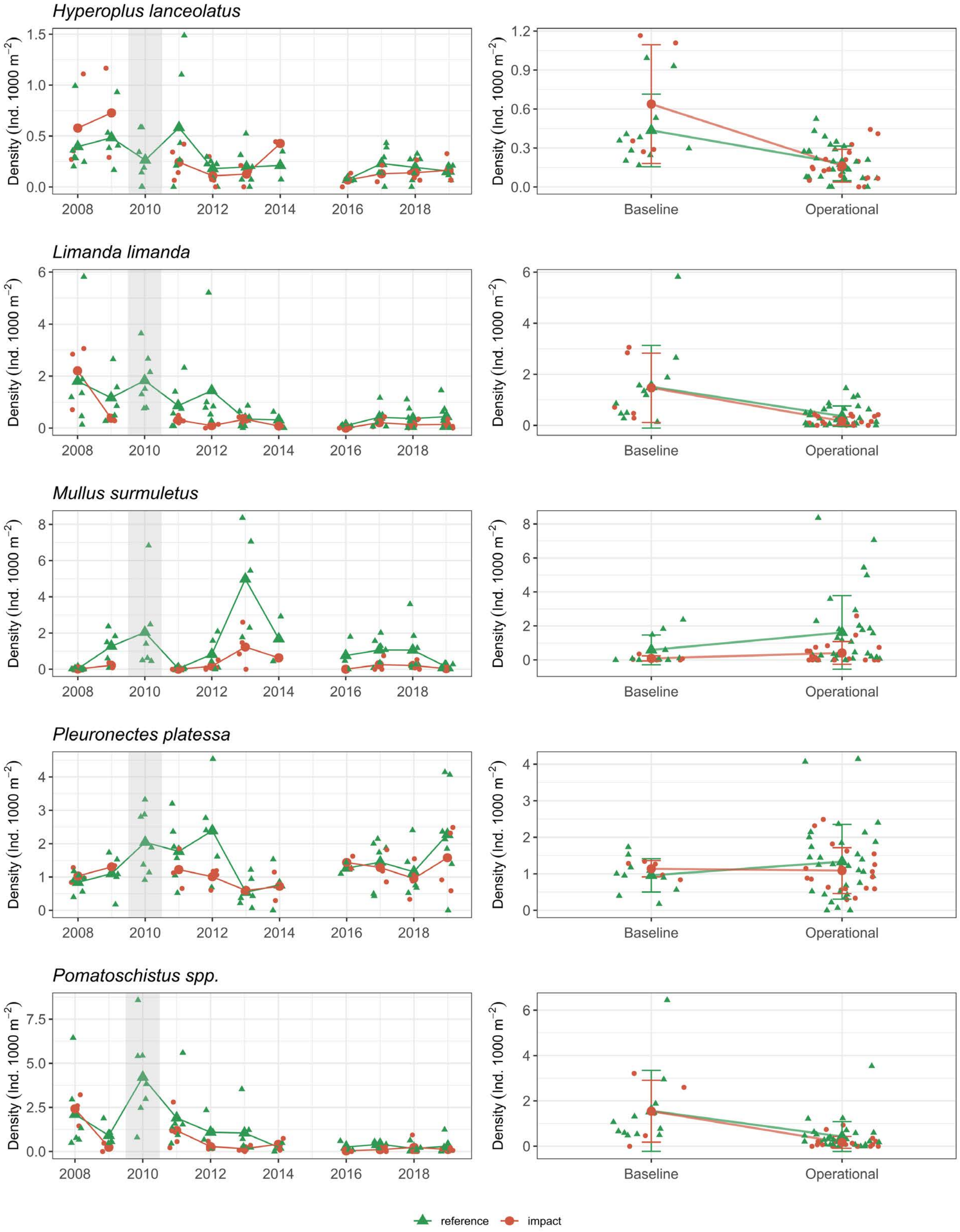
0
40
80
120
baseline
operational
reference
impact

4. Fish Belwind




Hyperoplus lanceolatus
Limanda limanda
Mullus surmuletus
Pleuronectes platessa
Pomatoschistus spp.
Density (Ind. 1000 m^{-2})
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2018
Baseline
Operational
reference
impact

CHAPTER 8

ON THE FOOD-WEB ECOLOGY IN OFFSHORE WIND FARMS AREAS: LESSONS FROM 4 YEARS OF RESEARCH

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Abstract

Offshore wind turbines are heavily colonised by fouling fauna that mainly consists of suspension feeders locally reducing phytoplankton and affecting the very basis of marine food webs. In this study, the effects of offshore wind turbines on the local food web properties were investigated at two levels: (a) detailed food web structure on one gravity-based foundation and (b) effects on primary productivity and fish on a local (Belgian part of the North Sea) scale. Fouling organisms and fish species were collected from a gravity-based foundation, while colonised PVC panels were used to perform an *ex situ* labelling experiment. Overall, the results of this study indicated that offshore wind farms (OWFs) influence the local food web properties, with the occurrence of fouling organisms slightly reducing the local annual primary producers but also being an important resource for organisms of higher trophic levels, *i.e.* fish. Furthermore, the key role of scour protection layers as newly introduced habitats was highlighted, since high food web complexity and provision of a wide range of resources for fouling and fish species was identified in the area. Trophic plasticity and resource partitioning were observed to be the main mechanisms allowing for the co-existence of multiple fouling species along the depth gradient of the gravity-based foundation. Benthic and benthopelagic fish species seem to utilize artificial structures, such as offshore wind turbines, as feeding grounds for a prolonged period, while pelagic fish either exploit them occasionally or not at all as feeding grounds. As the construction of more OWFs might lead to cumulative effects that are hard to predict, further research is needed to completely understand the effects of such structures on marine food webs.

1. Introduction

The offshore wind farm (OWF) industry is rapidly increasing worldwide, with new wind farms being licensed, under construction or planned to be constructed in the near future (Soma *et al.* 2019). Offshore wind turbines induce changes to the marine environment mainly by adding artificial hard substrates into (usually) soft sediment areas. These hard substrates are abundantly colonised by fouling organisms, which mainly consist of suspension feeders. Suspension feeding organisms filter the water column, reducing the phytoplankton densities in and near OWFs (Maar *et al.* 2009; Slavik *et al.* 2019).

Fouling organisms are zoned along zonation patterns along the depth gradient of offshore wind turbine foundations (fig. 1; Lindeboom *et al.* 2011; Krone *et al.* 2013a; De Mesel *et al.* 2015). In the southern North Sea, the intertidal zone comprises typical intertidal species, such as the macroalgae *Ulva* spp. and the midge *Telmatogeton japonicus*. The upper subtidal parts of the turbines are densely populated by the blue mussel *Mytilus edulis* (Krone *et al.* 2013a), creating the *Mytilus* zone. Below this zone, at approximately 8 m depth, there is a zone dominated by the amphipod *Jassa herdmani*, the *Jassa* zone, while at the lower parts of the turbines (*Metridium* zone), the anemone *Metridium senile* is abundantly present (Lindeboom *et al.* 2011; Krone *et al.* 2013a; De Mesel *et al.* 2015). Surrounding the turbines, the scour protection layer (SPL) and the soft sediments also host a rich and OWF-specific community (Langhamer 2012). Soft sediments near the turbine foundations exhibit increased abundance and species richness (Coates *et al.* 2014; Lefaible *et al.* 2018, 2019), which can be explained by the deposition of faecal pellets and detritus by fouling organisms (Coates *et al.* 2014).

The high macrofaunal species diversity within the fouling and natural (soft substrate) macrofaunal assemblages occurring on and close to offshore wind turbines leads to questions about the mechanisms that allow for their co-existence. Species co-existence and community structure are largely dependent on the ability of species to divide/share the available resources (Tilman 1982). In hard substrate communities, space has long been considered as the primary limiting resource (Buss & Jackson 1981; Ferguson *et al.* 2013; Dubois & Colombo 2014). However, food resource limitation may be another important factor affecting species co-existence (Buss & Jackson 1981; Dubois & Colombo 2014; Cresson *et al.* 2016). Trophic competition is only relevant when species with similar feeding preferences co-occur under limiting food conditions (López-Jamar *et al.* 1984; Cresson *et al.* 2016). Trophic competition can be reduced when co-existing organisms can exhibit trophic plasticity and/or resource partitioning (Lefebvre *et al.* 2009; Riera 2009). Trophic plasticity is the adaptation of an organism to exploit multiple resources (Lefebvre *et al.* 2009). Resource partitioning is the ability of organisms to divided (usually limited) resources (Ross 1986). Trophic plasticity may partly allow for resource partitioning (Ashton *et al.* 2010), although this does not necessarily mean that an organism exhibiting trophic plasticity will also partition the resources. Both of these mechanisms allow for minimizing trophic competition.

Apart from invertebrate organisms, the introduction of OWFs also influences vertebrates such as fish. The enhanced food availability (*i.e.* fouling fauna; Leitão *et al.* 2007; Reubens *et al.* 2011) and/or the increased structural complexity (*i.e.* provision of shelter against predators and currents; Bohnsack 1989) attract a variety of fish species. These can be categorised according to their ecology into: (a) benthic, living exclusively on and near the seafloor, *i.e.* sculpin (*Myoxocephalus scorpioides*; Gordon & Duncan 1985), (b) benthopelagic, living in close association with the bottom of the sea but capable of moving to the upper parts of the water column, *i.e.* cod (*Gadus morhua*) and pouting (*Trisopterus luscus*; Gordon & Duncan 1985), and (c) pelagic, occupying mid-water and surface water levels and being able to perform diel vertical migrations, *i.e.* horse mackerel (*Trachurus trachurus*) and mackerel (*Scomber scombrus*; Dale & Kaartvedt 2000). However, we now start

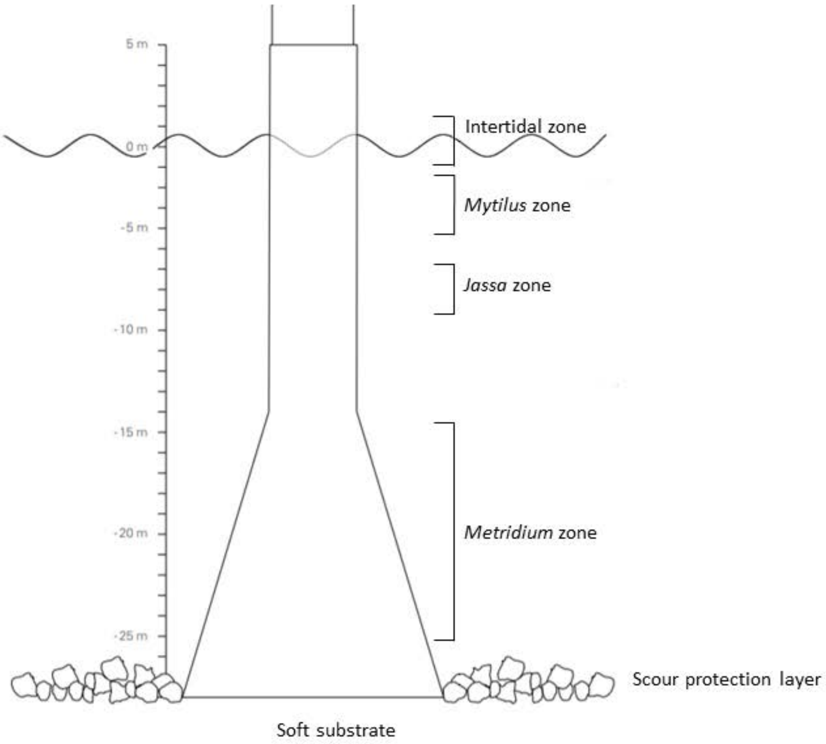

Figure 1. Vertical zonation patterns as formed by the fouling species and the different structures along the depth gradient of an offshore gravity-based wind turbine foundation (modified by the source: De Mesel *et al.* 2013).

getting insights into whether these species exploit artificial structures, such as OWFs, as feeding grounds for a prolonged period, and thus, whether they are attracted towards these installations by the unlimited food availability.

From the above, it is clear that the introduction of offshore wind turbines influences a variety of marine organisms, raising concerns about their impacts on marine ecosystems and ecosystem functioning (Lindeboom *et al.* 2011; Bailey *et al.* 2014). A food web study can provide a rigorous understanding of the ecological processes (Fry 2006) occurring on and near the offshore wind turbines.

This chapter overviews a series of studies (Mavraki 2020; Mavraki *et al.* 2020a, 2020b, 2020c, subm.) with the aim of investigating the impact of OWFs on the local food web properties, examining different aspects of the trophic niches of fouling communities, as well as a selection of macrofaunal and fish species. Trophic niches represent the overall trophic role of a community/species (Leibold 1995). The food web properties are examined both at a detailed level and at a larger scale, with each spatial scale answering different research questions. The detailed food web studies focused on the following questions:

- Are the differences in community composition also reflected in the food web structure?
- Is resource partitioning a mechanism promoting the co-existence of multiple species in the same community?
- Do species that occur in multiple depth zones exhibit trophic plasticity?

The research questions targeting the larger scale were:

- How much carbon is assimilated and grazed upon by fouling organisms?
- Do fish species that are attracted towards artificial reefs, such as OWFs, exploit them as feeding grounds for a prolonged period?

This paper is extracted from a doctoral dissertation and this report is its executive summary (Mavraki 2020). This PhD research was part of the project 'Functional biodiversity in a changing sedimentary environment: implication for biogeochemistry and food webs in a managerial setting', also known as FaCE-It. To highlight the impacts of OWFs on the food web structure along the depth gradient, insights of the food web properties of fouling communities from different zones along the depth gradient of an offshore wind turbine were investigated. Furthermore, the feeding habits of a selection of invertebrate species that occur in multiple zones along the depth gradient were examined to investigate whether trophic plasticity is the key mechanism allowing for their wide vertical distribution and survival in a competitive environment. Once trophic plasticity was confirmed, it was crucial to investigate which organisms were processing the highest amount of organic matter in the area, and thus, which species play a pivotal role in the reduction of primary producers. Finally, the high densities of fouling organisms on OWFs attract mobile predators, such as fish. Investigating whether fish exploit OWFs as feeding grounds for a prolonged period could indicate whether they remain in the area for a long time.

2. Material and methods

2.1. Detailed food-web studies

Fouling fauna samples were collected along the entire depth gradient of a gravity-based wind turbine (D6, coordinates: 51°33.04'N - 02°55.42'E) in the C-Power OWF in the BPNS. C-Power was the first OWF constructed in BPNS, with the installation of six gravity-based turbine foundations in 2008 (Degraer *et al.* 2010). The location and wind turbine were selected based on the wealth of data on the local natural (Coates *et al.* 2014) and fouling macrofauna (De Mesel *et al.* 2015), and on fish (Reubens *et al.* 2011). Furthermore, the age of the turbine foundation assured for the collection of communities of advanced successional stages.

Macrofaunal organisms living in the different depth zones (*i.e.* intertidal, *Mytilus*, *Jassa*, *Metridium*, SPL and soft substrate zones – see above), as well as benthopelagic, benthic and pelagic fish species near the turbine were collected. These organisms were processed for carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) stable isotopes to acquire the isotopic niches of the communities occurring in different depths, as well as of the consumer species occupying multiple depths. Isotopic niches provide information on the resource utilisation and the trophic level of species and are therefore, an approximation of their trophic niches (Newsome *et al.* 2007; Jackson *et al.* 2011). By analysing the isotopic niches of these organisms, we were able to examine the food web complexity along the depth gradient and investigate whether resource partitioning (measured as trophic diversity and redundancy) is a mechanism allowing for the co-existence of fouling fauna on turbine foundations.

At a next step, the contribution of resources in the diet of seven invertebrate species (*Diadumene cincta*, *Jassa herdmani*, *Metridium senile*, *Mytilus edulis*, *Necora puber*, *Ophiothrix fragilis* and *Pisidia longicornis*) occurring in multiple depths were examined using carbon and nitrogen stable isotopes. Differences of the isotopic niches of the same consumer species across depth zones (*i.e.* trophic plasticity) would indicate that these species are trophic generalists, capable of switching to other resources depending on their location, and hence, possibly showing a wide distribution and survival in a highly competitive environment.

2.2. Larger scale food web studies

A tripod with attached PVC panels (fig. 2) was deployed for one year within the C-Power OWF area to facilitate colonisation and allow for ex-situ community experiments. The colonised plates were incubated

Figure 2. The tripod with attached PVC panels to allow for colonisation by fouling organisms.

in experimental tanks, where ^{13}C-labelled lyophilised algal cells were provided. The plates were kept in the experimental tanks for three days to allow the organisms to feed. After that period, all the organisms were scraped off the PVC plates and analysed for δ^{13}C stable isotope to track the labelled carbon in their tissues. We calculated the carbon assimilation by fouling organisms and then we upscaled these results to the total number of offshore wind turbines in the BPNS to estimate the amount of primary producer standing stock that is grazed upon by these organisms. Moreover, we compared the amount of carbon that is not assimilated by the natural soft sediment macrofauna due to the construction and colonisation by fouling fauna of the three different types of foundations (monopiles, jackets and gravity-based foundations) in the BPNS. As such, we estimated the ratios turbine/sediment to evaluate the additional amount of carbon that is assimilated because of the presence of fouling fauna.

Finally, combined stomach content and stable isotope analyses on five fish species (benthic: sculpin; pelagic: mackerel and horse mackerel; and benthopelagic: pouting and juvenile cod) collected near the gravity-based foundation were conducted to investigate whether they utilise OWFs as feeding grounds for a prolonged period of time. Stomach content analysis (SCA) provides an indication of the short-term dietary preferences, while stable isotope analysis (SIA) serves as an estimation of the time-integrated diet of the fish species under study. Three initial assumptions (fig. 3) were made: (a) fish utilise artificial hard substrates as feeding grounds for

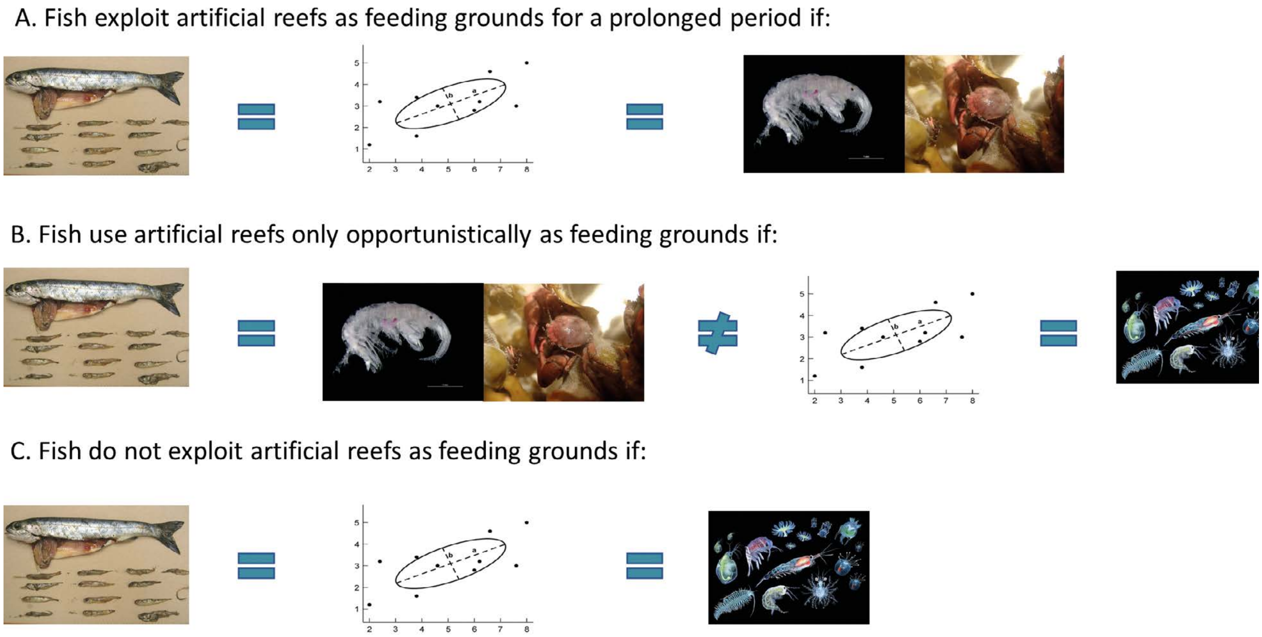

Figure 3. Schematic overview of the three initial assumptions for the fish dietary estimations. Ellipses represent the stable isotope analysis. The amphipod *Jassa herdmani* and the crab *Pisidia longicornis* (source for the images: H. Hillewaert) represent the artificial hard substrate associated fauna, while zooplankton is the pelagic food source.

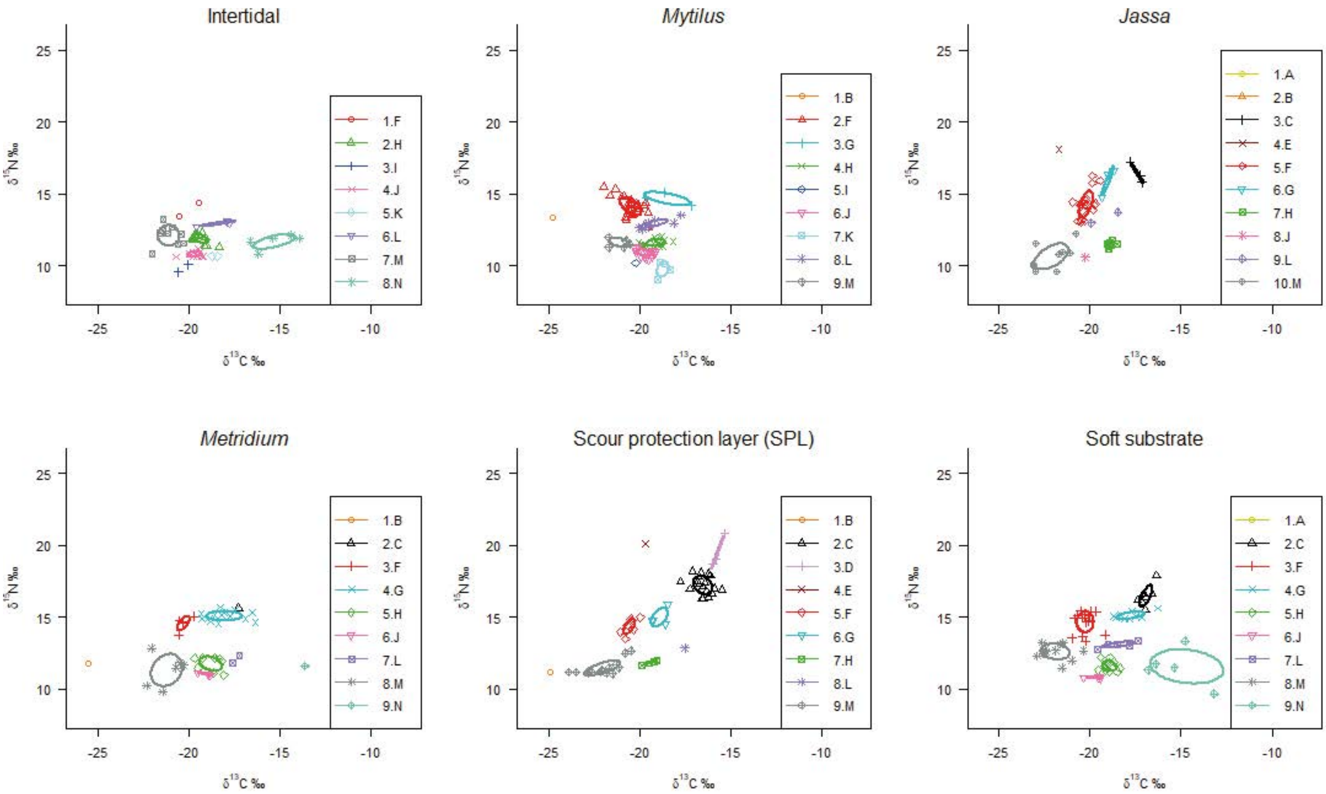

Figure 4. Trophic niches of the different groups of organisms samples at the different depth zones.

a prolonged period when both the short-term and the time integrated analyses show similar results and the fish's diets rely on fouling fauna that occurs only on hard substrates in the North Sea; (b) fish species occasionally use the artificial habitats as feeding grounds if the two analyses show contradicting results, with SCA reflecting a diet based on fouling fauna and SIA showing a diet composition on food items non-related to artificial hard substrates; and (c) fish probably do not exploit the artificial habitats as feeding grounds if both analyses reveal a diet based on food items that are not associated with artificial hard substrates. If the first assumption is confirmed, then fish species remain in the area for a prolonged time to feed, which could lead into better fitness conditions and result in increased production.

3. Results

3.1. Detailed level: community-scale

The $\delta^{13}C$ and $\delta^{15}N$ values of organisms obtained from all zones along the depth gradient indicated that the communities at the soft substrate and SPL contained organisms with the most dissimilar isotopic signatures (fig. 4). Trophic diversity and redundancy also differed among the zones. The largest trophic diversity and redundancy were found for the organisms found in the *Metridium* zone and the soft substrate. In contrast, the organisms occurring in the *Mytilus* zone had the lowest trophic diversity and redundancy.

The isotopic niche sizes differed between the different depth zones (table 1). The smallest isotopic niche was observed in the *Mytilus* zone (4.14‰^{2}), while the largest was observed for the soft substrate (12.63‰^{2}), followed by the SPL (9.54‰^{2}). The isotopic niche of the community at the soft substrate mainly reflected a broader range of $\delta^{13}C$ values and the isotopic niche of the SPL community showed the highest spread across $\delta^{15}N$ values.

3.2. Detailed level: species-specific study

The species-specific stable isotope study revealed depth-dependent differences in resource utilisation of species occurring at multiple depths (fig. 5). *Diadumene cincta* was the only species that showed stable dietary preferences among the different zones it occupied, mainly consuming suspended particulate organic matter. *Necora puber* was the only species that utilised roughly equal shares of multiple resources in every depth zone. All the other consumer species indicated high variability in their diets when occurring in different zones.

3.3. Carbon assimilation and grazing

The total carbon assimilation in the experimental tanks differed significantly among the fouling species (fig. 6). The local population of *Jassa herdmani* showed the highest total carbon assimilation, followed by *Mytilus edulis* and *Monocorophium acherusicum*. All the other species assimilated a significantly smaller amount of carbon.

Table 1. Isotopic niche size (‰^{2}) of the six sampling zones along the depth gradient of the gravity-based foundation

Zone	Isotopic niche size (‰^{2})
Intertidal	4.31
Mytilus	4.14
Jassa	8.09
Metridium	7.53
Scour protection layer	9.54
Soft substrate	12.63

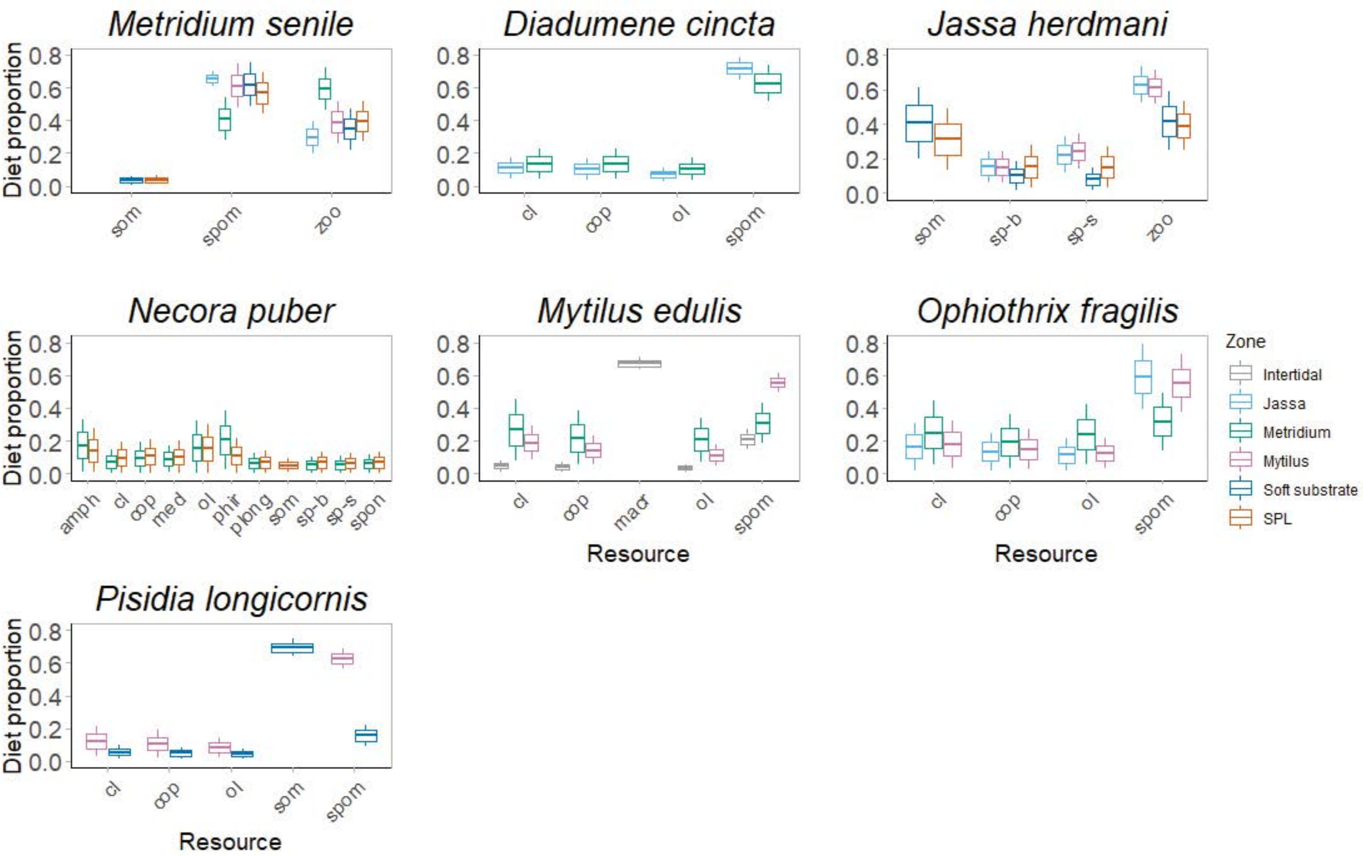

Figure 5. Contribution of the different resources to the diet of the seven invertebrate specie in the different zones. The different colours represent the zones. The coloured boxplots represent the 25th percentile, the median and the 75th percentile of the dietary proportions, while the whiskers represent the 1.5*Interquartile ranges (IQR) of the upper and lower quartile.

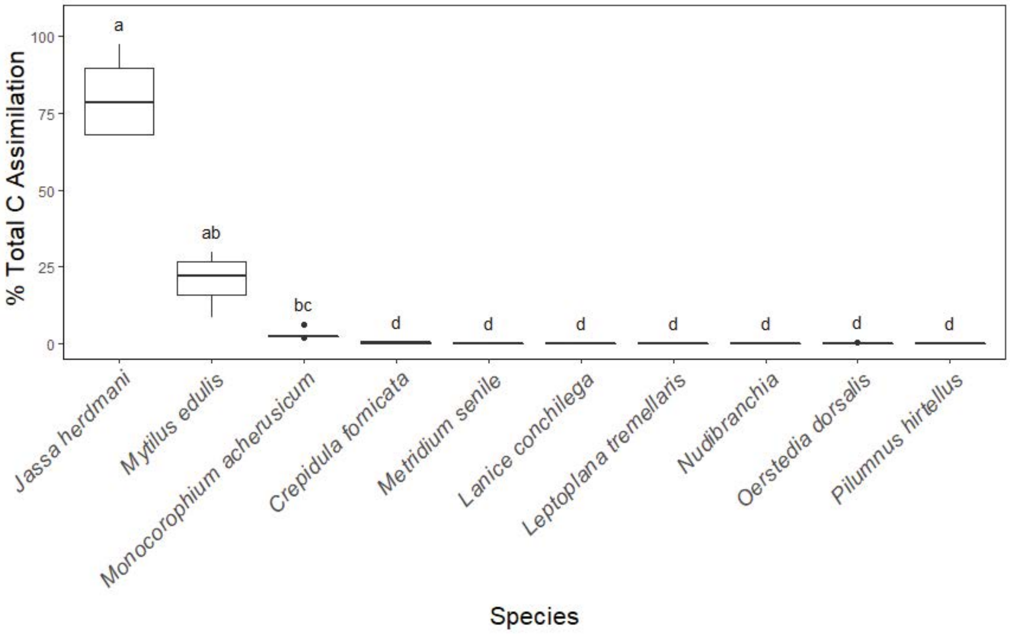

Figure 6. Total C assimilation (expressed as % of amount of added algae) of the different taxa found in the experimental tanks. The boxplots represent the 25th, median and 75th percentile and the whiskers show the 1.5*Interquartile ranges (IQR) of the upper and lower quantile.

The results of carbon assimilation by *J. herdmani* and *M. edulis* were upscaled to the total number of offshore wind turbines currently installed in the BPNS. The upscaling calculations indicated that these two species can assimilate up to 0.4% of the annual local primary producers in their tissues, while 1.3% of the primary producers in the BPNS is grazed upon by *J. herdmani* and *M. edulis*. Furthermore, the different types of foundations and the natural soft substrates were compared in terms of carbon assimilation. Monopiles and gravity-based foundations have large footprints (573 m^2 and 2227 m^2, respectively; Rumes *et al.* 2013), since their bases are surrounded by the rocks of SPLs. In contrast, jacket foundations have small footprints (10.5 m^2; Rumes *et al.* 2013), and they have higher structural complexity compared to monopiles and gravity-based foundations, since they consist of tubulars with multiple orientations (Krone *et al.* 2013b; Causon & Gill 2018). The comparison of carbon assimilation between the three different foundations types colonised by *J. herdmani* and *M. edulis* and the natural soft sediment macrofauna inhabiting the same surface area as the footprints showed that the introduction of jacket foundations and their subsequent colonisation cause the highest increase in carbon assimilation (ratio turbine/sediment: min: 14242 – max: 181259). The presence of monopiles results in the second-highest increase in carbon assimilation with the ratio turbine/sediment ranging from 38 to 485. Finally, the introduction of gravity-based foundations and their subsequent colonisation causes the smallest increase in carbon assimilation (ratio turbine/sediment: min: 9 – max: 116).

3.4. Fish feeding preferences

The SCA results indicated that the stomachs of pouting, cod and horse mackerel mainly contained the fouling amphipod *Jassa herdmani*, the stomach contents of sculpins contained both the crab *Pisidia longicornis* and fish, while the stomach contents of mackerels were dominated by zooplankton (fig. 7). SIA

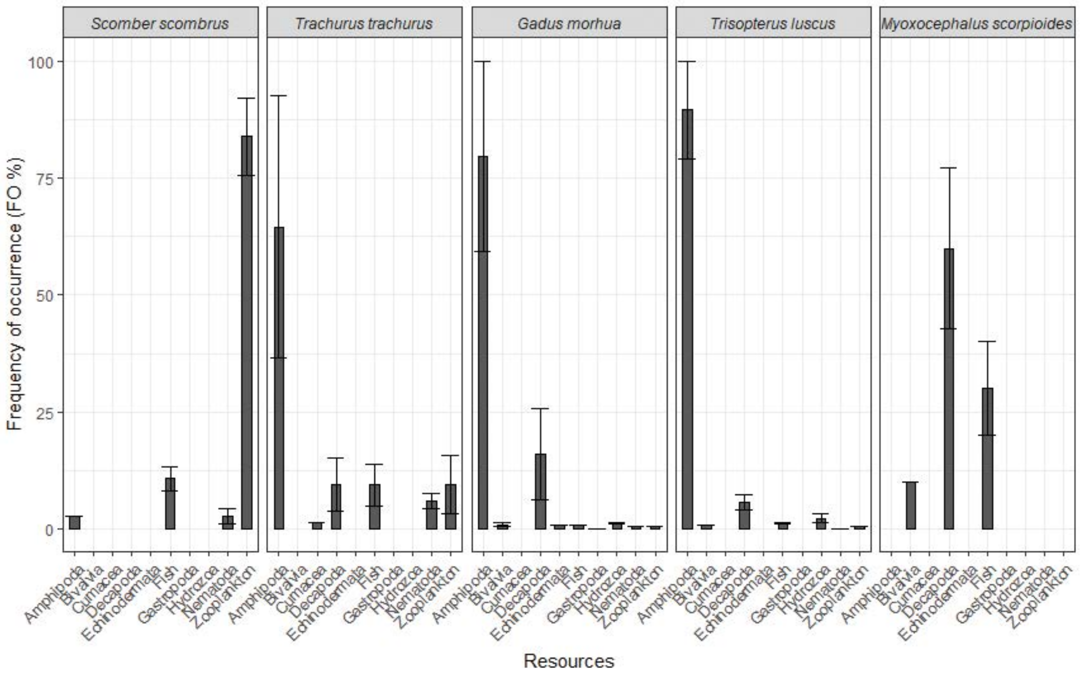

Figure 7. Percentage of abundance (%) of the prey items found in the stomach contents of each of the five fish species, the pelagic mackerel (*Scomber scombrus*) and horse mackerel (*Trachurus trachurus*), the benthopelagic cod (*Gadus morhua*) and pouting (*Trisopterus luscus*) and the benthic sculpin (*Myoxocephalus scorpioides*). The error bars represent the standard deviation.

confirmed the SCA result for most of the species, but not for horse mackerels that were found to feed on zooplankton on a long-term basis. Hence, the short-term diet of horse mackerels is dominated by prey items associated with artificial habitats, while the long-term dietary analysis indicated that this species exploits pelagic food sources.

4. Discussion

Overall, this study suggests that OWFs do influence the local food web properties from their basis, since fouling organisms slightly reduce the local (at the BPNS level) primary producers, to higher trophic levels, with several fish species exploiting these structures as feeding grounds for a prolonged period.

4.1. Detailed food web study

The detailed food web study at the community level indicated that larger trophic niches, and thus higher food web complexity are observed for the communities occurring in deeper zones and in zones where organic matter accumulation occurs (soft substrate and SPL) compared to the upper parts of the turbine. In contrast, the communities in the *Mytilus* and intertidal zones had the smallest trophic niches compared to the other parts, implying lower food web complexity at the upper parts of the turbine. Furthermore, the high trophic diversity and low redundancy in the *Metridium* zone, SPL and soft substrate suggested resource partitioning among and within the communities in these zones. This is an indication that resource partitioning is a mechanism that allows for the co-existence of a variety of species in these zones. Again, the *Mytilus* zone, consisting mainly of suspension feeders, indicated low trophic diversity and high redundancy, highlighting low levels of resource partitioning, contradicting previous findings on resource partitioning and resource selection mechanisms by suspension feeding organisms (Dubois & Colombo 2014).

The species-specific food web study along the depth gradient showed that six species (*Jassa herdmani*, *Metridium senile*, *Mytilus edulis*, *Necora puber*, *Ophiothrix fragilis* and *Pisidia longicornis*) are trophic generalists, exhibiting zone-specific resource use strategies and exploiting different resources in different zones. Such switching between resource use strategies can be a mechanism to avoid resource competition (Bolnick *et al.* 2003; Lefebvre *et al.* 2009) and indicates a large degree of trophic plasticity. Species occurring in deeper zones, such as the SPL and the soft substrate, had larger trophic niches, and thus, exploited a wider range of food sources compared to species occurring in shallower zones. Furthermore, the results of this study suggest that *J. herdmani* is a strong competitor for zooplankton, which was highlighted by the higher contribution of zooplankton in its diet compared to that of any other consumer species in every zone where *J. herdmani* occurred (fig. 4). *Diadumene cincta* was the only trophic specialist under study, and hence, the only species that did not show trophic plasticity. Our results support the hypothesis that trophic plasticity indeed is a mechanism allowing the co-existence and extended distribution of invertebrate organisms within and across zones at offshore wind turbines. This conclusion extends the well-recognized important role of trophic plasticity as a major driver of vertical zonation in marine ecosystems, *i.e.* shallow reef communities (Palardy *et al.* 2008; Fox *et al.* 2019) and deep sea communities (Carney *et al.* 2005) to hard substrate fauna in shallow coastal waters. Such zonation patterns in shallow coastal waters are commonly explained in relation to predation, space and food competition, and tolerance to unfavourable environmental conditions (Kaiser *et al.* 2011).

4.2. Wider effects

The carbon assimilation study indicated that different species assimilated significantly different amounts of carbon (fig. 5). Two of the most common species occurring on offshore wind turbines (the amphipod *J. herdmani* and the mussel *M. edulis*) play a key role in the carbon assimilation. Altogether, the distribution of carbon assimilation among the fouling species comes in agreement with previous studies performed in soft sediments, where total carbon assimilation was largely determined by the relative biomass of every taxon group (Middelburg *et al.* 2000; Kamp & Witte 2005; Woulds *et al.* 2007, 2016). The high total carbon assimilation by *J. herdmani* reflects its high abundance on the PVC panels and also its opportunistic feeding behaviour. The relative results for *M. edulis* imply that this species consumes a high share of the primary producers compared to the individuals of the other species. These findings are supported by the feeding habits of this species that largely consumes fine particulate macroalgal detritus (Dubois *et al.* 2007) and significantly decreases the net primary producer standing stock (Lemmen 2018; Slavik *et al.* 2019).

By upscaling the results of this experiment to the total number of offshore wind turbines that have been installed in the BPNS, we estimated that *J. herdmani* and *M. edulis* graze 1.3% of the annual local primary producers, with *J. herdmani* being responsible for 1.15% and *M. edulis* for 0.15% of the total reduction. This percentage is relatively small and has the same order of magnitude as an earlier model assessment for the entire southern North Sea (Slavik *et al.* 2019) and a study focusing on the German Exclusive Economic Zone (Joschko *et al.* 2008). Even though the reduction of the annual local primary producers estimated in this study is considered negligible, the filtering activities of the fouling fauna could possibly lead to an accumulation of biodeposits at the basis of the turbine foundations. These biodeposits are heavy particles, and thus, cannot be easily resuspended (Baeye & Fettweis 2015) and result in a local increase of organic matter in close proximity to the offshore wind turbines (Coates *et al.* 2014). Thus, the grazing activities have a minor effect on the primary production in the water column but the deposition of organic matter near the turbines likely has a considerable local effect on the sedimentary habitat. Furthermore, the installation of more OWFs in the future will naturally result in the provision of habitat for more fouling organisms and subsequently, in the increased consumption of the primary producers in the BPNS and local accretion of biodeposits.

Considering the footprint of the three different types of foundations in the BPNS and the amount of carbon that would be assimilated by the natural soft sediment macrofauna, we highlighted that the presence of such constructions colonised by fouling fauna significantly increases the carbon assimilation in the area. The occurrence of *J. herdmani* and *M. edulis* on a single jacket foundation causes the highest increase in carbon assimilation compared to the natural soft sediment macrofauna. This is probably caused by the very small footprint that jacket foundations have (10.5 m^2; Rumes *et al.* 2013) and also the heavy colonisation of these installations by fouling organisms. Carbon assimilation by these two organisms occurring on monopiles and gravity-based foundations was lower than that of jacket foundations but it was higher than the natural soft sediment macrofauna. This increased carbon assimilation may cause significant changes in nutrient dynamics and carbon export affecting mesopelagic and benthic processes (Letelier *et al.* 1996) and thus, influencing the entire marine food web.

Finally, the fish dietary analysis was based on a small number of fish individuals, and hence, the results need to be cautiously interpreted. However, some pattern were detected, which may feed into future research. The short-term and the time-integrated diet analysis of the five fish species indicated that the benthic and benthopelagic species (sculpin, pouting and juvenile cod) utilise artificial reefs, such as OWFs, as feeding grounds for a prolonged period. This was suggested by the diets of these species that were mainly based on fouling organisms (*i.e. J. herdmani* and *P. longicornis*) that occur abundantly on artificial habitats but not on natural hard substrates (Zintzen 2007). Horse mackerels, in contrary, feed on the abundant fouling organisms (*J. herdmani*) only on a short-term, while they exploit zooplankton on a long-term basis. Therefore, this species exploits the artificial habitats only occasionally as feeding grounds. Finally, mackerels were mainly feeding on zooplankton and hence, they did not utilise the artificial habitats as feeding grounds. The dietary results of this study in combination with previous findings corroborate the hypothesis that OWFs could potentially increase the local production of benthic and benthopelagic species. This is explained by their long-term feeding ecology, which suggests that they remain near artificial habitats for a long period of time to feed. On the contrary, pelagic species seem to be attracted by these structures for so far unknown non-trophic reasons and further research is needed to quanti- and qualify the potential effect on local production. Our findings are in line with previous studies stating that benthic and benthopelagic fish remain close to artificial reef habitats (Cresson *et al.* 2019), while the effect of artificial reefs on pelagic fish species is negligible (Powers *et al.* 2003; Cresson *et al.* 2019).

4.3. The larger picture

From the above, four main conclusions can be drawn: (a) SPLs play an important role as feeding grounds for the hard substrate associate species (both vertebrate and invertebrate), (b) soft substrates near the turbines show high food web complexity in terms of trophic diversity compared to other zones along the depth gradient, (c) *Jassa herdmani* is a strong competitor for zooplankton, while it also significantly contributes to the reduction of the primary producers, and (d) benthopelagic and benthic fish species exploit the artificial habitats, *i.e.* OWFs, as feeding grounds for a prolonged period.

The general pattern observed in this study was that SPLs play an important role in increasing habitat heterogeneity and promoting food web complexity at the base of the turbine foundations. More specifically, the SPL was found to accommodate species belonging to various trophic levels, demonstrating the second highest (after the soft substrate) food web complexity compared to the other zones along the depth gradient. The increased trophic complexity was further confirmed by the wide range of resources exploited by the species occurring in this zone. Finally, the pivotal role of SPLs on marine food webs was highlighted by the feeding ecology of fish species associated with these structures, *i.e.* benthic and benthopelagic species, which feed on fouling organisms for a prolonged period. The combination of these findings indicates that SPLs function as the main part of the OWF artificial reef compared to the turbines themselves, supporting predictions on the matter (Petersen & Malm 2006; Langhamer 2012). Indeed, Petersen and Malm (2006) had predicted that SPLs at the base of some artificial structures would act as the main part of the artificial reefs, given the provision of habitat heterogeneity. This heterogeneity would contribute to the increased species diversity and density, altering the nature of the soft sediments near offshore wind foundations and turning exposed and/or biodiversity poor soft substrates into species-rich ecosystems (Langhamer 2012). Finally, Langhamer (2012) predicted that food provisioning between the rocks of SPLs would get up to 60 times higher compared to the natural soft sediments, with our findings corroborating this hypothesis.

This study cannot provide concrete knowledge on whether benthic and benthopelagic species increase their local production since production cannot be easily measured. However, our findings do support the hypothesis that the local production of these fish species could potentially increase. Mechanisms that contribute to the increased production include food availability, increased feeding efficiency, provision of shelter against predators and currents, and provision of habitat for recruitment of settling organisms (Bohnsack 1989; Reubens *et al.* 2014). This study clearly shows that OWFs provide increased feeding activity and efficiency. Cod has shown strong residency and site fidelity towards offshore wind turbines (Reubens *et al.* 2013a), while pouting individuals were observed to have larger length and fitness proxies inside the OWFs than the individuals occurring in sandy areas (Reubens *et al.* 2013b). Even though these findings support the production hypothesis, further research is needed to prove that production of fish species locally and regionally actually occurs.

Our results cannot be seen isolated from the larger picture of the proliferation of OWFs in the BPNS and beyond. In the future, more OWFs will be installed, covering a larger surface area and providing habitat for fouling organisms. The introduction of more OWFs will lead to increased environmental carrying capacity at the base of the turbines, which could act as 'trophic-diversity hot spots'. More habitat will be available to be colonised by trophic generalists, which have more trophic links than trophic specialists, which can result in more stable food webs, as food-web stability is highly dependent on the most connected species (Dunne *et al.* 2002), which almost by definition are trophic generalists. More OWFs and their subsequent colonisation will naturally lead to a further local reduction of the primary producers influencing the very basis of the marine food webs and biogeochemical cycling (Slavik *et al.* 2019). However, in nutrient-rich regions, the filtering activities of fouling fauna can sustain longer phytoplankton blooming periods through faster nutrient recycling, which could support high productivity (Slavik *et al.* 2019). Altogether, the installation of more offshore wind turbines will further affect the food web properties, due to the cascading trophic effects caused by the fouling fauna colonising these structures. However, further research is needed to completely understand the cumulative effects of these artificial structures on marine food webs.

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