2022

MEMOIRS
on the Marine Environment

SCIENTIFIC REPORTS SERIES

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

GETTING READY FOR OFFSHORE
WIND FARM EXPANSION
IN THE NORTH SEA

Edited by
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This report should be cited as:

Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds). 2022. *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Getting ready for offshore wind farm expansion in the North Sea*. *Memoirs on the Marine Environment.* Brussels: Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, 106 pp.

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This publication has been peer-reviewed.

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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. On 31 December 2019, Belgium submitted a National Energy and Climate Plan to 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. 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. In that zone, eight wind farms are operational with a combined installed capacity of 2.262 MW. A second area for renewable energy of 285 km² is foreseen by the new Belgian marine spatial plan (2020-2026) with the government aiming for an installed capacity of 3,15 to 3,5 GW in this zone.

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 2021.

DEGRAER Steven, BRABANT Robin, RUMES Bob and VIGIN Laurence

EXECUTIVE SUMMARY

GETTING READY FOR OFFSHORE WIND FARM EXPANSION IN THE NORTH SEA

DEGRAER Steven, BRABANT Robin, RUMES Bob, VIGIN Laurence, BAETENS Jan, BLOMME Ellen, BRAECKMAN Ulrike, COURTENS Wouter, DE BACKER Annelies, DEBUSSCHERE Elisabeth, DE PAUW Lukas, HOSTENS Kris, KAPASAKALI Danae-Athena, KERCKHOF Francis, KERKHOVE Thomas R.H., LEFAIBLE Nene, MEYS Joris, MOENS Tom, STIENEN Eric, VAN DE WALLE Marc, VANERMEN Nicolas, VAN HOEY Gert, VERSTRAETE Hilbran& WITTOECK Jan

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Insights into the nature and magnitude of the impact of offshore wind on the marine ecosystem are becoming ever more pressing given the anticipated tenfold expansion of the offshore wind sector in the North Sea area in the coming decades. 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 260 Gigawatt (GW) by 2050, with intermediate targets of at least 76 GW by 2030 and 193 GW by 2040.

Currently, eight offshore wind farms are operational in the Belgian part of the North Sea (BPNS), totaling an installed capacity of 2.26 GW and consisting of 399 offshore wind turbines (Chapter 1). They produce an average of 8 TWh annually, accounting for ~1/3 of gross electricity production from renewable energy sources in Belgium. An additional zone for offshore renewable energy – the Princess Elisabeth Zone – has been designated in the marine spatial plan 2020-2026 and is anticipating an installed capacity ranging between 3.15 and 3.5 GW. As 'Blue Economy' matures to a sustainable blue economy, it has been tasked with ensuring the environmental sustainability of the natural capital of the oceans and seas. With 523 km² reserved for operational and planned offshore wind farms in Belgium, 344 km² in the adjacent Dutch Borssele zone, and 122 km² in the French Dunkerque zone, cumulative ecological impacts continue to be a major concern.

Worldwide, a plethora of monitoring and research programs target mapping the ecological impacts. They all contribute to building that knowledge base needed for advice on an environment-friendly design and operation of offshore wind farms. They hence provide the basis for combatting the biodiversity, climate and energy crises going hand in glove. The necessary knowledge base encompasses both baseline information on the ecosystem at stake and insight into the impacts at various scales in space and time. Since 2005, the Belgian offshore wind farm environmental impact monitoring program, WinMon.BE, generates baseline ecological information in the Belgian offshore renewable energy zone and beyond, and investigates the impacts at various spatio-temporal scales. Its two-fold aim is to quantify both the anticipated and unanticipated impacts and understand the cause-effect relationships behind a selection of these ecological impacts. The knowledge gained so far has served fine-tuning offshore

wind farm construction and operation practices in the existing offshore renewables zone and will serve an environment-friendly design and operation of the Princess Elisabeth Zone.

This WinMon.BE report based on data collected up to and including 2021, focuses on selected topics to get ready for offshore wind farm expansion in the Belgian part of the North Sea, by touching upon new insights into (1) spatial distribution patterns (epibenthos, hyperbenthos and fish of sandy bottoms) and the identification of areas sensitive to offshore wind farms (seabirds) and (2) 'promoting the good' (artificial hard substrate fouling communities) and 'mitigating the bad' (seabird collision and harbor porpoise disturbance).

To get ready for the future epibenthos and demersal fish monitoring of potential spillover effects of the fully developed existing offshore renewables zone and for the future assessment of the potential impacts of the new offshore renewables zone, a community analysis on epibenthos and demersal fish abundance data in the entire BPNS (83 locations; 540 samples; 2008-2020) was performed (Chapter 2). Both epibenthos and fish communities largely follow similar spatial distribution patterns with a clear distinction between the coastal and the offshore area. Within the coastal area, a first community dominated by brown shrimp *Crangon crangon* and gobies *Pomatoschistus* spp., occurs in the muddy sediments near the eastern part of the Belgian coast. A second community occurring in fine sands, is dominated by serpent star *Ophiura ophiura* and dab *Limanda limanda*. The offshore communities are additionally structured by sand bank topography. The most widespread offshore coarse sand community is characterized by the bivalves *Spisula solida* and *Spisula elliptica*, the crab *Liocarcinus marmoreus* and the squid species *Loligo vulgaris* and *Sepiola atlantica* as typical epibenthic species. For fish, this community is dominated by lesser weever *Echiichthys vipera*, with solenette *Buglossidium luteum*, scaldfish *Arnoglossus laterna*, red mullet

Mullus surmuletus and reticulated dragonet *Callionymus reticulata* as additional characteristic fish species. A species-poor version of this community occurs on top of the steep sand banks. For fish, a third community could be discerned around the 12 nautical miles consisting of a mixture of coastal and offshore species, making this the most diverse community. The existing offshore renewables zone largely overlaps with the spatial distribution of the offshore epibenthos and fish communities, for which monitoring in between the turbines did not show meaningful impacts in the wind farm. Potential spillover effects cannot be excluded however and will be investigated further by focusing on the offshore coarse sand community, thereby following a gradient design based on distance from the wind farm. The wider surroundings of the less surveyed new offshore renewables zone suggest that a coarse sand epibenthos and demersal fish community is to be expected within the south-western zone, yet in a topographically heterogeneous context represented by steep sand banks and gravel beds. A thorough before-impact survey will be essential to assess the potential impacts of this new offshore renewables zone on epibenthos and fish.

This year's WinMon.BE monitoring program for the first time shed a light on the hyperbenthos ecosystem component (Chapter 3). We hypothesized that the impact of turbine presence ("artificial reef effect") and the ceasing of fishery activities ("fisheries exclusion effect") would result in enriched hyperbenthic communities within the offshore wind farms. Three replicate hyperbenthos samples were collected inside and outside two Belgian offshore wind farms, each with specific local habitat conditions, foundation type and time of construction. In the wind farm on the Thornton Bank, the hyperbenthos community had features of both offshore and transitional (nearshore → offshore) communities. The community at the wind farm closer to the coast generally corresponded with transitional communities, but also harbored species that are found in

more nearshore areas. Total densities and species richness were higher at the impact site compared to the control site for the wind farm on the Thornton Bank (on average 1856 vs.1399 ind. 100 m^{-3} and 35 vs. 28 spp. sample^{-1}, respectively). This corroborates the enrichment hypothesis. Also, the community structure showed differences between the control and impact sites. The hyperbenthos community in the wind farm closest to the coast was characterized by higher densities. Also here, densities were higher at the impact site compared to the reference site (4483 vs. 2105 ind. 100 m^{-3} on average), but no differences were found for species richness (22 vs. 28 spp. sample^{-1} on average), nor for community structure. Despite the valuable insights gained within this study, it also revealed that increased sampling efforts will be needed to enhance the ability to fully characterize the hyperbenthic communities and strengthen the statistical power to detect offshore wind farm-related impacts.

While detailed knowledge on distribution patterns is important to design adequate monitoring programs, this information may further provide insights into area-specific sensitivity of different species to offshore wind farms. In Chapter 4, we developed species distribution models, intended to feed into a sensitivity map regarding offshore wind farm development. We focused on four seabird species known to be sensitive to wind farm-induced displacement, i.e., red-throated diver *Gavia stellata*, northern gannet *Morus bassanus*, common guillemot *Uria aalge* and razorbill *Alca torda*, and proposed an integrated displacement sensitivity index based on their cumulative occurrence. In this analysis, ship-based seabird counts collected across the BPNS in the period 2000-2018 were used. As explanatory variables for the species distribution modelling, water depth, variation in water depth, salinity, distance to the coast and wind farm presence were considered. The offshore wind farm factor was retained in the distribution models of all four species, in line with their sensitivity to disturbance by wind turbines. The species distribution models

further allow us to quantify the numbers of seabirds expected to be impacted by wind farm displacement. In absolute numbers, common guillemot is the most impacted species, with about 1600 individuals being displaced by the existing and future offshore renewables zones. Mapping the displacement sensitivity index highlighted one area as particularly sensitive to offshore wind farm development, situated in front of the western part of the Belgian coast between 5 and 12 nautical miles offshore. This area is located well outside all existing and future Belgian wind farms, giving the opportunity to avoid future developments in that zone or otherwise to take compensating measures. To ultimately inform the marine spatial planning process, further finetuning of the modelling process and taking account of additional seabird species and anthropogenic pressures will be needed.

Getting ready for offshore wind farm expansion in the North Sea not only involves species distribution and sensitivity mapping, but also entails a continued effort in gathering knowledge for an environment-friendly design of offshore wind farms. In this context, the introduction of hard substrates is taking a prominent role, known as the "artificial reef effect", with fouling communities being at its basis. Foundations of offshore wind turbines can have a 35-fold higher biomass compared to surrounding soft sediments, which can influence local food web dynamics. These fouling species can be indigenous, but the establishment of non-indigenous species is an increasing concern. To better understand the potential effects of large-scale colonization of offshore wind turbines by fouling species, we compared the fouling communities on the turbine foundations and surrounding scour protection in Belgian offshore wind farms to the fouling communities on long-existing artificial hard structures, *in casu* shipwrecks (Chapter 5). Therefore, we used the Belgian artificial hard substrate database, holding all species records of macrobenthic (> 1 mm) species associated with different artificial hard substrates in the Belgian part

of the North Sea. In total, 209 species were listed. Shipwrecks were characterized by a higher fouling species richness compared to offshore wind farms (165 vs. 114 spp.). Species identity was also different between both artificial hard substrates, with 95 unique fouling species for shipwrecks and 44 unique fouling species for offshore wind farms. The differences in biodiversity between both structures may be attributed to the older age and the higher structural complexity of shipwrecks. Increasing the complexity of the scour protection layer surrounding turbine foundations might increase species richness on the artificial hard substrate, which often is considered an asset of offshore wind farms.

While the above is an example of knowledge gathering with the aim of optimally 'promoting the good', we also need to continue getting ready for maximally 'mitigating the bad'. A prominent and long-standing issue with offshore wind farms is how to adequately mitigate seabird collisions. As the offshore wind energy technology is rapidly progressing and because wind turbines at sea have a relatively short life span, repowering scenarios are already being discussed for the oldest wind farms. Ongoing technological developments result in larger wind turbines and an increased open airspace between turbines. Despite taller towers having larger rotor swept zones and therefore, a higher collision risk area compared to smaller-sized turbines, there is increasing evidence that fewer but larger, more power-efficient turbines may have a lower collision rate per installed megawatt. As such, turbine size can offer an opportunity to mitigate seabird fatalities by increasing the clearance below the lower rotor tip. We assessed the seabird collision risk for a hypothetical repowering scenario of the first offshore wind farm zone in Belgian waters with larger turbines and the effect of an additional increase in hub height (i.e., distance from the water surface to the center of the turbine's rotor) on that theoretical collision risk (Chapter 6). For all considered bird species, the estimated collision risk decreased in a repowering scenario with 15

MW turbines (40% reduction on average) because of higher clearance between the lower tip of the turbine rotor and the sea level, and the need for a lower number of turbines per km². Increasing the hub height (and thus the clearance) of those 15 MW turbines with 10 m, further decreased the expected number of seabird collisions with another 37% on average. Terrestrial birds and bats also migrate at sea, but the effect of larger turbines on these taxa is less clear. Bird radar data show that a higher number of nocturnal migrants fly at rotor height of larger turbines resulting in a 9% increased collision risk for songbirds in the repowering scenario compared to the current wind farms. The response of bats to offshore turbines is still poorly understood making it difficult at this point to predict how bats will be impacted by fewer, larger turbines. So, while larger turbines and increasing the hub height can be beneficial for seabirds, this likely needs to be applied in combination with curtailment strategies, which stop the turbines during heavy terrestrial bird and bat migration events, as such reducing the impact on these species groups.

A second prominent and long-standing issue with offshore wind farms is the disturbance of marine mammals during piling activities, producing excessive impulsive sound levels. In the southern North Sea, offshore wind farm construction usually entailed hydraulic pile driving resulting in high levels of impulsive sound. Despite recent advances in noise-mitigation technology, harbor porpoises (*Phocoena phocoena*) respond to this pile driving over a period of hours to days per driven pile, depending on the distance at which the animals were disturbed. We used passive acoustic monitoring datasets from 2018 to 2020, including the construction periods of three offshore wind farms, to determine the factors which influenced the likelihood of detecting harbor porpoises before, during and after pile driving in the Belgian part of the North Sea (Chapter 7). During pile driving and in the 24 hours after pile driving, mean detection rates of porpoises reduced up to 20 km from

the pile driving location although both the magnitude and duration of this reduction decreased markedly with increasing distance. By means of generalized additive modelling, we found distance to the construction site (as a proxy for received sound level) to be the main driver for porpoise response to pile driving, whereas seasonality, time of day and type of sound mitigation had limited but significant effects on the spatial and temporal extent of avoidance of the construction area by porpoises. In the immediate vicinity of the construction site, the reduction in porpoise detection rates started even prior to the pile driving, suggesting the presence of other sources of disturbance in this area. Our results suggest that efforts to reduce the impact of underwater noise generated by future offshore wind farm construction on marine life should aim to limit not only the generated noise levels but also the overall duration of the construction period.

In conclusion, getting ready for offshore wind farm expansion in the North Sea and beyond entails a continued effort to gain ever more knowledge needed for fine-tuning an environment-friendly design and operation of offshore wind farms. Substantial progress has been made and has proved to be applicable for a sound management of offshore renewable energy. However, many unknowns remain to be tackled. A coordinated and flexible approach to monitoring and research across offshore wind farms, like WinMon.BE, and at the relevant ecological scale, will be the best guarantee for an efficient and effective knowledge gathering. Efforts in that sense are being made in other countries as well (e.g., New England coast of the U.S. and Atlantic coast of France) and should continue to be made; this also in the wider North Sea region, where country boundaries have tended to hamper this regional approach.

CHAPTER 1

OFFSHORE RENEWABLE ENERGY IN THE BELGIAN PART OF THE NORTH SEA

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Abstract

Eight offshore wind farms are operational in the Belgian part of the North Sea (BPNS), totaling an installed capacity of 2.26 Gigawatt (GW) and consisting of 399 offshore wind turbines. They produce an average of 8 TWh annually, accounting for ~1/3rd of gross electricity production from renewable energy sources in Belgium (FPS Economy, 2022). An additional zone for offshore renewable energy has been designated in the marine spatial plan 2020-2026 and is anticipating an installed capacity ranging between 3.15 and 3.5 GW. As "Blue Growth" matures to a sustainable blue economy, it has been tasked with ensuring the environmental sustainability of the natural capital of the oceans and seas (EU, 2021).

With 523 km² reserved for operational and planned offshore wind farms in Belgium, 344 km² in the adjacent Dutch Borssele zone, and 122 km² in the French Dunkerque zone, cumulative ecological impacts continue to be a major concern. These anticipated impacts, both positive and negative, triggered an environmental monitoring program focusing on various aspects of the marine ecosystem components, but also on the human appreciation of offshore wind farms. This

introductory chapter provides an overview of the status of offshore renewable energy development in the BPNS.

1. Offshore wind energy development in Belgium

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 revised marine spatial plan that came in force on March 20th, 2020.

The European Directive 2009/28/EC on the promotion of the use of energy produced from renewable sources, 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
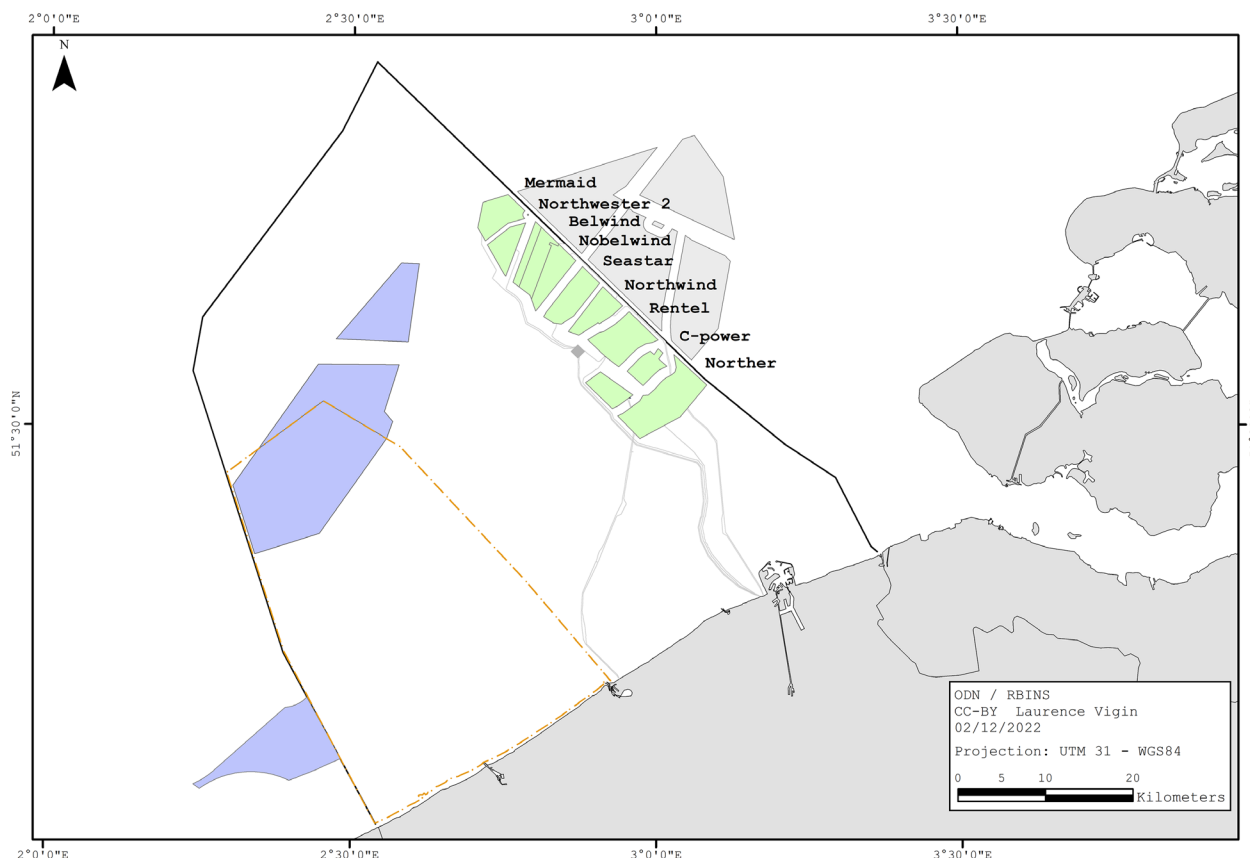
Figure 1. Current and planned zones for renewable energy in and around the Belgian Part of the North Sea. Operational wind farms in Belgian waters are shown in green. Operational wind farms in the Dutch Borssele area are in grey. The blue areas in the NW of the Belgian part of the North Sea are the Princess Elisabeth zone, an area for renewable energy development as delineated in the revised marine spatial plan 2020-2026. Also in blue is the proposed Dunkerque offshore wind farm in French waters. The orange dashed line is the Belgian Natura 2000 area 'Vlaamse banken'.

consumption, which had to be achieved by the end of 2020. Offshore wind farms in the BPNS make an important contribution to that goal.

On 31 December 2019, Belgium submitted a National Energy and Climate Plan to 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 includes 4 GW of operational offshore wind energy by 2030 (Belgische Overheid, 2019).

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

a wind farm, even if a domain concession was granted.

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

future project to the Minister responsible for the marine environment. MUMM's advice includes an environmental impact assessment, based on the EIS. The Minister then grants or denies the environmental permit in a duly motivated decree.

At present, nine projects were granted a domain concession and an environmental permit (from South to North: Norther, C-Power, Rentel, Northwind, Seastar, Nobelwind, Belwind, Northwester II & Mermaid (Table 1 and Fig. 1). On July 20th 2018, the merger between the Seastar and Mermaid projects was finalized and the resulting merged project was named Seamade NV. 399 wind turbines are operational in the Belgian part of the North Sea (Fig. 2). The entire first area has 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. Over the last decade, turbine size, rotor diameter and installed capacity per turbine has gradually increased (Table 1) 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 mitigate and/or minimize 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.

On 20 March 2020, the second marine spatial plan for the BPNS (Royal Decree of May 22nd, 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, indicators and targets addressing marine protected areas and the management of human uses including commercial fishing,
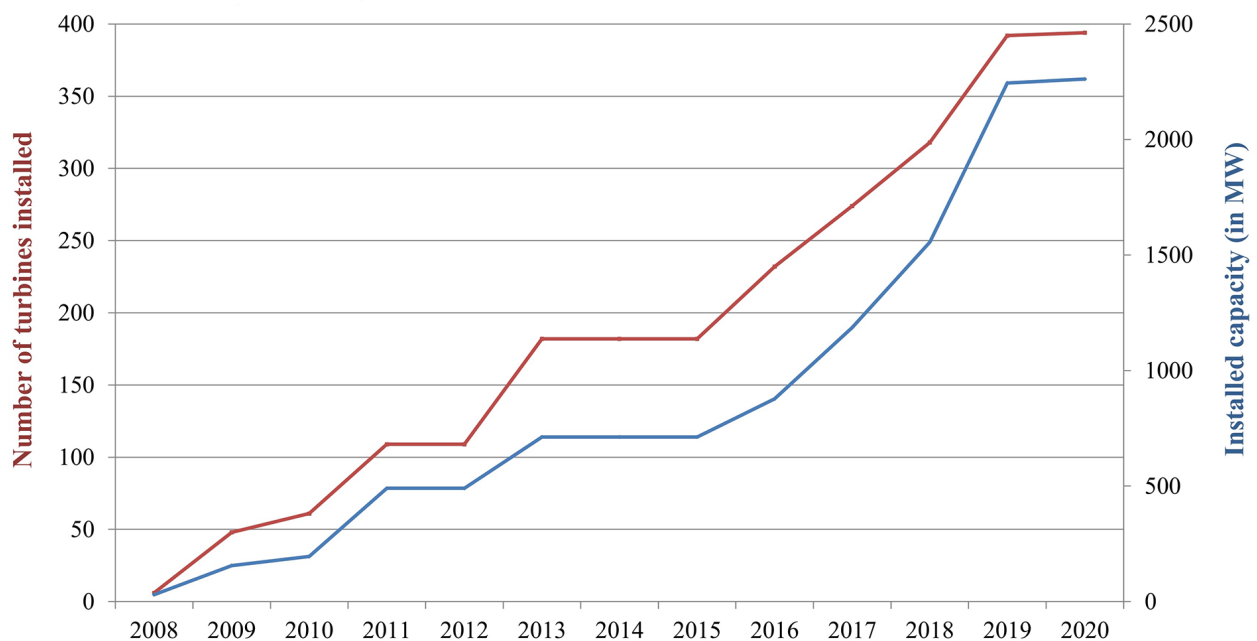
Figure 2. Number of offshore wind turbines installed and installed capacity in the Belgian Part of the North Sea since 2008.

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

Project		Number of turbines	Capacity (MW)	Foundation type	Rotor diameter (m)	Hub height (m LAT*)	Total capacity (MW)	Operational since
Norther		44	8.4	monopile	164	107	370	2019
C-Power	phase 1	6	5	gravity based	126	94	325	2009
	phases 2 & 3	48	6.2	jacket	126	94		2013
Rentel		42	7.4	monopile	154	106	309	2019
Northwind		72	3	monopile	90	72	216	2014
SeaMade		58	8.4	monopile	167	109	487	2020
Belwind	phase 1	55	3.1	monopile	90	72	171	2011
	Alstom Demo project	1	6	jacket	150	100		2013
Nobelwind		50	3.3	monopile	90	72	165	2017
Northwester 2		23	9.5	monopile	164	106	219	2020

*lowest astronomical tide
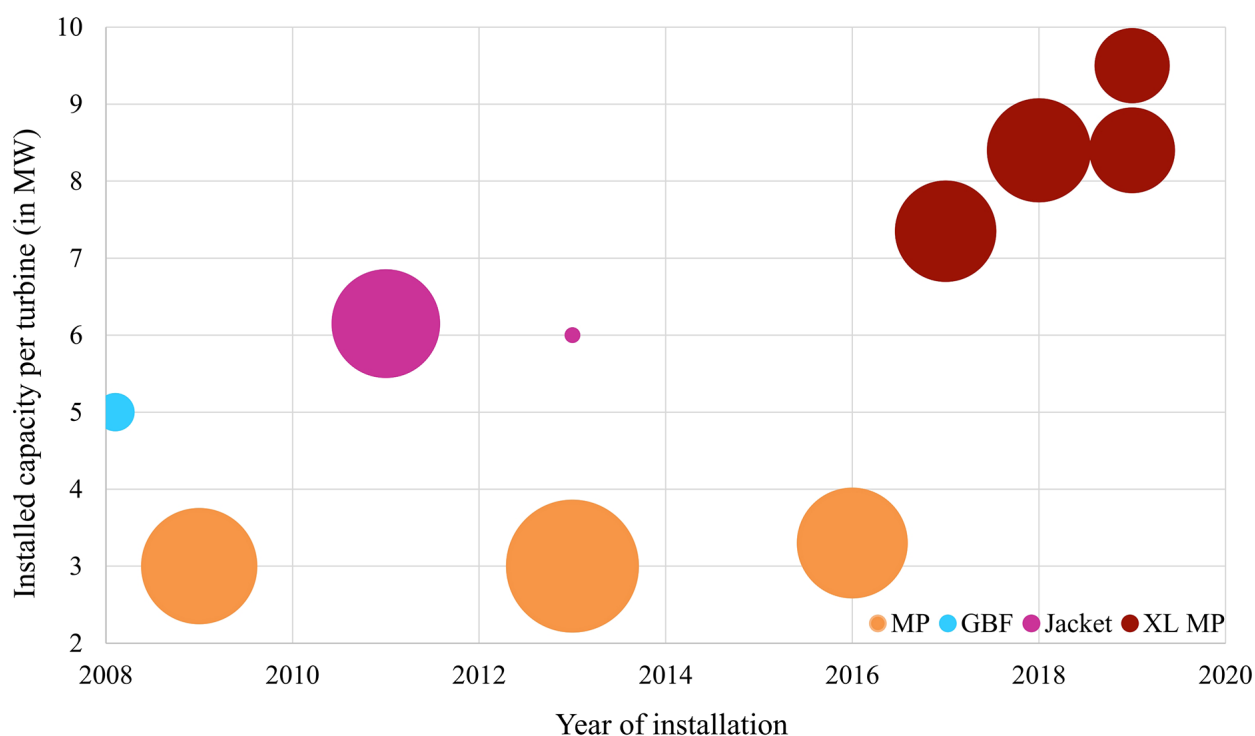
Figure 3. Overview of the timing, individual capacity and foundation type of offshore wind turbines installed in the Belgian Part of the North Sea since 2008. The size of the bubbles is proportional to the number of turbines installed per project of phase (Table 1). MP: monopile foundation, GBF: Gravity based foundation, Jacket: Jacket foundation, XL MP: monopile foundations exceeding approximately 7 m in diameter.

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 a second zone for renewable energy of 285 km² located at 35-40 km offshore (Fig. 1). This second zone would be suitable for an additional 3.15-3.5 GW of installed capacity. Storage of energy and grid reinforcement continue to be major hindrances to the further integration of renewables into the electricity grid and locations are foreseen for reinforcing the offshore electricity grid.

This second Belgian zone for marine renewable energy is partly located inside the designated Natura 2000 area 'Vlaamse banken'. 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.

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CHAPTER 2

DESCRIBING THE EPIBENTHOS AND DEMERSAL FISH COMMUNITIES IN THE BELGIAN PART OF THE NORTH SEA IN VIEW OF FUTURE OFFSHORE WIND FARM MONITORING

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Abstract

Future epibenthos and demersal fish monitoring of potential 'spillover' effects of the fully operational 238 km² Belgian offshore wind farm (OWF) area together with future evaluation of the potential effects of the newly designated Princess Elisabeth area for renewable energy requires sound knowledge on the epibenthos and demersal fish communities of the BPNS. To this end, a community analysis on epibenthos and demersal fish abundance data (2008-2020), covering 540 beam trawl sampling events in autumn spread over 83 locations on the BPNS, was performed.

Both epibenthos and fish communities largely follow similar spatial distribution patterns with a clear distinction between the coastal and the offshore area. Within the coastal area, we distinguish a mud community occurring in the muddy sediments near the eastern part of the coast which is dominated by brown shrimp *Crangon crangon* and gobies *Pomatoschistus* spp., for resp. epibenthos and fish. The fine sand community is correlated with the fine sandy sediments in the coastal area and is highly dominated by resp. the serpent star *Ophiura ophiura* and dab *Limanda limanda.* Distinction between the different offshore communities seems to be structured by sand bank topography. For both epibenthos and fish, a widespread offshore coarse sand community is observed with *Spisula solida*, *Spisula elliptica*, *Liocarcinus marmoreus* and squid species *Loligo vulgaris* and *Sepiola atlantica* as typical epibenthic species. For fish, this community is dominated by lesser weever *Echiichthys vipera*, with solenette *Buglossidium luteum*, scaldfish *Arnoglossus laterna*, red mullet *Mullus surmuletus* and reticulated dragonet *Callionymus reticulata* as additional characteristic fish species. For both epibenthos and fish, a species-poor version of this community occurs on top of the steep sand banks dominated by *Pagurus bernhardus* for epibenthos and dominated completely by lesser weever for fish. For fish, a clear third transitional community could be discerned around the 12 NM consisting of a mixture of coastal and offshore species making this the most diverse community.

The current operational Belgian OWF area largely overlaps with the spatial distribution of the offshore epibenthos and fish communities. Locations inside the OWF concessions cluster nicely together with all non-concession locations confirming the conclusion from previous studies that epibenthos and fish assemblages on the soft sediments in between the turbines underwent no drastic changes. For studying potential future spillover effects, the offshore coarse sand community locations are the best candidates to be included in a gradient design.

For the newly designated Princess Elisabeth area, data on soft sediment communities is scarce with regard to the current beam trawl sampling locations. The wider surroundings suggest that for the sandy areas the coarse sand community is to be expected within the south-western zone. However, the topography of this area is very heterogeneous with steep sandbanks, and gravel beds occurring in between these sand banks, so a thorough before-impact monitoring will be essential to enable assessing the future OWF effects on epibenthos and fish. Here as well, inclusion of a gradient design is preferred and the outcomes of the community analyses will help in defining the best possible reference locations.

1. Introduction

The Belgian OWF area in the eastern part of the Belgian Part of the North Sea (BPNS) is fully operational since the end of 2020, making this a contiguous zone of 238 km² with an installed capacity of 2.26 GW renewable energy (Rumes & Brabant 2021). All fishery activities are excluded in the OWFs. Up till now, effects on soft sediment epibenthos and demersal fish have been focused on the two oldest wind farms C-Power and Belwind (De Backer & Hostens 2018a; De Backer et al. 2020). The BACI design, with 'impact' and 'near control' beam trawl samples, was mainly focussed on potential changes in the OWF compared to the surrounding sandy environment outside the concession areas.

Now that all concessions are combined in one large contiguous OWF area, we expect that the fisheries exclusion or 'refugium' effect (Handley et al. 2014) might become more prominent and lead to 'spillover' or fringe effects (export of biomass to surrounding habitats by recruitment or migration out of the area). To capture these potential refugium and 'spillover' effects, the sampling design need to be changed towards a gradient design, including sample locations within and at different distances from the OWF area. In that respect, it is key to delineate distinct epibenthos and demersal fish communities within the BPNS, to ensure that gradient sampling locations are situated within similar communities, allowing for a proper comparison.

Macrobenthos communities are already well described for the BPNS (Van Hoey et al. 2004; Breine et al. 2018), but for epibenthos and demersal fish such well-defined communities based on high resolution sampling are still to be described. Moreover, the Belgian federal government has delineated a second area of 285 km² for renewable energy (i.e. the Princess Elisabeth area) located at 35-40 km offshore in the northwestern part of the BPNS (Rumes & Brabant 2021). This created a second reason for a thorough analysis and description of the epibenthos and demersal fish communities in the BPNS, allowing for a good monitoring design from the start.

As such, the main objective of this study was to perform a community analysis on mid-term epibenthos and demersal fish data of the BPNS, to be able to determine a proper gradient design to investigate 'spillover' effects in the first Belgian OWF area, and to propose a future environmental monitoring design for the Princess Elisabeth Zone (PEZ).

2. Material and methods

2.1. Study area

The BPNS is situated in the southern part of the North Sea and only covers 0.5 %

(3,454 km2) of the North Sea basin. It is a shallow sea with an average water depth of 20 m and a maximum depth of 46 m and it is characterized by numerous sand bank systems: (1) Coastal Banks, parallel to the coastline, (2) Flemish Banks, about 10-30 km offshore of the western Belgian coast, (3) Zeeland Banks, some 15-30 km offshore of the eastern Belgian coast, and (4) Hinderbanks, about 35-60 km offshore (Van Hoey *et al.* 2004). Due to the presence of these sandbanks, a highly variable and complex topography is observed and sedimentological diversity is high as well. Fine sand occurs along the coastline, high mud content near the mouth of the river Scheldt in the eastern part of the BPNS, while further offshore, grain size increases to medium and coarse sand (Verfaillie *et al.* 2006) (Figure 1). Moreover, subtidal natural hard substrates, i.e. gravel beds occur in the area as well in between the sandbanks (Van Lancker *et al.* 2007; Montereale-Gavazzi *et al.* 2021).

For the sandbank habitat, five distinct macrobenthic assemblages have been described that are strongly related to sediment type and bottom topography (Van Hoey *et al.* 2004; Degraer *et al.* 2008; Breine *et al.* 2018). Three are situated within the coastal area: the *Macoma balthica* community occurring in the muddy sediments at the east coast, the *Abra alba* community associated with fine coastal sediments and the *Magelona - Ensis leei* community in the shallow, nearshore area. Offshore, two communities are present: the *Nephtys cirrosa* community, occurring in medium sands, and the *Hesionura elongata* community typical for coarser sands.

The gravel beds are suitable for an array of species that cannot occur in soft-bottom habitats (Houziaux *et al.* 2008). They naturally host rich macro- and epibenthos communities that include sessile and/or long-lived species (i.e. >5 yrs) (e.g. *Pomatoceros triqeter*, *Sabellaria spinulosa*, *Haliclona oculata*, *Flustra foliacea*, *Alcyonium digitatum*, *Sertularia cupressina*, *Ostrea edulis*, *Buccinum undatum*) (Houziaux *et al.* 2008). Therefore, gravel beds are very

valuable habitats and hotspots of biodiversity, but they are highly pressurized and mostly in a deteriorated state putting them high on the agenda for conservation and restoration measures.

2.2. Sampling and biological data

A dataset was compiled with beam trawl samples (540 in total) that were collected in the BPNS in autumn over the period 2008-2020. Samples were collected in the framework of the ILVO long-term environmental impact monitoring programs in relation to different human activities. Samples with a direct impact of dredge disposal and sand extraction were excluded, while samples taken in the framework of the offshore wind farm (WinMon.BE) monitoring program (both impact and control) have been included, since the aim of the analyses was to find suitable reference locations at a gradient from the wind farms. In total, 540 sampling events at 83 locations (Figure 1) have been included in the dataset.

On these locations, demersal 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. Till 2009, the net was towed for a total of 30 minutes at an average speed of 4 knots over the bottom along with the current (approx. 2 NM). From 2010 onwards, tow duration was reduced to 15 minutes (approx. 1 NM). A comparative field study revealed no difference in catch composition for the different tow durations (Derweduwen *et al.* 2010). Data on time, start and stop coordinates, trajectory and sampling depth were recorded to enable a correct conversion towards sampled surface units. Epibenthos and fish were identified to the lowest possible taxonomic level, mostly species level, counted, measured (all fish, crabs and shrimps) and wet weighted (all epibenthos) onboard. Some epibenthos samples that could not be fully processed onboard, were frozen and further processed in the lab.

Figure 1. Overview map with beam trawl sampling locations on top of the median grain size map with indication of the epibenthic (left) and fish (right) community to which they belong (see further). Grey lines are locations which could not be properly assigned to one of the communities. Depth contours provided are 10 m (light grey) and 20 m (dark grey).
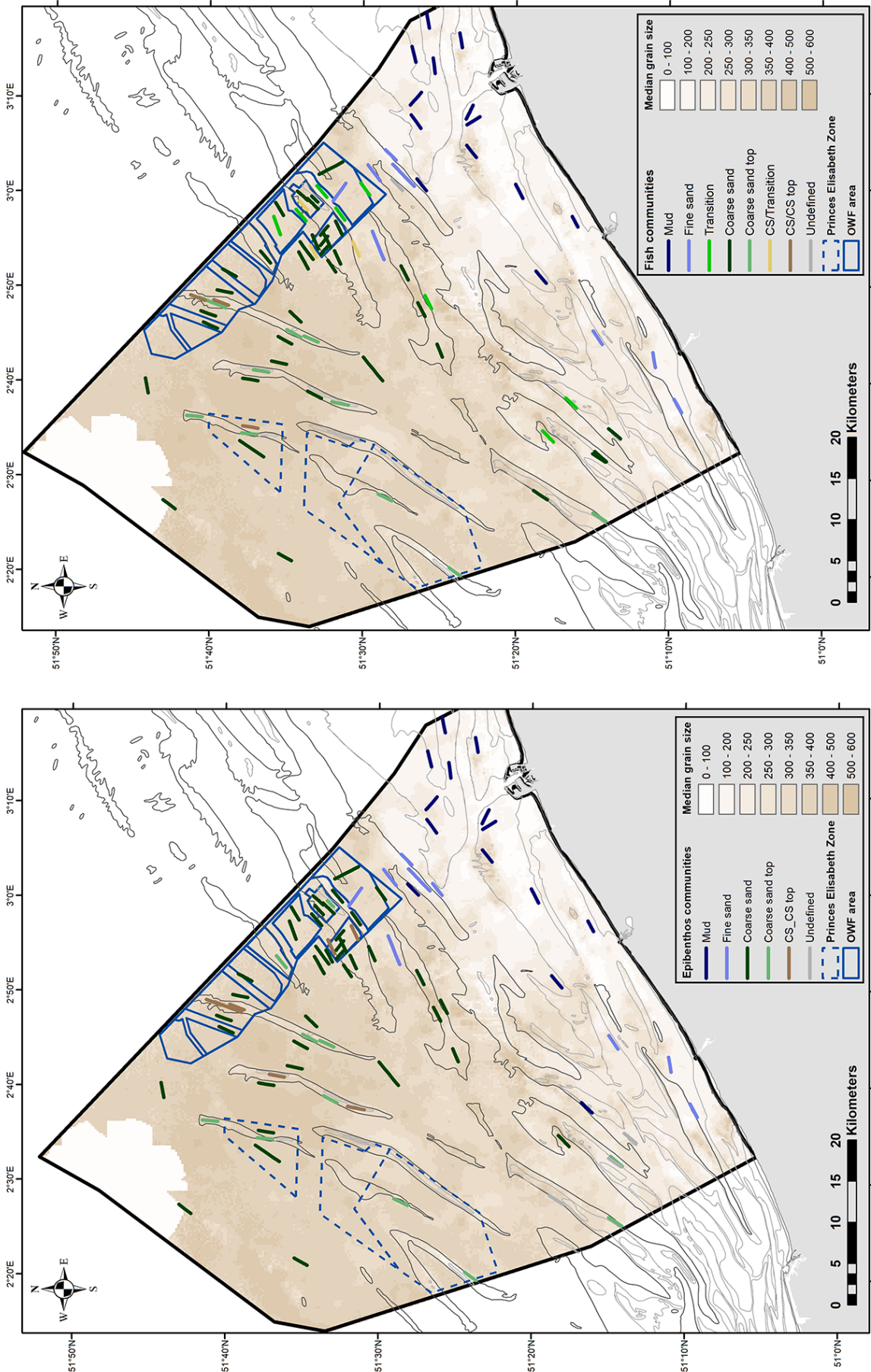
Count and wet weight data were converted to densities/wet weight based on the trawled surface area for standardization to individuals per 1000m² or gWW per 1000 m². Pelagic species (based on www.fishbase.org) such as *Sprattus sprattus*, *Trachurus trachurus*, *Scomber scombrus*, along with jellyfish, certain benthic bivalves (such as *Abra alba*) and polychaetes were excluded from the analyses, since these are not quantitatively sampled with a beam trawl. Given that data was gathered over different monitoring programs, and by different persons over different years, species difficult to identify (e.g. *Pomatoschistus* spp., *Macropodia* spp.) were lumped at a higher taxonomic level to exclude taxonomic errors. Because of much higher abundances of epibenthos compared to fish (sometimes up to $100 \times$ higher densities), the dataset was split in two, i.e. epibenthos and demersal and bentho-pelagic fish, further referred to as fish. In this way, fish patterns are not blurred by the dominance of certain epibenthos species.

2.3. Data analysis

All data analyses were performed in Primer version 7 with PERMANOVA add-on software (Clarke & Gorley 2015; Anderson *et al.* 2008). The analyses were done for each ecosystem component (i.e. epibenthos and fish) separately.

For each ecosystem component, two different datasets were used for two types of analyses, i.e. (1) identification of the different communities, and (2) structural characterization of the communities. To identify the different epibenthos and fish communities in the BPNS, a dataset containing resp. 35 and 37 taxa was used, excluding taxa occurring in less than 3% of the sampling events to rule out the influence of different sampling effort in a specific community (a higher sampling effort enhances the chance of finding rare species). Cluster analyses was performed to identify the epibenthos and fish communities, based on a Bray Curtis resemblance matrix after resp. fourth-root

and square-root transformation of species densities. The choice of transformation was aided by visualising the data matrix through shade plots (Clarke & Gorley 2015). Since certain epibenthos taxa (e.g. *Ophiura* spp.) tend to dominate in huge numbers, we opted for fourth-root, while for fish this is less the case and a square root transformation was sufficient. Cluster groups were visualised using non-parametric Multidimensional Scaling Ordination (nMDS). A similarity Percentages (SIMPER) routine with a cut-off level of 90% was applied to identify the species that contributed most to the within-group similarity of the cluster groups (i.e. the communities). Furthermore, shade plots have been used to visualise species occurrence in the different cluster groups by ordering samples per cluster group and by clustering species which tend to have similar patterns of abundance across the samples (Clarke & Gorley 2015).

Distribution of the cluster groups over the BPNS was visualised using ArcMAP 10.4. Each sampling location was attributed to a cluster group if it clustered for the majority of the sampling events in the same group. Otherwise, a sampling location was attributed to a transition group if it switched between two cluster groups, or it remained undefined if it switched between more than 2 groups.

In the second analysis, the identified communities were characterised by means of structural variables using the DIVERSE module (density, biomass for epibenthos, number of taxa (S), Margalef's diversity (d), Shannon index (H′(loge)) and Simpson index (1-λ)). For these analyses, the entire datasets, containing resp. 50 and 51 taxa, were used, with density and biomass standardised to 1000 m². To test for significant differences in these variables between cluster groups, a univariate one-way Permanova based on the Euclidean distance resemblance matrix (with unrestricted permutation of raw data) was performed (Anderson & Robinson 2003), followed by pairwise tests to situate the differences between cluster groups.
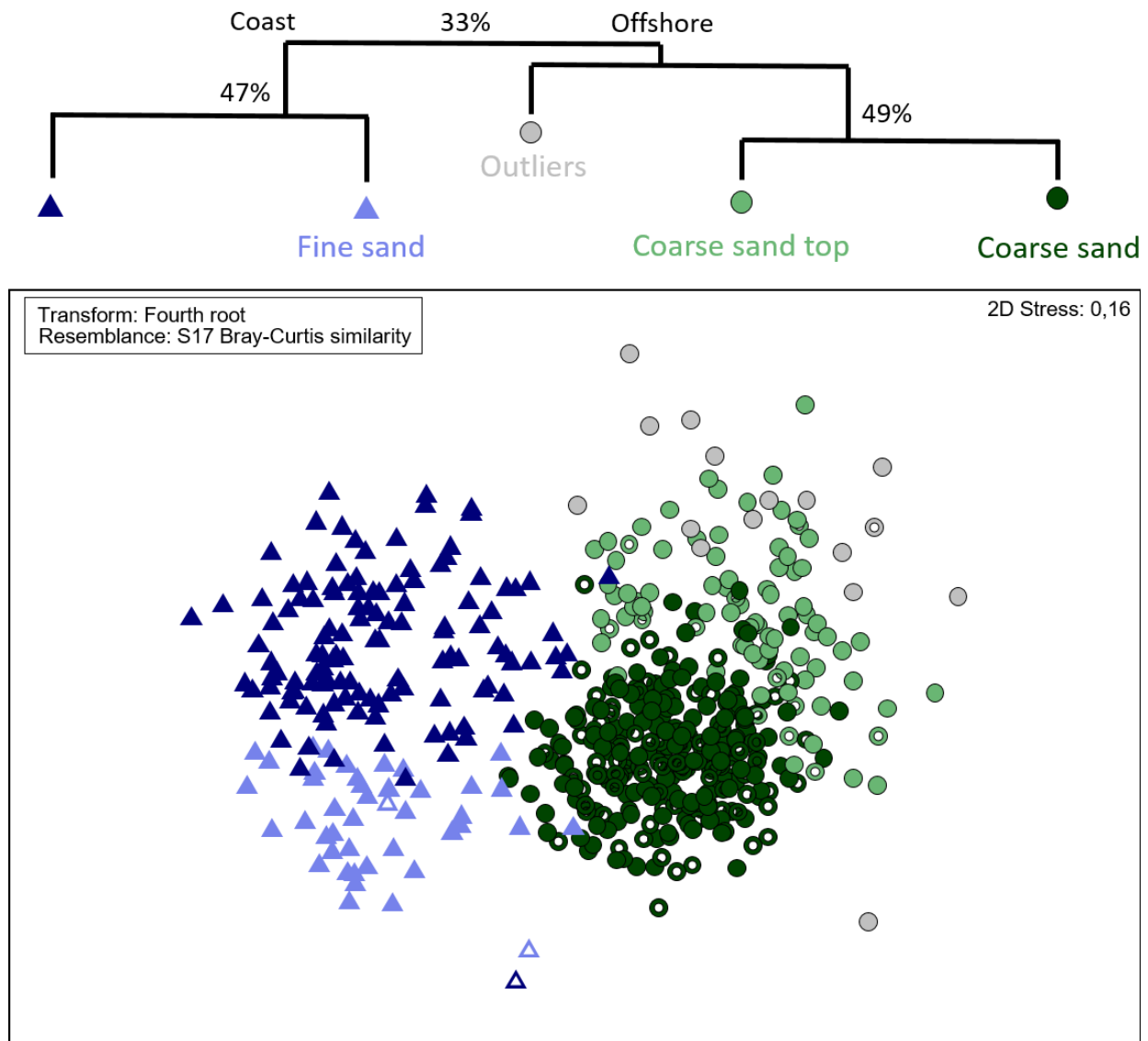
Figure 2. Top: simplified representation of cluster analysis on fourth root transformed species abundance data for epibenthos, only representing the position of the main groups and with indication of split-off similarity level. Below: nMDS ordination. Every point represents a sampling event assigned to a certain community based on the cluster analysis, open symbols represent sampling events within OWFs.

3. Results

3.1. Epibenthos

3.1.1. Communities and spatial distribution

The cluster analyses showed a first separation in two broad clusters at the 33% similarity level, i.e. a coastal and an offshore cluster (Figure 2). Further split off at the 49% similarity level identified four main groups, i.e. two within the coastal cluster and two within the offshore cluster, which have been included in further analyses (Figure 2). In total,

16 samples (all situated offshore) could not be assigned to one of the four main clusters, as they split off at a lower similarity level. The four main cluster groups were defined as the Mud (57% avg. within-group similarity), Fine sand (60% avg. w/i-group sim.), Coarse sand (61% avg. w/i-group sim.) and Coarse sand-top (54% avg. w/i-group sim.) communities, based on the habitat/sediment type of the area where they are located, sometimes in combination with the position on the sand bank.

Table 1. Characterization of the epibenthic communities by indication of the average 'within-group' similarity (SIMPER); species listed account for 90% of cumulative contribution of the 'within group' similarity (in % and average abundance (N, ind 1000 m⁻²)), the average (±SD) for a number of univariate parameters is provided per community.

Assemblage	Mud			Fine sand			Coarse sand			Coarse sand top		
W/i group similarity	57%			60%			61%			54%		
Species	Species	%	N	Species	%	N	Species	%	N	Species	%	N
	Crangon crangon	33.1	488	Ophiura ophiura	24.1	1402	Pagurus bernhardus	15.2	5.8	Pagurus bernhardus	25.2	2.07
	Ophiura ophiura	22.8	200	Crangon crangon	10.8	109	Ophiura albida	12.5	4.4	Asterias rubens	13.9	0.47
	Liocarcinus holsatus	17.1	39	Asterias rubens	10.3	93	Asterias rubens	11.4	2.8	Loliginidae juv	13.8	0.50
	Pagurus bernhardus	7.4	2.3	Liocarcinus holsatus	9.9	52	Ophiura ophiura	10.3	2.4	Ophiura ophiura	13.2	0.32
	Asterias rubens	5.9	2,1	Ophiura albida	7.8	40	Liocarcinus holsatus	7	0.6	Liocarcinus holsatus	8.1	0.10
	Anthozoa spp.	2.5	0.3	Pagurus bernhardus	7.7	15	Spisula solida	5.8	0.4	Sepiola atlantica	4.9	0.02
	Tritia reticulata	2.3	0.4	Tritia reticulata	5.6	12	Spisula elliptica	4.3	0.2	Macropodia spp.	3.5	0.01
				Crepidula fornicata	4.1	10	Macropodia spp.	4.3	0.1	Ophiura albida	3.1	0.03
				Loliginidae juv	3.8	3	Loliginidae juv	4	0.2	Loligo vulgaris	2.4	0.01
				Liocarcinus depurator	3	1.2	Liocarcinus marmoreus	3.6	0.1	Crangon crangon	2.1	0.03
				Macropodia spp.	2.9	1.3	Sepiola atlantica	3.4	0.1			
							Loligo vulgaris	3.4	0.1			
							Psammechinus miliaris	3.1	0.1			
							Anthozoa spp.	2.8	0.1			
# samples	44			121			272			87		
N (ind/1000 m²)	1776.3 ± 2533.3			3732 ± 3007			47.2 ± 76.8			11.5 ± 14.6		
Biomass (g/1000m²)	2652.2 ± 2933.9			6860.6 ± 5244.9			156.8 ± 172.7			43.6 ± 60.6		
S	10.1 ± 3.3			14.4 ± 2.8			15.3 ± 2.9			9.8 ± 2.1		
Margalevs diversity	1.4 ± 0.7			1.8 ± 0.5			4.4 ± 1.1			5.2 ± 2.9		
Shannon diversity	0.9 ± 0.4			1.1 ± 0.4			1.8 ± 0.3			1.5 ± 0.3		
Simpson diversity	0.4 ± 0.2			0.5 ± 0.2			0.8 ± 0.1			0.9 ± 0.4		

Regarding the coastal communities, the Mud community is mainly restricted to the nearshore eastern part of the coast, while the Fine sand community has a broader onshore distribution, close to the shoreline in the west and a bit further away in the east (Figure 1). Most offshore samples (±75%) were assigned to the Coarse sand community and are widely distributed offshore. Most locations within the OWF area do belong to this community. The offshore Coarse sand top community is mainly restricted to the top of the offshore steeper sand banks (Figure 1). The outliers are also most related to this group (Figure 2).

Overall, structural characteristics differed significantly among the four communities (one-way Permanova, for all main tests p=0.0001). Although, number of species did not differ between Mud and Coarse sand top and not between Fine sand and Coarse sand (resp. p=0.47 and p=0.09). Simpson index was not different (both low values) between the coastal communities Mud and Fine sand (p=0.18), indicating high dominance of a few species. Density and biomass were at least an order of magnitude higher in the coastal communities compared to the offshore communities, whereas diversity measures (Margalev's, Shannon and Simpson) were higher in the offshore communities. (Table 1).

3.1.2. Community characteristics

Based on the shade plot (Figure 3), it is clear that six species (*Asterias rubens*, *Pagurus bernhardus*, *Ophiura albida*, *Liocarcinus*
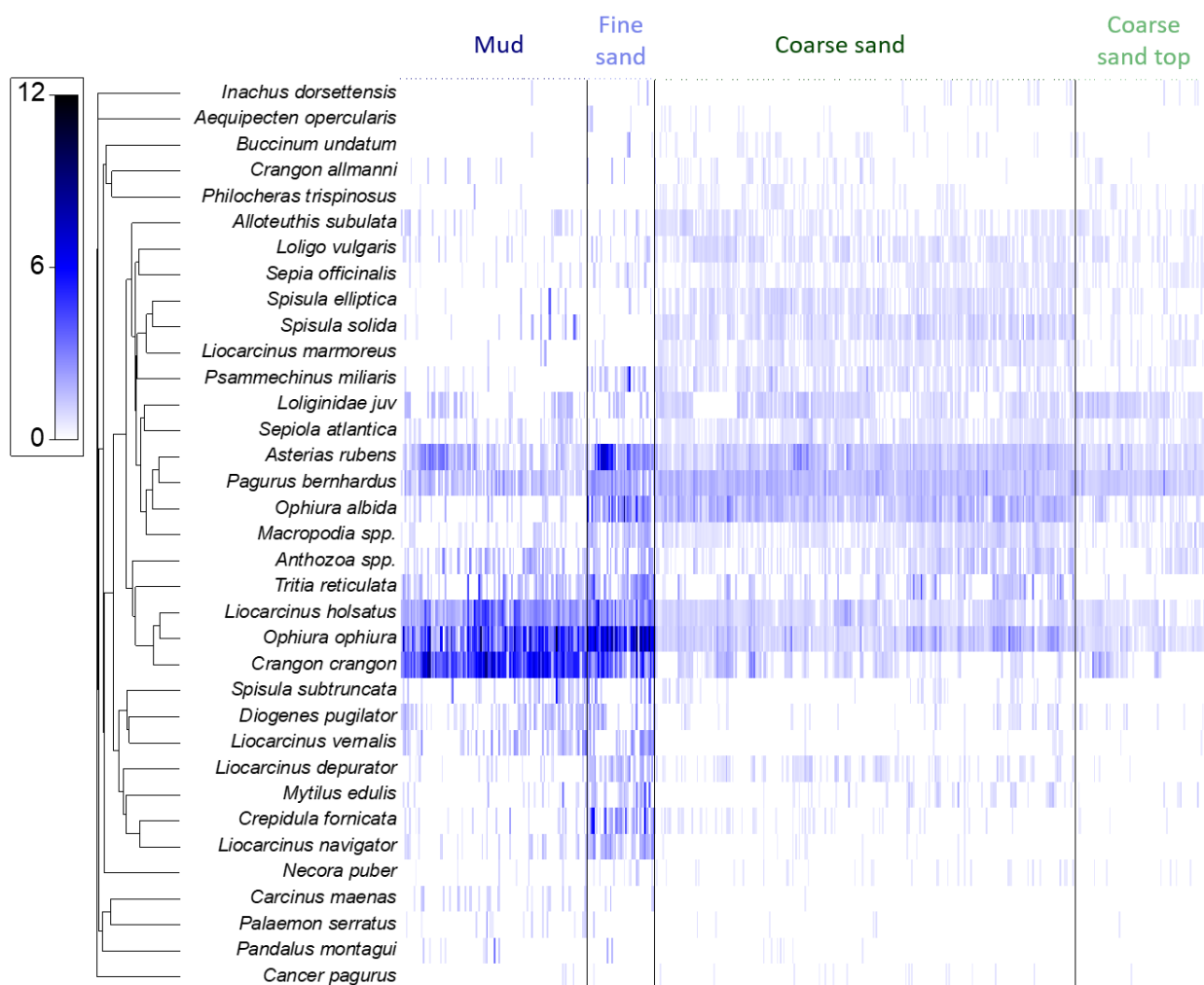
Figure 3. Shade plot showing averaged, fourth-root transformed abundance (ind. 1000 m^{-2}) of the main epibenthos species in the delineated epibenthic communities.

holsatus, *Ophiura ophiura* and *Crangon crangon*) occur commonly on the BPNS across all communities, but at much lower abundances offshore compared to coastal. Seven other species are more common in the coastal communities (*Spisula subtruncata*, *Diogenes pugilator*, *Liocarcinus vernalis*, *Liocarcinus navigator*, *Liocarcinus depurator*, *Mytilus edulis* and *Crepidula fornicata*) albeit with different abundances in both coastal communities. Another nine species (*Alloteuthis subulata*, *Loligo vulgaris*, *Sepia officinalis*, *Spisula elliptica*, *Spisula solida*, *Liocarcinus marmoreus*, *Psammechinus miliaris*, *Loliginidae* juv. and *Sepiola atlantica*) are generally more common and abundant in the offshore communities (Figure 3).

3.1.2.1. Coastal communities

Mud

This community is mainly dominated by the brown shrimp *C. crangon* (33% contribution to within-group similarity), followed by *O. ophiura* (23%) and *L. holsatus* (17%) (Table 1 and Figure 3). It is characterized by high densities (N=1776±2533 ind. 1000 m⁻²) and biomass (2652±2934 g 1000 m⁻²) and a low number of species (10±3, d=1.4±0.7). Diversity measures Simpson (0.4±0.2) and Shannon (0.9±0.4) are very low compared to the other communities.

Fine sand

This community is dominated by the serpent star *O. ophiura* (24% contribution to within-group similarity), followed by *C. crangon*, *Asterias rubens* and *L. holsatus* (each 10 %). Other characteristic species are *Tritia reticulata* (6 %) and *Crepidula fornicata* (4 %) (Table 1; Figure 3). Density (N=3732±3007 ind. 1000 m⁻²) and biomass (6861±5245) are very high, while species richness is average (S=14±3, d=1.8±0.5). Shannon (1.1±0.4) and Simpson (0.5±0.2) diversity are again low compared to both offshore communities.

3.1.2.2. Offshore communities

Coarse sand

This community is not dominated by a single species, but characterized by a more or less equal contribution to within-group similarity of *P. bernhardus* (15%), *O. albida* (13%), *A. rubens* (11%) and *O. ophiura* (10%). Other typical species/taxa are *S. solida* (6%) and *S. elliptica (4%)*, *Macropodia* spp. (4%) and representatives of the squid family i.e. *Sepiola atlantica* (3%), *Loligo vulgaris* (3%) and juvenile squids *Loliginidae* juv. (4%) (Table 1; Figure 3). Density (47±77 ind. 1000 m⁻²) and biomass (157±173 g 1000 m⁻²) are very low compared to the coastal communities but still three times higher compared to the offshore Coarse sand top community (Table 1). Species richness on the other hand is higher compared to the coastal communities (S=15±3, d=4±1) as are the diversity measures Shannon (1.8±0.3) and Simpson (0.8±0.1).

Coarse sand top

The common hermit crab *P. bernhardus* (25% contribution to within-group similarity) dominates this community, followed by *A. rubens* (14%), *Loliginidae* juv. (14%) and *O. ophiura* (13%). It has the lowest density (11±15 ind. 1000 m⁻²) and biomass (44±61 g 1000 m⁻²) amongst all communities. Number of species is low (10±2), while Margalef's diversity, which takes into account the number of individuals, is highest across communities (5±3). Diversity measures Shannon (1.5±0.3) and Simpson (0.9±0.4) are higher than for both coastal communities, while comparable with the offshore Coarse sand community.

3.2. Fish

3.2.1. Fish communities and spatial distribution

At the 27% similarity level, hierarchical cluster analysis on fish abundance data identified a coastal and an offshore group (Figure 4). Within the coastal cluster, a further subdivision at the 42% similarity level discerned a 'mud' (60% avg. w/i-group
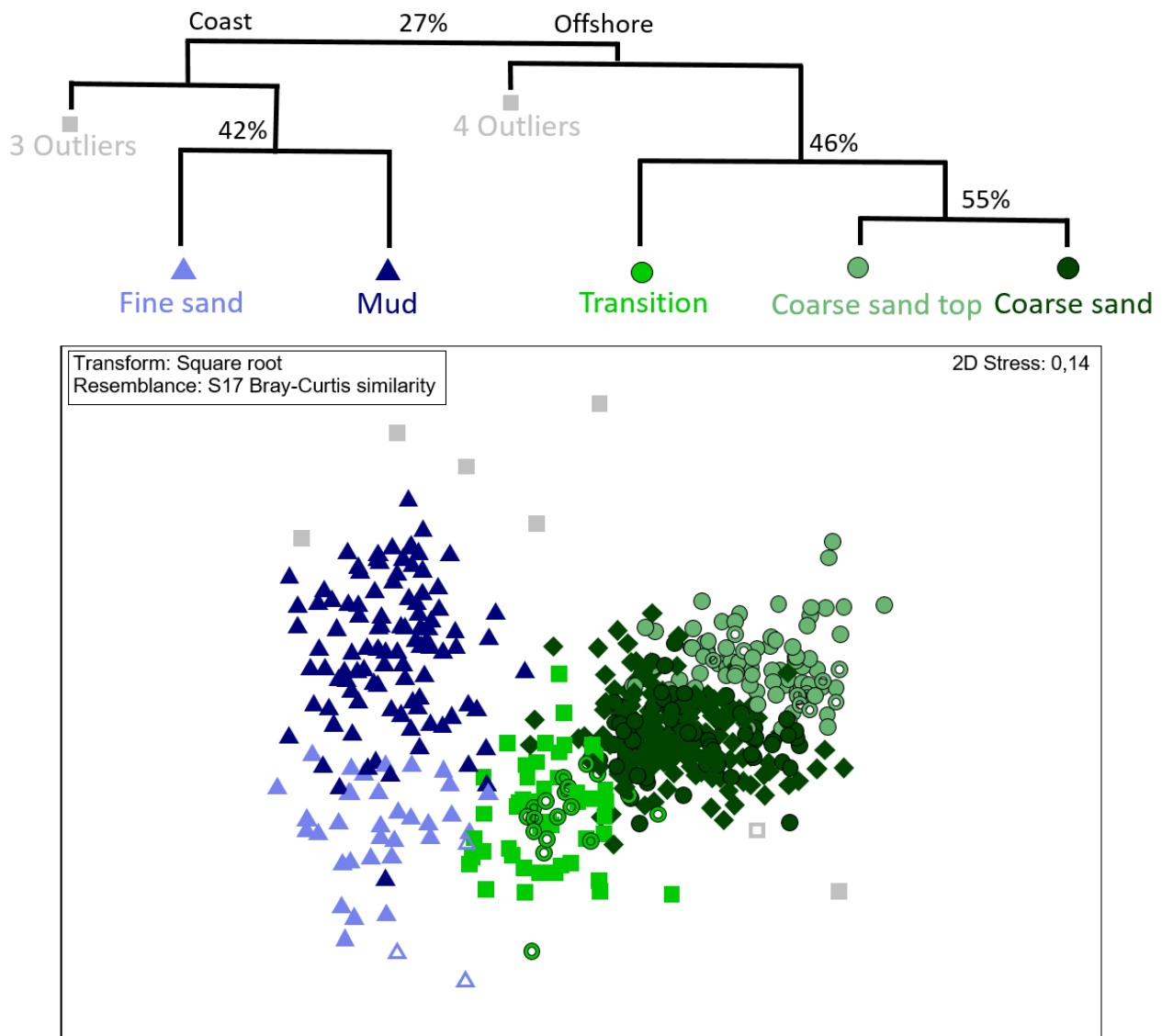
Figure 4. Top: simplified representation of cluster analysis on square root transformed species abundance data for fish, only representing the position of the main groups and with indication of split-off similarity level. Below: nMDS ordination. Every point represents a sampling event assigned to a certain community based on the cluster analysis, open symbols represent sampling events within OWFs.

sim.) and a 'fine sand' (61% avg. w/i-group sim.) community. Within the offshore cluster, three cluster groups could be distinguished: a 'coarse sand' (64% avg. w/i-group sim.) and 'coarse sand top' (67% avg. w/i-group sim.) community at the 55% similarity level, and a 'transition' community (63% avg. w/i-group sim.) splitting off at the 46% similarity level (Figure 4). Only 7 samples could not be assigned to one of the five clusters and were omitted from all further analyses.

The mud community is mainly restricted to locations near the eastern part of the coast, while the fine sand community has a wider onshore distribution (Figure 1). Within the offshore cluster, most locations (around 37) belong to the coarse sand community, which has a very broad offshore distribution. The ten locations of the 'coarse sand top' community are situated on top of the steep offshore sandbanks. A fifth cluster is distinguished as a separate fish community, the transition

community, which is mainly restricted to the slope/gully locations in the transitional area around the 12 NM border) between the coastal fine sand community and the offshore locations (Figure 1).

Overall, structural characteristics differed significantly among the five communities (one way Permanova, all main tests p=0.0001), although the fine sand and coarse sand community did not differ for number of species (p=0.1). Also, for Margalef's species diversity, no significant difference was observed between both coastal communities (mud and fine sand, p=0.5) nor between the coarse sand and transition community (p=0.15). Density was not significantly different between coarse sand

top and transition communities (p=0.1). In general, density was at least twice as high in the coastal communities compared to the offshore communities.

3.2.2. Community characteristics

The shade plot shows a transition in species composition from coastal to offshore. Three species occur commonly across all communities in the entire BPNS, i.e. gobies *Pomatoschistus* spp., plaice *Pleuronectes platessa* and dab *Limanda limanda* (Figure 5). Pouting *Trisopterus luscus*, hooknose *Agonus cataphractus*, sole *Solea solea* and whiting *Merlangius merlangus* are more characteristic of the coastal area, while the offshore communities are dominated by
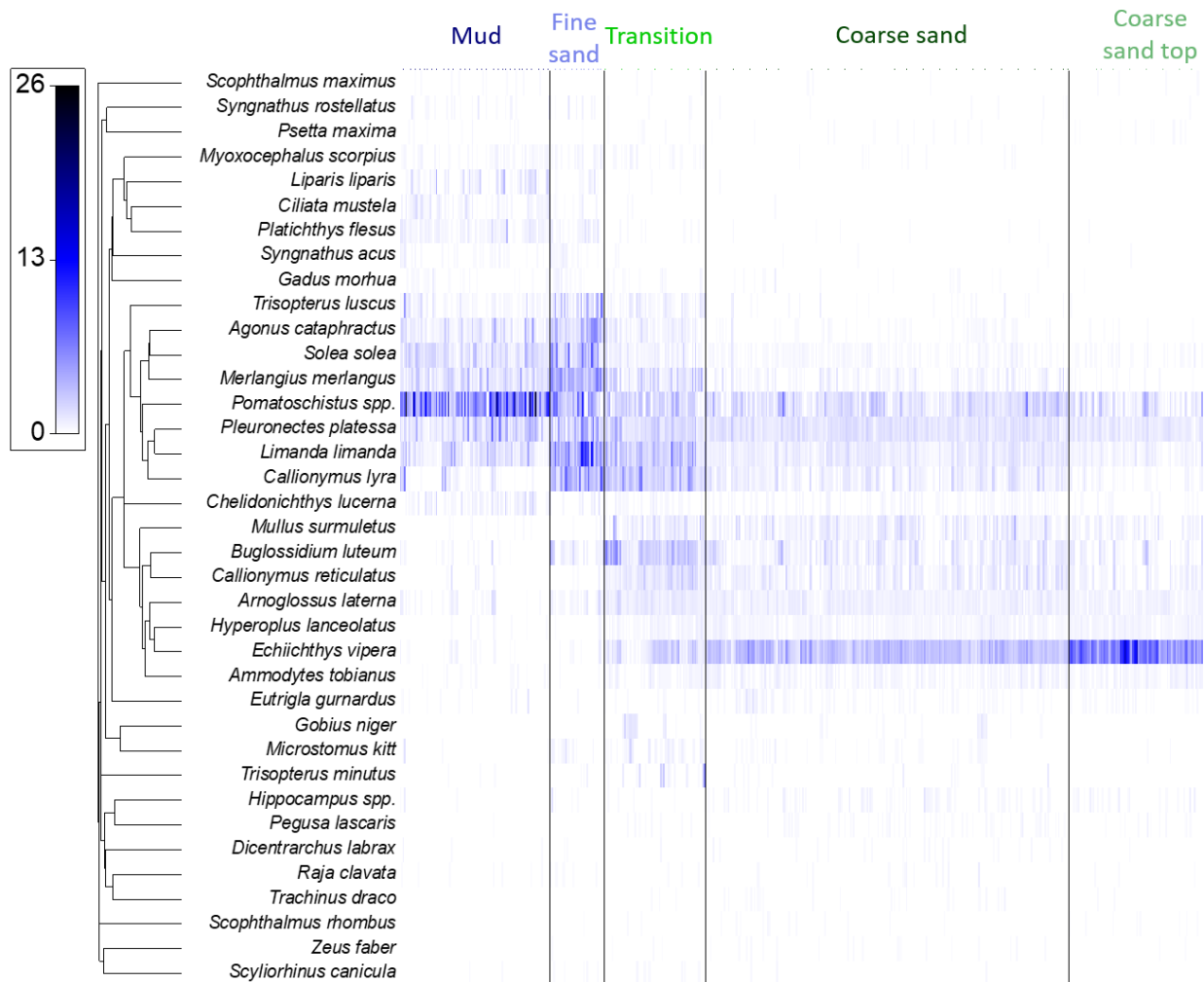
Figure 5. Shade plot showing averaged, square-root transformed abundance (ind. 1000 m^{-2}) of the main fish species in the delineated fish communities.

Table 2. Characterization of the fish communities by indication of the average 'within-group' similarity (SIMPER); species listed account for 90% of cumulative contribution of the 'within group' similarity (in % and average abundance (N, ind 1000 m^{-2})), the average ($\pm$ SD) for a number of univariate parameters is provided per community.

Assemblage	Mud			Fine sand			Transition			Coarse sand			Coarse sand top		
W/I group similarity	60			61			63			64			67		
	Species	%	N	Species	%	N	Species	%	N	Species	%	N	Species	%	N
	Pomatoschistus spp.	54.3	68.6	Limanda limanda	19.1	28.3	Callionymus lyra	14.3	6.8	Echiichthys vipera	32.7	9.1	Echiichthys vipera	64.6	47.3
	Pleuronectes platessa	8.9	3.3	Merlangius merlangus	17.1	16.6	Limanda limanda	14.0	7.2	Pleuronectes platessa	13.9	1.7	Pleuronectes platessa	9.7	1.0
	Solea solea	7.8	1.8	Callionymus lyra	15.2	15.1	Buglossidium luteum	12.7	6.4	Pomatoschistus spp.	12.2	2.4	Pomatoschistus spp.	5.3	1.1
	Limanda limanda	7.3	1.6	Pomatoschistus spp.	11.4	11.9	Pomatoschistus spp.	10.8	4.3	Limanda limanda	7.9	0.8	Arnoglossus laterna	5.0	0.3
	Merlangius merlangus	7.3	1.7	Agonus cataphractus	9.3	7.7	Pleuronectes platessa	10.3	3.5	Arnoglossus laterna	6.7	0.5	Hyperoplus lanceolatus	4.0	0.2
	Agonus cataphractus	4.4	1.2	Solea solea	9.2	9.4	Callionymus reticulatus	6.8	1.7	Mullus surmuletus	4.5	0.5	Limanda limanda	2.9	0.2
	Platichthys flesus	2.3	0.3	Pleuronectes platessa	9.1	7.8	Echiichthys vipera	6.6	2.4	Callionymus lyra	4.4	0.5			
							Arnoglossus laterna	5.8	0.9	Buglossidium luteum					
							Merlangius merlangus	5.4	1.8	Callionymus reticulatus					
							Mullus surmuletus	2.9	0.6						
							Agonus cataphractus	2.6	0.4						

	Mud	Fine sand	Transition	Coarse sand	Coarse sand top
# samples	100	36	67	241	89
N (ind/1000 m²)	85.9 ± 109.2	142.5 ± 87.4	50.1 ± 32.4	21.8 ± 10.6	60.4 ± 47.7
S	12 ± 2.5	13.9 ± 2.5	16.3 ± 2.3	13.1 ± 2.8	10.2 ± 2.7
Margalefs diversity	2.8 ± 0.7	2.8 ± 0.6	4.2 ± 0.9	4.1 ± 0.9	2.4 ± 0.7
Shannon diversity	1.1 ± 0.5	1.7 ± 0.2	2 ± 0.2	1.6 ± 0.4	0.5 ± 0.3
Simpson diversity	0.5 ± 0.2	0.8 ± 0.1	0.8 ± 0.1	0.7 ± 0.2	0.2 ± 0.1

lesser weever *Echiichthys vipera*. Four other species are more typical for the offshore area namely red mullet *Mullus surmuletus*, solenette *Buglossidium luteum*, reticulated dragonet *Callionymus reticulatus* and scaldfish *Arnoglossus laterna*. Also, greater and smaller sandeel (*Hyperoplus lanceolatus* and *Ammodytes tobianus*) are characteristic of the offshore clusters, although in lower densities (Figure 5).

3.2.2.1. Coastal communities
Mud

The mud community is dominated by *Pomatoschistus* spp. (54% contribution to w/i group similarity), followed by *P. platessa* (9%) and *S. solea* (8%). Density (86±109 ind. 1000 m^{-2}) is lower compared to the fine sand community but quite higher than the offshore/coarse sand communities. This is one of the least diverse fish communities in the BPNS: Number of species (12±2), Margalefs diversity (3±0.7), Shannon (1±0.5) and Simpson (0.5±0.2) are all low (Table 2).

Fine sand

Four species contribute for around 60% to within-group similarity i.e. *L. limanda* (19%), *M. merlangus* (17%), *Callionymus lyra* (15%) and *Pomatoschistus* spp. (11%). This community has the highest densities (142 ±87 ind. 1000 m^{-2}) and on average 14 (± 2.5) fish species are present per sample (Table 2). Shannon (1.7± 0.2) and Simpson (0.8±0.1) are among the highest compared to the other communities.

3.2.2.2. Offshore communities
Coarse sand

This community is dominated by *E. vipera* (33% contribution to w/i group similarity) with an average density of 9 ind. 1000 m^{-2}. Other species contributing to within-group similarity are *P. platessa* (14%), *Pomatoschistus* spp. (12%), *L. limanda* (8%) and *Arnoglossus laterna* (7%) (Table 2). Density (22±11 ind.

1000 m^{-2}) is very low compared to the other communities. Number of species (13±3) and diversity measures Shannon (1.6±0.4) and Simpson (0.7±0.2) have average values in comparison to the other communities, while Margalefs diversity (4.1±0.9) is among the highest.

Coarse sand top

The coarse sand top community is spatially restricted to the tops of the steep offshore sandbanks (Figure 1) and completely dominated by *E. vipera* (65% contribution to w/i group similarity), with average density of 47 ind. 1000 m^{-2}. Few other species are occurring and only in very low densities, of which *P. platessa* (10%) contributes most to within-group similarity. Average density (60±48 ind. 1000 m^{-2}) is relatively high for this offshore community due to the high abundance of lesser weever, but in terms of biodiversity this is a very poor community, with a very low number of species (10±3) and very low values for Margalefs diversity (2.4±0.7), Shannon (0.5±0.3) and Simpson (0.2±0.1) (Table 2).

Transition

Callionymus lyra (14%), *L. limanda* (14%), *B. luteum* (13%), *Pomatoschistus* spp. (11%) and *P. platessa* (10%) contribute more or less evenly to the first 60% of within-group similarity in this transitional fish community. Species composition is a mixture between species more characteristic for coastal and species more typical for offshore locations (Table 2; Figure 5), making it the most diverse fish community. Density (50±32 ind. 1000 m^{-2}) is average, while number of species (16±2) and Margalefs diversity (4±1) are highest compared to the other communities. Also, Shannon (2±0.2) and Simpson (0.8±0.1) are highest among all fish communities.

4. Discussion

4.1. Epibenthos and fish communities in the BPNS

Both fish and epibenthos communities largely follow similar spatial distribution patterns in the BPNS. For both ecosystem components, there is a clear distinction in communities between the coastal and the more offshore area, situated more or less around 3 NM off the south-western part of the coast and 12 NM off the north-eastern part of the coast. Densities and biomass (the latter only available for epibenthos) are very high within the coastal communities compared to the offshore clusters. For epibenthos, the main species are ubiquitous present in the entire BPNS, i.e. *P. bernhardus*, *C. crangon*, *O. albida*, *O. ophiura*, *A. rubens* and *L. holsatus* occur in all communities. These are indeed the species known to be common in the southern North Sea (Callaway *et al.* 2002). The division between epibenthic communities is mainly due to differences in abundance and/or dominance of one or more of these common species, next to the occurrence of some less abundant but characteristic species, like *Spisula* spp. and squids, that are more common offshore. For fish, there is more a gradient in species from coast to offshore with sole *S. solea*, hooknose *A. cataphractus* and pouting *T. luscus* more characteristic in the coastal communities and lesser weever *E. vipera* dominating the offshore clusters, while dab *L. limanda* and plaice *P. platessa* occur all over the BPNS with varying abundances.

The coastal community of both ecosystem components is further subdivided in a community occurring in more muddy sediments located at the north-eastern part of the coast. These communities largely coincide with the macrobenthic *Limecola balthica* community as described in Breine *et al.* (2018). Number of species and diversity is low within these communities and they are dominated by the brown shrimp, *C. crangon* and gobies, *Pomatoschistus* spp., for resp. epibenthos and fish. These Mud communities are most probably under influence of the outflow of the Scheldt estuary as well. The other community distinguished within the coastal area for both epibenthos and fish seems to be correlated with fine sandy sediments, and largely overlaps with the macrobenthic *A. alba* community (Breine *et al.* 2018). The fine sand communities are the most species rich and diverse communities both for epibenthos and fish in the coastal zone, but highly dominated by resp. the serpent star *O. ophiura* and dab *L. limanda*. Other characteristic species for epibenthos are the netted dog whelk *Tritia reticulata*, the slipper limpet *Crepidula fornicata* and the crab species *L. navigator* and *L. depurator*. For fish, whiting *M. merlangus*, hooknose *A. cataphractus* and common dragonet *C. lyra* are seen as characteristic species in this fine sand community.

The offshore area is characterised by coarser, permeable sediments, harbouring two macrobenthic communities *Nephtys cirrosa* in medium sands and *Hesionura elongata* in coarse sand (Breine *et al.* 2018). Fish and epibenthic communities do not really follow this delineation in spatial distribution. A distinction between the different offshore communities seems to be structured by sand bank topography, and as such probably associated with current patterns. For epibenthos, two offshore communities are observed: a widespread 'offshore coarse sand' community occurring on the northern sandbank systems of the Zeeland banks and Hinderbanks, and a 'coarse sand top' community that only occurs on top of the steeper offshore sandbanks (i.e. Hinderbanks and offshore Flemish banks). The offshore coarse sand community has the highest number of species and is the most diverse epibenthic community with *S. solida*, *S. elliptica*, *L. marmoreus* and squid species *Loligo vulgaris* and *Sepiola atlantica* as typical species. The coarse sand top epibenthos community can be seen as a species-poor version of the offshore coarse sand community, characterised by very low density and biomass and dominated by the hermit crab *P. bernhardus*.

For fish, the most widespread offshore coarse sand fish communities is characterized by intermediate diversity and number of species, and dominated by lesser weever *E. vipera*, with solenette *B. luteum*, scaldfish *A. laterna*, red mullet *M. surmuletus* and reticulated dragonet *C. reticulata* as additional characteristic species. In analogy with epibenthos, a species-poor version of this community, i.e. the coarse sand top community, occurs on top of the steeper offshore sand banks. This is the least diverse community, which is entirely dominated by lesser weever. In contrast to the epibenthos, we observed a slightly different subdivision in communities within the offshore area, with a clear third transitional community. It occurs around the 12 NM area in between the coastal fine sand and the offshore coarse sand fish communities, mainly on the slopes of the northern Zeeland banks and the more coastal southwest Flemish banks. This Transition community is the most diverse fish community with the highest number of species, consisting of a mixture of coastal (e.g. *C. lyra*) and offshore species (e.g. *Buglossidium luteum*).

The importance of sandbank topography in structuring communities on smaller-scale has been shown by previous studies (Ellis *et al.* 2010; Mestdagh *et al.* 2020). In a previous analysis of fish abundances, Buyse *et al.* (2022) also observed the two coastal communities (mud and fine sand), but only one offshore community, namely the coarse sand community. The fact that in our study, different offshore communities were discerned associated with sandbank topography, is most probably related to the higher number of sampling locations, which enabled us to distinguish communities at a higher spatial resolution.

4.2. Implications for future OWF monitoring of epibenthos and fish

The current OWF area in the eastern part of the BPNS largely overlaps with the spatial distribution of the offshore epibenthos and fish communities, except for the most southern concession of Norther, where the coastal fine sand communities occur. In a previous study, we already concluded that Norther exhibited a different epibenthos and fish assemblage than the other concession areas (De Backer & Hostens 2018b). Within the actual concession areas, the majority of the current sampling locations belong to the coarse sand community for both epibenthos and fish. Few locations tend towards the species-poor coarse sand top community, especially on top of the Bligh Bank in the Belwind concession zone. The more diverse transitional fish community occurs at the slope and gully locations neighbouring C-Power, thus in the southern part of the OWF area.

From our analysis, it is clear that the locations inside the OWF concessions cluster nicely together with all non-concession locations in the same respective epibenthos and fish communities. They do not form a separate community or assemblage. This corroborates the conclusion of De Backer *et al.* (2020) that epibenthos and fish assemblages on the soft sediments in between the turbines underwent no drastic changes. Secondary effects suggesting an expansion of the artificial reef effect and some refugium effects due to fisheries exclusion have been picked up already at the species level (De Backer *et al.* 2020; Buyse *et al.* accepted). However, these are not yet at a magnitude to be picked up as such in the overall community analysis.

Nevertheless, the larger contiguous OWF area where fisheries are excluded, is expected to act as a refugium after a certain time and potentially enhance biomass or length of certain species or even change assemblage composition (Handley *et al.* 2008). This refugium effect might potentially lead to 'spillover' (export of biomass to surrounding habitats by recruitment or migration out of the area). The current BACI design used for monitoring the potential impacts of OWFs on epibenthos and fish – with impact and near control locations – is, however, not optimal

to detect such spillover effects. A gradient design would be more suited. This type of design has been used to evaluate the patterns of fish distributions and to elucidate spillover effects in and around marine protected areas (Methratta 2020). One of the advantages of BAG (Before-After Gradient) designs is that there is no need to identify suitable control areas, but instead focus the effort on sampling multiple locations along a distance gradient (Methratta 2020). Nevertheless, it is important that these sampling locations are characterized by similar epibenthos and fish communities, to avoid that differences in abundance or species diversity only reflect differences in communities due to environmental differences rather than real effects of the OWFs.

For sure, it will be important that the difference in community is accounted for in statistical models. In that respect, the analyses conducted in this study are important to select suitable sampling locations. Based on our results, the coarse sand community locations are the best candidates to be included in a gradient design, e.g. locations on the Gootebank, the Hinderbanks, but also the most offshore locations that are currently monitored already by ILVO. Even some of the coarse sand top community locations, the species-poor version of the offshore coarse sand community, may be included since some of the OWF locations (within Belwind) tend towards this community. Of course, then it will be important to account for this in the statistical models, e.g. by introducing a 'community' factor. Another important aspect when studying spillover/refuge effects is to not only focus on fish abundances, but also to look at fish life history and population characteristics, such as age, length, weight and sex ratio to enhance the detection of potential refuge effects (Florin *et al.* 2013). Including this in future gradient monitoring for a selection of commercially important species, such as plaice and dab, will help to further elucidate OWF effects.

Regarding the newly designated Princess Elisabeth area, few sampling locations have been investigated by means of an 8 m beam trawl in the past. Especially the largest south western zone is data poor. The locations that are currently sampled within the ILVO environmental monitoring programs are mainly located on top of steeper sandbanks, which harbour the species-poor coarse sand top epibenthos and fish communities. Based on the known locations from the wider surroundings (e.g. in the north-eastern offshore zone), the coarse sand community is certainly to be expected within the south-western zone of the Princess Elisabeth area. For sure, the topography of this area is very heterogeneous with steep sandbanks, so it will be important to take this into account in future monitoring and modeling. Additionally, relict gravel beds with high biological value are to be expected in the area, as suggested by the potential gravel distribution map of the BPNS (Van Lancker *et al.* 2007; Pecceu *et al.* 2021). Due to the high heterogeneity and poor data availability, a thorough before-impact monitoring will be essential to enable assessing the future effects. Follow-up of these gravel beds may require a different approach, but to properly evaluate the potential effects of OWFs on the soft sediment epibenthos and fish communities, a gradient design from the start will be the most suitable solution, since this will improve our understanding of how OWFs influence species distribution patterns (Methratta 2020).

Acknowledgements

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CHAPTER 3

INCORPORATING HYPERBENTHOS SAMPLING IN OWF MONITORING SURVEYS: CURRENT KNOWLEDGE, CHALLENGES AND OPPORTUNITIES

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Abstract

In this study, findings are reported from the first large-scale hyperbenthic monitoring survey conducted in two Belgian offshore windfarms (OWFs) that differ in terms of local habitat conditions, turbine foundation type and timing of construction. Three replicates (tracks) were collected inside and outside each OWF and the applied sampling design was used to describe the general hyperbenthic communities within these areas and to assess potential OWF-related impacts through a spatial analysis (inside vs outside tracks) as it is proposed that the cumulative effects of turbine presence ("artificial reef effect") and cease of fishery activities ("fisheries exclusion effect") might result in enriched hyperbenthic communities within the OWFs. Hyperbenthic distributions at the C-Power study site can be defined as a "mixture community", receiving influences from offshore and transitional communities. Communities at the Norther study site corresponded with transitional communities, but also harboured species that are found in more nearshore areas and exhibited considerable variation within the concession zone. Total densities, diversity and community composition differed between the inside and outside areas at C-Power and while these trends are aligned with the enrichment hypothesis, it remains unclear whether these spatial differences can be attributed to the presence of the OWF. Results at the Norther study site did not indicate similar trends and it is suggested that the lack of OWF-related effects may be attributed to the "young" lifespan of the Norther site (operational since 2019–2020) and the habitat heterogeneity that characterizes this OWF. Despite the valuable insights gained within this study, it also revealed that increased sampling efforts are needed to enhance the ability to fully characterize the hyperbenthic communities and strengthen the statistical power to detect OWF-related impacts, especially for the Norther OWF. It is also recommended to include relevant abiotic variables such as the near-bottom water pigment concentration, seabed granulometry and organic matter content within the sampling design.

1. Introduction

Research on hyperbenthos is a relatively "young" discipline and this group of animals in the water column, living on or associated with

the seabed aroused general interest among researchers only from the late 1950s (Mees & Jones 1997). Over time, hyperbenthos has been known by a variety of names such as "suprabenthos", "demersal zooplankton" or "benthopelagic plankton", leading to a lack of synthesis across studies (Mees & Jones 1997). Mees & Jones (1997) were the first to introduce a general classification of this group into mero- and holohyperbenthos, based on their time spent within the hyperbenthic zone. Holohyperbenthos are animals that spend variable periods of their adult life within this zone and comprise groups such as peracarids, copepods, chaetognaths and hydromedusae (Mees & Jones 1997; Dewicke *et al.* 2003). In contrast, merohyperbenthos species are only found within the hyperbenthal zone during early life stages and include larval decapods, polychaetes and fishes (Mees & Jones 1997; Dewicke *et al.* 2003). Organisms of this ecosystem component are also found in the dynamic, lowest layers of the water column, comprise a broad assemblage of species with diverse traits (*e.g.* morphology, vertical position and mobility) and are known to exhibit a variable distribution, both in space and time. All of these aspects complicate efficient sampling, and specialized sampling devices such as the hyperbenthic sledge have only been developed and used since a few decades (Mees & Jones 1997; Dewicke *et al.* 2003). This type of equipment and its success also strongly depend on local conditions such as ship capacities, weather conditions, depth and seafloor topography, where a relatively flat and even seabed is required for a successful deployment (Mees & Jones 1997; Lefaible *et al.* 2019a). This was confirmed during a feasibility study to sample hyperbenthos in two Belgian offshore wind farms (OWFs) constructed within sandbank systems (Lefaible *et al.* 2019b), which revealed that samples taken along previously used epibenthic tracks (1 km) resulted in several non-representative samples due to high accumulation of sediment in the hyperbenthic sledge collectors because of the relative position of the tracks to the sand ridges (Lefaible *et al.* 2019b).

The issues associated with disparate terminology, the intrinsic dynamic characteristics of these organisms and the difficulties to perform a representative quantitative sampling have resulted in a low number of descriptive baseline studies on hyperbenthic distributions (Mees & Jones 1997; Parry *et al.* 2021). Within the available literature, there is also a strong focus on two habitat types, namely estuaries and the surf zone on beaches, while there is little information on hyperbenthic communities in deeper, offshore areas (Mees & Jones 1997). A study by Dewicke *et al.* (2003) represents one of a few extensive surveys on hyperbenthic spatial patterns in the Belgian part of the North Sea (BPNS). This study detected six biological communities that could be linked with the different subtidal sandbank systems. Trends in community density and biomass principally followed an onshore-offshore gradient, while an east-west gradient was observed for diversity, which was most pronounced in the onshore areas (Dewicke *et al.* 2003). In general, hyperbenthic abundance and diversity were significantly lower at the offshore Hinder Banks compared to the Flemish and Zeeland Banks located closer to the coast, and it was concluded that distance to the coastline, current direction and habitat heterogeneity were the most important structuring factors for larger-scale hyperbenthic distribution patterns (Dewicke *et al.* 2003).

Despite the limited knowledge of this ecosystem component, its importance in the functioning of marine ecosystems has been widely recognised (Mees & Jones 1997; Dewicke *et al.* 2003; McGovern *et al.* 2018; De Neve *et al.* 2020). Due to their vertical position within the marine system, at the interface between sediment and water column, it is expected that these organisms play an important role in the exchange of energy, nutrients and biomass between the seabed and the water column (Mees & Jones 1997; De Neve *et al.* 2020). Besides their importance for bentho-pelagic coupling, hyperbenthos also fulfil a supporting role as a

food source for higher trophic levels (Mees & Jones 1997; De Neve *et al.* 2020). For example, stomach content analysis revealed that several hyperbenthic taxa, and certainly mysids, constitute a major component of the diet of many demersal fish and epibenthic crustaceans throughout different stages of their life cycles (post-larval, juvenile and adult), which also implies that the "hyperbenthic zone" functions as a nursery area for these often commercially important species (Mees & Jones 1997; De Neve *et al.* 2020). Because of the strong analogy between hyperbenthic distribution patterns and those of other benthic assemblages such as macrobenthos, it is believed that these organisms might also be influenced by the altered habitat characteristics created by the presence of wind turbines (Dannheim *et al.* 2020; Lefaible *et al.* 2019b, 2021). On the one hand, altered hydrodynamics around the turbines create "refugium" areas with decreased current velocities, which might increase the settlement of passively dispersing, planktonic species within those areas and provide sheltered against hydrological forcing for actively swimming hyperbenthic organism (Mees & Jones 1997). On the other hand, the combined effects of organic enrichment and increased deposition of organic matter by the turbine-associated epifauna (Dannheim *et al.* 2020) may attract more mobile species that can actively migrate to these areas with higher food availability, as has been demonstrated for several demersal and benthic fish species (Vandendriessche *et al.* 2013; Reubens *et al.* 2014). It is therefore hypothesised that these turbine-related effects on the sediment could create more favourable conditions and the concomitant establishment of richer hyperbenthic communities within the OWFs. Moreover, it is not allowed to trawl in these areas, and thereby the OWFs function as de facto marine protected areas (MPAs). Bottom fishing activities negatively affect benthic and pelagic ecosystem components and can be considered as one of the most important disturbances within the marine realm (Johnson 2002). It is therefore

expected that the removal of this pressure within the concession zones could allow the recovery of seabed-water interface habitats and communities, with concomitant positive impacts on higher trophic levels due to their important function as a nursery area and food supply for many fish and crustaceans (Vandendriessche *et al.* 2013; Reubens *et al.* 2014). As a result, monitoring hyperbenthos could also be a propitious method to assess long-term impacts of the "fishery exclusion effect" of OWFs.

The main objective of this study is to investigate potential OWF-related impacts on hyperbenthic communities for two concession zones that differ in terms of local habitat characteristics, turbine foundation types and timing of construction. To achieve this, samples were collected from areas located inside and outside the OWFs to perform a spatial analysis in which we want to test whether OWF areas harbor enriched hyperbenthic communities. Furthermore, it will also be verified whether the sampling method (*i.e.* sample collection and processing) resulted in a qualitative description of the hyperbenthic communities within these areas. Through these findings, we want to contribute to the general knowledge of this poorly studied ecosystem component and highlight the importance of their inclusion within OWF monitoring surveys.

2. Material and methods

2.1. Study areas

Sampling was performed in two operational OWFs situated in the eastern concession zone of the Belgian Part of the North Sea (BPNS), namely C-Power and Norther. Whereas these concession zones are located relatively close to each other, they differ in terms of local habitat characteristics, turbine foundation types and timing of construction. C-Power was constructed on the Thornton bank (TB) at an intermediate distance from the coastline (30 km) relative to the most offshore situated OWF (Rumes & Brabant 2017). This OWF is composed of 6 gravity-based and 48 jacket

foundations and became fully operational in 2013 (Rumes & Brabant 2017). Norther is located 23 km from the Belgian coastline, southeast from the TB and represents the most nearshore OWF within the concession area (Lefaible *et al.* 2021). The construction phase of the Norther OWF ended in 2019, with the installation of 44 monopiles, which also makes this OWF the most recent operational park (Lefaible *et al.* 2021).

2.2. Sampling strategy

A feasibility study performed in 2018 revealed that sampling locations based on existing epibenthic tracks (1 km towing distance) at C-Power did not result in adequate hyperbenthos samples due to a high accumulation of sediment within the sample collectors (Lefaible *et al.* 2018). Therefore, it was proposed to perform several (3 tracks), shorter (150 m towing distance) replicate tracks in which local topography (*i.e.* position of sand ridges) is taken into account, based on digital terrain models for the C-Power study site provided by the Federal Public Service Economy (FOD Economie). The new sampling strategy was applied during a one-day sampling cruise (3/11/2021) on board the *Simon Stevin* in the C-Power and Norther concession areas where three replicate tracks were taken within each concession area and reference areas outside the OWFs (Fig. 1). The hyperbenthic sledge on the *Simon Stevin* consists of two nets: the lower net samples water depths from 0.2 to 0.5 m, and the upper net from 0.5 to 1 m above the bottom; both nets have a 1 mm mesh size. A flowmeter was installed in the lower net to calculate the volume of water filtered during each sampling event. An average volume of 71 ± 21 m³ was filtered through the nets during the sampling in and around the C-Power OWF. Average volumes were lower in the inside the OWF (n = 3, 65 ± 19 m³) compared to the reference area (n = 3, 77 ± 23 m³). For the Norther study site, an average volume of 40 ± 5 m³ of water was filtered through the nets at the Norther study site and the average amount of water flow was comparable between areas inside

and outside the OWF (38 ± 5 m³ and 42 ± 5 m³). Tows were carried out during daytime conditions and hauled counter current at the lowest towing speed possible (1.5 knots) according to the strategy applied by Dewicke *et al.* (2003) to allow a descriptive comparison with results found in this study.

2.3. Sample processing and analysis

Samples obtained from the lower and upper nets were immediately rinsed on board with seawater over a 1 mm sieve, collected into separate 1L pots and preserved with seawater-buffered 4% formaldehyde. In the laboratory, organisms were sorted, counted and identified to the lowest possible taxonomic level. Further specification on the life stage of the organism (megalopa, larva and juvenile) was also applied for certain classes such Polychaeta, Malacostraca (infraorder: Caridea, Anomura, Brachyura) and Actinopterygii. Following DeWicke *et al* (2003), typical macrobenthic and non-hyperbenthic species (juvenile and adult fish and decapods, fish eggs and ectoparasitic organisms) were removed from the dataset. Total densities for each sample were obtained by standardizing the data to individuals per 100 m³ as follows: ind./100 m³ = number individuals /(surface net * number of turns flowmeter*0.3) * 100. The multiplication by 0.3 within this formula was added to account for the fact that the flowmeter used in this study increased by one unit for every three rounds. Due to time limitations, biomass could not be determined and the upper-net samples collected in the Norther OWF still need to be processed. Diversity indices were calculated based on raw count data and included species richness (S), Shannon-Wiener diversity index (H') and Pielou's evenness (J'). Species reported on a higher taxonomic level were considered as "unique" if no other representative of the same taxon level was present or if they were distinctly different (morphospecies).

Relative contributions of the major observed high-level taxonomic groups (Class and Order) and species to overall total
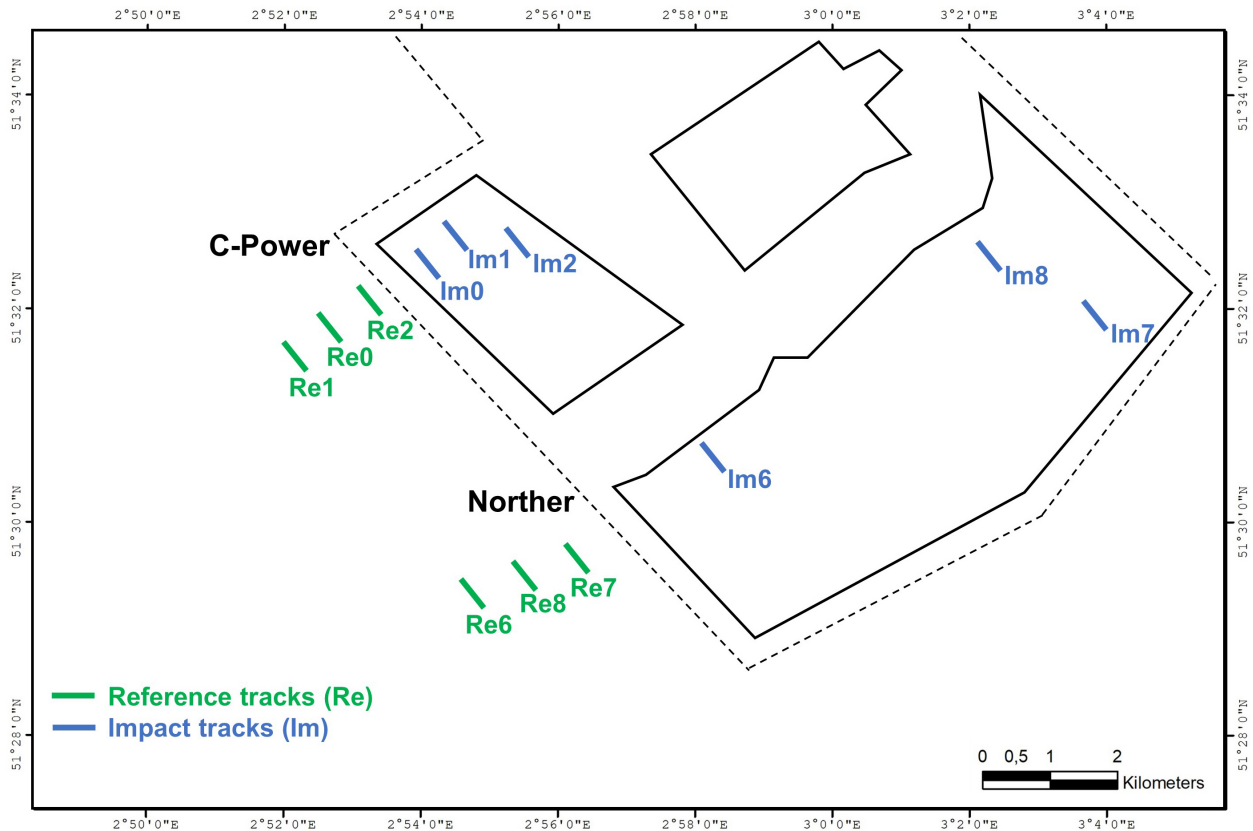
Figure 1. Location of the two windfarm concession areas under study (upper: C-Power, lower: Norther) sampled during the 2021 monitoring campaign. Positions of the tracks are depicted with green lines for the reference tracks (Re: Re0-3 and Re6-7) taken outside the OWFs and blue lines for the impact tracks (Im; Im0-2 and Im6-7) taken inside each OWF.

densities for each sample were calculated and used to describe the hyperbenthic community composition patterns for the inside and outside areas at each OWF. In addition, Species Accumulation Curves (SACs) were constructed for the lower net samples from each area (inside and outside the OWFs) for C-Power and Norther to assess the level of accuracy of the monitoring survey. Only the lower net samples were used to allow a comparison between both OWFs as the upper net samples of the Norther OWF could not be included within this study. Moreover, several non-parametric richness estimators (Chao1, Jacknife1, Bootstrap) were also calculated to estimate the number of unrecorded species.

OWF-related effects were investigated by means of a spatial comparison to test differences between the sampled areas (inside OWF vs outside OWF) for the structural univariate indices under study. For the C-Power study site, this was done by a two-way ANOVA (Factors: "Position" and "Area" with levels: "lower", "upper" and "inside", "outside"), while a one-way ANOVA (Factor: "Area" with levels "inside", "outside") was used for the Norther study site. Assumptions of normality and homogeneity of variances were tested by Shapiro-Wilk and Levene tests, respectively, and log transformations were performed if these assumptions were not met. Moreover, a posteriori power analyses were performed to assess the statistical power of the applied sampling design to detect differences between the areas inside and outside the OWFs for two important univariate community descriptors (total densities and Shannon-Wiener diversity). This was done by means of the pwr.anova.

test (k = , n = , f = , sig.level = , power =) function, based on the fitted models. In order to allow a correct comparison between the two OWFs under study, these models were fitted for the lower net samples only. The power analysis function was based on four parameters, namely the sample size (k and n), the effect size (f), the significance level (default of 0.05) and the power or probability to detect an effect. Within this analysis, the sample size (n) corresponds to the three lower samples, taken within each subgroup (k = 2, inside vs outside). The effect sizes (f) were calculated through the etaSquared() function, for the sum of square values obtained from the fitted models. This allowed to calculate the required sample size per subgroup to detect effects with a power of 0.80 (*i.e.* 80% likelihood to detect an effect). In addition, the statistical power was calculated for the currently applied sampling design (k =2 with n = 3) and to predict the statistical power under two theoretical scenarios of increased sampling effort (k = 2 with n = 6, k = 2 with n = 12). Comparable to the univariate analysis, a two-way and one-way Permanova test was performed to assess OWF-related effects on the community composition for C-Power and Norther, respectively. Homogeneity of

multivariate dispersions was tested using the PERMDISP routine (distances among centroids). Species Indicator Analysis (SIA, package "indicspecies") to test which species were most responsible for the differences in community composition. Patterns in terms of community composition were visualized by means of non-metric multidimensional scaling (nMDS) plots, constructed through a Bray-Curtis matrix. All the multivariate analyses were based on relative abundance data, to exclude effects of differences in total abundances between samples.

3. Results

3.1. Hyperbenthic distribution patterns

Within the C-Power study site, a total of 54 species was found, originating from 11 higher taxa (Class/Order) and total densities per sample ranged from 930 ind. 100 m^{-3} to 2197 ind. 100 m^{-3} (Fig. 2). Relative contribution calculations showed that community compositions were clearly dominated by three major groups. Cnidaria, identified as medusa of the class Hydrozoa, accounted for 50 % of total densities. Malacostracan crustaceans of the order Amphipoda also contributed meaningfully to overall abundances (29 %)
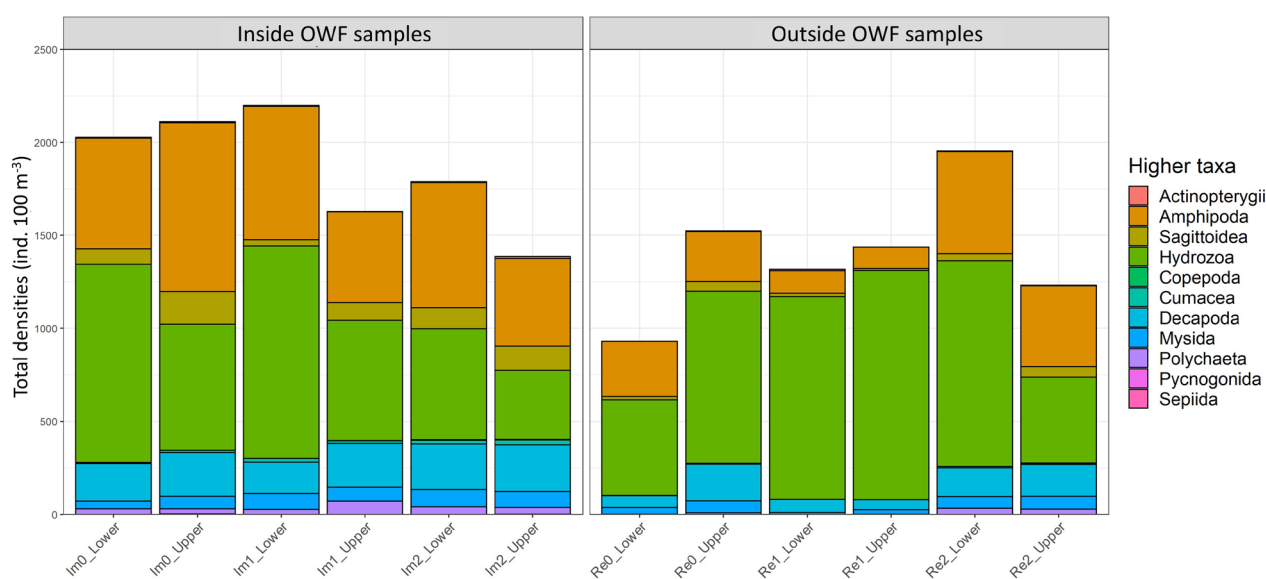
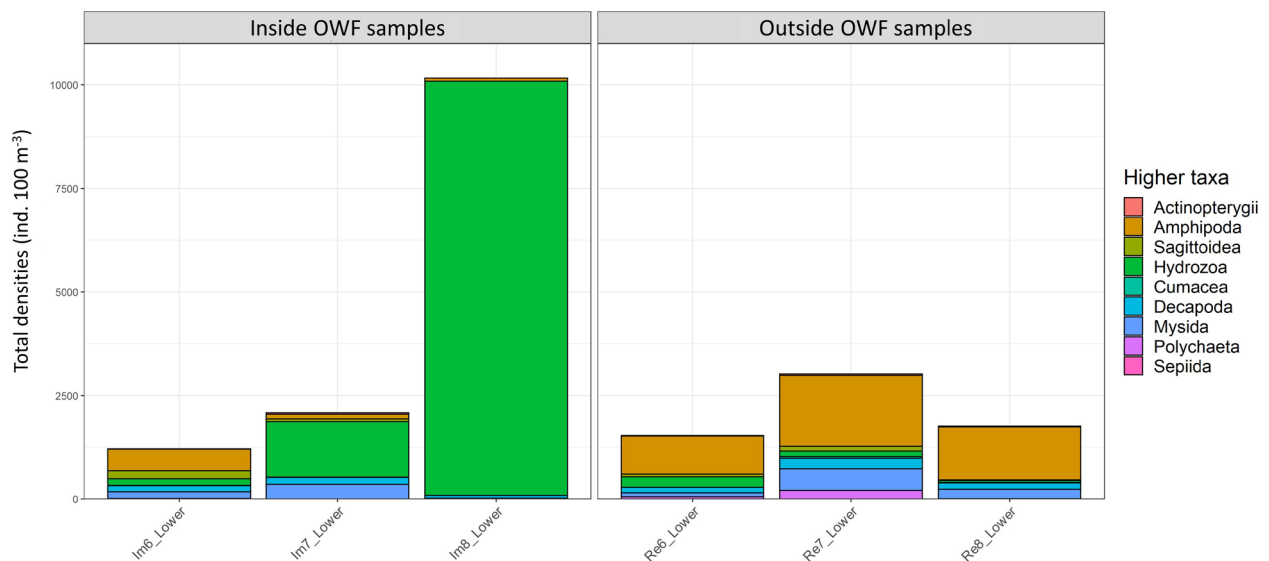**Figure 2.** Total hyperbenthic densities (ind. 100 m^{-3}) per sample for the areas located inside (impact) and outside (reference) the C-Power study area. Relative contributions of the major lower-level taxa to overall densities are also included.

and this group was mainly represented by organisms of the family Atylidae (*Nototropis* sp.). A third dominant group was larval and megalopa life stages of different Decapoda, such as *Pisidia longicornis* and *Macropodia rostrata*, making up about 11 % of total densities. Sagittoidea (*Sagitta elegans*) and Mysida (mainly *Mesopodopsis slabberi*, *Acanthomysis longicornis* and *Gastrosaccus spinifer*), contributed to a lesser extent (4%) to overall densities. All of these species were detected in each sample and are therefore considered as characteristic hyperbenthic species within the C-Power samples.

The lower net samples of the Norther OWF contained a total number of 40 species, divided over 9 higher level class and order taxa (Fig. 3). Total densities showed considerable variation and ranged from 1205 ind. 100 m^{-3} to 10157 ind. 100 m^{-3}. One sample (Im8) taken inside the OWF, clearly deviated from the other ones in terms of total densities, which could be attributed to the extreme abundance of hydromedusae (9990 ind. 100 m^{-3}). If this group was not considered, this sample actually proved to be the most impoverished one in terms of total abundance (166 ind. 100 m^{-3}) and diversity

(S: 17, H: 0.12). Community composition at the Norther site was dominated by Hydrozoa (Hydromedusa) and Amphipoda (*Nototropis* sp.), which collectively contributed about 80% to total abundances. However, Mysida also comprised about 7% and were mainly represented by the species *Gastrosaccus spinifer*, *Schistomysis* sp. and *Mesopodopsis slabberi*. Megalopa and larval life stages of the Decapoda group (*Pisidia longicornis* and morphospecies 1) comprised another 5% to overall abundances, while many other groups contributed to a lesser extent: Sagittoidea (*Sagitta elegans*, 2%), Polychaeta (*Lanice conchilega* larva, 1.5%) and Actinopterygii (fish larvae, 1%). All of the above-mentioned species were shared among all samples except for the sample Im8 sample. The latter sample did not contain the species *Sagitta elegans*, *Schistomysis* sp. and fish larva, and overall community composition was comparable to the one described for the C-Power site.

Figure 4 visualizes the cumulative number of species recorded as a function of the sampling effort (lower net samples, n = 3) for the different areas under study. The resulting SACs indicate that the largest share of newly recorded species are found

Figure 3. Total hyperbenthic densities (ind. 100 m^{-3}) per sample for the areas located inside (impact) and outside (reference) the Norther study area. Relative contributions of the major lower-level taxa to overall densities are also included.

between the first and second sample, while the curves seem to already level off slightly towards n = 3, especially for the samples taken inside the C-Power OWF (Fig. 4A) and the reference area outside the Norther OWF (Fig. 4D). Based on the richness estimator values, it appears the number of "undetected" species was found to be rather low, but some variation was seen between the sampled areas. Highest absolute differences between the estimated richness values and the recorded species richness (S) were found for the samples inside the Norther OWF and those

collected outside the C-Power OWF, ranging between 4 and 8 species. For the other two sampling areas (inside the C-Power OWF and outside the Norther OWF) these values ranged between 1 and 5 species.

3.2. OWF-related impacts: inside vs outside areas

Results from the two-way ANOVA analysis for the C-Power study site revealed that the average values for all the univariate variables were comparable between the lower

S: 40
Chao 1: 41 ± 2
Jack1: 45 ± 4
Bootstrap: 42 ± 3
(A)

S: 37
Chao 1: 41 ± 3
Jack1: 45 ± 6
Bootstrap: 41 ± 4
(B)

S: 32
Chao 1: 36 ± 3
Jack1: 39 ± 5
Bootstrap: 36 ± 3
(C)

S: 36
Chao 1: 38 ± 2
Jack1: 41 ± 4
Bootstrap: 39 ± 3
(D)
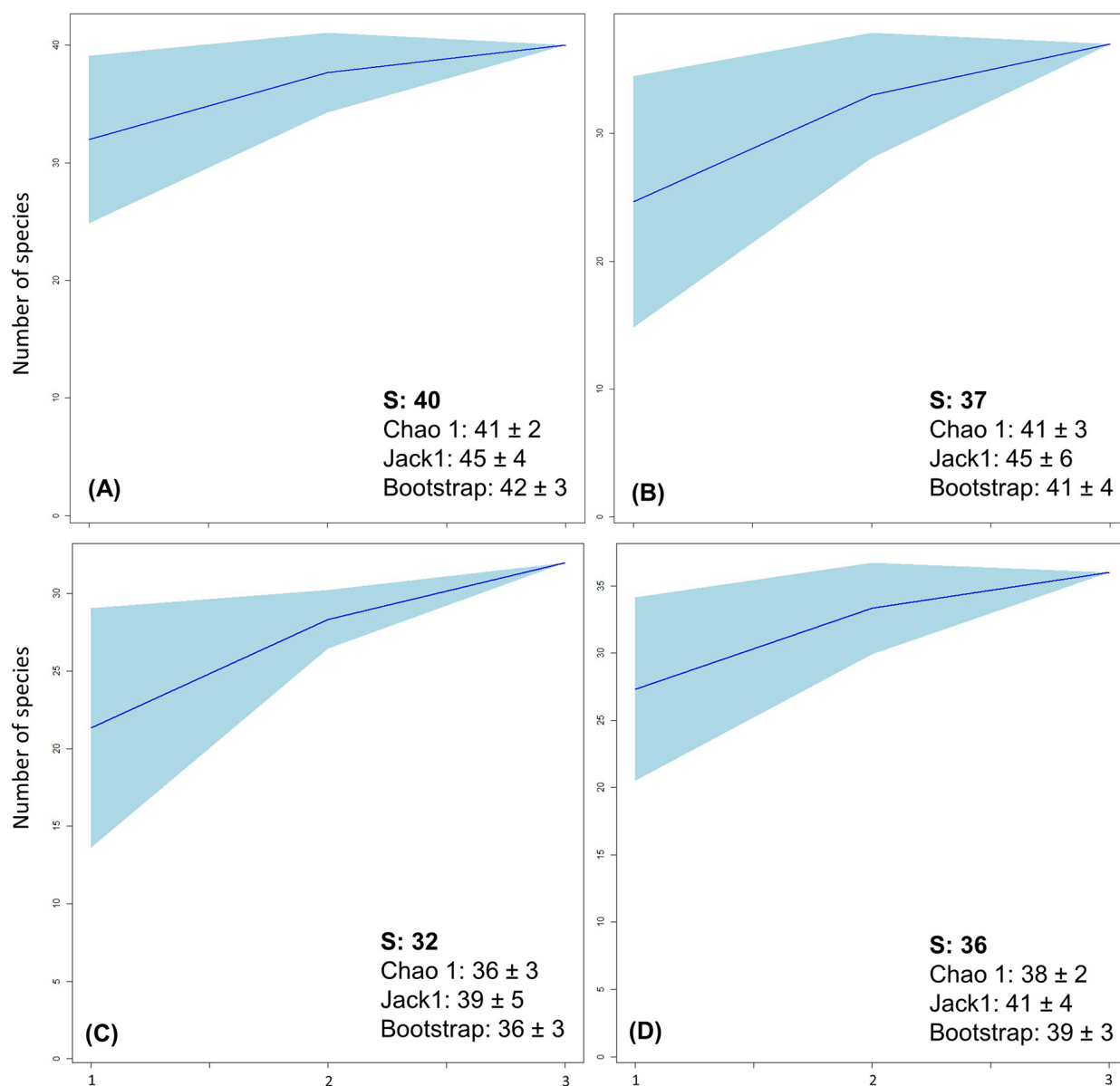
Figure 4. Species Accumulation Curves (SACs) for the samples taken inside (graphs A and C) and outside (graphs B and D) the C-Power (upper) and Norther (lower) study site. For every area, the species richness (S) is given, together with several richness estimators: Chao1, Jacknife1 and Bootstrap (mean ± SE).

and upper net samples (TN/S/H'/J', factor "Position": p > 0.05) and that no significant interactions were found (TH/S/h'/J', factors "Area" and "Position": p > 0.05). In contrast, all of the univariate community descriptors showed higher average values inside the OWF compared to the reference area located outside the concession zone (Table 1) and significant spatial differences were detected between the inside and outside areas for the total abundance (TN, factor "Area": p = 0.049), the Shannon-Wiener diversity (H', factor "Area": p = 0.024) and the Pielou's evenness (J', factor "Area": p = 0.033). The statistical power analysis was performed for the sample size estimation, based on the models to compare the total abundance (TN) and Shannon-Wiener diversity (H') between the inside and outside areas at C-Power. The effect sizes were found to be 0.94 and 1.04 for TN and H' respectively. Given these sample sizes, a significance level of 0.05 and a power of 0.80, the projected sample size per subgroup or area (inside vs outside) would be n = 6 for TN and n = 5 for H'. Based on the applied model, with n = 3 (3 lower samples for each subgroup) and the calculated effect sizes, the computed power proved to be 0.42 for TN and 0.49 for H'. If the number of samples per subgroup would be hypothetically multiplied by two (n = 6), the estimated power would increase to 0.83 and 0.90 for TN and H respectively, while an additional increase to n = 12 would further increase the estimated power to approximately 1 (0.99) for both variables.

A similar trend was found for the community composition, which was significantly different between the areas under investigation at C-Power (two-way Permanova, factor "Area": p = 0.030), which is also visualized through an nMDS plot in Figure 5. Indicator Species Analysis, revealed that a total of 8 species were significantly associated with the "impact group" and the majority of them corresponded to the highest ranked species from the relative abundance analysis. The most distinct species associated with the impact samples (p-value < 0.01, '**') included: *Bodotria* sp. (Cumacea) and *Crangonidae* sp. (Decapoda, Caridea) and other species (p-value < 0.05, '*') included *Sagitta elegans* (Chaetognatha), *Macropodia rostrata* megalopa (Decapoda), *Nototropis* sp. (Amphipoda), *Lanice conchilega* larva (Polychaeta), *Pseudocuma* sp. (Cumacea) and *Pariambus typicus* (Amphipoda). For the reference samples, the SIA only revealed a strong association with one taxon, namely *Hydromedusa* (Cnidaria, p-value < 0.05, *).

While average total densities were higher for the samples taken inside the Norther OWF compared to the reference samples, an opposite trend was found for the diversity indices (Table 2). Except for the average total abundance (TN, factor "Area": p = 0.046), the one-way ANOVA analysis did not reveal any significant differences for the other structural community descriptors between the impact and reference samples (S/H'/J', factor "Area: p > 0.05, Table 2).

Table 1. Overview of calculated community descriptors (mean ± SE and p-values) for the spatial comparison between samples taken inside and outside the C-Power OWF.

C-Power – univariate results	Inside OWF	Outside OWF	Main effect (Area)
Total densities (N, ind. 100 m^{-3})	1856 ± 140	1399 ± 152	p = 0.049 (*)
Number of species (S)	35 ± 3	28 ± 2	p = 0.058 (NS)
Shannon-Wiener (H')	1.80 ± 0.10	1.26 ± 0.20	p = 0.024 (*)
Pielou's evenness (J')	0.51 ± 0.03	0.37 ± 0.05	p = 0.033 (*)

*Signif. codes: '***'p < 0.001, '**'p < 0.01, '*'p < 0.05, 'NS'p > 0.05*

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Table 2. Overview of calculated community descriptors (mean ± SE and p-values) for the spatial comparison between samples taken inside and outside the Norther OWF.

Norther – univariate results	Inside OWF	Outside OWF	Main effect (Type)
Total densities (N, ind. 100 m^{-3})	4483 ± 3488	2105 ± 567	p = 0.046 (*)
Number of species (S)	22 ± 3	28 ± 2	p = 0.174 (NS)
Shannon-Wiener (H')	1.20 ± 0.69	1.55 ± 0.16	p = 0.565 (NS)
Pielou's evenness (J')	0.37 ± 0.21	0.47 ± 0.04	p = 0.622 (NS)

*Signif. codes: '***'p < 0.001, '**'p < 0.01, '*'p < 0.05, 'NS'p > 0.05*

Comparable to the power analysis conducted for C-Power, sample size estimations were performed for the total abundance (TN) and Shannon-Wiener diversity (H') between the inside and outside areas at the Norther OWF. These analyses revealed that the effect sizes were considerably lower (TN: 0.41 and H': 0.32) compared to C-Power and that approximately 24 samples and 39 samples would be necessary for each subgroup (inside vs outside) given a significance level of 0.05 and a power of 0.80. Based on the applied model, with n = 3 (3 lower samples for each subgroup) and the calculated effect sizes, the computed power proved to be 0.12 for TN and 0.09 for H'. If the number of samples per
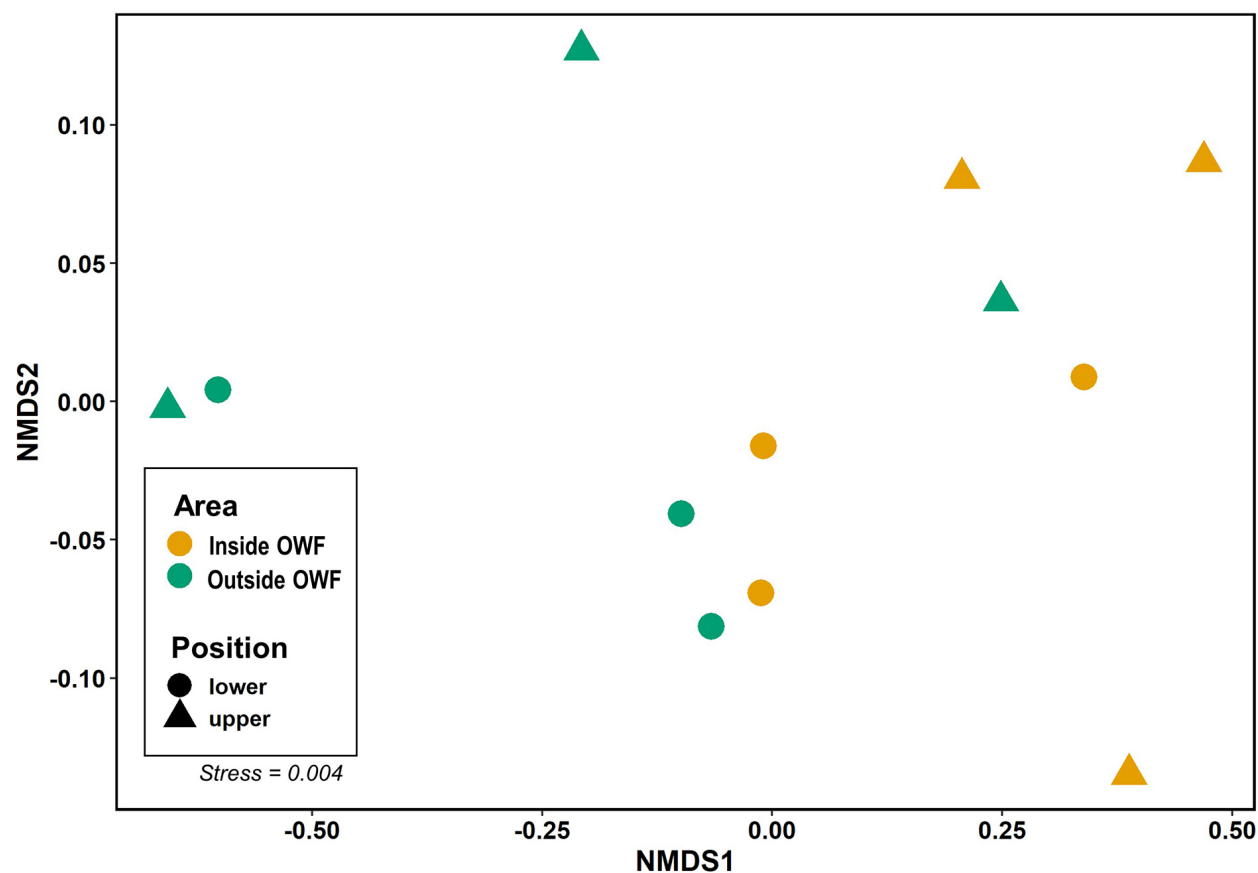
Figure 5. Non-metric multidimensional scaling (NMDS) plot for the lower and upper samples collected for the area inside (orange) and outside (green) the C-Power OWF, based on relative abundance data.

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subgroup would be hypothetically multiplied by two (n = 6), the estimated power would increase to 0.25 and 0.17 for TN and H respectively, while an additional increase to a sample size of n = 12 would further increase the estimated power to approximately 0.48 and 0.32 for TN and H' respectively. Community composition analysis showed no significant differences between both areas at Norther (one-way Permanova, p > 0.05), and the Species Indicator Analysis did not show a selected number of species for any of the groups under study.

4. Discussion

4.1. Spatial distribution patterns of hyperbenthic communities

Extensive sampling by Dewicke *et al.* (2003) in the BPNS revealed the presence of a clear onshore-offshore gradient in terms of hyperbenthic distributions and identified a total of six hyperbenthic communities situated within onshore, transitional and offshore clusters. It was concluded that hydrodynamical forces and habitat heterogeneity can be considered as the most important structuring factors for hyperbenthic distributions in this North Sea area (Dewicke *et al.* 2003). Offshore areas are controlled by strong ebb-dominated currents (SW-direction) and are characterized by coarser sediments and lower settlement rates of suspended material (Dewicke *et al.* 2003). Hyperbenthic communities in these areas such as the Hinder offshore and Zeeland transitional communities are therefore mainly composed of planktonic taxa, which are less dependent on deposited organic matter, such as hydromedusae, chaetognaths (Sagittoidea) and early life stages of many benthic organisms (merohyperbenthos) which also have a planktonic lifestyle (Dewicke *et al.* 2003). In terms of geographic location, the C-Power OWF is situated at the boundary between the transitional and offshore clusters and its presence within this convergence zone could also lead to passive accumulation of organisms and concomitant higher abundances of merohyperbenthic taxa (Mees & Jones 1997;

Dewicke *et al.* 2003). These patterns were observed in C-Power where densities were dominated by the planktonic hydromedusae and, to a lesser extent, chaetognaths, together with individuals of *Nototropis* sp., belonging to the merohyperbenthos. Other important representatives of the latter group were larval stages of the decapod crustaceans *Pisidia longicornis* and *Macropodia rostrata*, in agreement with results of Dewicke *et al.* (2003). It can therefore be concluded that hyperbenthic communities at C-Power are located at a node receiving influences from the three surrounding communities (Hinder offshore, Flemish and Zeeland transitional) described by Dewicke *et al.* (2003).

The Norther OWF can also be situated within the Zeeland transitional area, but is located more nearshore compared to C-Power and is not characterized by the presence of a subtidal sandbank (Lefaible *et al.* 2021). Nearshore areas are mainly influenced by flood-dominated currents and are believed to experience less intense hydrological forcing (Dewicke *et al.* 2003). This also implies that these areas experience higher suspended matter concentrations compared to offshore areas, with locally increased levels of organic matter deposition (Dewicke *et al.* 2003). The majority of the samples collected at the Norther study site were clearly less dominated by the typical planktonic groups such as hydromedusae and Chaetognatha, and showed higher abundances of merohyperbenthic groups, such as amphipods or larval decapods, compared to the samples collected at C-Power. In addition, organisms of the order Mysida contributed more to total abundances in Norther compared to C-Power. Mysids are motile, omnivorous organisms that often reach high densities in areas with increased food availability and are important representatives of nearshore hyperbenthic communities as their distribution is also strongly linked to the existing onshore-offshore gradient (Mees & Jones 1997; Dewicke *et al.* 2003; Parry *et al.* 2021). The importance of this group and especially the occurrence of the "characteristic" species

Schistomysis sp., which is typically associated with more onshore located hyperbenthic communities, implies that the community composition at the Norther study site differs from more offshore situated OWFs (Dewicke *et al.* 2003). Previous studies within the Norther OWF already revealed that this area is very heterogeneous both in terms of abiotic and biotic conditions and constitutes a mosaic of habitat types (Lefaible *et al.* 2021). This finding was again confirmed by the strong variation in terms of hyperbenthic distributions that was found in this study within the impact area. Two impact tracks (Im6 and Im7) were taken within an area that has been categorized as Habitat Type 1 (HT1), which is characterized by fine, organically enriched sediments that contain significant amounts of coarser material (Lefaible *et al.* 2021). In contrast, the third impact sample (Im8) was collected in the Habitat Type 3 (HT3), which corresponds with medium-coarse sands with relatively low organic matter and is typically associated with subtidal sandbank systems (Lefaible *et al.* 2021).

4.2. OWF-related effects in the hyperbenthic zone

The introduction of underwater structures within mobile, soft-sediments alters hydrological conditions in the wake of the turbine, resulting in the creation of sheltered areas with the deposition of finer sediments and increased retention of deposited organic matter (Danheim *et al.* 2020. The turbines also provide new habitat for the colonization of hard substrate species which enhances overall habitat complexity and biodiversity through the so called artificial reef effect (Danheim *et al.* 2020; Degraer *et al.* 2020). Established epifouling communities (> 6 years) are mainly composed by suspension feeders such as *Mytilus edulis* and have the ability to organically enrich the surrounding sediment through the deposition of faecal pellets (Degraer *et al.* 2020). Moreover, biofouling "drop-offs" from the turbines can occur, which might expand the impact footprint

of the artificial reef effect to areas located further away from the turbines (Lefaible *et al.* in prep.). Previous studies within the C-Power OWF have already revealed that the long-term presence of the jacket foundations and combined impacts of the artificial reef effect and altered hydrological conditions (sediment fining and organic enrichment), have led to a shift towards richer macrobenthic communities at distances of approximately 38 meters from the turbines (Lefaible *et al.* 2019a; Braeckman *et al.* 2020; Lefaible *et al.* in prep.).

It is therefore considered that certain results within this study such as the lower average water flow together with a higher hyperbenthic abundance and diversity inside the C-Power OWF compared to the reference area, could be a consequence of these turbine-related impacts. Moreover, significantly different hyperbenthic communities were found between the areas inside and outside the OWF and these findings also seem to support the hypothesis that the creation of more favourable conditions related to turbine-induced habitat changes could result in enriched hyperbenthic communities (Lefaible *et al.* 2021). For example, a relative abundance analysis showed that mysids such as *Mesopodopsis slabberi* and *Gastrosaccus spinifer* occurred more frequently inside the OWF. These motile and omnivorous/detrivorous organisms reach higher densities in areas of increased food availability, which might be a reason for their higher presence within the OWF (Mees & Jones 1997; Dewicke *et al.* 2003). A species indicator analysis also revealed that some holohyperbenthic groups such as cumaceans (*Diastylis* sp., *Bodotria* sp.) and amphipods (*Parambius typicus*) were strongly associated with the samples collected inside the C-Power concession zone. These surface deposit feeding species are, however, not typical for the transitional areas but are rather characteristic for onshore communities (Dewicke *et al.* 2003). Moreover, organisms of *Parambius typicus* are also often attached to substrata such as large-bodied infauna, algae and hydroids.

Their increased abundance could therefore also be a result of the cease of fisheries activities within C-Power. Bottom fishing physically disturbs the seafloor in various ways and benthic communities are strongly affected through the removal of epibenthic organisms and structures (Johnson 2002). This, together with increased abundances of larvae of the tube-building polychaete *Lanice conchilega* and fish larvae, could indicate that the seabed-inhabiting fauna has been freed of the disturbances associated with commercial fishing. These findings are in line with the proposed hypothesis that cumulative effects of the long-term (> 10 years) presence of the jackets and cease of fishery activities ("fisheries exclusion effect") might result in enriched hyperbenthic communities within the OWF area. However, it should be emphasized that this hypothesis remains suggestive and should therefore be interpreted with caution. Subtidal sandbank systems are highly dynamic habitats and are known to exhibit strong small-scale heterogeneity both in terms of abiotic and biotic conditions (Mestdagh *et al.* 2020; Cheng *et al.* 2021). Therefore, future research is highly recommended to determine whether these differences are induced by the OWF presence, or simply a result of natural spatial variation between the areas sampled inside and outside C-Power.

In contrast to C-Power, no clear spatial differences were detected between the areas located outside and inside the Norther OWF, while strong variation was found among the samples taken inside the concession zone. This lack of conclusive results may of course be related to the fact that this OWF has been built quite recently and has only been operational for two years. It is therefore likely that the communities within this OWF probably still have to recover from the short-term effects associated with the construction phase, which would also explain why the samples collected inside the OWF were generally poorer compared to the reference samples in terms of density and diversity. An additional explanation is the fact that the presence of strong natural spatial variability

(habitat types) in combination with a relatively low number of replicates (3 tracks) is insufficient to test OWF-related effects in a robust way, which was confirmed by the power analysis conducted within this study. Future monitoring studies are therefore important to investigate potential long-term effects, in which the applied sampling design should be able to incorporate the habitat heterogeneity that is found within this OWF.

4.3. Sampling quality assessment

Horizontal hauling devices such as the hyperbenthic sledge used in this study, are often preferred as the equipment of choice when sampling for hyperbenthic communities (Hamerlynck & Mees 1991; Mees & Jones 1997; Dewicke *et al.* 2003). They are designed to sample at least the lower one metre of the water column such that no contamination with the sediment occurs (Mees & Jones 1997). The level of success, however, strongly depends on local factors such as depth, degree of exposure, bottom topography and weather conditions, which also implies that heavier and more robust equipment (ship size and power, lifting gear) is necessary to sample deeper and offshore habitats in which the OWFs under study are located (Mees & Jones 1997). Compared to the feasibility study performed in 2018, the extent of the sampling effort within this study did however indicate that the adjusted sampling strategy can be considered effective: both OWFs were sampled during a one-day sampling campaign, for which each towing event proved to be successful, resulting in the desired number of quantitative samples.

Besides the actual sample collection, it is also important to assess the sampling quality or "catch efficiency" of the obtained samples (Del Vecchio *et al.* 2019). Species found within the hyperbenthal zone are often highly mobile and migrate through this zone at specific times (day/year) or at certain life-history stages (Mees & Jones 1997; Dewicke *et al.* 2003). Consequently, it has proven rather difficult to provide complete population

assessments of hyperbenthic communities as they comprise a wide array of different species and exhibit dynamic distributions in space and time (Mees & Jones 1997). Due to a lack of scientific literature on hyperbenthic spatial patterns within these areas and the absence of true baseline studies, it is also challenging to verify whether the obtained dataset can be considered as "representative". The samples in this study turned out to be quite rich in terms of abundance and diversity, and sample processing (sorting, counting, identifications) proved to be a very time-consuming process, which even resulted in an incomplete dataset for the Norther study site. The very high number of hydromedusae specimens that were found in this study may be attributed to the timing of the sampling (*i.e.* daytime conditions) as these organisms are often concentrated within the hyperbenthic zone during the day (Mees & Jones 1997). In terms of community composition, the hyperbenthic descriptions within this study corresponded to the communities described by Dewicke *et al*. (2003), which currently is the only one that can be used as a "baseline" study. The SACs and species richness estimators did, however, show that an asymptote had not yet been reached and a fraction of undetected species could still be expected, especially within the Norther OWF. Assuming that these communities have similarities with more onshore situated clusters and the strong habitat heterogeneity within this OWF, it is likely that increased sampling effort will be required to fully characterize the hyperbenthic communities within this concession zone.

A final aspect to consider is the level of the vertical distinction that was applied for the hyperbenthic sampling in this study. Two-level (lower and upper nets) hyperbenthic sledges are often used to account for potential varying hyperbenthic distributions throughout the sampled water column (Mees & Jones 1997). The effective height that marks the distinction between the lower and upper samples can be variable and also depends on the water depth, but the majority of hyperbenthic studies use a sledge with a lower net up to about

50 cm, comparable to our study (Mees & Jones 1997). Within the currently available literature, findings regarding vertical segregation of hyperbenthic communities are rather inconclusive as some studies show that much higher densities are found within the lower net samples, while other studies report homogenous hyperbenthic distributions between the different nets (Mees & Jones 1997). Results in our study correspond to the latter, as no significant positional differences were found in terms of densities, diversity and composition between the lower and upper nets for the C-Power study site. However, it must also be considered that the lower net of the hyperbenthic sledge used in this study was situated at 20 cm, which means that the lowermost centimetres of the water column (0–20 cm) were not sampled. Certain taxa such as mysids are known to be epibenthic during the day and reside near the bottom, while they move up to the water column during the night (Mees & Jones 1997; Parry *et al*. 2021). As a result, the densities and diversity of mysids, which are characteristic hyperbenthos, could be under-represented with the applied sampling methodology.

4.4. Challenges and opportunities

The results obtained from this study proved to be valuable as they allowed to enhance our knowledge on the distribution of hyperbenthic communities within these areas, and to assess the strength of the applied sampling design to investigate the hyperbenthic enrichment hypothesis. Obtaining complete, qualitative data appeared to be a time-consuming process due to the high densities and diversity that characterized these samples. Sample processing activities such as sorting, counting and especially species-level identification were fairly labour intensive and required specific taxonomic expertise. Evidently, this constraint creates a trade-off between the sampling effort and the number of replicates that can be processed per study site. We therefore conclude that a sampling design in which 12 tracks are sampled (24 lower and upper samples) represents the maximum

number of replicates that can be processed qualitatively within an annual monitoring program.

The sampling quality assessment performed within this study revealed that an increase of the sampling effort would strongly enhance the ability to fully characterize the hyperbenthic communities and the statistical power to detect any spatial differences between the areas inside and outside the OWFs. This appears to be especially true for the Norther study site, which can be attributed to the fact that several habitat types are found within this concession zone. It is therefore proposed to increase the sampling effort to 6 tracks inside and 6 tracks outside the OWFs and to revise the positions and number of tracks for the Norther study site based on previous macrobenthos studies to account for the habitat heterogeneity (Lefaible *et al.* 2021). It can then be further discussed how the actual sample processing can be spread over a longer period of time depending on the focus of the research objectives and reporting. Given the fact that no vertical segregation was detected between the lower and upper net samples within this study, it might also be considered to limit the actual

reporting to the lower net samples only to decrease the amount of time spend on sample processing.

While the spatial differences and trends found for the C-Power study area support the proposed hypothesis of hyperbenthic enrichment, it remains unclear whether this can also be linked to aberrant abiotic conditions between the two areas due to actual turbine-related impacts. We therefore strongly recommend including relevant abiotic variables at the sediment-water column boundary within the design to strengthen our ability to assess this hypothesis more thoroughly. This could be achieved by applying a similar methodology used by Dewicke *et al*. (2003), in which every tow was preceded by the deployment of a multi-corer. From the obtained cores, the near-bottom water will be collected and used for pigment analysis. In addition, the upper sediment layers (0–3 cm) could be used to determine seabed conditions in terms of granulometry and organic matter. With these proposed adjustments future studies will be able to obtain the most cost-effective methodology to maximize the efficient use of monitoring resources and research objectives outcomes.

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CHAPTER 4

SEABIRDS AND OFFSHORE WIND FARMS – THE POTENTIAL VALUE OF SENSITIVITY MAPPING IN MARINE SPATIAL PLANNING

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Abstract

In this study we developed species distribution models, intended to feed into a sensitivity map regarding offshore wind farm development. We focused on four species known to be sensitive to wind farm disturbance, i.e., red-throated diver (*Gavia stellata*), northern gannet (*Morus bassanus*), common guillemot (*Uria aalge*) and razorbill (*Alca torda*), and proposed an integrated 'displacement sensitivity index' based on their cumulative distribution. Interestingly, the species distribution models allow to quantify the numbers of seabirds expected to be impacted by wind farm displacement and thus to flag potential conflicts with conservation objectives defined within the European Marine Strategy Framework and/or Birds Directive. Mapping our 'displacement sensitivity index' further highlighted one area as particularly sensitive to wind farm development, situated in front of the western part of the Belgian coast between 5 and 12 nautical miles offshore. While provisional, the results of this study are highly promising, distinguishing one compact area which is historically known as important seabird habitat. Also, it is located well outside all current and planned wind farms, giving the opportunity to avoid future developments there or otherwise to install compensating measures. To ultimately inform the marine spatial planning process, however, we advise finetuning the modelling process and taking in account additional seabird species and anthropogenic pressures.

1. Introduction

Current and planned wind farm developments will soon occupy about 15 % of the Belgian part of the North Sea (BPNS). Knowing that certain seabird species tend to avoid areas occupied by turbines raises concerns regarding the cumulative impact of such extensive developments on seabird population demographics. By informing the marine spatial planning process, well-founded sensitivity maps may serve as a tool to avoid or compensate offshore wind farm (OWF) impacts on seabirds. As such, this study is intended as a first step in outlining a suitable method to map seabird sensitivity related to OWFs across the BPNS.

2. Material and methods

2.1. Seabird monitoring: in the field

In this analysis we used the results of ship-based seabird counts collected across the BPNS in the period 2000-2018. Except for the zone further than 25 nautical miles from the coast there has been good coverage of the BPNS during this timeframe (see Figure 1). The frequency and geographical focus of the monitoring routes, however, strongly varied through time.

Ship-based seabird counts have always been conducted according to a standardised and internationally applied method, combining a 'transect count' for birds on the water and repeated 'snapshot counts' for flying birds (Tasker *et al.* 1984). We focus on a 300 m wide transect along one side of the ship's track, and while steaming at a speed of about 10 knots, all birds in touch with the water (swimming, pecking, diving) observed within this transect are counted ('transect count'). Importantly, the perpendicular distance of each observed bird (group) to the ship is estimated, allowing to correct for decreasing detectability with increasing distance afterwards (distance analysis, see §2.2). The transect is therefore divided in four distance categories (A = 0-50 m, B = 50-100 m, C = 100-200 m and D = 200-300 m). Counting all flying birds inside this transect, however, would cause an overestimation and would be a measure of bird flux rather than bird density. As such, the density of flying birds is assessed through one-minute interval counts of birds flying within a quadrant of 300 by 300 m inside the transect ('snapshot counts'). As the ship covers a distance of approximately 300 m per minute when sailing the prescribed speed of 10 knots, the full transect is covered by means of these subsequent 'snapshots'.
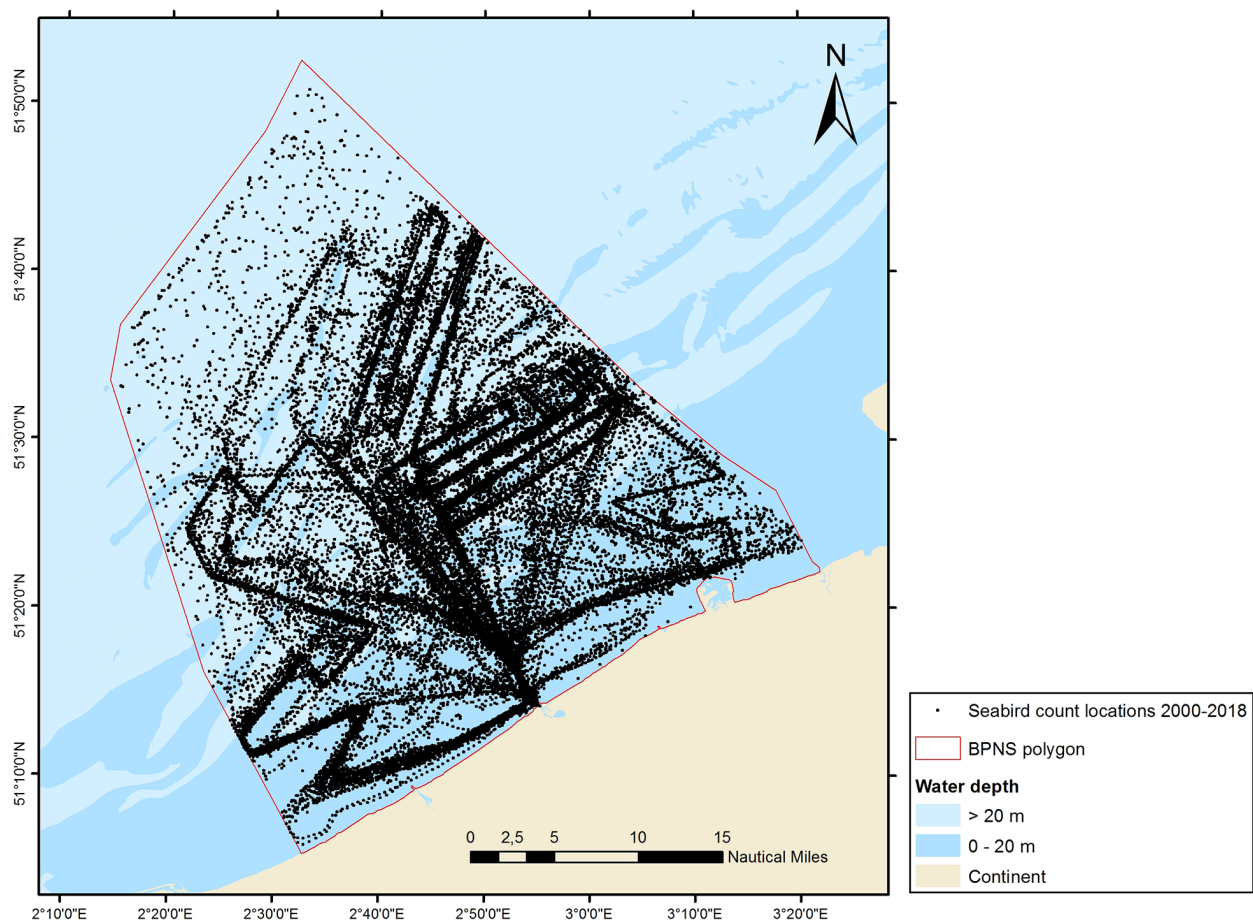
Figure 1. Count locations included in the OWF displacement sensitivity analysis for the period 2000-2018.

2.2. Seabird monitoring: aftermath

We corrected our transect count numbers for the decreasing probability of detecting birds with increasing distance to the ship (Buckland *et al.* 2001; Thomas *et al.* 2010). The exact relation between distance and detection probability is expected to be species-specific, and further likely to depend on bird group size and observation conditions (Marques & Buckland 2003). Observation conditions were included in the detection models as 'wind force' (Beaufort scale) or 'wave height' (categorised as 0-0.5 m / 0.5-1.0 m / 1.0-2.0 m / 2.0-3.0 m…), both variables being assessed visually throughout the surveys. To look for suitable species-specific detection models, we fitted each of the following four 'full models' alternatively with a half-normal and a hazard-rate detection function:

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

We did not add cosine or polynomial adjustments to the models as doing so sometimes appeared to result in non-monotonic functions. This would imply that the detection probability *increases* with distance, which is assumed to be highly improbable. For each species, the best fitting full model was chosen based on the 'Akaike Information Criterion' (AIC), and a manual backward covariate selection was then performed to obtain a parsimonious detection model. The resulting models were used to estimate detection probabilities, varying with the observed species and selected covariates. Next, the counted numbers were 'distance-
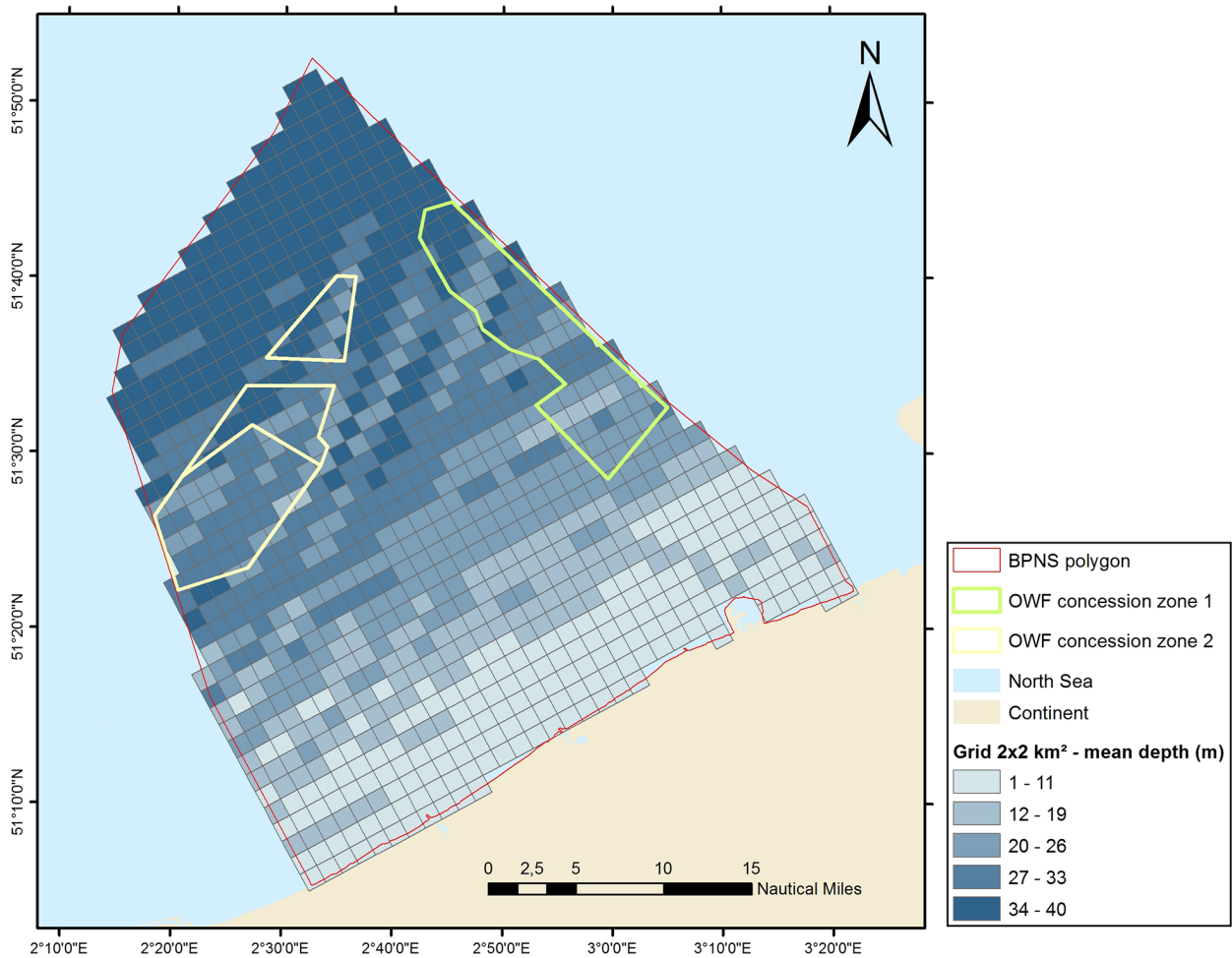
Figure 2. Mean depth parameter values over a 2×2 km² grid across the BPNS (geometrical interval scale).

corrected' by dividing them by the predicted detection probabilities.

In this analysis we considered four species, i.e., red-throated diver (*Gavia stellata*), northern gannet (*Morus bassanus*), common guillemot (*Uria aalge*) and razorbill (*Alca torda*), all regarded to be sensitive to OWF displacement (e.g., Vanermen & Stienen 2019). Their 'distance-corrected' numbers were eventually summed per year per month over a 2×2 km² grid across the BPNS, to obtain our response variable. Along with the seabird numbers, the area counted (the transect width of 300 m multiplied by the distance travelled) was summed as well and was used as an offset variable in the models.

2.3. Model parameters

For species distribution modelling (SDM) we considered several abiotic parameters, i.e.,

water depth, variation in water depth, salinity, distance to the coast and OWF presence.

Water depth data were taken from Van Lancker *et al*. (2007). The mean and standard deviation of water depth were calculated per grid cell of 2×2 km² (see Figures 2-3) to obtain the parameters 'mean depth' and 'SD depth' applied in the SDM.

Salinity data were downloaded from the Copernicus website (Copernicus 2022). There, we obtained hourly sea surface salinity figures for the period 2000-2021 at a 7 km resolution. We transformed this data file to a raster with interpolated values, which in turn were averaged over the forementioned 2×2 km² grid cells (see Figure 4).

Lastly, grid cells including at least one of the offshore wind turbines of the Belwind (2011-2018), C-Power (2013-2018), or
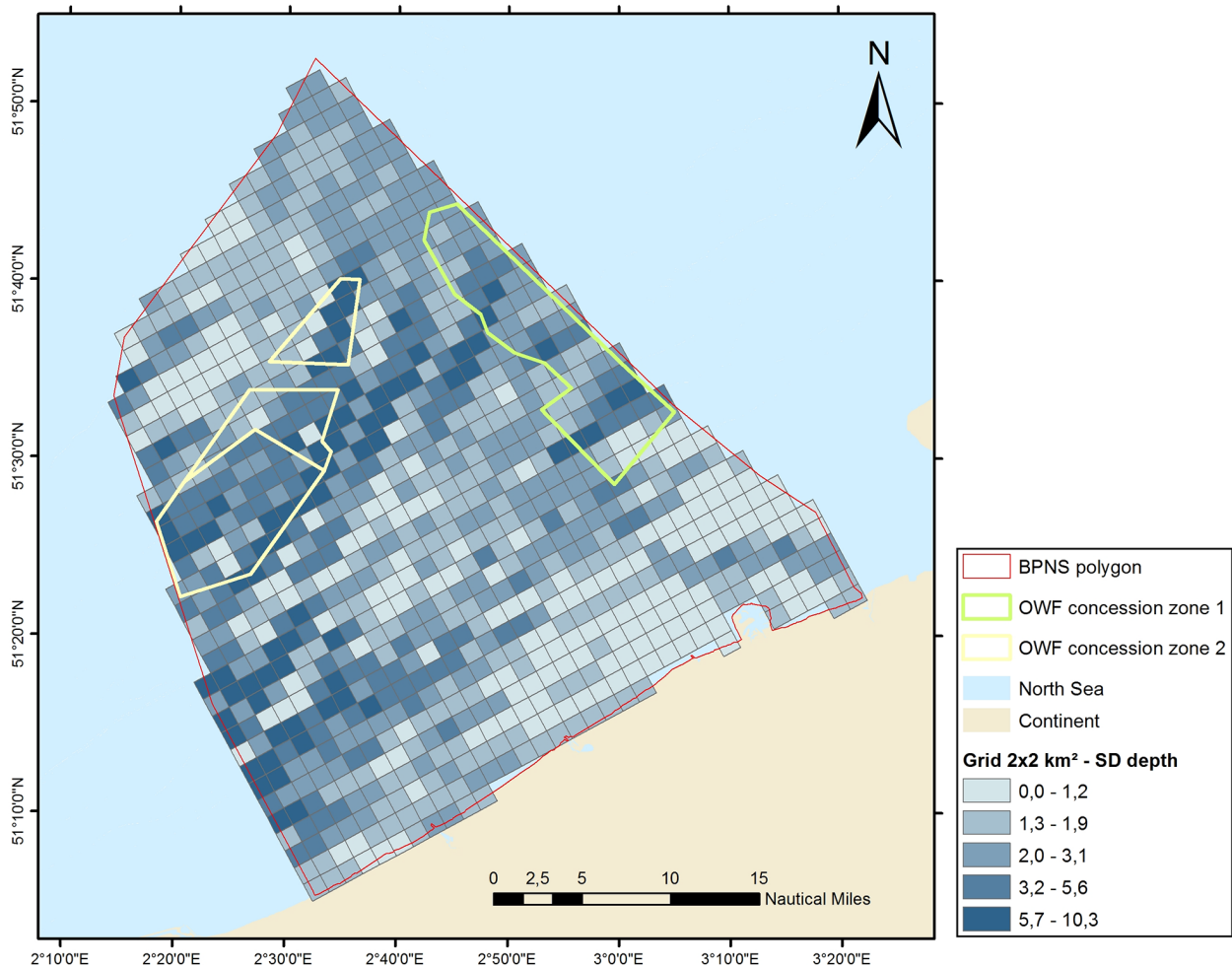
Figure 3. SD depth parameter values over a 2×2 km² grid across the BPNS (geometrical interval scale).

Norther (2015-2018) wind farms were set at TRUE for the Boolean OWF parameter, for the indicated periods in which these wind farms were operational.

2.4. Species distribution models

We modelled our response variable (number counted per year per month per grid cell) using area (i.e., the area counted) as an offset, mean depth, SD depth, salinity and month as thin-plate smoothers, OWF as a factor variable and year as a random intercept, the full (fixed) part of the model thus according to:

$N \sim offset(area) + s(mean\ depth) + s(SD\ depth) + s(salinity) + s(month) + OWF$

All smoothers were limited to 6 knots to avoid overfitting, while the smoother of month was further defined as a cyclic smoother.

We chose between a Poisson and negative binomial distribution based on the AIC, after which we performed backward model selection until the AIC reached its minimum or alternatively, until all parameters were significant ($P < 0.05$).

2.5. Displacement sensitivity

To come to a measure of displacement sensitivity, the predicted densities (assuming a scenario without OWFs) of the four species considered in this analysis were standardised to a value between 0 and 1 by dividing the prediction per grid cell by the maximum predicted value for a specific month (the one with highest overall occurrence). This way we ensure that all species contribute equally, independent of the variation in densities between species. Next, the standardised values for all four species were summed per
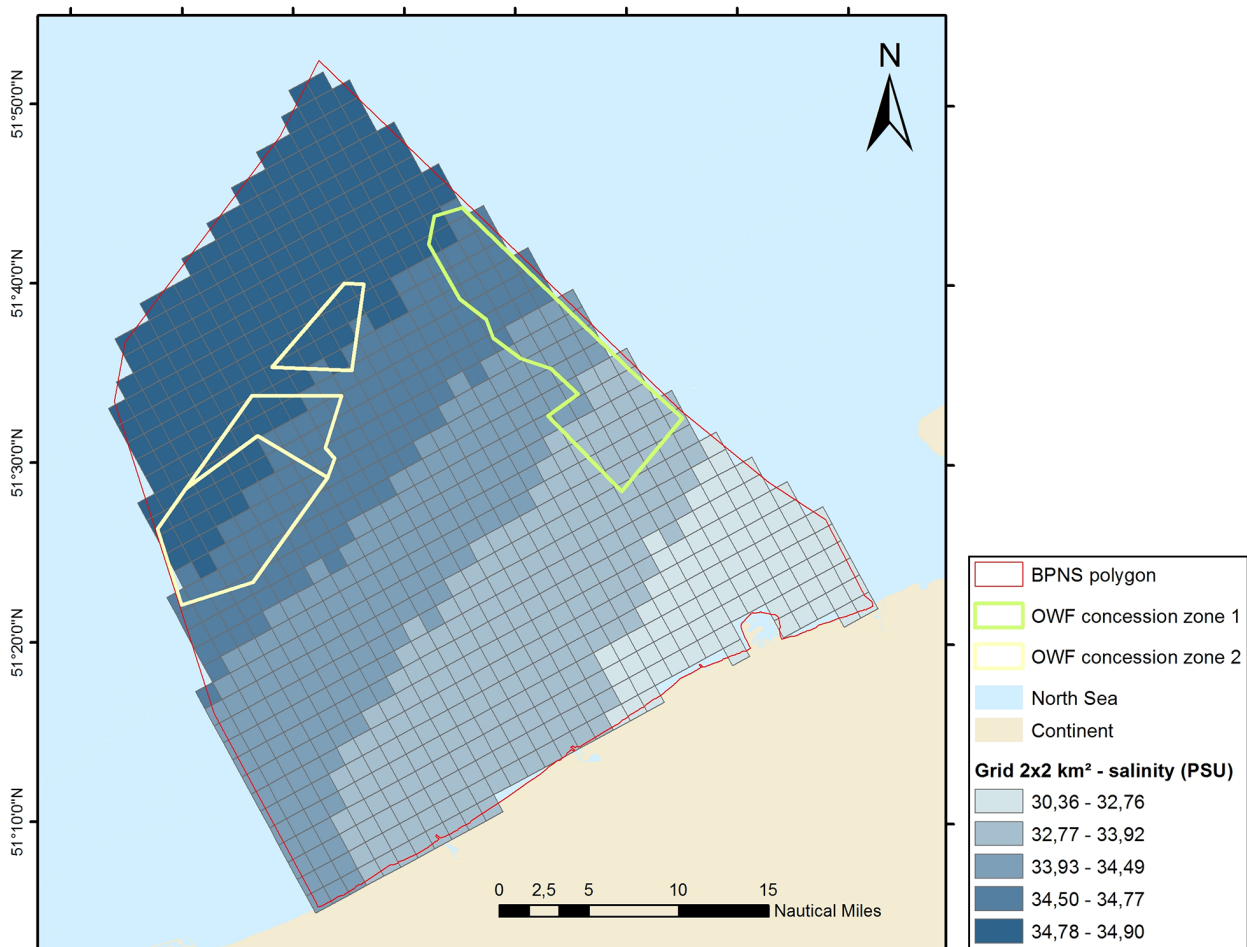
Figure 4. Salinity parameter values over a 2 × 2 km² grid across the BPNS (geometrical interval scale).

grid cell, thus obtaining a value which in theory could vary between 0 and 4, hereby called the displacement sensitivity index (DSI). Grid cells with a high DSI thus imply high numbers (relative to their maximum predicted densities) of at least some of the four displacement-sensitive species.

3. Results

3.1. Red-throated diver

All variables were retained in the model except for salinity. Looking at the predicted distribution we see that red-throated divers are expected to occur in highest numbers in an area 2 to 8 nautical miles offshore, where densities of 0.2 to 0.5 birds/km² are reached during midwinter (Figure 5). The current

OWF concession zones do not overlap with this area of highest occurrence.

3.2. Northern gannet

For northern gannet, only SD depth was discarded from the model. Highest densities are reached in October, with predicted densities up to 5 birds/km² during autumn migration. The species' distribution is oriented offshore, with a clear influence of the saline gradient on top. Highest predicted densities are reached between the ridges of the 'Hinderbanken', and also in the far north-western corner of the BPNS (Figure 6). Yet, considering the limited number of observations in the latter area (Figure 1) we should be careful in emphasizing the high predictions there. For northern gannet, the OWF concession
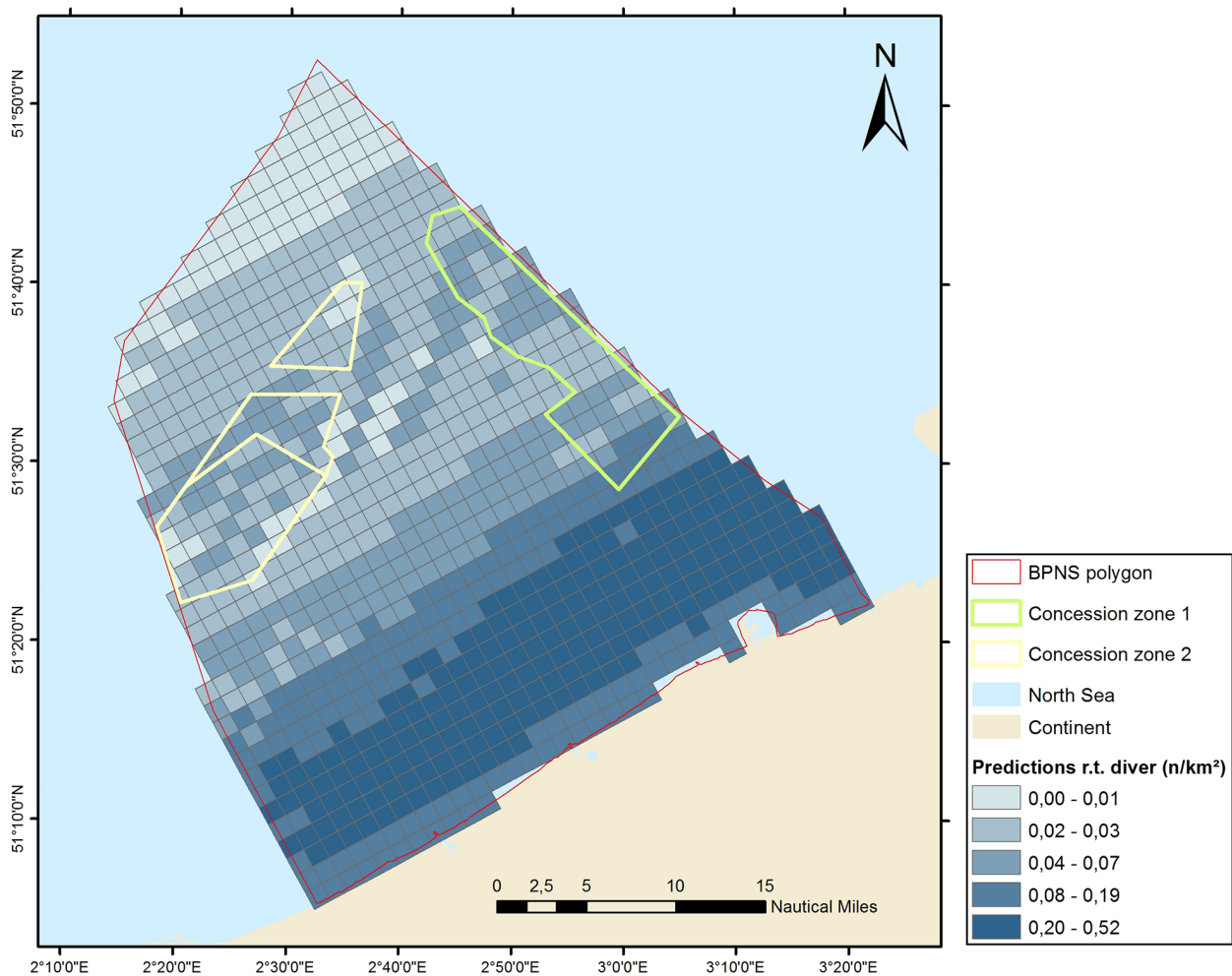
Figure 5. Predicted distribution of red-throated diver for the month December with the OWF factor set at FALSE (geometrical interval scale).
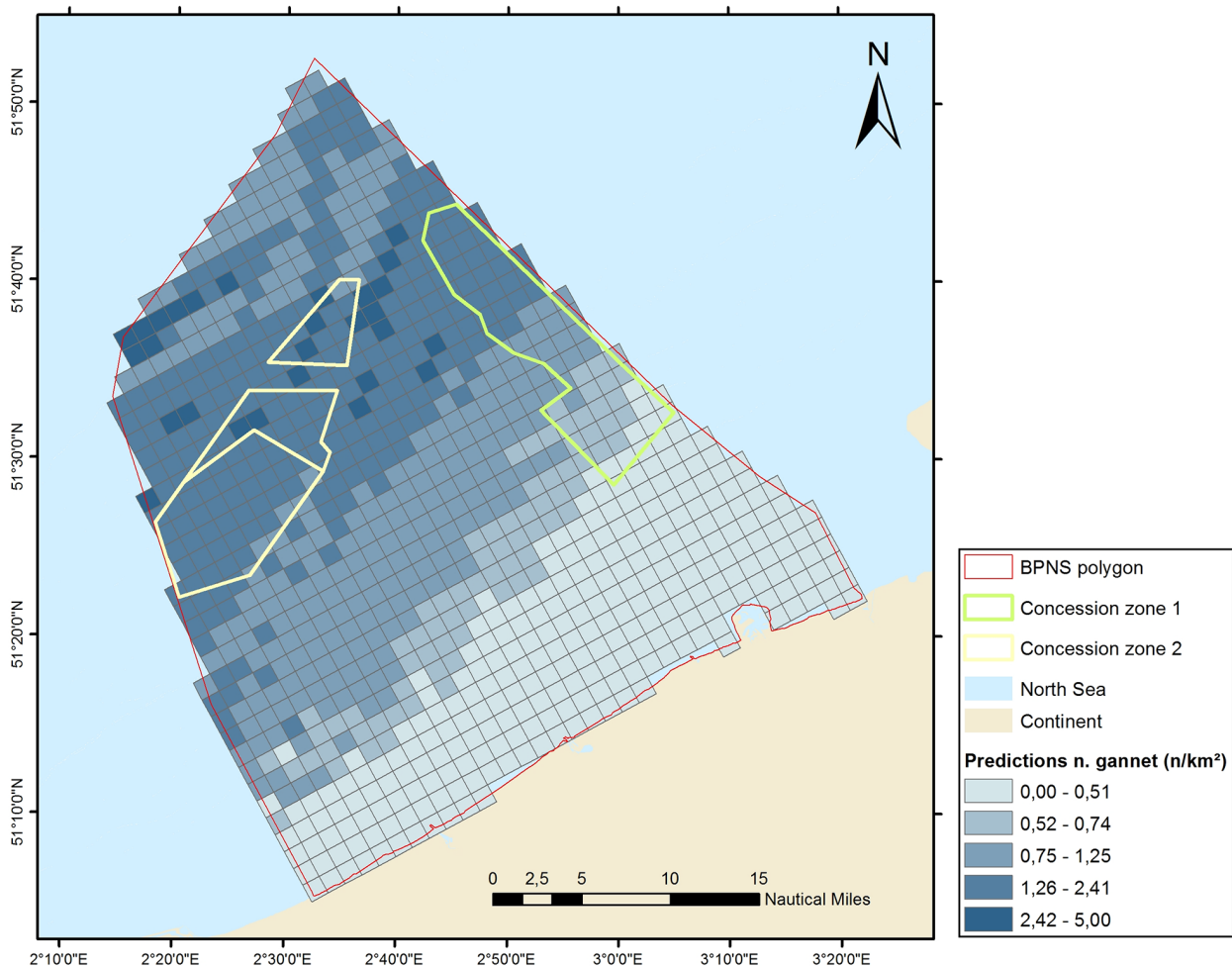
Figure 6. Predicted distribution of northern gannet for the month October with the OWF factor set at FALSE (geometrical interval scale).

zones clearly overlap with areas of abundant occurrence, which is especially the case for concession zone 2.

3.3. Common guillemot

All parameters were retained in the model for common guillemot, which explained 58 % of the deviance of the data. During midwinter, predicted densities go up to 10.5 birds/km². The birds clearly avoid the low-saline waters in front of the eastern coast and reach highest densities on top of the ridges of the 'Vlaamse Banken' (Figure 7). There is large overlap between high-density areas and the OWF concession zones, especially in case of concession zone 2.

3.4. Razorbill

As for common guillemot, we retained all parameters in the model, which achieved to explain 52 % of the deviance in the data. The species reaches its highest densities on the 'Vlaamse Banken', with locally 1.9-2.6 birds/km² in the month November, and lowest densities in the low-saline waters near the Westerschelde estuary (Figure 8). Razorbill has a distinct seasonal pattern, with generally increased numbers in the winter half year, yet with secondary peaks in numbers in February and November, illustrating that a certain part of the birds only migrates through. There appears to be limited overlap between areas with high abundance of razorbill on the one hand and the OWF concession zones on the other hand.
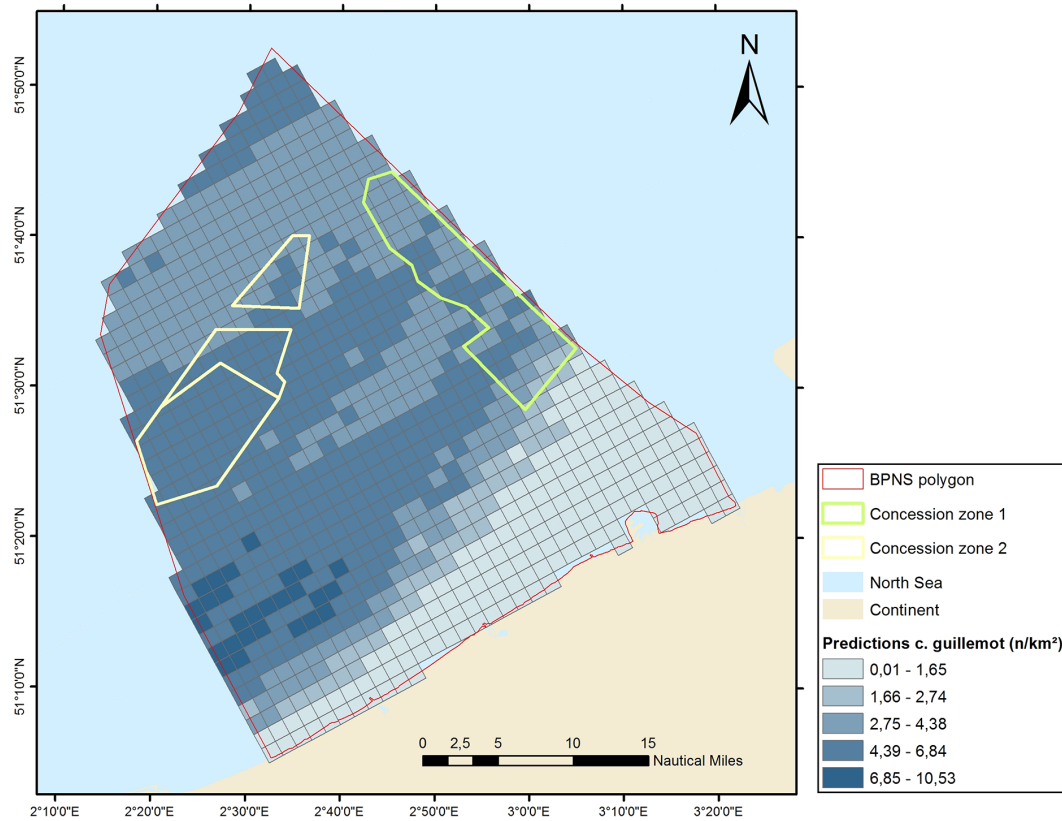
Figure 7. Predicted distribution of common guillemot for the month January with the OWF factor set at FALSE (geometrical interval scale).
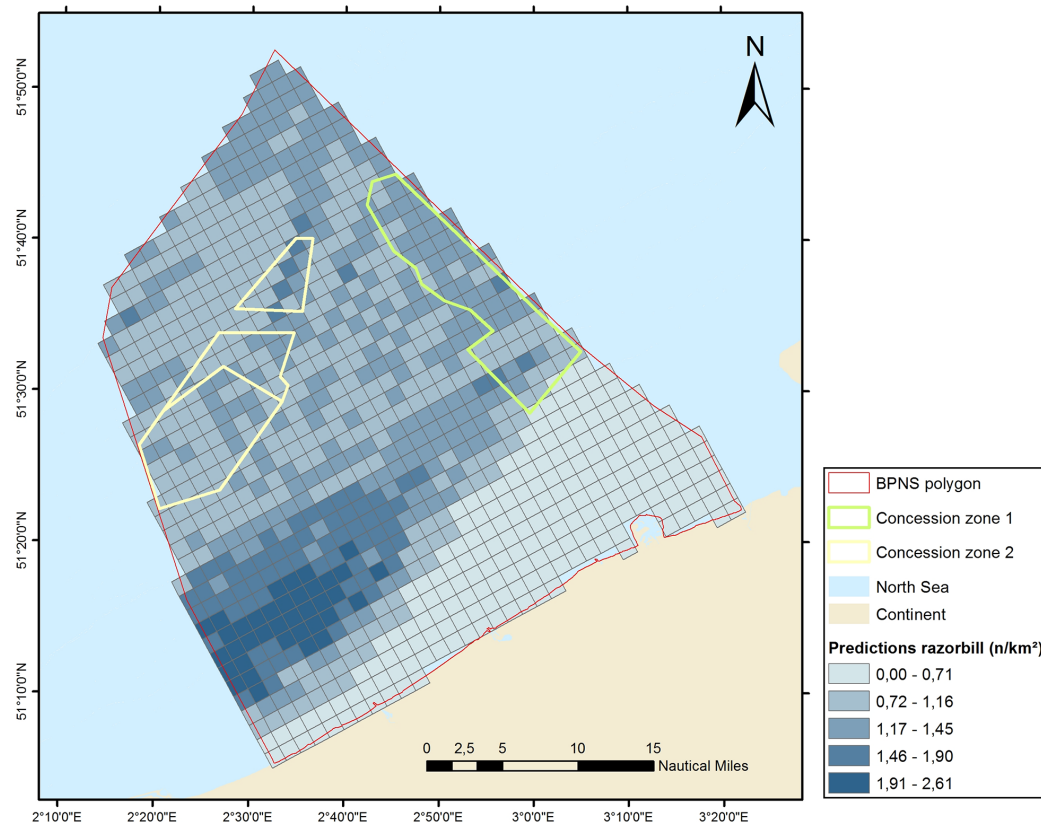
Figure 8. Predicted distribution of razorbill for the month November with the OWF factor set at FALSE (geometrical interval scale).

Table 1. Estimated OWF coefficients and accompanying expected decreases in numbers.

	OWF coefficient	P-value	Expected decrease inside OWFs
Red-throated diver	-1.66	0.105	81 %
Northern gannet	-1.85	< 0.001	84 %
Common guillemot	-1.13	< 0.001	68 %
Razorbill	-0.53	0.007	41 %

3.5. OWF effect

Interestingly, in all models the OWF factor was retained. Though not statistically significant in case of red-throated diver, the OWF factor did contribute positively to this model as well based on the AIC. The fact that the main distribution of divers does not overlap with the current wind farm developments clearly makes it hard for statistical evidence to occur. For the three other species, the coefficient was both negative and significant, underpinning the negative effect of OWFs on their presence. The estimated coefficients and associated P values for the OWF factor are summarised in Table 1. The third column shows the expected decrease in numbers in the wind farm concession zones (calculated as 1 minus the exponentiation of the OWF coefficient).

The SDM results further allow to estimate the species' total numbers residing at the BPNS, as well as the number of birds that are expected to be impacted by current and planned OWFs (Table 2). In absolute numbers, common guillemot is the most impacted species, with about 1600 individuals being displaced by the (future) OWFs in concession zones 1 and 2. The strongest relative impact,

however, was found for northern gannet with 17.4 % of 3340 individuals expected to be displaced.

3.6. Displacement sensitivity

By summing the standardised density predictions of four displacement-sensitive species per grid cell and mapping the resulting DSI values (see §2.5), we obtained the displacement sensitivity map as shown in Figure 9. One zone with DSI values higher than 1.8 jumps out clearly, and is situated in front of the western part of the Belgian coast between 5 and 12 nautical miles offshore. This area is often referred to as 'Vlaamse Banken', and more precisely, it includes part of the 'Oostdyck' and most of the sandbanks 'Buitenratel', 'Kwintebank' and 'Middelkerkebank'.

Assessing the contribution of the different species to the DSI values across three areas of interest (OWF concession zone 1, OWF concession zone 2 and the aforementioned area with high displacement sensitivity) resulted in the bar plot below (Figure 10). This plot illustrates how the species contribution at 'Vlaamse Banken' is quite different from those in the wind farm concession zones,

Table 2. Predicted numbers at the BPNS for scenarios with and without OWFs, in the month with maximum densities.

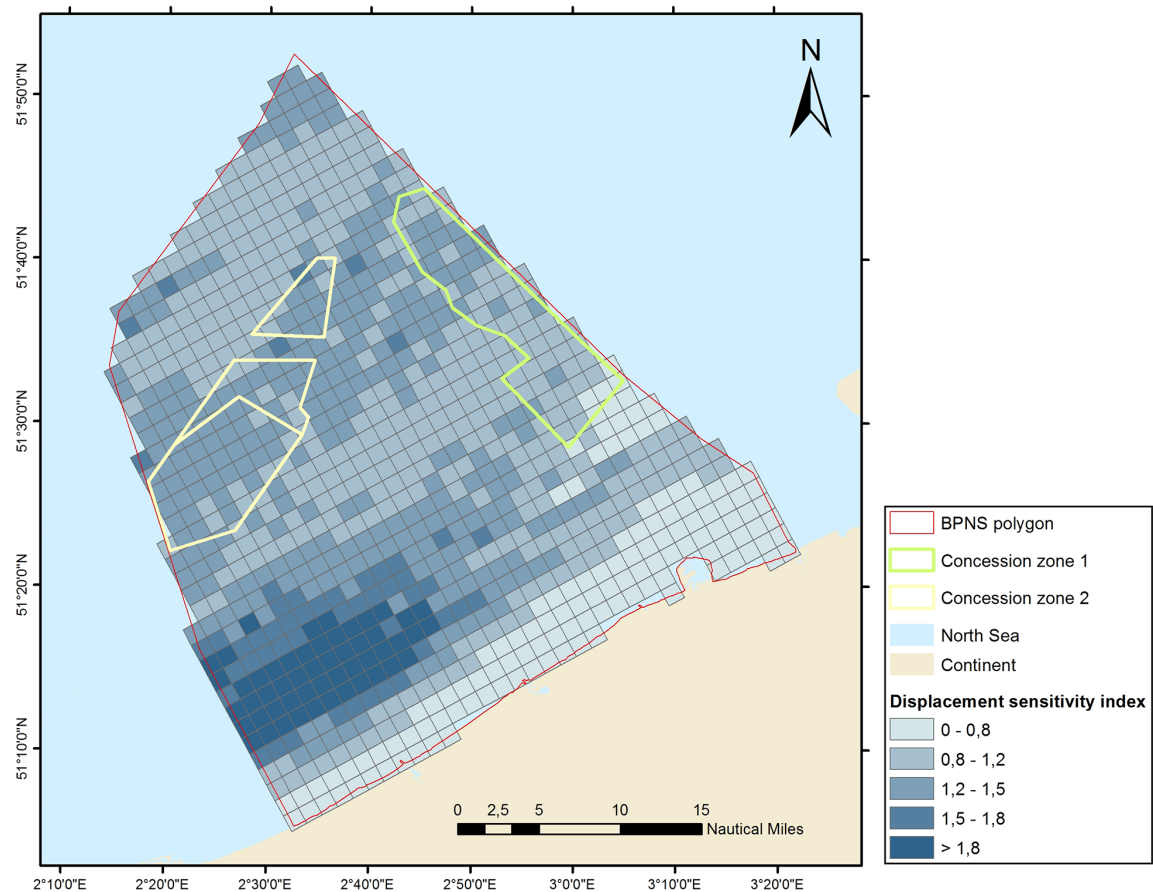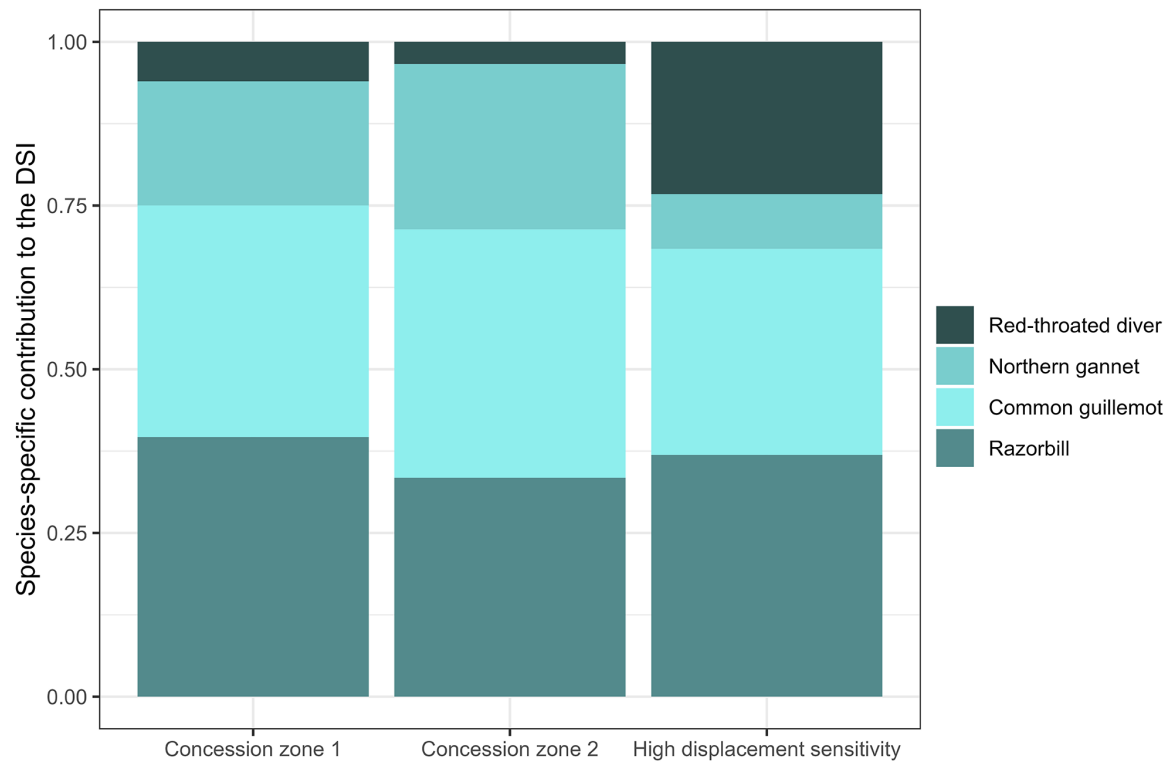|	Total predicted numbers at BPNS *without* OWFs	Total predicted numbers at BPNS *with* OWFs	Predicted % of numbers impacted by OWFs
Red-throated diver (December)	414	403	2.8 %
Northern gannet (October)	3340	2760	17.4 %
Common guillemot (January)	13233	11629	12.1 %
Razorbill (November)	3535	3298	6.7 %

Figure 9. OWF displacement sensitivity map.

Figure 10. Contribution of the four selected species to the DSI values in the OWF concession zones 1 and 2 on the one hand and the 'high displacement sensitivity' area on the other hand.

mainly because of high predicted densities of red-throated diver, compared to relatively low densities of northern gannet. The wind farm concession zones on the other hand show a quite comparable species contribution.

4. Discussion

This study intended to look for a suitable method to produce an integrated sensitivity map regarding OWF-induced seabird displacement. For this purpose, we first corrected observational data for distance-related bias and then linked the corrected seabird numbers with a range of explanatory environmental variables to produce species distribution models (Waggit *et al.* 2020; Mercker *et al.* 2021) of four species known to be sensitive to OWF disturbance (red-throated diver, northern gannet, common guillemot and razorbill).

Compared to the targeted BACI analyses reported throughout the WinMon.BE research program, this analysis was less focussing on wind farms alone, including the whole BPNS. Nevertheless, our SDM revealed strong effects of the presence of OWFs in all 4 species. With predicted decreases of 84 % for northern gannet and 68 % for common guillemot, the results are highly comparable to the decreases reported in Vanermen *et al.* (2019), i.e., 82-98 % for northern gannet and 63-75 % for common guillemot. For razorbill the predicted decrease found here (41 %) is lower compared to the one reported earlier (67-75 %), while for red-throated diver we never reported any estimates due to very little overlap between this species' distribution at the BPNS and OWFs. Interestingly, the SDMs provide quantitative insight in the numbers of seabirds expected to be impacted by OWF developments in the BPNS and thus allow to flag potential conflicts with conservation objectives defined within the Marine Strategy Framework and/or Birds Directive.

It is important to note that this is a first explorative analysis, and the SDMs can be finetuned in various ways. One way would

be incorporating distance to the nearest OWF instead of including the OWF effect as a Boolean factor. The wind farm effect could in theory also interact with the other parameters, which was not investigated here. Clearly, other human pressures too may influence seabird distribution, such as fishing activities and ship traffic (Mercker *et al.* 2021), parameters that were not included in the SDM here.

In a next step, we cumulated the standardised model predictions to obtain our intended sensitivity map. In this map one compact area in front of the western part of the Belgian coast clearly stands out due to particularly high DSI values. Interestingly, roughly the same area has always been conceived as good seabird habitat during ship-based surveying and was already highlighted as being sensitive to seabird disturbance and oil pollution by Seys (2001). The area is further enclosed entirely by the special area for conservation 'The Flemish Banks' (Habitats Directive), yet shows very little overlap with the special areas for the protection of birds 'SPA 1', 'SPA 2' and 'SPA 3' (Birds Directive) (Figure 11). This can easily be explained by the fact that the latter were delineated based on an entirely different set of species, namely great crested grebe (*Podiceps cristatus*), common scoter (*Melanitta nigra*), little gull (*Hydrocoloeus minutus*) and two tern species (*Sterna hirundo* and *Thalasseus sanvicensis*) (Haelters *et al.* 2004), none of which were included in this analysis.

For a more thorough displacement sensitivity mapping, ideally, we should also include common scoter, as this species is also known to be sensitive to displacement by wind farms. At the BPNS, the numbers of common scoters are monitored yearly through aerial instead ship-based surveys, yet these surveys only cover a rather narrow strip along the Belgian coast, thus hampering reliable SDM across the full extent of the BPNS.

To conclude, well-founded sensitivity maps can be an important tool in informing marine spatial planning. It allows to avoid developments in areas with large numbers
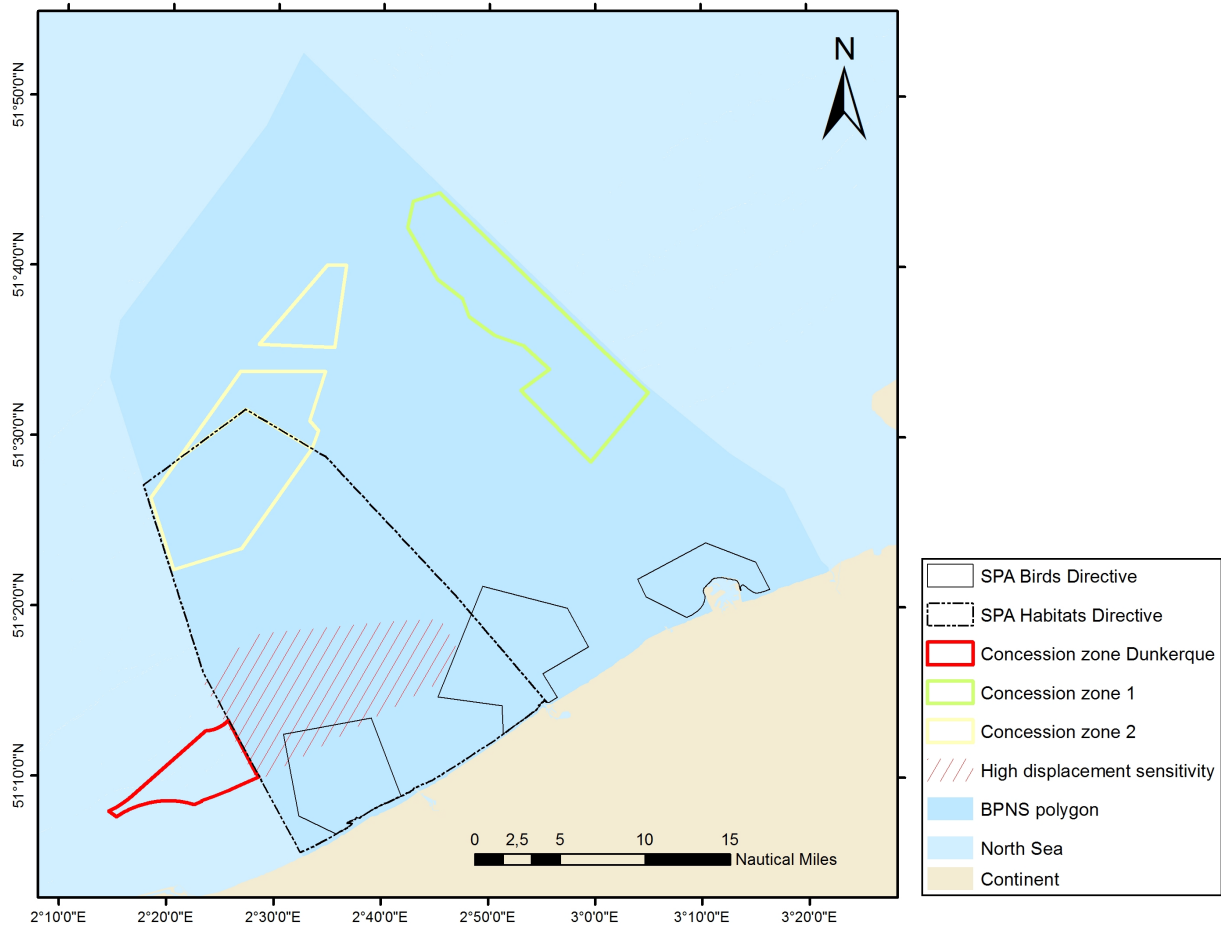
Figure 11. Location of the area with high DSI values relative to the Habitats and Birds Directive areas and planned and operational OWF concession zones.

of birds that are sensitive to wind farm disturbance, or alternatively, to install protective measures in sensitive areas in order to compensate for wind farm impacts elsewhere. In the highly dynamic marine environment, it may prove hard to find areas that compensate the same species and numbers that are impacted, as for example illustrated by the limited contribution of northern gannet in the area with high DSI values (Figure 10) compared to the concession zones. Likewise, it can be difficult to quantify the benefits of

protective measure to any given species. All this, however, should not impede the implementation of compensating measures since species protection laws are not only intended to protect single species, but also to conserve their habitat and all other species associated with that habitat. In accordance to this 'umbrella' concept of nature conservation, installing a marine protected area aimed at compensating the loss of suitable seabird habitat caused by offshore wind farming at the BPNS should be given consideration.

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CHAPTER 5

A COMPARISON BETWEEN THE EPIFAUNAL BIODIVERSITY OF SHIPWRECKS AND OFFSHORE WIND FARMS IN THE BELGIAN PART OF THE NORTH SEA

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Abstract

In this contribution we compared the epifaunal biodiversity of shipwrecks with turbine foundations and surrounding scour protection layers of offshore wind farms in the Belgian part of the North Sea. Shipwrecks were characterized by a higher epifaunal species richness compared to offshore wind farms (165 vs 114). Species identity was also different between both artificial hard substrates, with 95 unique epifaunal species for shipwrecks and 44 unique epifaunal species for offshore wind farms. The differences in biodiversity between both structures may be attributed to the older age and the higher structural complexity of shipwrecks. Increasing the structural complexity of turbine foundations and surrounding scour protection layers might increase the epifaunal biodiversity of offshore wind farms, leading to more similar epifaunal communities as those found at shipwrecks.

1. Introduction

Several marine activities are adding a variety of artificial hard structures to the ocean environment. These activities range from shipping (in the form of shipwrecks), coastal defence and development (harbour walls, groynes, breakwaters, etc.), oil and gas extraction (platforms) to renewable energy production (offshore wind turbine foundations and surrounding scour protection layer). Also, nature conservation and restoration can actively add artificial hard structures in the marine environment in the form of (primary) artificial reefs, while the other structures can be regarded as secondary artificial reefs (Krone 2012). All these structures provide artificial hard substrates, which are colonised by epifaunal (fouling) communities (e.g., Whomersley & Picken 2003; Zintzen & Massin 2010; Kerckhof *et al*. 2012; Van Moorsel 2014; Wetzel *et al*. 2014; Schutter *et al*. 2019; Coolen *et al*. 2020).

In the North Sea, artificial hard substrates were historically mostly present in the form of shipwrecks and coastal infrastructure, and in the form of oil and gas platforms. The massive rollout of offshore wind farms (OWFs) is currently the highest contributor to new artificial hard substrates in several

countries bordering the North Sea, thereby vastly surpassing the amount of other artificial hard substrates. This evolution will only accelerate, driven by the increasing demand for renewable energy production (Ellabban *et al.* 2014; Wilding *et al.* 2017). The effects on ecosystem structure and functioning of this proliferation of offshore wind can be substantial (Degraer *et al.* 2020). The structures are quickly colonised by high numbers of hemi-sessile animals such as anemones, bivalves and filter-feeding amphipods which may influence particle and nutrient fluxes, and potentially affect plankton production (Newell 2004; Maar *et al.* 2007). Foundations of offshore wind turbines can have a 35-fold higher biomass compared to surrounding soft sediments (Krone *et al.* 2013), and can influence local food web dynamics (Mavraki *et al.* 2020). The production of (pseudo-)fecal pellets by these colonising organisms also affects the surrounding soft sediments (Krone *et al.* 2013; Coates *et al.* 2014; De Borger *et al.* 2021; Ivanov *et al.* 2021). Additionally, colonising bivalves such as the blue mussel (*Mytilus edulis*) form on their turn a secondary hard substrate for the settlement of other species (Rumes *et al.* 2021). Another consequence of the addition of artificial hard substrates in soft sediment areas is the increased dispersal potential of hard substrate associated species, which may use these substrates as stepping stones to expand and establish new areas (Connell 2001; Bulleri & Chapman 2010). These species can be indigenous or non-indigenous, with the establishment of non-indigenous species becoming an increasing concern (Langhamer 2012; Mineur *et al.*, 2012; Adams *et al.* 2014; de Mesel 2015; Kerckhof *et al.* 2016).

To better understand the potential effects of large-scale colonisation of offshore wind turbines by epifaunal species, including non-indigenous ones, a comparison with the epifaunal communities of long-existing artificial hard structures such as shipwrecks is a logical first step. Most shipwrecks in the North Sea have been there for decades or even longer, enabling the development of mature epifaunal communities. Because their biodiversity is much larger than that of the surrounding soft sediments, they are regarded as 'hotspots' for biodiversity (Zintzen *et al.* 2006). The question is whether this can also be the fate for the foundations and surrounding scour protection layer of offshore wind turbines. The comparison of the fouling communities of both artificial hard structures can potentially be of importance for the decommissioning discussion of offshore wind farms (Fowler *et al.* 2020).

In this chapter, we compare the epifaunal communities between offshore wind farms and shipwrecks in the Belgian part of the North Sea (BPNS), with special attention to the presence of non-indigenous species. The comparison will lead to a better understanding of these offshore artificial hard substrate communities and their fate throughout time.

2. Material and methods

For the qualitative comparison of the epifauna between offshore wind farms and shipwrecks in the BPNS, with all data exploration and visualization performed in R (R Core Team 2022), we used our Artificial Hard Substrate database. In this database, species records of all macrobenthic (>1 mm) species that are associated with different artificial hard substrates in the BPNS are recorded (Kapasakali *et al.* 2019). The taxonomic nomenclature was based on the World Register of Marine Species (WoRMS 2022). Soft-sediment species are occasionally found in hard substrate communities, but are not actually part of it, so these were removed from the database, as was the case for pelagic species. Furthermore, only full species records were considered; records on higher taxonomic levels such as genus, family or phylum were removed. Since we focused on hard substrate communities of shipwrecks and offshore wind farms, all records from other structures, such as groynes, harbour walls and buoys, were excluded from the analysis. As no intertidal shipwrecks are present (anymore) in the

Table 1. List of terms and definitions concerning the (non-)native status of species, adapted from Kapasakali *et al.* (2019).

Term	Definition
Indigenous	A biogeographical status indication, meaning those species that occur naturally (unaided by human action) within a particularly defined area. *Synonyms: native, autochthonous*
Non-indigenous	A biogeographical status indication, meaning those species that did not occur geographically within a particularly defined region prior to some predetermined period (after Les & Mehrhoff 1999). *Synonyms: non-native, allochthonous*
Cryptogenic	A species that is not demonstrably native or introduced (after Carlton 1996).
Introduced	A subset of non-indigenous species, whose presence in a region is attributable to human actions that enabled them to overcome fundamental biogeographical barriers (i.e., human-mediated extra-range dispersal) (modified from Richardson *et al.* 2011).
Range-expanding	A subset of non-indigenous species, whose presence into a novel region is attributable to natural dispersal; such expansion may be assisted or primarily driven by human-mediated changes to the environment (modified from Richardson *et al.* 2011).
Established	Species with a self-sustaining population in a non-indigenous region (modified from Les & Mehrhoff 1999). *Synonyms: naturalised*

BPNS, the focus was only on subtidal species. The native status of the species records in the database is also considered, and is defined according to Kapasakali *et al.* (2019 and Table 1). Species whose status is unclear, are indicated as cryptogenic (Carlton 1996).

In the BPNS, there are around 300 shipwrecks (Afdeling Kust 2022), of which at least 55 shipwrecks are older than 100 years (Demerre *et al.* 2020). Our species records were extracted from a subset of 10 shipwrecks, all at least 40 years old, as described in Zintzen (2007). There are currently nine OWFs in the BPNS, with a total of 399 wind turbines. Fouling data from the foundations and surrounding scour protection layer of the offshore wind turbines originate from samples taken at the C-Power OWF (2008-2020) and the Belwind OWF (2010-2020) and are recorded in the WinMon.BE database (see Kerckhof *et al.* 2019 and references therein). While the shipwrecks are distributed across the BPNS, with sampling sites in coastal, transitional and offshore water masses, the OWFs are situated in the Northeastern part of the BPNS, in transitional and offshore waters only.

3. Results

We retained a total of 209 species, of which 44 unique for OWFs and 95 unique for shipwrecks. OWFs and shipwrecks furthermore share 70 species (Fig. 1). Shipwrecks are more diverse, with a total of 165 species, while in OWFs, 114 species were recorded (Fig. 2).

For some higher taxa, we observed remarkable differences between habitats. OWFs and shipwrecks share one bryozoan species, while 7 bryozoan species are unique for OWFs, and 10 species unique for shipwrecks. A similar pattern is observed with the gastropods, for which both habitats share 9 species, while 8 species are unique to OWFs and 12 species unique to shipwrecks. Polychaetes are the most numerous taxon, with 18 shared species, 8 unique to OWFs and 30 unique to shipwrecks. Cumaceans, entoprocts and mysids (all represented by one species) were only found at OWFs, while pycnogonids (two species) were only present on shipwrecks. Sponges (Porifera) reached a high diversity at shipwrecks (9 unique species), but only one unique species

Fouling species richness of artificial hard substrates
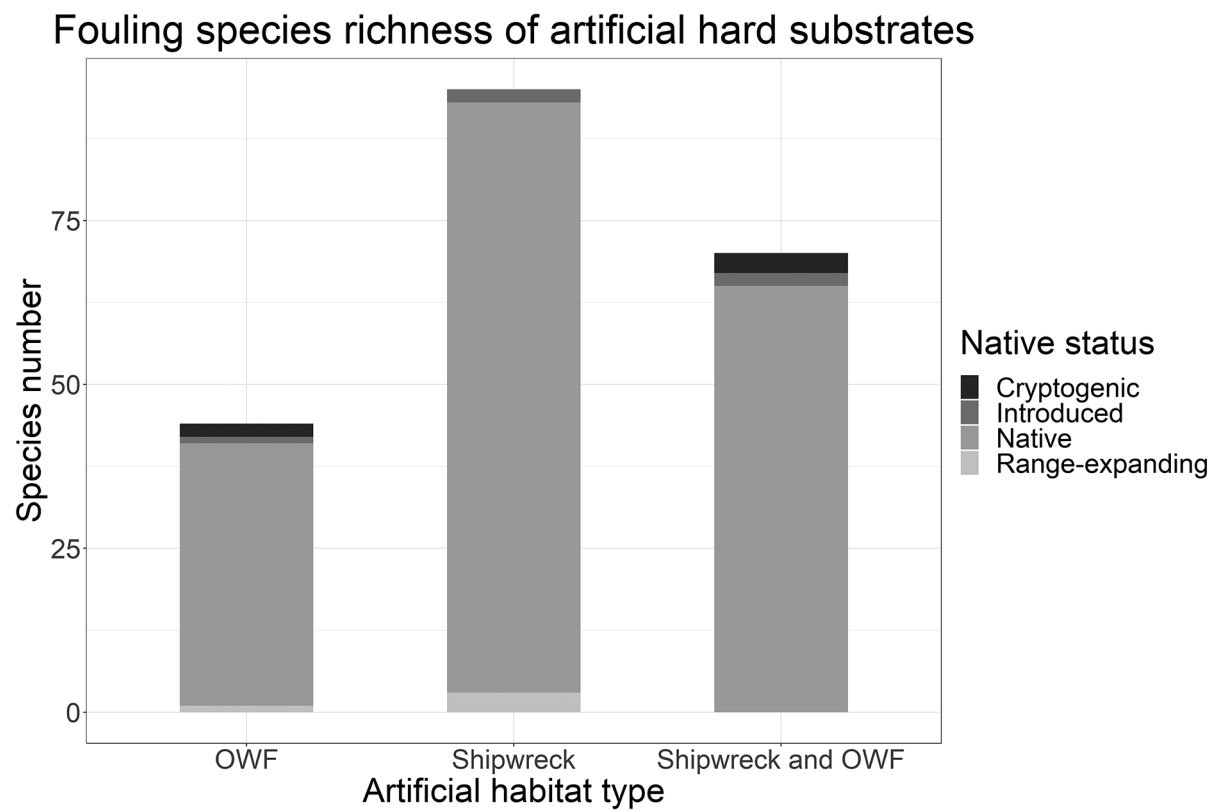
Figure 1. Number of unique native and non-indigenous macrobenthic species at offshore wind farms (OWF), shipwrecks; and present at both habitats.

Fouling species richness of artificial hard substrates
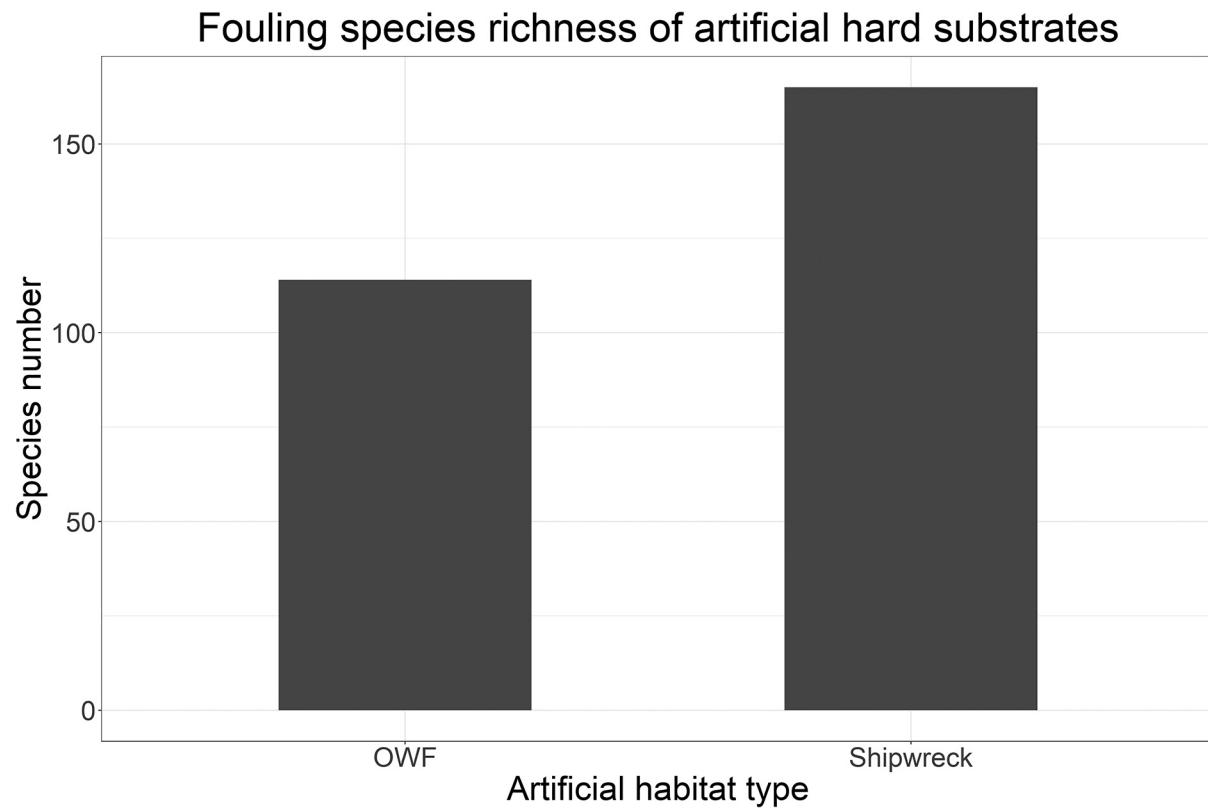
Figure 2. Number of hard-substrate macrobenthic species (subtidal only) recorded at offshore wind farms (OWF) and at shipwrecks.

Table 2. Non-indigenous species (subtidal only) at offshore wind farms (OWF) and shipwrecks.

Species	Higher Taxon	Non-indigenous	Habitat
Diplosoma listerianum	Ascidiacea	Cryptogenic	OWF
Lysianassa ceratina	Amphipoda	Range-expanding	Shipwreck
Monocorophium acherusicum	Amphipoda	Cryptogenic	Shipwreck and OWF
Monocorophium sextonae	Amphipoda	Introduced	Shipwreck and OWF
Fenestrulina delicia	Bryozoa	Introduced	OWF
Amphibalanus improvisus	Cirripedia	Cryptogenic	OWF
Perforatus perforatus	Cirripedia	Range-expanding	OWF
Crepidula fornicata	Gastropoda	Introduced	Shipwreck and OWF
Janira maculosa	Isopoda	Introduced	Shipwreck
Eulalia aurea	Polychaeta	Range-expanding	Shipwreck
Lysidice ninetta	Polychaeta	Range-expanding	Shipwreck
Hymeniacidon perlevis	Porifera	Introduced	Shipwreck

at OWFs, while another species was shared in both habitats.

The number of non-indigenous species (12, Table 2, Fig. 3) remains low compared to indigenous species (197) (Fig. 1). Of the non-indigenous species, five are species introduced through human activities, four are range-expanding species because of climate change, and for the remaining five, their true origin is uncertain. There is no clear difference in non-indigenous species richness between OWFs and shipwrecks.

4. Discussion

The current and future massive rollout of offshore wind in the North Sea will lead to a huge proliferation of artificial hard substrates. These provide additional habitat for hard substrate associated fauna in a largely soft bottom environment, and will attract species that would otherwise not be able to colonise the area. This increase in biodiversity is thus not unexpected or remarkable, but what is observed all around the North Sea (e.g., Whomersley & Picken 2003; Zintzen & Massin 2010; Kerckhof *et al.* 2012; Krone 2012; Van Moorsel 2014; Wetzel *et al.* 2014; Coolen *et al.* 2020).

We found that species richness was markedly higher on shipwrecks than on offshore wind farm foundations and the surrounding scour protection layers in Belgian waters. This can be attributed to the age of the structures, which is significantly older for shipwrecks. For example, the species richness of sponges is much higher on shipwrecks than on OWFs (10 vs 1). These slow-growing, fragile species are characteristic for 'mature' and undisturbed communities on hard substrates (Wahl 2009; Hiddink *et al.* 2017; Malecha & Heifetz 2017). In the Netherlands, species richness on older oil and gas platforms was also higher than on younger wind turbine foundations (Coolen *et al.* 2020), although total species richness remained below of what we have observed for the shipwrecks and OWFs in the BPNS. As offshore wind farms age, we may thus expect that the species composition might become more similar to the one observed at shipwrecks.

At the same time, shipwrecks provide more structural heterogeneity than offshore wind farms, possibly enabling higher species richness at shipwrecks. Zintzen *et al.* (2006), for example, observed a clear differentiation in species composition between horizontal and vertical sections of shipwrecks. Wind turbines lack this structural complexity, especially the foundations which are smooth, nearly vertical structures rising from the seabed. Generally, turbine foundations are massively covered by a shallow subtidal *M. edulis* zone and a deeper
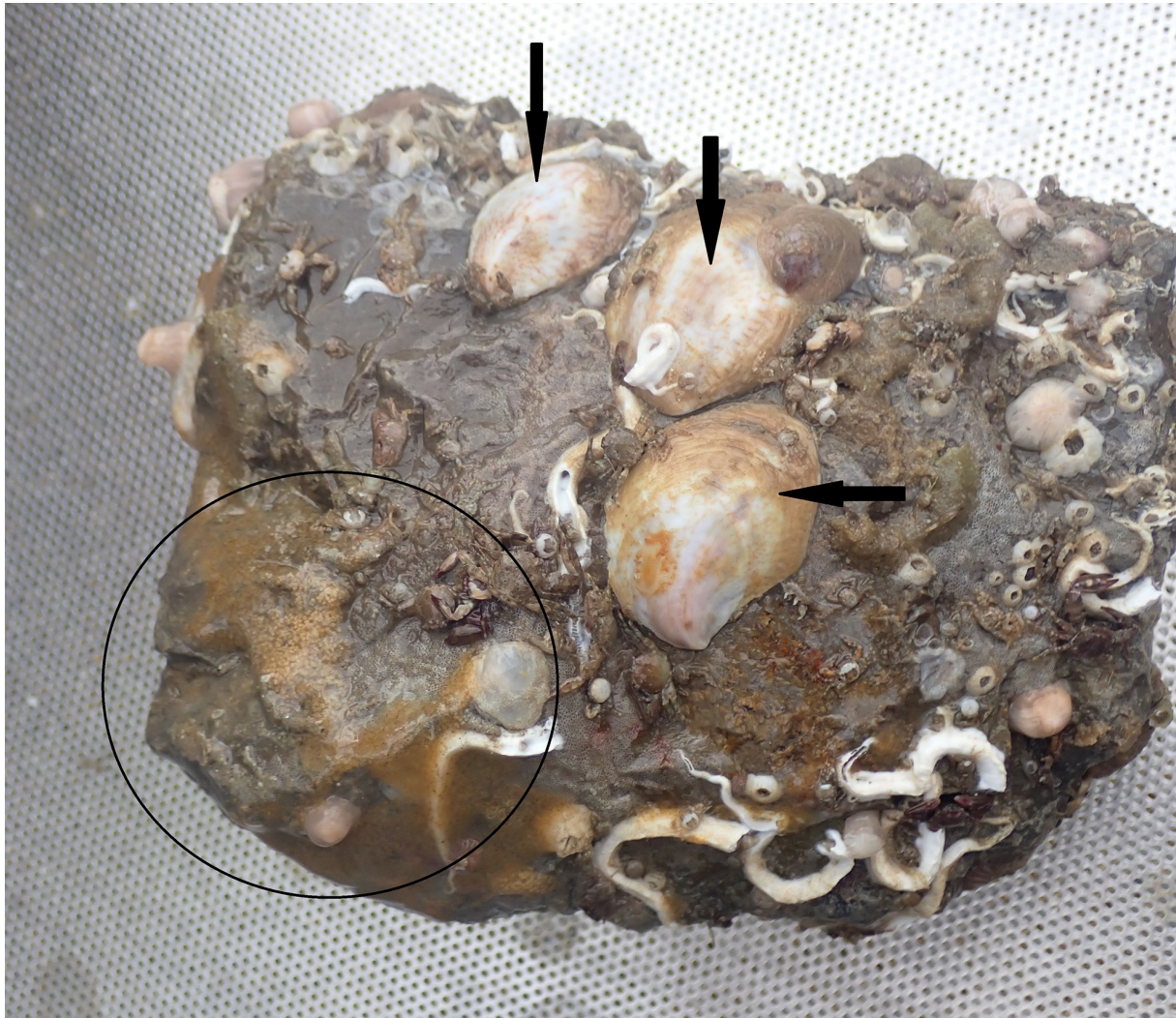
Figure 3. Scour protection rock with the non-indigenous species *Crepidula fornicata* (arrows) and *Diplosoma listerianum* (circle) (©RBINS, F. Kerckhof).

Metridium senile zone, with tubes of *Jassa* amphipods interspersed (Krone *et al*. 2013; De Mesel *et al*. 2015; Degraer *et al*. 2020). This dominance of only a few species can likely be attributed to the lack of structural complexity. Increasing the structural complexity of both the turbine foundations and the scour protection layers might increase species diversity and thus give rise to more diverse communities.

Next to a difference in species richness between both artificial hard substrates, also the species composition differs between them, with OWFs having almost 40% of unique species, and shipwrecks almost 60%. This might indicate that OWFs, which are spanning

the entire water column, represent a different habitat than shipwrecks, which are only extending a few meters above the seafloor. However, the OWFs in the present study are also monitored more intensively than the shipwrecks, and their unique species might just not be recorded yet from shipwrecks.

The proportion of non-indigenous species found on the subtidal artificial hard substrates in this study is lower compared to other artificial hard substrates in intertidal and/or coastal areas. For example, seven non-indigenous species are found in the intertidal zone of OWFs, accounting for 23% of the species found (Kerckhof *et al*. 2016), while subtidally, non-indigenous species account

for only 6.1% of the species (also seven species), despite the much larger substrate surface available for subtidal species. Some of these species are range-expanding species, arriving naturally in our areas as a result warming waters and the presence of suitable, previous non-existent hard substrates. Other species are introduced by human activities and would otherwise not have made it to our seas. Infamous examples of this are the subtidal slipper limpet *Crepidula fornicata*, originating from the North-West Atlantic, and the Japanese oyster *Crassostrea gigas*, coming from the West Pacific. *Crepidula fornicata* is a competitor for space, and can inhibit settlement of epifaunal species, including reef-forming species such as the European flat oyster *Ostrea edulis* and the Ross worm *Sabellaria spinulosa*. *Crassostrea gigas* can also compete for space but is more restricted to the intertidal and shallow subtidal, therefore coming less into competition with indigenous species (except with blue mussel *M. edulis*). The competition for space is, however, not only restricted to introduced non-indigenous species, also the indigenous anemone *M. senile* is a competitor for space. This 'dominant native' thrives on artificial hard substrates and can lead to less diverse communities (Kerckhof *et al.* 2019).

Hard substrate associated species, both indigenous and non-indigenous, can use shipwrecks and turbine foundations as stepping stones to strategically position themselves in the soft-sediment dominated North Sea and to colonise new areas herein (Zintzen & Massin 2010). The stepping stone effect of both artificial hard substrates can be regarded as synergistic. On the one hand, turbine foundations offer an intertidal and shallow subtidal zone, colonised by species such as *M. edulis* and *C. gigas* (Kerckhof *et al.* 2016), which are generally absent on deeper subtidal shipwrecks (Krone *et al.* 2013). On the other hand, shipwrecks have a higher structural complexity, providing opportunities for a more diverse set of colonising species, while both structures are strengthening the position of species such as *M. senile*, *C. fornicata* and

Tubularia spp, which are thriving on both these artificial substrates.

Despite the high species richness and varied species composition of the artificial hard substrates investigated in this study, these substrates harbour different communities than natural hard substrates (Zintzen 2007; Krone *et al.* 2013; Kerckhof *et al.* 2017). Even if left undisturbed for decades, artificial hard substrates cannot serve as a replacement for the loss and ecological decline of natural hard substrates. Although biodiversity is expected to increase over time on artificial hard substrates, they will form their own typical assemblages, as currently evaluated by the EUNIS Habitat Classification: '*Faunal Communities on Atlantic Circalittoral Artificial Hard Substrate*' (code MC1228, EUNIS habitat classification, updated version March 2022). If, however, biodiversity is a criterium in the decommissioning debate of OWFs (Fowler *et al.* 2020), it might be an option to leave at least part of the turbine foundation and the surrounding scour protection layer in place.

In conclusion, the subtidal epifaunal hard substrate communities of shipwrecks and offshore wind farms are different, with a higher species richness at shipwrecks and a marked difference in species composition between both habitats. Higher structural complexity and older age might be reasons for the higher species richness of shipwrecks. It is unclear if OWFs will reach similar species richness as shipwrecks over time, if structural complexity is not increased. Increasing the complexity of the scour protection layer surrounding turbine foundations might increase species richness and thus support biodiversity, an approach called nature-inclusive design of marine infrastructure. This is currently investigated in research projects such as the EU Horizon 2020 project UNITED or the EDEN2000 project, financed by the Belgian Federal Public Service Environment.

Acknowledgements

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CHAPTER 6

TURBINE SIZE IMPACTS THE NUMBER OF SEABIRD COLLISIONS PER INSTALLED MEGAWATT AND OFFERS POSSIBILITIES FOR MITIGATION

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Abstract

As the offshore wind energy technology is rapidly progressing and because wind turbines at sea have a relatively short life span, repowering scenarios are already being discussed for the oldest wind farms. Ongoing developments result in larger wind turbines and an increased open airspace between turbines. Despite taller towers having larger rotor swept zones and therefore a higher collision risk area compared to smaller-sized turbines, there is increasing evidence that fewer but larger, more power-efficient turbines may have a lower collision rate per installed megawatt. As such, turbine size can offer an opportunity to mitigate seabird fatalities by increasing the clearance below the lower rotor tip.

We assessed the seabird collision risk for a hypothetical repowering scenario of the first offshore wind farm zone in Belgian waters with larger turbines and the effect of an additional increase in hub height on that theoretical collision risk.

For all species included in this exercise, the estimated collision risk decreased in a repowering scenario with 15 MW turbines (40.4% reduction on average) because of higher clearance between the lower tip of the turbine rotor and the sea level, and the need for a lower number of turbines per km². Increasing the hub height of those 15 MW turbines with 10 m, further decreases the expected number of seabird collisions with another 37% on average.

However, terrestrial birds and bats also migrate at sea and the effect of larger turbines on these taxa is less clear. Possibly even more terrestrial birds and bats are at risk of collision compared to the current turbines. So, while larger turbines and increasing the hub height can be beneficial for seabirds, this likely needs to be applied in combination with curtailment strategies, which stop the turbines

during heavy migration events, to reduce the impact on other species groups.

1. Introduction

At present, 399 wind turbines, with capacities ranging between 3.3 and 9.5 MW, are operational in an area designated for renewable energy in Belgian waters. This totals an installed capacity of 2.26 GW (chapter 1 of this report). The first turbines were constructed in 2008 and the last ones in 2020. As the offshore wind energy technology is rapidly progressing and because wind turbines have a relatively short life span of ca 20-25 years (Bonou *et al*. 2016), repowering scenarios are already being considered. A second zone for offshore renewable energy in Belgian waters, the princess Elisabeth zone, was designated in the revised marine spatial plan in 2020 and is anticipating an installed capacity ranging between 3.15 and 3.5 GW. These developments reduce our dependence of fossil fuels but on the other hand can pose negative effects on birds. Birds are affected directly through collision with structures, and indirectly through behavioural responses to the altered habitat (Drewitt & Langston 2006; Fox *et al*. 2006). To estimate the direct seabird mortality caused by collisions with offshore wind turbines, avian collision risk models (CRM) are used. These models integrate a variety of bird and turbine parameters to calculate the theoretical collision risk. The model outputs should be handled with care and not be interpreted as absolute figures considering the uncertainty around specific input parameters. They are however very useful to compare development scenarios and to identify the species most at risk (Brabant *et al.* 2020). CRMs are therefore routinely used in environmental impact assessments and the outputs may have actual consequences for wind farm developments, weighing on consenting decisions (Masden *et al.* 2021).

The developments in wind energy technology result in increasingly large wind turbines, with several types already exceeding

a capacity of 10 MW (De Kooning *et al.* 2021). Currently, the largest turbines on the market are the 14 MW Siemens Gamesa 14-222 DD, the 14 MW GE Renewable Energy's Haliade X and the 10 MW Vestas V164. In February 2021, Vestas announced a new 15 MW V236 offshore turbine to be available by 2022, thus pushing the size of wind turbines even further with a rotor diameter of 236 m. The V236-15.0 MW was selected for the He Dreiht offshore wind farm (900 MW) in the German North Sea in 2025 (De Kooning *et al.* 2021).

Larger turbines have higher rotor planes and an increased open airspace between neighbouring turbines. Despite taller towers having larger rotor swept zones and therefore a higher collision risk area compared to smaller sized turbine, there is increasing evidence that fewer but larger, more power-efficient turbines have a lower collision rate per megawatt (Marques *et al.* 2014; Shimada 2021). This technological evolution towards larger turbines can thus have a positive side-effect as they may result in fewer seabird collisions.

Turbine size also offers the opportunity to mitigate seabird fatalities (Arnett & May 2016), as increasing the draught height of turbines, i.e., the height between the water surface and the lower tip of the rotor, by 5 or 10 meter can be applied as a measure to reduce collision risk. In the UK this measure is being tested in three OWFs (Hornsea 2, East Anglia 3 and Vanguard; MacArthur Green 2019) and is recommended by Natural England for all future projects in the North Sea. It is also being considered in the Netherlands (personal comment Witteveen & Bos).

The aim of this report is to assess the seabird collision risk for a hypothetical repowering scenario of the first offshore wind farm zone in Belgian waters with larger turbines and to simulate the effect of an increased draught height on that theoretical collision risk.

2. Material and methods

2.1. Research strategy

The first offshore wind farm zone in Belgian waters has a surface of 238 km². The area is fully occupied by nine operational wind farms with a total installed capacity of 2.26 GW, resulting in an average capacity of 9.5 MW/km². New developments and repowering scenarios, however, are aiming to increase the capacity per km². In the second zone for renewable energy, the Princess Elisabeth zone (PEZ), the aim is an installed capacity ranging between 3.15 and 3.5GW, which would accord to 11.1 and 12.3 MW/km² respectively and an average turbine capacity between 15 and 17 MW. Repowering the first zone to 11.7 MW/km², the average of the goal for the PEZ, would imply 181 turbines of 15 MW and an installed capacity of 2.71 GW.

In this study we calculated the collision risk of six seabird species for (1) the current operational wind farms in the first zone for renewable energy in Belgian waters, (2) a repowering scenario of that first zone with 15 MW turbines with a standard height and (3) a repowering scenario with 15 MW turbines with a 10 m increased turbine hub height.

2.2. Collision risk modelling

Estimating bird collisions at sea is routinely done using theoretical CRMs, which calculate species-specific collision risks based on wind farm and turbine specifications, bird-related parameters and bird densities (Masden & Cook 2016). The CRM most frequently used is the one developed by Band (2012). The Band model (Band 2012) has undergone several modifications 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 flight height between the lowest and the highest level of the rotor swept area. 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) CRM, providing a more robust and transparent method of accounting for uncertainty in the estimation of seabird collision rates, also including the four model options developed by Band.

The stochastic CRM (sCRM) is available in two ways: as an online Shiny application (https://dmpstats.shinyapps.io/avian_stochcrm/) and as an R package that can be downloaded and run locally (https://github.com/dmpstats/stochCRM). We calculated the collision risk with the model option 3 of the sCRM in the online application. Option 3 of the model is considered the most realistic calculation (McGregor *et al.* 2018). The input parameters needed for the sCRM are further described in the paragraphs below.

The sCRM was run for 3000 iterations for the three different scenarios, resulting in species-specific numbers (± standard deviation) of collision victims per year.

2.3. Turbine related input data

For the second scenario with next generation wind turbines at sea, we used the specifications of the Vestas v236 15 MW turbine prototype. This turbine has a rotor diameter of 236 m, while the hub height is site specific (Vestas website). The height of offshore turbines is determined by the height of the transition piece (TP) which is determined by the wave regime in the area. On top of the transition piece, a safety clearance is needed so the lowest tip of the rotor can move freely above the TP platform, crane, etc. In Belgian waters the TP height would be around 20 m above the lowest astronomical tide (LAT). Adding 10 to 15 m clearance above the TP and the radius of the rotor (118 m), results in a hub height of about 150 m LAT for this Vestas v236 15 MW turbine (pers. comm. Belgian windfarm developer Parkwind).

Table 1. Wind farm and turbine related input data for the stochastic CRM for the currently operational wind farms and both repowering scenarios in the first renewable energy zone in Belgian waters.

	N of turbines	Width (km)	Latitude (°)	Tidal offset (m)**	Turbine model (MW)	Number 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	21	5.4	10.95	5.2
C-Power	54	4.4	51.55	4.3	6.15	3	63	27	5	12.22	5.6
Rentel	42	4.7	51.59	4.3	7.35	3	77	24.5	5	11.62	5.4
Northwind	72	3.1	51.62	4.3	3.3	3	56	23	4	14.85	6
Seastar	30	2.8	51.64	4.3	8.4	3	83.5	21.5	5.4	10.95	5.2
(No)Belwind	106	5.1	51.67	4.3	3.3	3	56	23	4	14.85	6
Northwester II	23	4.2	51.69	4.3	9.5	3	82	20.4	5.4	10.52	5.1
Mermaid	28	3.6	51.71	4.3	8.4	3	83.5	21.4	5.4	10.95	5.2
Repowering scenario	181	35	51.62	2.0	15	3	118	28.0	7.8	10.52	5.1
Repowering scenario +10m	181	35	51.62	2.0	15	3	118	38.0	7.8	10.52	5.1

*scenario 1 is the sum of the collisions calculated for the nine wind farms currently installed. 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.

**tidal offset is the difference between mean sea-level (MSL) and highest astronomical tide (HAT).

***air gap is the distance between the lowest tip of the rotor and the sea-level measured as HAT (Masden *et al*. 2015).

The CRM was applied on the current OWFs in Belgian waters and on two repowering scenarios, the first one with the Vestas v236 15 MW turbine with a 150 m LAT hub height and the second one with an increased hub height of 160 m LAT.

The input data for the currently installed OWFs (scenario 'as is') and the two repowering scenarios are shown in Table 1. For the latter we considered the first OWF zone as one homogenous wind farm with Vestas v236 turbines (Table 1). As the Vestas v236 turbine is under development some parameters remain unknown because of non-disclosure agreements. Therefore, rotor speed and pitch were taken from Gyimesi *et al*. (2018) for the biggest turbine currently operational in Belgian waters, i.e., the Vestas v164 turbine. Blade width of the Vestas v236 is also unknown, so we extrapolated the blade width from the Vestas v164. Information on turbine activity per month were taken from Masden *et al*. (2015).

2.4. Species selection

The focus of this study was on the six most abundant seabird species inside the Belgian offshore wind farms: northern gannet *Morus bassanus*, common gull *Larus canus*, lesser black-backed gull *Larus fuscus*, herring gull *Larus argentatus*, great black-backed gull *Larus marinus* and black-legged kittiwake *Rissa tridactyla* (Vanermen *et al*. 2019). Other species were not selected because of insignificant post-construction densities inside the wind farms or because they are at low risk of collision because of 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 Thorntonbank (Vanermen *et al*. 2019). Strangely, however, this species is rarely recorded flying inside the wind farm, resulting in negligible densities of flying birds.

Table 2. Bird related input data for the stochastic CRM.

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).

2.5. Bird related input data

Avoidance rates and flight speed data are taken from the empirical study of Skov *et al*. (2018). Body length and wingspan are taken from Snow & Perrins (1998). Nocturnal activity data for lesser black-backed and herring gull are described by Gyimesi *et al*. (2017), based on telemetry data from birds in Dutch, Belgian and English colonies. For the other species the assumptions of Garthe & Hüppop (2004) are adopted. 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 (Table 2).

2.6. Bird density data

Monthly post-construction bird surveys started in 2010 and 2013 on the Bligh Bank and Thorntonbank offshore wind farms

respectively and were continued for five years. Details on the applied methodology and sampling scheme can be consulted in Vanermen *et al*. (2016, 2019). During these surveys flying birds and birds on the water were counted separately and we selected only the flying birds to calculate seasonal densities as input for the sCRM.

The post-construction density data of the Bligh Bank and Thorntonbank offshore wind farms were averaged and used to calculate the collision risk for the different scenarios of the first renewable energy zone (Table 3).

3. Results

For the current nine wind farms in the first Belgian zone for renewable energy (scenario 1) a total of 60.7 ± 236.4 collisions

Table 3. Average 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 Thorntonbank in winter (December-February), spring (March-May), summer (June-August) and autumn (September-November).

Season	Northern gannet	Common gull	Lesser black-backed gull	Herring gull	Great black-backed gull	Black-legged kittiwake
Winter	0.00 ± 0.00	0.29 ± 0.50	0.03 ± 0.06	0.03 ± 0.11	0.07 ± 0.13	0.56 ± 0.61
Spring	0.01 ± 0.03	0.01 ± 0.02	0.25 ± 0.39	0.01 ± 0.04	0.01 ± 0.04	0.07 ± 0.17
Summer	0.01 ± 0.03	0.00 ± 0.00	0.18 ± 0.17	0.00 ± 0.00	0.03 ± 0.05	0.00 ± 0.00
Autumn	0.01 ± 0.04	0.02 ± 0.07	0.04 ± 0.10	0.01 ± 0.02	0.17 ± 0.25	0.12 ± 0.22

Table 4. sCRM option 3 output (3000 iterations) resulting in an estimated number of collisions per species per year (± SD) for the three different scenarios.

Species	Scenario 1: current OWFs (n collisions/year ± SD)	Scenario 2: repowering scenario (n collisions/year ± SD)	Scenario 3: repowering scenario +10 m (n collisions/year (± SD)
Northern gannet	0.2 ± 2.5	0.0 ± 0.02	0.0 ± 0.26
Common gull	6.9 ± 94.8	3.9 ± 62.5	1.5 ± 33.2
Lesser black-backed gull	16.9 ± 133.9	5.8 ± 142.0	4.9 ± 125.2
Herring gull	0.8 ± 24.8	0.5 ± 15.4	0.1 ± 5.8
Great black-backed gull	32.3 ± 202.8	25.7 ± 346.6	6.7 ± 105.9
Black-legged kittiwake	3.6 ± 26.6	0.3 ± 8.4	0.5 ± 11.9
Total	60.7 ± 263.4	36.2 ± 380.1	13.7 ± 167.8

per year are expected for the six selected seabird species (Table 4). Lesser black-backed and great black-backed gull have the highest risk of collision in the current OWFs and account for 81.5% of the total number of collisions. Only 0.7% of the total number of expected collisions are attributed to northern gannets. The total number of collisions per year decreases with 40.4% in a development scenario with 181 15 MW turbines (scenario 2). This reduced collision risk is species-specific and for the species with the highest collision risk the reduction varies between 20.4% for great black-backed gull, 43.5% for common gull and 65.7% for lesser black-backed gull. Increasing the hub height with 10 m (scenario 3) further decreases the expected number of collisions in the current OWFs with another 37% on average, relative to scenario 2.

Repowering the first zone for renewable energy in Belgian waters with 15 MW turbines reduces the number of collisions per MW installed capacity with an average of 50.4% for the six selected seabird species (Table 5).

Increasing the hub height of these 15 MW turbines with 10 m results in 81.2% less collisions per MW compared to the current OWFs.

4. Discussion

Lesser and great black-backed gull have the highest risk of collision in the current OWFs. For the other four species the risk is limited which is explained by their lower densities inside the wind farms (Vanermen *et al*. 2019) and their lower flight altitude (Johnston *et al*. 2014).

For all species included in this exercise, the estimated collision risk decreases in a repowering scenario with 15 MW turbines (40.4% reduction on average). The observed reduction is a combination of higher clearance between the lower tip of the turbine rotor and the sea level, and the lower number of turbines per km². Seabird flight height profiles indicate that most birds at risk of collision are flying in the lower part of the swept area (Johnston *et al*. 2014). Thus, a higher clearance above

Table 5. Number of collisions of the six selected species per year per MW installed capacity for the three different scenarios.

	Installed capacity (MW)	Number of collisions/year ± SD	Number of collisions/(year*MW)
Current OWFs	2260	60.7 ± 263.4	0.027
Repowering scenario	2715	36.2 ± 380.1	0.013
Repowering scenario +10 m	2715	13.7 ± 167.8	0.005

the water surface can significantly reduce the number of birds exposed to collision risk. The average clearance of the current OWFs is 22.7 m HAT. For the 15 MW turbines considered in the repowering scenario this would be 28.0 m HAT. Also, because the capacity of the individual turbines is higher, less turbines are needed which further reduces the collision risk per megawatt. It is important to note that it is unclear how repowering scenarios with fewer but bigger turbines will affect the seabird densities inside the wind farms. We used the post-construction seabird densities observed in the current wind farms. The increased spacing between larger turbines might result in a reduced avoidance response of seabirds and thus higher densities, which in turn would increase the collision risk again. This trade-off needs to be considered on a case-by-case basis (Harwood & Perrow 2019). Possibly also the flight height distribution of seabirds might be altered in wind farms with increased spacing between turbines. Telemetry studies could provide valuable data on flight height distributions in wind farms with different configurations.

Davies & Band (2012) consider turbine height as a management tool for bird collision risk at offshore wind farms. As the flight height distribution of seabirds is skewed to relatively low height above the sea surface (Furness *et al.* 2013; Johnston *et al.* 2014), increasing the clearance of the rotor-swept area above sea-level is likely to reduce the risk of seabird collisions through a reduction of bird densities at risk height (Harwood & Perrow 2019). This is confirmed by the further reduction in collision risk through a 10 m increase of the hub height of the 15 MW turbines. This third scenario in our model calculations reduces the total number of collisions with 77.4% compared to the current OWFs, but also with 62.2% compared to the standard repowering scenario. The outcome of our model calculations are in line with the calculations of MacArthur Green (2019) which yielded a reduction in predicted collision risk of 41% on average, for an increase in turbine clearance of 5 m (from 22 to 27 m HAT) for

the Norfolk Vanguard offshore wind farm. Increasing the turbine hub height can thus be an effective mitigation measure to reduce seabird collisions and should be considered in future developments and repowering scenarios, although this creates additional engineering and material costs.

Aside from seabirds, also several hundred million terrestrial birds of approximately 250 species migrate over the North Sea every year, many of them being nocturnal migrants (Lack 1963; Lensink *et al.* 1999; Hüppop *et al.* 2006) and are as such at risk of collision with offshore wind turbines (Hüppop *et al.* 2019). Especially Blackbird *Turdus merula*, Song Thrush *Turdus philomelos*, Redwing *Turdus iliacus* and Robin *Erithacus rubecula* migrate in high numbers across the North Sea at night (Krijgsveld *et al.* 2011; Fijn *et al.* 2015). During migration, birds fly at greater altitudes than when foraging or commuting between sites (Garthe & Hüppop 2004) and choose the altitude stratum in which their energy costs are lowest (Hüppop *et al.* 2006), ranging from sea level up to 10 km. Passerine migrants tend to fly relatively low in the atmosphere (<1000 m) during most nights (Gauthreaux 1991), and their flight altitude is affected by wind and other atmospheric variables (Bruderer *et al.* 1995).

Bird radar data from a location inside one of the OWFs in Belgian waters demonstrate that large proportions of nocturnal migrants fly at rotor height (Brabant *et al.* 2021a). 44.3% of the radar recordings during autumn migration in 2019 were at rotor height of the current turbines (Brabant *et al.* 2021a). This figure would increase to 58.3 % for the Vestas v236 turbines that were used in the repowering scenarios in this study. As an example, on 29 October 2019 there was heavy nocturnal migration. The migration traffic rate (MTR) from 7 to 8 pm was 613 birdtracks per km per hour at rotor height. For the new Vestas v236 turbines this MTR at rotor height would have been 967 birdtracks per km per hour, as this type of turbine has a larger rotor pane. This means that more migrating songbirds will

be at risk with bigger compared to smaller turbines. With this MTR, a scenario with 181 Vestas v236 turbines would result in 9% more collisions of songbirds than the current 399 wind turbines. So, the technological developments towards bigger turbines might benefit seabirds, but can result in more songbird collisions.

Bats are also known to migrate above the North Sea (Brabant *et al*. 2021b; Lagerveld *et al*. 2021). The flight height distribution of bats at sea remains unclear. It was thought that they generally fly at low altitudes, but bats were also detected acoustically at hub height in OWFs in Belgian waters (Brabant *et al*. 2019). The number of detections at nacelle height was around 10 % of the detections at low altitude (15m above sea level). Hatch *et al*. (2013) visually observed bats at higher altitudes at sea (>100 m). So, at least a part of the migrating bats are at risk of collision with offshore wind turbines. The behavioral response of bats to offshore turbines is also poorly understood and some studies indicate that bats are attracted to the turbines lighting (Voigt *et al*. 2018). It is therefore difficult at this point to predict how bats will be impacted by larger turbines. Taller turbines may even result in greater mortality for bats (Barclay *et al*. 2007). Curtailment strategies that idle turbines during intense migration events are therefore probably the most effective measure to mitigate the collision risk for migratory songbirds and bats (Cook *et al*. 2011; Marques *et al*. 2014; May 2017; Boonman 2018). To conclude, while larger turbines and increasing the hub height can be beneficial for seabirds, they likely need to be applied in combination with curtailment strategies to reduce the impact on other species groups.

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CHAPTER 7

WHAT DRIVES HARBOUR PURPOISE (*PHOCOENA PHOCOENA*) RESPONSE TO PILE DRIVING SOUND?

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Abstract

In the southern North Sea, offshore wind farm construction usually requires hydraulic pile driving resulting in high levels of impulsive sound. Despite recent advances in noise-mitigation technology, harbour porpoises (*Phocoena phocoena*) respond to this pile driving over a period of hours to days per driven pile, depending on the distance at which the animals were disturbed. We used passive acoustic monitoring (PAM) datasets from 2018 to 2020, including the construction periods of three offshore wind farms (Norther, Northwester 2 and SeaMade), to determine the factors which influenced the likelihood of detecting harbour porpoise (*Phocoena phocoena*) before, during and following pile driving in the Belgian part of the North Sea (BPNS). During pile driving and in the 24 hours after pile driving, mean detection rates of porpoises reduced up to 20 km from the pile driving location although both the magnitude and duration of this reduction decreased markedly with increasing distance. Generalized Additive Modelling (GAM) found distance to the construction site (as a proxy found received sound level) to be the main driver for porpoise response to pile driving with seasonality, time of day and type of sound mitigation having a limited but significant effect on the spatial and temporal extent of avoidance of the construction area by porpoises. In the immediate vicinity of the construction site, the reduction in porpoise detection rates starts even prior to the pile driving suggesting the presence other sources of disturbance in this area. Our results suggest that efforts to reduce the impact of underwater noise generated by future offshore wind farm construction on marine life should aim to limit not only the noise levels generated but also the overall duration of the construction period.

1. Introduction

The harbour porpoise (*Phocoena phocoena*) is by far the most common marine mammal in the BPNS, after several years of virtual absence (Haelters *et al.* 2011). Despite interannual variation, harbour porpoises show a distinct spatial and temporal distribution in Belgian waters with relatively high densities from January to April and lower numbers from May to August, plus they tend to concentrate in more northerly and offshore waters (Haelters *et al.* 2011, 2016; Augustijns 2018). The animals present in Belgian waters do not form an isolated population, but are part of a much larger population, which extends into the southern and central North Sea. In the greater North Sea, the harbour porpoise is considered vulnerable because of high bycatch levels (Kaschner 2003) and its exposure to increasing levels of noise pollution ranging from continuous shipping noise (Wisniewska *et al.* 2018) to impulsive noise from, e.g., pile driving (Brandt *et al.* 2018), and seismic surveys (Van Beest *et al.* 2018). Nonetheless, the species is protected by both national (Belgian Government 2001) and EU law (European Union 1992), and consequently deliberate actions of killing, disturbing, injuring, or habitat deterioration are prohibited throughout its range. In the absence of mitigating measures, the high levels of impulsive underwater sound generated during pile driving can potentially kill, injure and disturb marine mammals depending on their distances from the source (see, e.g., Carstensen *et al.* 2006; Bailey *et al.* 2010) with some studies indicating potential negative cumulative impacts on the harbour porpoise population the North Sea as a result of planned wind farm development over the next decade (de Jong *et al.* 2019).

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 260 GW by 2050, with intermediate targets of at least 76 GW by 2030 and 193 GW by 2040 (North Seas Energy Cooperation 2022). Concerns over the possible impact of high intensity impulsive sound generated during the construction of these offshore wind farms on harbour porpoise have been a driving force in determining national impulsive noise regulations in North Sea countries with Germany, the Netherlands and Belgium all formulating different, but similar, underwater sound thresholds (see Rumes *et al.* 2016 for a comparison). In Belgium, this concern over the high levels of underwater noise being generated during pile driving operations for the building of the first offshore wind farms (Norro *et al.* 2010, 2013) and the observed large-scale avoidance of the construction zone by porpoises (Haelters *et al.* 2011, 2013) led to the formulation of strict mitigating measures which included both seasonal pile driving restrictions (Rumes *et al.* 2013), and a threshold for impulsive underwater sound in the Belgian part of the North Sea (BPNS) at 185 dB re 1 μPa (Sound Pressure Level, zero to peak) at 750 m from the source (Anonymous 2012). This led offshore wind farm developers in the BPNS to apply noise mitigation systems which made incremental progress in complying with the national threshold (Rumes & Degraer 2020). When effective noise mitigation was applied, reductions to the spatial and temporal extent of avoidance of the construction area by porpoises were observed (Rumes & Zupan 2021). Nonetheless, and especially in the immediate vicinity of the pile driving site, a prolonged reduction in porpoise detection rates was observed.

In this chapter, we applied a GAM model to data from the construction of three wind farms to determine the factors which influence the observed spatial and temporal extent of harbour porpoise avoidance during pile driving and thereby provide an improved basis for formulating effective mitigating measures.

2. Material and methods

2.1. Study area and sites

The Southern Bight of the North Sea includes the BPNS with a surface of approximately 3450 km². The BPNS only covers 0.5% of the entire area of the North Sea and is characterized by shallow waters with a maximum depth of 45 m and a complex system of sandbanks. In 2004, in the western part of the BPNS, a 264 km² zone was designated for renewable energy. In 2011, this zone was adjusted on its Northern and Southern side to ensure safe shipping traffic thereby reducing the area to 238 km². Between 2009 and 2020, nine projects have constructed wind farms in this part of the BPNS.

For this study we focused on three wind farms constructed between 2018 and 2020: Norther, Northwester 2 and SeaMade (Figure 1).

Norther NV obtained an environmental permit on January 18th 2012 to build and operate its offshore wind farm. The windfarm was built at a distance of 20 km from the coastline to the south of the Thornton bank. The total capacity of this wind farm of 370 MW is provided by 45 turbines, each with a capacity of 8.4 MW. Pile driving for the Norther wind farm comprised 45 piling events from June 8th up to November 12th, 2018. Pile diameter ranged from 7.2 to 7.8 m, penetration depth lay between 24 to 47 m and total piling time varied between 52 min and 3h43 min. All piles were installed using an S-3500 Hydraulic Hammer (maximum energy per pile 3028 ± 456 kJ). The contractor was legally obliged to turn on an acoustic deterrent device one hour before the start of piling. Construction logs show that the acoustic deterrent device was often switched on much earlier, in casu between 60 to 490 minutes (on
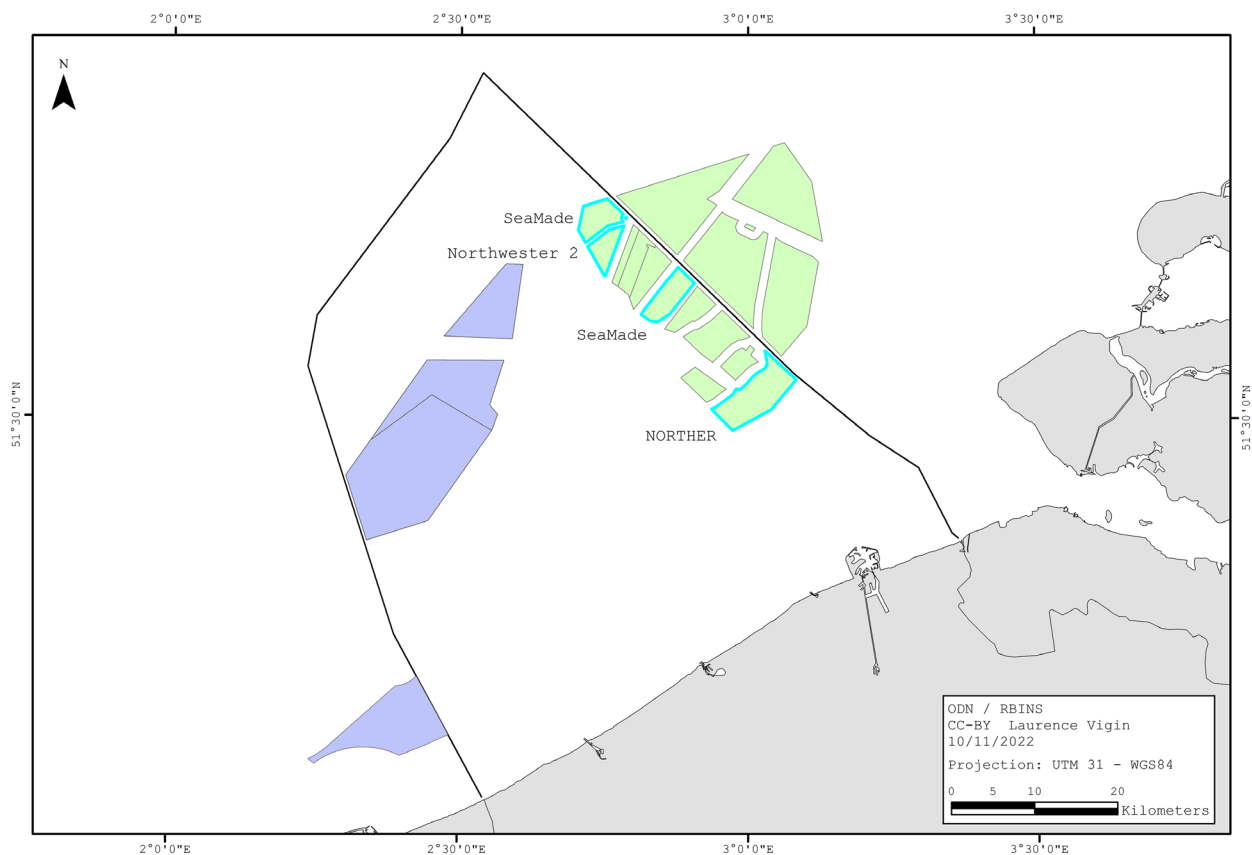
Figure 1. Operational (green) and planned (blue) offshore wind farm zones in the Belgian part of the North Sea and adjacent Dutch and French waters showing the location of the three wind farms constructed between 2018 and 2020: Norther, Northwester 2 and SeaMade.

average 150 minutes) before the start of pile driving (Rumes & Degraer 2020).

The second wind farm, NV Northwester 2, is located at 51 km off the coast of Zeebrugge to the northwest of Nobelwind, was granted an environmental permit on 18 December 2015. The total capacity of this wind farm of 219 MW is provided by 23 turbines, each with a capacity of 9.5 MW. Pile driving for the Northwester 2 wind farm comprised 24 piling events (23 turbines and one offshore high voltage station) from July 29th up to November 13th, 2019. Pile diameter ranged from 7.4 to 8.0 m, penetration depth lay between 29 to 39 m and total piling time varied between 1 h 36 min and 3h40 min. All piles were installed using an S-3000 Hydraulic Hammer (maximum energy per pile 1942 ± 406 kJ). The contractor was legally obliged to turn on an acoustic deterrent device 30 minutes before the start of piling. Construction logs show that the acoustic deterrent device was switched on between 32 to 342 minutes (on average 66 minutes) before the start of pile driving (Rumes & Degraer 2020).

The third wind farm, SeaMade, is comprised of two separate sections located at 40 and 54 km off the coast of Zeebrugge and was granted an environmental permit on 13 April 2015. The total capacity of this wind farm of 487 MW is provided by 58 turbines, each with a capacity of 8.4 MW. Pile driving for the SeaMade wind farm comprised 60 piling events (58 turbines and two offshore high voltage station s) from September 8th, 2019, up to January 2nd, 2020. Pile diameter ranged from 7.5 to 8.0 m, penetration depth lay between 27 to 41 m and total piling time varied between 1h5 min and 3h26 min. All piles were installed using an S-4000 Hydraulic Hammer (maximum energy per pile 1930 ± 423 kJ). The contractor was legally obliged to turn on an acoustic deterrent device 30 minutes before the start of piling. Construction logs show that the acoustic deterrent device was switched on between 24 to 185 minutes (on average 42 minutes) before the start of pile driving (Rumes & Degraer 2020).

All three wind farms used hydraulic pile driving to install monopile foundations. At Norther, pile driving using a single big bubble curtain (SBBC) took place in 2018. A SBBC consists of one ring of perforated pipes positioned on the sea floor around the foundation to be piled. Compressors located on the construction vessel or on a separate platform feed air into the pipes. The air passes into the water column by regularly arranged holes. Freely rising bubbles form a large curtain around the entire structure, thus shielding the environment from the noise source (Koschinski & Lüdemann 2013). Noise reductions of 10-15 dB SEL have been found for SBBC (Bellman et al. 2015). In a DBBC, used at Northwester 2 and SeaMade, a second ring of perforated pipes is positioned on the sea floor around the foundation to be piled which, according to Bellman et al. (2015) should result in additional noise reduction of ~3dB (or a further halving of the noise emissions). Northwester 2 was the only project to successfully use noise mitigation measures that limit the transmission of noise pollution to the marine environment to the extent that the in-situ measured sound level (SPL_{z-p}) remained below the national threshold of 185 dB re 1 µPa at 750 m from the source (Norro 2020). Other measures taken with the aim of reducing the impact of pile driving on harbour porpoise included the use of an ADD to deter porpoises from the immediate vicinity of the construction site and the obligation to halt pile driving when a porpoise is detected near the construction site (see Rumes et al. 2020 for an overview).

2.2. Study set up

Harbour porpoises use echolocation for navigation, foraging, and social communication (Berta et al. 2015; Au 2018; Read 2018). This makes it their most important sensory perception and they have been shown to use this echolocation system almost continuously (Akamatsu et al. 2007; Wisniewska et al. 2016). This allows a correlation between detection rates of porpoise clicks by passive acoustic monitoring

devices and porpoise density in a marine area. Passive acoustic monitoring of porpoises was conducted using the Continuous Porpoise Detector (C-PoD, further indicated as PoD). PoDs consist of a hydrophone, a processor, batteries and a digital timing and logging system. They continuously monitor sounds between 20 kHz and 160 kHz, and can detect all odontocetes except sperm whales (*Physeter macrocephalus*). A PoD does not record sound itself, but stores the sound parameters of each click instead, such as time of occurrence, duration, dominant frequency, bandwidth and sound pressure level. Using dedicated software (CPOD.exe,; Tregenza 2014), the clicks are processed and a detector generates click trains which are then classified into trains produced by odontocetes and trains that originate from other sources such as boat SONAR. Distinction can be made between harbour porpoises, a species producing narrow-band, high frequency clicks, and dolphins, producing more broadband clicks with a lower frequency. The maximum detection range for porpoises is approximately 400 meters. PoDs have autonomy of up to 200 days (www.chelonia.co.uk). As porpoise click sounds are emitted in frontal direction with a beam angle of 16.5° maximum (Au *et al*. 1999), PoDs are only able to detect porpoises if they are facing towards the hydrophone.

For this study, we used data from PoDs deployed at 19 locations in the BPNS: 11 of which were specifically deployed for this study and the other 8 are part of the Cetacean passive acoustic network, (Flanders Marine Institute 2021). PoD locations need to be visited every 3-4 months to replace the batteries and memory card. This was not always possible due to logistical issues (incl. COVID-19) leading to gaps in the dataset. In addition, certain mooring locations were changed over time in function of ongoing construction activities.

2.3. Data selection and dataset preparation

PoD data (merged high and moderate quality click train detections) were downloaded via LifeWatch data R package (Flanders Marine Institute, 2021; ; Hernandez *et al*., 2021). The selected PoD data ranged from the 1st of July 2018 to the 30th of June 2021. Detections were aggregated per hour to Detection Positive Hours (i.e., 0/1; DPH). We only used data where the PoD recorded a full hour (60 minutes). Minutes where the number of clicks exceeded the upper detection limit (4096 clicks per minute) were removed from the dataset. As in Brandt *et al* (2016), hourly data was disregarded when data for more than two minutes needed to be removed. In total, 53 % of the original hourly data was kept.

At least 30 minutes before pile driving an ADD is to be activated in order to deter porpoises from the immediate vicinity of the construction site and to protect them from the acute effects of construction noise. However, due to operational uncertainties, the actual interval between ADD activation and the start of pile driving is quite variable (Rumes & Degraer 2020) and for these analyses, the start of pile driving was provided by the developers in daily reports on piling activities. As in Brandt *et al*. (2016), hours when deterrence took place before or after the piling itself were excluded from the dataset in an attempt to exclude the effects of acoustic deterrence devices on porpoise presence (Figure 2). To align the (per hour) DPH information on detections with the (per minute) information on pile driving, the latter was rounded to the nearest hour, and for each hour the following information was generated: time relative to the acoustic disturbance in hours and location of the most recent disturbance. We calculated the minimum time since acoustic disturbance (in hours) per PoD station and per hour and combined it with the information on the distance to the individual piling events.

2.4. Modelling

Hourly Porpoise presence was modelled with the goal of identifying patterns in the porpoise presence before, during and after pile driving in the BPNS (Table 1). A Generalized Additive Model (GAM) was

Table 1. Variables tested for incorporation into the model (variables retained in the final model are indicated with *).

Variable	Type	Description
Pile driving related		
Distance*	Thin plate regression spline	Distance between a piling event and its respective PoD station
Pile Project*	Random effect	A combination of Pile ID and Project
Relative Time*	Thin plate regression spline	Hour relative to the piling event (ranging from -48 to +120)
Sound Mitigation*	Fixed effect	Type of sound mitigation used for the respective piling event
Time-related		
Hour	Cyclic cubic regression spline	Hour of the day
Month*	Fixed or random effect	Month of the year (1-12)
Space-related		
Latitude Receiver*	Random effect	Latitude coordinates of the receiver
Longitude Receiver*	Random effect	Longitude coordinates of the receiver
Interaction		
Distance & Relative Time*	Interaction smoother	Interaction smoother between Distance and Relative Time
Hour by month*	Cyclic by fixed effect	Hour spline is redefined per level of the month

All data analyses were performed in R version 4.2.1 (R Core Team 2022).
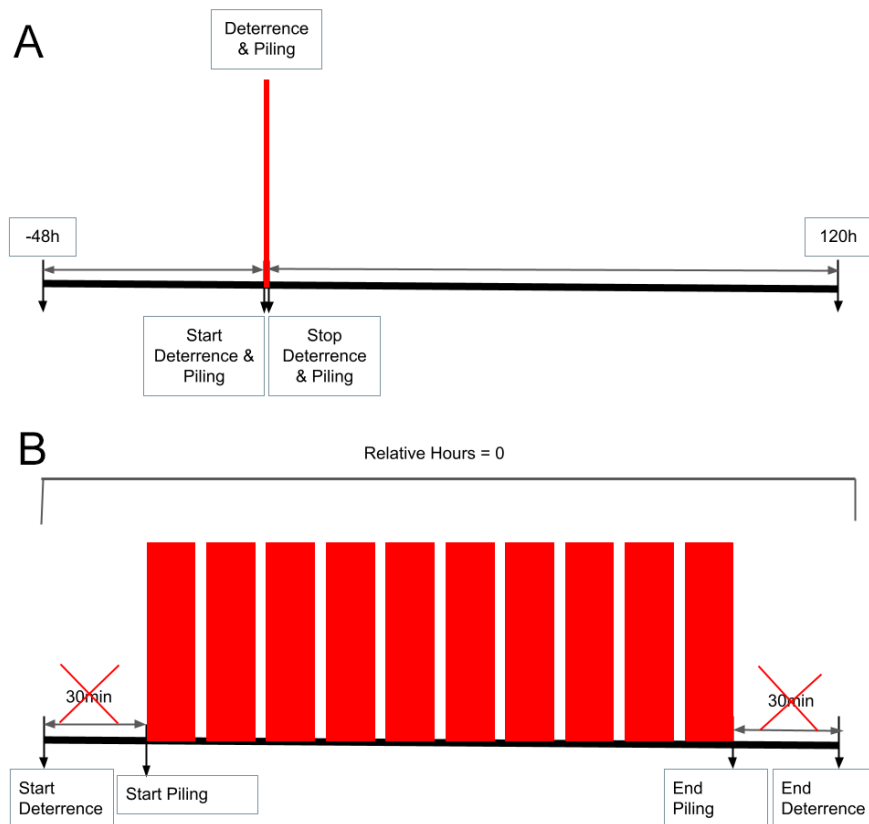
Figure 2. A. Schematic of the timeline displaying how the relative times of the hours before, during and after one pile driving event are implemented. **B**. The pile driving event itself. The moments of deterrence before and after pile driving are crossed out to indicate that these were excluded from the model data.

fitted and evaluated using the R package mgcv, version 1.8.40 (Wood 2017). Both piling- and noise-related variables (to account for noise exposure and applied mitigation) were included. Time-and space- related variables were added to account for temporal autocorrelation and inherent temporal and spatial patterns such as seasonality and habitat suitability. The Akaike information criterion (AIC) was be used as a guide to decide on the best suitable combination of these variables. More complex models were disregarded in favor of simpler models if the more complex model didn't result in a meaningful improvement in AIC (see De Pauw 2022 for a detailed overview of the models tested and their AIC-scores).

In our dataset, pile driving events only occurred from July to early January, and thus the full effect of seasonality on porpoise response to pile driving will not be incorporated into the models. However, porpoise seasonal occurrence in the BPNS is known vary greatly between July and January (Haelters *et al*. 2016; Augustijns 2018), and thus it is justified to incorporate the month as a proxy for seasonal effect in the models.

3. Results

Nearly all tested variables were incorporated into the final GAM model which had the lowest AIC value (Table 1). The smooth and parametric effects of the selected model are shown in Figure 3. Note that effects are not
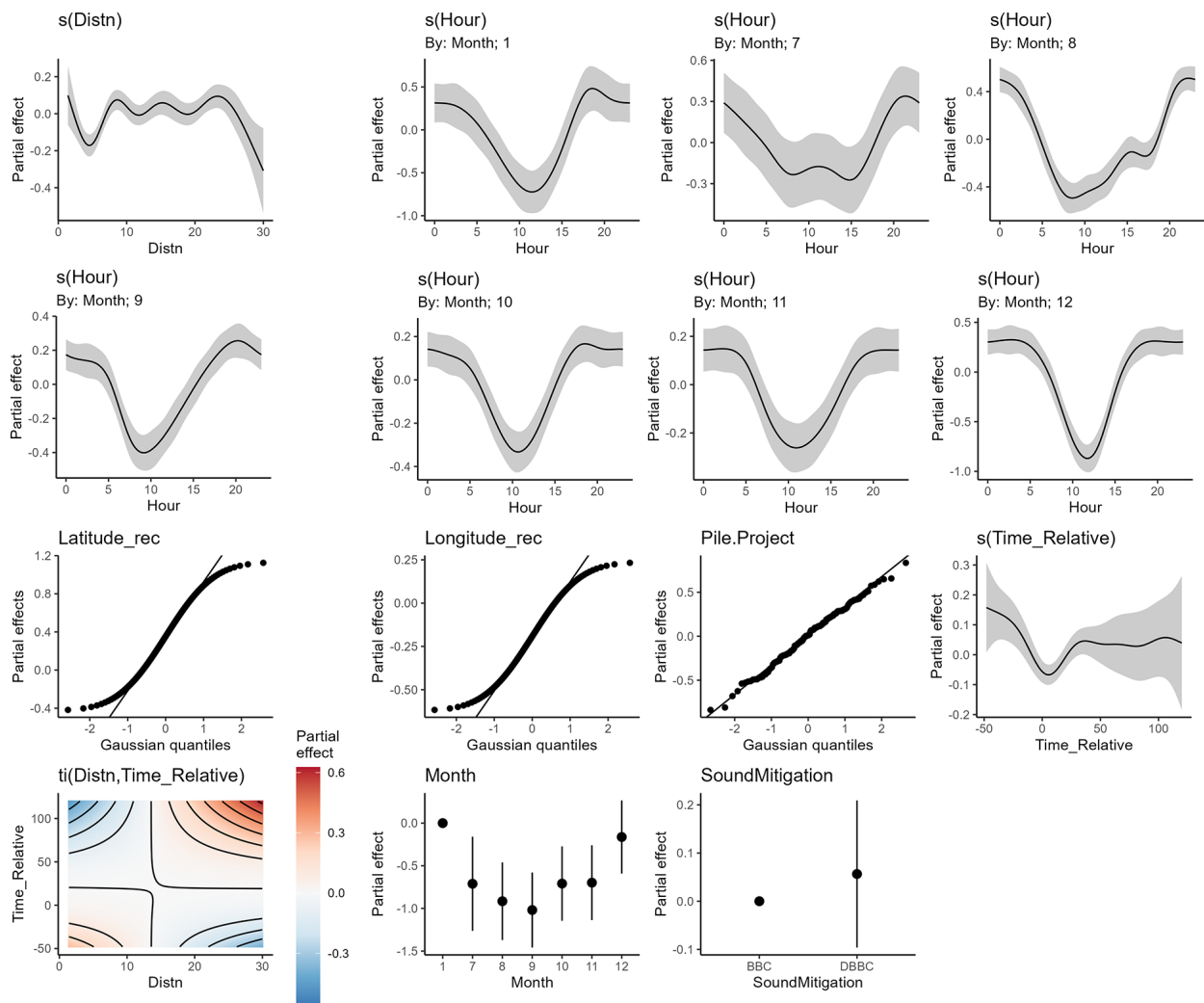
Figure 3. Model smooth and parametric effects.

absolute but relative, as the smoothers are centered to ensure identifiability of the model.

The effect of Hour of the day (per month) reflects the known diel pattern of porpoise activity in Belgian waters and its seasonal changes (see Augustijns *et al*. 2018).

The seasonal effects, which are shown by the month partials, show the known decline in presence over the summer months.

The partial effect of Sound Mitigation shows a lot of variation and almost no difference in effects with DBBC scoring on average only slightly better than SBBC. It should be noted here that the degree in success of DBBC application varied significantly between different projects (see Norro 2020).

Due to the presence of an interaction smoother, the effects of Relative Time and Distance can only be evaluated in combination. The relative combined effect of Distance, Relative Time and their interaction term is shown in Figure 4A. Before the pile driving takes place (-48h to ~-4h), a positive combined effect can be seen on porpoise detections across all distances. The decrease in effect on porpoise detections can be noticed during and to a limited extent shortly before

the pile driving (at Relative Time = ~-4h -0) this decrease tends to be less pronounced at larger distances from the source (from ~20 km). After pile driving, this negative effect continues at small distances (< 10 km), but relatively quickly (~12h) bounces back to a positive effect for longer distances (> 10 km).

Figure 4B shows that most of the observations occur during pile driving. There are also more frequent observations 48 hours after the piling than 48 hours before the piling. Two reasons for the lower number of observations in the hours before the pile driving are 1) the rule that baseline hours (before pile driving) were only counted if at least 48 hours had passed from the previous piling event which was relatively rare, 2) due to the exclusion of hours with ADD use (see 2.3).

The uneven spread of the data is also the likely cause for the wiggly nature of the modelled effect of Relative Time and Distance. The used thin plate regression splines will be anchored by dense regions of data, and the polynomial nature of those splines allows the response surface to wiggle between those anchor points. The used penalty terms in the fitting process are not sufficient to dampen
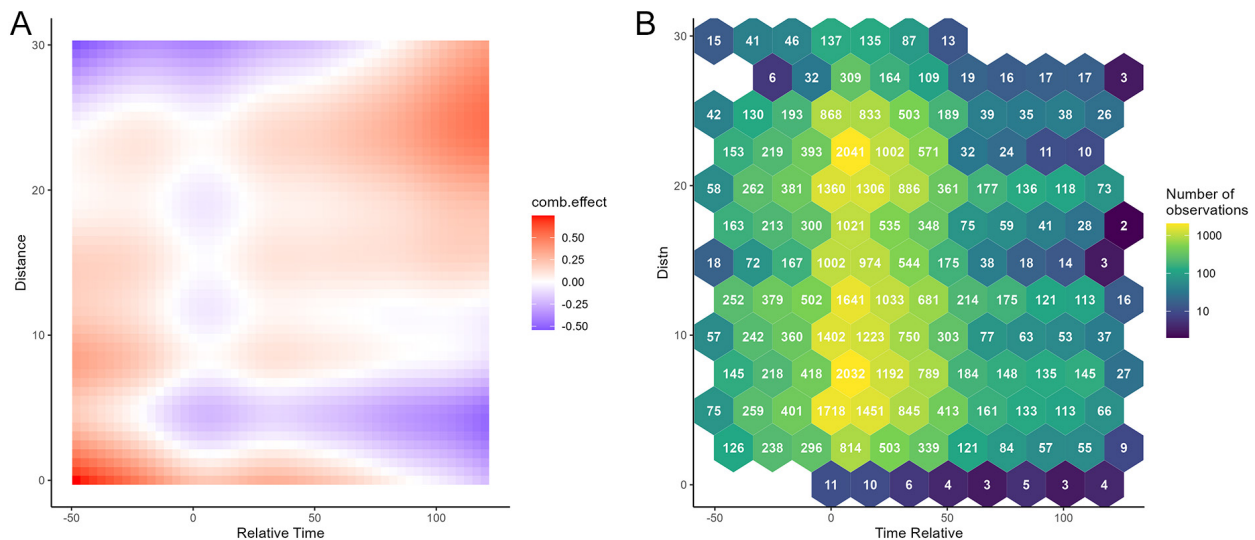
Figure 4. Comparison of combined effect of Distance from the sound source and Time Relative to pile driving on porpoise detection positive hours (Dph) (A) with their respective amounts of datapoints (B). **A**. Combined effect Time Relative and Distance on Dph. **B**. Number of recorded hours spread through Time Relative and Distance (Distn).

Table 2. Variance components of the random effects of the model with all random effects included.

Component	Variance	Standard deviation	Lower Confidence Interval	Upper Confidence Interval
Latitude Receiver	0.0846	0.291	0.0725	1.170
Longitude Receiver	0.0233	0.153	0.0378	0.616
Pile Project	0.1340	0.365	0.3140	0.425

this effect when the data is spread unevenly. Hence the model output should be interpreted with care in the most general terms.

Table 2 shows the variance of the three random effects in the model. Pile Project has the highest variance, which implies that between the piling events there is more variance on the effect on porpoise presence than that of the spatial variance explained by Latitude and Longitude of the receiver.

4. Discussion

4.1. Key model findings and conse-quences for effective mitigation of impul-sive underwater sound

After correction for other sources of variation, the GAM model predicts a moderate reduction in odds for detection of porpoises during pile driving compared to 48 hours before pile driving at distances up to 20 km from the source (Figure 4A). This relative decline becomes less marked with increasing distance from the piling event. This is in line with porpoise response to lower levels of pile driving sound (Rose *et al*. 2019). From 25 km onwards the model hints at an opposite effect, with relatively more detections after pile driving than before pile driving. Dähne *et al*. (2013) reported a similar increase in detections beyond 25 km and suggested that this could be due to the displacement of affected porpoises towards these areas. However, due to the limited availability of data (see Figure 4B), effects beyond 20 km are predicted with large uncertainties and hence should be interpreted with care.

At distances up to around 5 km from the piling event, the model also showed a

reduction in porpoise detections starting several hours before the piling event. As previously suggested, this could be due to elevated levels of shipping noise and other preparatory works (Brandt *et al*. 2016; Rumes *et al*. 2017; Rumes & Zupan 2021).

At these lower distances, the effect remains negative for the entirety of the modelled period (up to 120 hours after the piling event). As data becomes scarce when the piling event is further away in time, trends after 48 hours are modelled with large uncertainties and hence should be interpreted with caution. A possible explanation for the extended negative effect could be that due to consecutive piling events porpoises learn to avoid the wind farm construction zone, as argued in Rumes *et al*. (2017). However, it could also be linked to the seasonality of porpoise distribution in Belgian waters (Haelters *et al*. 2016) and the uneven distribution of pile driving events with longer intervals between subsequent pile driving events (which are more often observed at the start of the construction period, here: in periods of lower porpoise densities). In general, effects of seasonality and time of the day on the likelihood of detecting porpoises aligned well with known information on porpoise behavior (Augustijns *et al*. 2018) and seasonality (Haelters *et al*. 2011) in Belgian waters.

The type of sound mitigation used does not seem to have an effect according to our model (Figure 3). It is unclear whether this is due to the difference in sound mitigation between SBBC and DBBC being only in the order of a few dB (Bellman 2014) or because of the uneven way in which the latter sound

mitigation technique was applied (Norro 2020). In this study all pile driving events were accompanied by some form of sound mitigation, which made it impossible to study the effect of sound mitigation on harbour porpoises' response to pile driving per se. Previous studies have shown the effects of unmitigated pile driving on porpoise to reach much farther (26 km [s.e.: 22-30 km]) than those of mitigated pile driving (11 km [s.e.: 10-12 km]) (Rose *et al.* 2019; Rumes *et al.* 2021).

4.2. Limitations to the current study and future work

Even though AIC is a well-known model selection criterion it also comes with some disadvantages. It only measures the relative quality of a model, so even though AIC tells which model fits better, the best model could still fit the data poorly (Zajic 2019). Furthermore, the lack of a framework for formal hypothesis testing doesn't allow to decide whether the improvement in AIC is substantial enough to be relevant. Alas, the complexity of GAM incorporating random terms doesn't allow to use, e.g., likelihood-ratio tests typically used with GLMs for this purpose. Hence, choosing between models solely on AIC score remains a fairly subjective matter. For our model, effects beyond 20 km should be interpreted with care, as only limited data was available at these distances (see Figure 4B) which can result in artifacts of the smoother.

As noted previously (Rumes *et al.* 2017), even during pile driving, harbour porpoises are not completely absent from sites in the vicinity of pile driving. Lacking information on the movement on individual porpoisesand the amount of underwater sound these animals are exposed to, it is impossible to draw conclusions about causal relationships based on the presented model. Detections in

the vicinity of the construction zone can be due to both the continued presence of animals which tolerate higher levels of underwater sound and animals which are moving away from the sound source. A future comparison of the proportion of feeding buzzes to total porpoise click trains (sensu Nuuttila 2013; Zein *et al.* 2019) during and after acoustic disturbance can provide more information on their response to acoustic disturbance.

In the last 20 years ~25 GW of offshore wind has been constructed in the North Sea (WindEurope 2021). Over the next 20 years, construction rate is expected to increase nearly tenfold resulting in an increased exposure of marine life to harmful levels of impulsive underwater sound from wind farm construction. Avoiding potential negative cumulative impacts on local cetacean populations will require coordinating construction efforts and/or formulating coherent mitigation measures at North Sea scale. Current mitigation efforts vary strongly between individual countries, but, in general, have focused on reducing transmission and the lowering of sound levels of individual pile driving events to comply with national impulsive noise regulations for impulsive underwater sound. Our results show received sound level (here distance of the PoD to the construction site) to indeed be the main driver for the magnitude of the porpoise response to pile driving. However, the relatively long duration of disturbance (and consequent reduction in porpoise detections) in the vicinity of the construction site (< 10 km) highlights the potential environmental benefits of measures aimed at a) reducing the overall duration of the construction period (e.g., by installing fewer foundations or further reducing the time between piling events) b) avoiding pile driving during periods with elevated porpoise presence (e.g., by implementing a seasonal pile driving ban).

Acknowledgements

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