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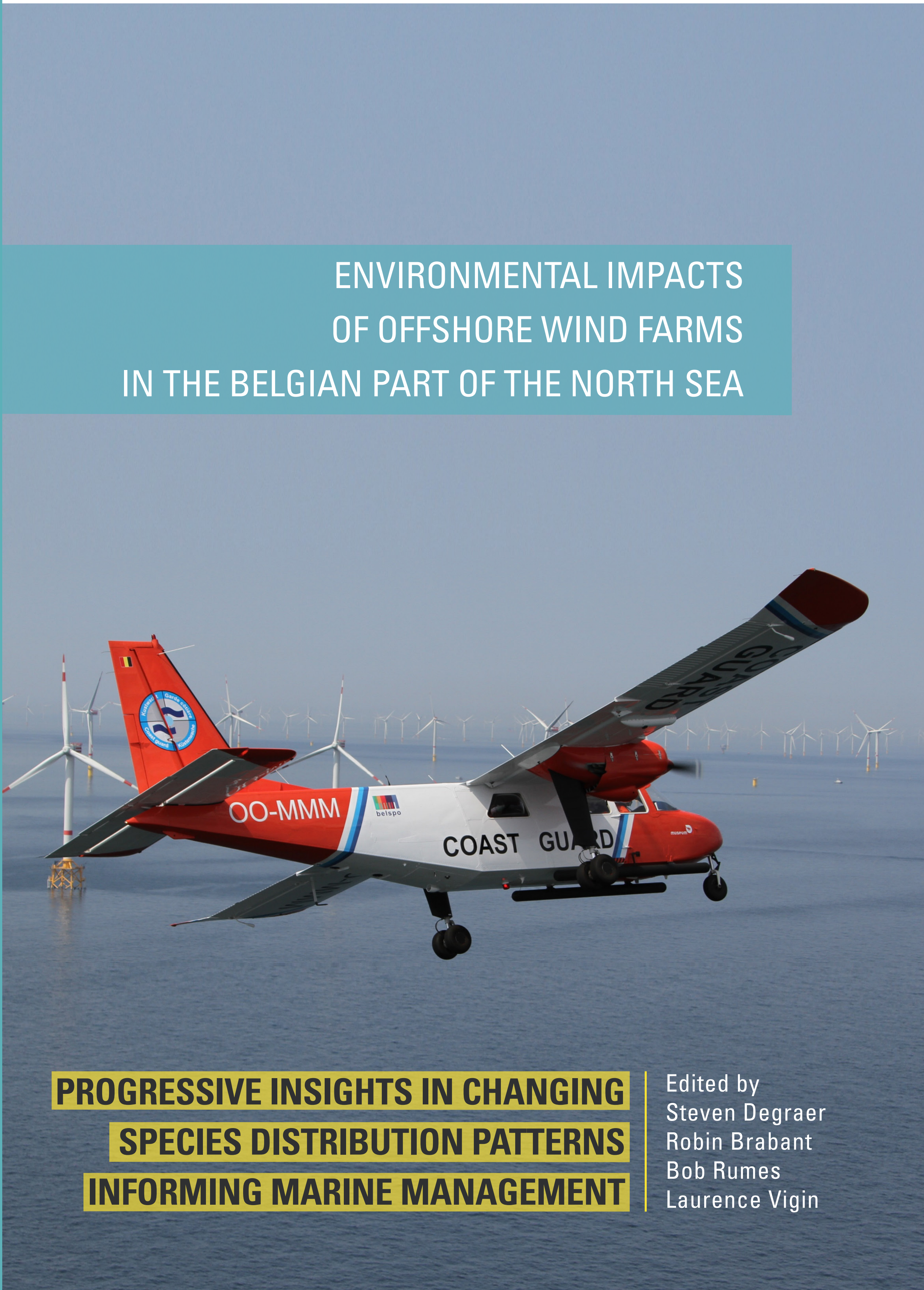

ENVIRONMENTAL IMPACTS OF OFFSHORE WIND FARMS IN THE BELGIAN PART OF THE NORTH SEA

PROGRESSIVE INSIGHTS IN CHANGING SPECIES DISTRIBUTION PATTERNS INFORMING MARINE MANAGEMENT

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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 2022.

DEGRAER Steven, BRABANT Robin, RUMES Bob and VIGIN Laurence

EXECUTIVE SUMMARY

PROGRESSIVE INSIGHTS IN CHANGING SPECIES DISTRIBUTION PATTERNS INFORMING MARINE MANAGEMENT

Steven DEGRAER [*], Robin BRABANT, Bob RUMES, Laurence VIGIN, Ulrike BRAECKMAN, Jolien BUYSE, Wouter COURTENS, Annelies DE BACKER, Jan HAELTERS, Kris HOSTENS, Christelle JAMMAR, Nene LEFAIBLE, Tom MOENS, Silvia PAOLETTI, Abril REYNES-CARDONA, Eric STIENEN, Jan VANAVERBEKE, Marc VAN DE WALLE, Nicolas VANERMEN, Tine Miet VAN MAELE & Hilbran VERSTRAETE

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At present, eight offshore wind farms are operational in the Belgian part of the North Sea, totalling an installed capacity of 2.26 Gigawatt (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, anticipating an installed capacity ranging between 3.15 and 3.5 GW, has been designated in the marine spatial plan 2020-2026. With 523 km² reserved for offshore wind farms in Belgium, 344 km² in the adjacent Dutch Borssele zone and 50 km² in the French Dunkerque zone, cumulative ecological impacts remain a major concern. These anticipated impacts, both positive and negative, are investigated through the WinMon.BE environmental monitoring programme focusing on various aspects of the marine ecosystem components.

Most environmental monitoring programmes for offshore wind farms are halted five years after installation. However, research has shown that this period is way too short and consequently these programmes do not provide the insight needed to manage offshore wind farms in an evidence-based manner. With the Belgian offshore wind farm environmental impact monitoring and research programme, WinMon.BE, we show that fifteen years after the first installation of offshore wind turbines in the Belgian part of the North Sea, progressive wind farm-induced changes in the marine ecosystem are still observed, underlining the importance of long-term research for a sound offshore wind farm management. The WinMon.BE programme has adopted a philosophy of long-term investigation, spanning the full life cycle of offshore wind farms, i.e. from construction to decommissioning. The progressive insights have not only informed the management and development of the first Belgian offshore wind farm zone, which was gradually constructed between 2008 and 2020. Our scientific insights also guided the design of the second Belgian offshore wind farm zone, i.e. the Princess Elisabeth Zone, in an environment-sensitive manner, through the currently ongoing Environmental Impact Assessment procedure.

Long-term impacts of offshore wind farms on the macrobenthic communities inhabiting the surrounding natural soft sediments were investigated over a time span of 13 years (2008–2020; Chapter 2). Our analyses support what is already generally accepted regarding turbine-related impacts. Higher macrobenthos abundance, species richness and diversity are observed in sediments with higher fine particles fractions and total organic matter content. They also confirmed the common pattern of higher abundances in the gullies between sandbanks. Climate-related predictors (sea surface temperature and Atlantic Multi-decadal Oscillation) were significantly correlated with macrobenthic diversity, abundance and species richness. For future studies, it remains important to incorporate local environmental variables that are affected by the turbine presence (like sediment characteristics and organic matter), aside from water depth and climate-related variables. Our study further revealed that no stable state has yet been reached after 13 years of offshore wind farm operations. These findings clearly highlight the importance of long-term studies, as more time is needed (1) for the impacts to get gradually established and (2) to collect sufficient data to be able to detect and observe trends in the response of macrobenthic communities to the presence of offshore wind farms.

Changes in species distribution patterns were identified for demersal fish, as exemplified for plaice *Pleuronectes platessa*, a species extensively studied in terms of its spatial distribution, diet and movement patterns in relation to offshore wind farms (Chapter 3). A combination of visual diving transects (at the turbine scale), beam trawl samples (at the wind farm scale) and acoustic telemetry demonstrated the significance of the scour protection layer and the sandy patches in between the rocks as a feeding habitat for plaice. Plaice benefits from the increased food availability at the hard substrates, as indicated by a trophic analysis combining gut content analysis with a biomarker approach (fatty acid analysis). Despite the increased prey availability, morphometric (i.e., Fulton's K index) and organosomatic condition indices (i.e. fullness-, gonadosomatic-, hepatosomatic and digestive-somatic index) did not (yet) reveal evidence of a better condition in plaice, potentially due to the sampling size being too small to detect differences. Our findings suggest that offshore wind farms serve as a refuge for plaice, potentially mitigating direct fishing mortality and likely enhancing plaice production. It remains to be investigated whether this translates to spillover effects into the adjacent areas where fishing is permitted and how such effects may affect fisheries, given the anticipated large-scale expansion of offshore renewable energy zones in the broader North Sea.

Altered species distribution patterns in relation to the presence of offshore wind farms are not independent of other human activities, such as shipping, fisheries and mariculture. This is particularly the case for highly mobile species like marine mammals. We used aerial survey data collected between 2009 and 2022, and analysed the spatio-temporal distribution patterns of the harbour porpoise *Phocoena phocoena* in function of selected environmental drivers and anthropogenic stressors (Chapter 4). The distribution of harbour porpoise followed a consistent seasonal pattern, with the highest densities in spring, but with high interannual variability, with abundance peaks in 2011, 2014 and 2018. Harbour porpoise distribution correlated with latitude and longitude, with the species preferring the western part of the Belgian part of the North Sea, revealing a strong overlap with the Vlaamse Banken Special Area of Conservation (SAC). The distribution was also significantly negatively correlated with marine traffic intensity and distance to the closest offshore wind farm. However, it is essential to exercise caution to avoid overinterpreting these correlations. Further monitoring and research is recommended to better understand the interaction between natural factors, such as prey availability, and anthropogenic stressors, driving the spatial distribution of harbour porpoises.

With an ever increasing number of offshore wind farms in Belgian waters, monitoring programmes need to be adaptive to ensure gaining the best knowledge on changing species distribution patterns. The adapted monitoring strategy for seabirds not only aims to detect displacement responses, it is also designed to detect avoidance distances and the effect of turbine density on seabird displacement (Chapter 5). The results presented at this stage (count data from February 2021 to April 2023) need to be considered as indicative since more data and advanced spatial modelling are needed to detect potential seabird avoidance or attraction effects with sufficient confidence. Nevertheless, based on this limited dataset and mean values, it is interesting to see that for several species, the observed responses are in line with what has been found before and/or elsewhere. As such, our results indicate an attraction effect for great black-backed gull *Larus marinus* and great cormorant *Phalacrocorax carbo*, and an avoidance effect for northern gannet *Morus bassanus*. On the other hand, we no longer noticed a (strong) avoidance of common guillemots *Uria aalge* and even observed an increased number of razorbills *Alca torda* in the wind farms, possibly indicating habituation or specific habitat preferences. The revised monitoring design aims at informing future planning decisions regarding wind farm configuration to mitigate the impact on seabirds.

Aside a bird-sensitive wind farm design, mitigating the impacts on birds may also entail measures to reduce bird collision numbers. The southern North Sea is one of the main migration flyways in Europe. The highest flight intensities at sea are recorded at night during spring and autumn migration, mainly of migrating passerines, which normally migrate at high altitudes, up to several kilometres. However, a portion of these songbirds flies at rotor height of the wind turbines and are thus at risk of collision. Temporarily stopping the turbine operation during high collision risk events for songbirds, e.g. when adverse weather conditions bring large numbers of passerines into the range of the turbine rotors, may substantially prevent collision mortality. However, this management measure has not yet been applied regularly (Chapter 6). The Netherlands are pioneering curtailment measures in offshore wind farms and, Germany and France are starting to perform tests, while other countries are open for discussions on the topic. Temporarily turbine shutdowns may be highly effective for reducing collision mortalities in certain scenarios, but site-specific monitoring programmes remain necessary to assess the effectiveness and the finetuning of the measure. Furthermore, a regional approach may be most appropriate to maximize the efficiency and ecological benefits of such measure.

In conclusion, the results presented in the present WinMon.BE monitoring and research report demonstrate the importance to progress our insights in changing species distribution patterns in relation to offshore wind farms. Fifteen years past the installation of the first turbines in Belgian waters, the marine ecosystem has not yet reached a new equilibrium, as demonstrated for the soft sediment macrobenthos communities inhabiting the sandy sediments surrounding the turbines and scour protection layers. Continued, new and detailed research is indispensable to further our understanding on how marine ecosystems respond to wind farms. This research should not only focus on the attraction of hard substrate species, but also on species that are less evidently impacted by offshore wind farms, such as plaice and other demersal (flat)fish. We need to critically reflect on the efficiency and effectiveness of our and other regional monitoring and research programmes to ensure collecting the best data, as shown with the re-designed monitoring programme for seabirds. As demonstrated for marine mammals, we need to address the most pertinent questions, e.g. the contextualization of offshore wind farm effects. Progressive insights are necessary to feed evidence-based, efficient and effective mitigation measures – such as regional curtailment programmes - and to develop and design eco-friendly offshore wind farms.

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 ~1/3 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 with an expansion of the Belgian offshore transmission network scheduled to start in 2024. To reduce spatial competition in the marine economy options for sustainable offshore multi-use are being investigated. 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 50 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 and recent developments in offshore renewable energy in the BPNS.

1. Offshore wind energy 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. A second area of 285 km² is reserved in the revised marine spatial plan that came in force on March 20th, 2020. In the neighboring Dutch Borssele zone two wind farms are operational and totaling an installed capacity of 1.5 GW on an area of 344 km². In front of Dunkerque, in the French part of the North Sea, 50 km² is reserved for offshore wind development (Fig. 1). On 24 April 2023, a North Sea coalition of nine countries committed to combined targets for offshore wind of at least 300 GW by 2050 (https://northseasummit23.be/en/ostend-declaration).

The European Directive 2018/2001 on the promotion of the use of energy produced from renewable sources, imposes a binding target of 32% for the overall share of energy from renewable sources in the EU's gross final consumption of energy in 2030. In 2021, the share of energy from renewable energy in Belgium was 13%, up from 1.9% and 7.7% in 2004 and 2013 respectively (Eurostat, 2023)

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

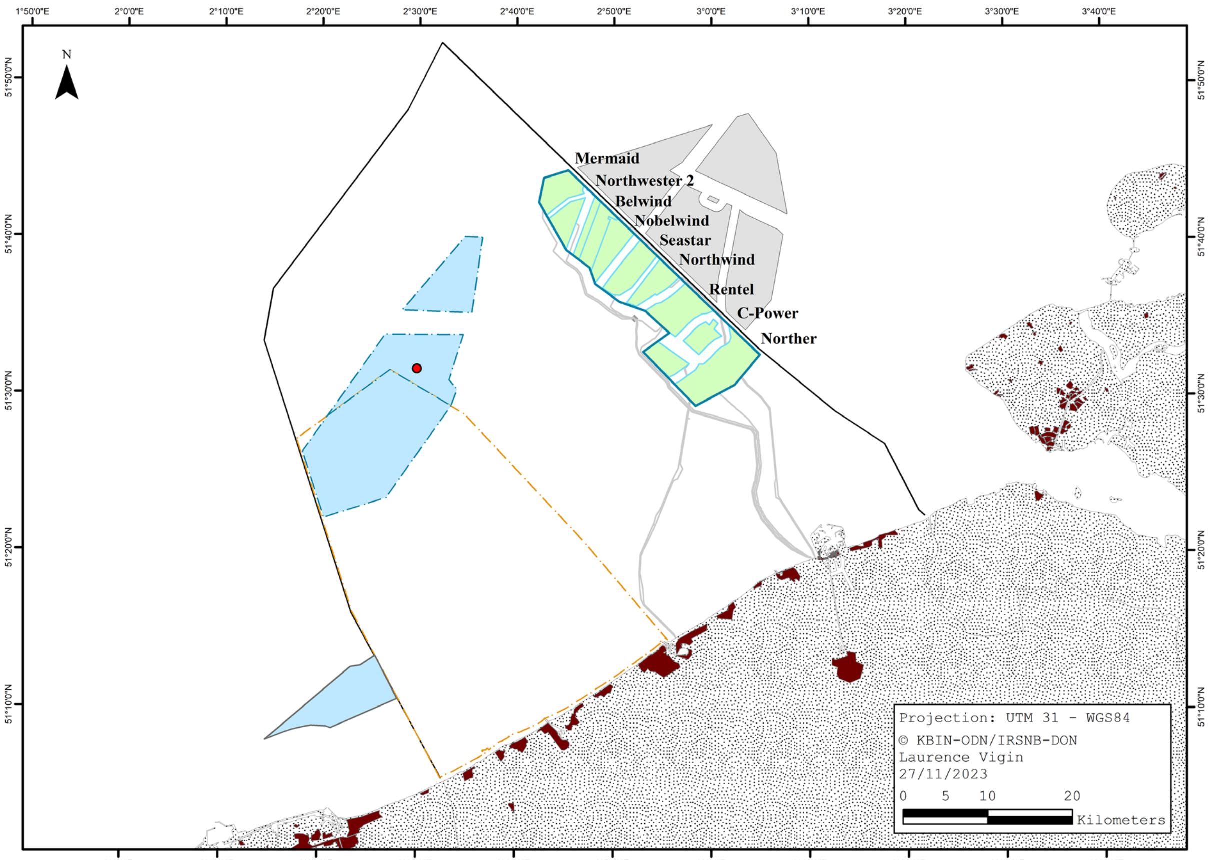

Figure 1. Current and planned zones for renewable energy in and around the Belgian Part of the North Sea. Operational wind farms in Belgian waters are shown in green. Operational wind farms in the Dutch Borssele area are in grey. The blue areas in the NW of the Belgian part of the North Sea are the Princess 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'. The red dot is the location of the modular offshore grid 2.

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. 2).

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

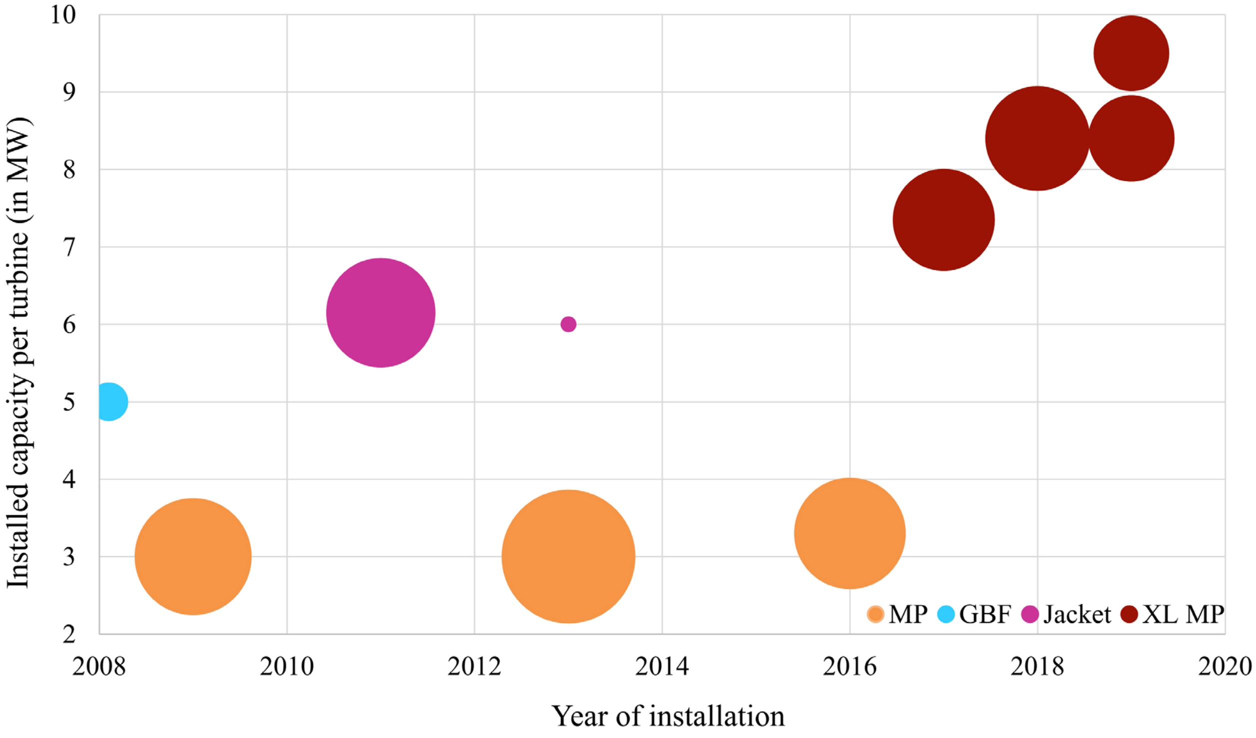

Figure 2. Overview of the timing, individual capacity and foundation type of offshore wind turbines installed in the Belgian Part of the North Sea since 2008. The size of the bubbles is proportional to the number of turbines installed per project of phase (Table 1). Abbreviations: GBF = Gravity based foundation; Jacket = Jacket foundation; MP = monopile foundation; XL MP = monopile foundations exceeding approximately 7 m in diameter.

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, 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, called the Princess Elisabeth zone (PEZ), will 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 marine renewables into the electricity grid. In 2023, Elia was granted an environmental permit for the Modular Offshore Grid 2 to reinforce the offshore electricity grid (see below).

The PEZ is partly located inside the designated Natura 2000 marine protected area ‘Vlaamse banken’ (Fig. 1). To determine whether and how the new offshore wind farms can be designed and operated with respect to the existing and aspired ecological values as defined by the conservation objectives for the area, a targeted research programme was set up. The EDEN2000 project on “Exploring options for a nature-proof DEvelopment of offshore wind farms inside a Natura 2000 area” aimed at filling knowledge gaps of prime relevance to and advice for an environment friendly development of the spatial overlap of the PEZ and the Special Area of Conservation “Vlaamse Banken”. These knowledge gaps linking societal concerns with research questions, were identified based on iterative roundtable consultations of environmental NGOs active in the BPNS, the Belgian Offshore Platform representing the Belgian offshore renewables industry, the Federal Public Service Environment responsible for the implementation of environmental policies in the BPNS, the Cabinet of the Minister of the North Sea and the Royal Belgian Institute of Natural Sciences. Filling these gaps necessitated either a summary of existing knowledge, a dedicated analysis of existing data or newly designed research. The EDEN2000 studies span the mitigation of negative impacts and the promotion of positive impacts, and touch upon (1) the ecological context of the area, (2) the artificial reef effect in its widest sense, (3) the effects of fisheries exclusion inside the area and (4) the effects of the introduction of energy. This programme commenced in 2019 and results were published in 2023 (Degraer *et al.* 2023). All studies, conclusions and recommendations are publicly available at https://www.health.belgium.be/nl/eden2000-studies.

2. Elia Modular Offshore Grid 2

On January 9th, 2023, ELIA applied for an environmental license for the construction and operation of Modular Offshore Grid 2 (MOG2). This MOG2 project aims to expand the Belgian offshore transmission network by developing and constructing additional offshore substations and export cables. MOG2 thus provides the connection between the new wind farms in the PEZ and the onshore Belgian transmission network. Additionally, facilities will be created for new HVDC interconnections, such as the Nautilus project with the United Kingdom and the Triton Link project with Denmark. The MOG2 project comprises the construction of an artificial island or multiple platforms for AC (Alternating Current) and HVDC (High Voltage Direct Current) substations (Fig. 3)

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	Depth (m LAT)	Monopile diameter (m)	Average distance to the coast (km)
Norther		44	8.4	monopile	164	107	370	2019	20–35	8–9 m	25
C-Power	phase 1	6	5	gravity based	126	94	325	2009	14–18	NA	27
	phases 2 & 3	48	6.2	jacket	126	94		2013		NA	
Rentel		42	7.4	monopile	154	106	309	2019	22–36	8 m	34
Northwind		72	3	monopile	90	72	216	2014	16–29	6–7 m	38
SeaMade	Seastar	30	8.4	monopile	167	109	487	2020	22–38	8.3	38
	Mermaid	28							24–40		53
Belwind	phase 1	55	3.1	monopile	90	72	171	2011	15–37	4.3	49
	Alstom Demo project	1	6	jacket	150	100		2013		NA	
Nobelwind		50	3.3	monopile	90	72	165	2017	26–38	6–7	46
Northwester 2		23	9.5	monopile	164	106	219	2020	25–40	8–9	51

* lowest astronomical tide

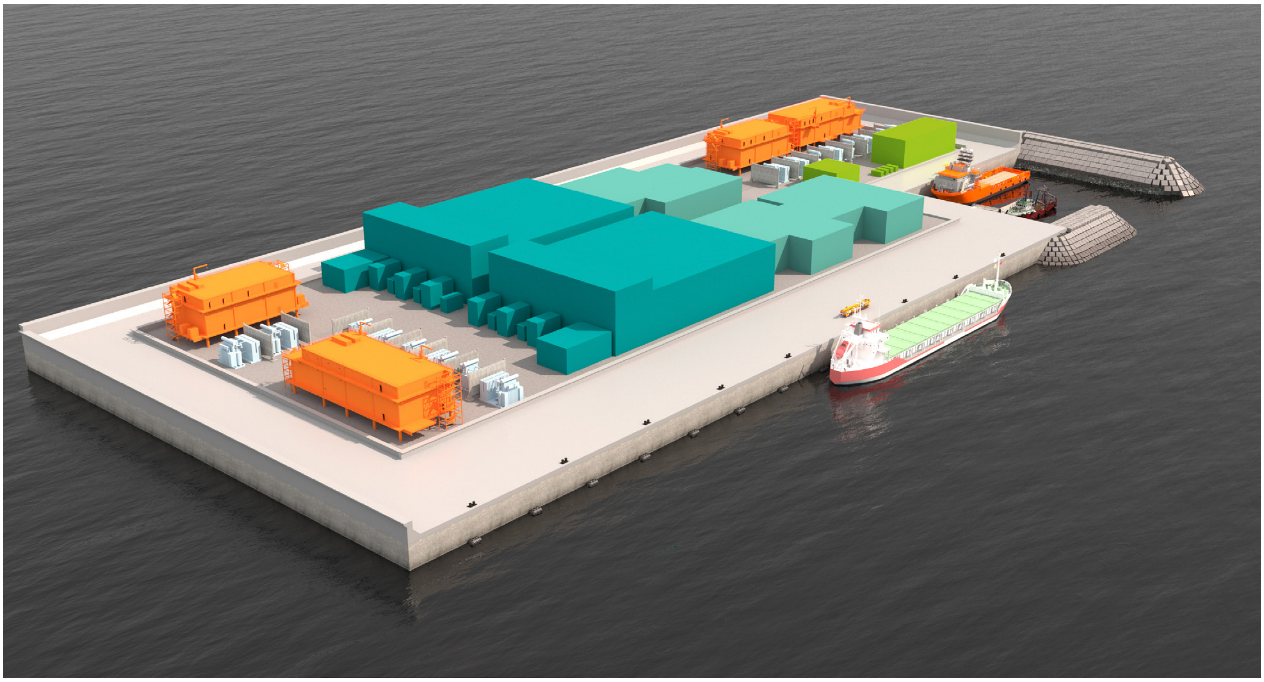

Figure 3. Visualisation of the energy-island MOG2 (Source: Elia).

and the connection of these substations to the mainland through six 220 kV three-phase AC export cables and one HVDC cable system. In the EIA, the environmental impacts of various alternatives regarding location, cable route and execution were investigated and the alternative with multiple platforms rather than a caisson-type artificial island was found to have the smallest impact for most aspects of the marine environment (Haelters *et al.*, 2023). On September 26th, 2023, ELIA was granted their environmental license for the construction and operation of MOG 2 and construction is expected to start in summer 2024. Because of the additional functionalities of an island compared to platforms, the island alternative will be constructed. The location is shown on Fig. 1. A dedicated environmental monitoring programme was drafted to validate the predicted impacts of this project. This will also be the first offshore construction project in Belgium where Nature Inclusive Design measures are broadly applied aimed at offsetting some of its negative impacts on the marine environment.

3. Decommissioning of the first offshore wind farms

The first offshore wind farms in Belgian waters were constructed in 2008 and 2009. As the decommissioning of this first generation of wind farms is approaching, many questions are raised about the phased decommissioning process in the period 2034–2047. On the one hand, new technologies are providing new options for decommissioning. For example, options for repurposing and recycling blades and ways to remove foundations in their entirety from the ground are being explored. On the other hand, new insights surrounding the interaction between wind farms and biodiversity are constantly emerging. Monitoring the ecological effects of wind farms has shown that additional biodiversity has been created in and around offshore wind turbines, the so-called artificial reef effect. These new hard substrates underpin a rich underwater fauna of invertebrates, which in turn attracts various fish species, bird species and possibly marine mammals.

To decommission offshore wind farm infrastructure, there are theoretically several options. The foundations can either be

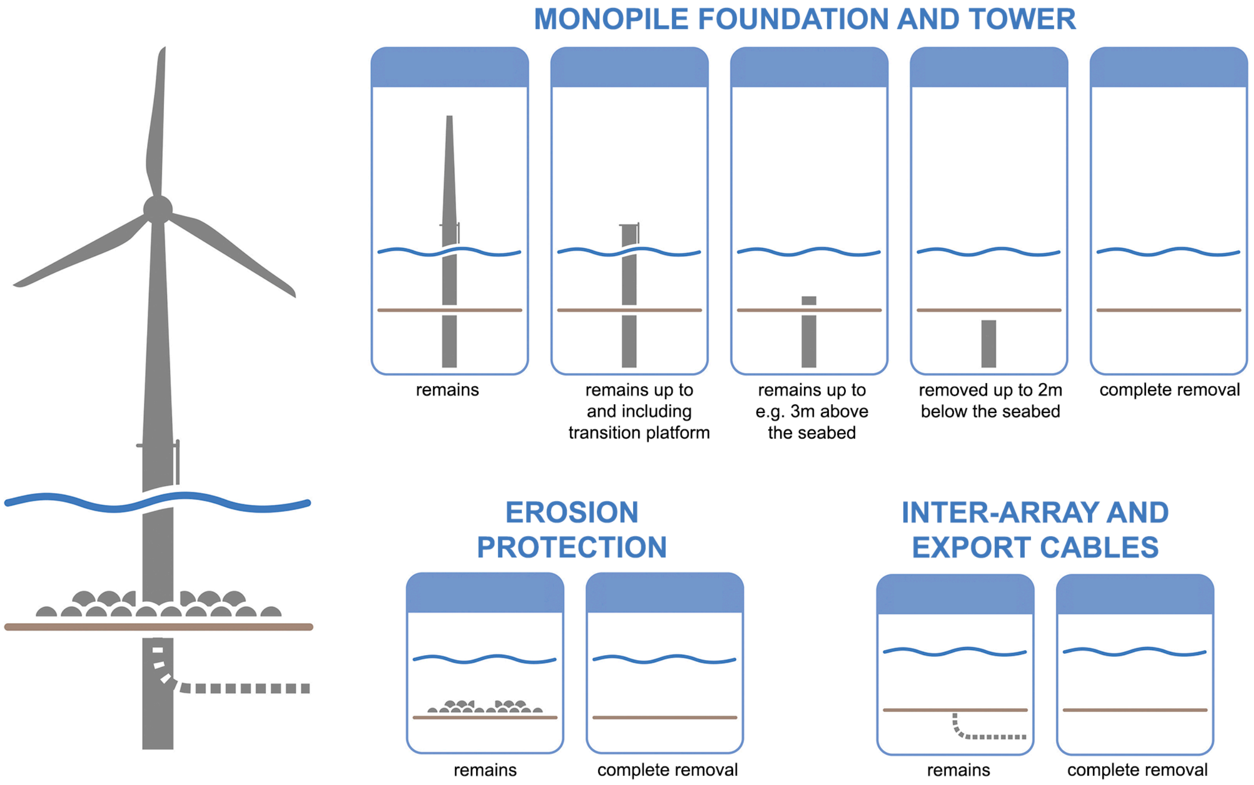

Figure 4. Different options of offshore wind farm infrastructure decommissioning (after Van Maele *et al.* 2023a).

completely or partially removed or remain entirely on site. Erosion protection layers and cabling can also be removed or remain on site (Fig. 4). Of the more than 40 organisations that were consulted during a stakeholder participation process, the majority favoured complete removal of all man-made structures.

The naturally occurring and desirable fauna of dynamic sandy substrates is adapted to high dynamics, allowing it to withstand and quickly recover from temporary disturbance caused by decommissioning activities.The new biodiversity created as a result of the artificial reef effect is not considered of such interest in a naturally dynamic sandy-bank ecosystem to be left untouched because it is a habitat that does not naturally occur at that site. Moreover, decommissioning in the context of repowering will re-provide hard substrate in the form of a new wind farm, so that those additional habitat, shelter and resting opportunities will recover in the short term and in phases.

Leaving some of the infrastructure in place could be useful for attaching structures for aquaculture, passive fishing or as a research base (sensors, testing new technologies, etc.), for example, but these functionalities could equally be envisaged for yet-to-be-built wind turbines. In addition, retaining (part of) the foundation and leaving erosion barriers and cables in place do not outweigh the disadvantages such as e.g. insecurity and the missed opportunity to reuse materials.

The wind farm operators, on the other hand, who have to carry out and pay for the decommissioning, are rightly concerned whether it will be both feasible and affordable in engineering terms to completely remove the foundations. Also, removing the erosion protection, even if it is to be reused for the same purpose when repowering, is a costly and time-consuming activity. Thus, further research and consultations still appear necessary to identify the feasibility and the advantages and disadvantages of the alternative decommissioning scenarios (complete or partial removal of wind farm infrastructure, including erosion protection layers). By starting this in time, there is time for the public and private partners involved to prepare.

The findings from the stakeholder consultation process also offer insights into how future wind farms can be optimally designed, taking into account the decommissioning phase. In particular, promoting circular use of materials offers sustainability opportunities.

The Princess Elisabeth zone contains zones with natural hard substrate, a low-dynamic habitat with high ecological value. Decommissioning activities will therefore have a greater impact here than on the dynamic sandy soils where the current wind farms are located. On the other hand, in the gravel beds of the Princess Elisabeth zone, many win-wins can be achieved by implanting artificial hard substrate such as wind turbines and erosion protection layers. Whereas in the first zone it is advised, for reasons of natural value, to remove everything when decommissioning, in the Princess Elisabeth zone it remains to be seen how to avoid disturbance of the gravel beds during decommissioning as much as possible, and how to preserve the natural value of the artificial hard substrate in the vicinity of the gravel beds as much as possible.

4. Aquaculture

For the stakeholders consulted in the participation project, the primary goal of aquaculture in the Belgian part of the North Sea is sustainable food production. Despite a current focus on oysters and mussels, a broad spectrum of organisms is suitable for this purpose including algae, whelks, scallops, fish, jellyfish, sea cucumbers, sea urchins, sea grasses and even bacteria. Whether offshore (integrated) multitrophic aquaculture is technically possible in the Belgian North Sea needs further investigation.

Given the limited extent of the BPNS, optimal use of available space remains one of the main concerns highlighting the opportunities for multiple use of space. The stakeholder process for aquaculture in the Belgian part of the North Sea, highlights the potential for aquaculture in the zones for wind energy, though linked to predefined basic and boundary conditions. However, where possible, facilitating aquaculture in these offshore wind farms, should be taken into account already at the design stage of the wind farms which is unlikely to happen without policy support and a suitable regulatory framework (Van Maele *et al.* 2023b)

References

Belgische Overheid, 2019. Nationale Klimaatcommissie, Belgisch geïntegreerd Nationaal Energie en Klimaatplan (NEKP, 2021-2030), Afdeling A: Nationaal plan.

Degraer, S., Brabant, R. & Vanaverbeke, J. (eds). 2023. *EDEN 2000 – Exploring Options for a Nature-proof Development of Offshore Wind Farms inside a Natura 2000 Area*. Memoirs on the Marine Environment. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 440 pp.

EU, 2021. Communication from the commission to the European parliament, the council, the European economic and social committee and the committee of the regions on a new approach for a sustainable blue economy in the EU Transforming the EU's Blue Economy for a Sustainable Future, COM/2021/240 final.

Eurostat, 2023. Renewable energy statistics. https://ec.europa.eu/eurostat/statistics-explained/index.php?title=Renewable_energy_statistics#Share_of_renewable_energy_more_than_doubled_between_2004_and_2019 [accessed October 2023].

FPS Economy, 2022. ENERGY - Key Data - Edition February 2022. 39 pp.

Haelters, J., Brabant, R., De Cauwer, K., Degraer, S., Desmit, X., Devolder, M., Kerkhove, T.R.H., Legrand, S., Norro, A., Merveille, J-B., Parmentier, K., Rumes, B., Schallier, R., Van den Eynde, D., Van Lancker, V., Van Roy, W. & Lauwaert, B., 2023. *Milieueffectenbeoordeling van de aanvraag van Elia Asset NV voor het verkrijgen van een milieuvergunning voor de installatie en exploitatie van het Modular Offshore Grid 2*. BMM, Koninklijk Belgisch Instituut voor Natuurwetenschappen, Brussel. 129 pp.

Van Maele, T.M., Desplenter, N., Van Aken, I. & Degraer, S. 2023a. *Visievorming ONTMANTELING OFFSHORE WINDPARKEN in het Belgisch deel van de Noordzee*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 40 pp.

Van Maele, T.M., Desplenter, N. & Degraer, S. 2023b. *Visievorming AQUACULTUUR in het Belgisch deel van de Noordzee*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 75 pp.

CHAPTER 2

LONG-TERM IMPACT OF TURBINE PRESENCE ON MACROBENTHIC COMMUNITIES IN THE BELGIAN PART OF THE NORTH SEA

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Abstract

The present study investigates the long-term impacts of offshore wind farms (OWFs) on macrobenthic communities in Belwind and C-Power, in the Belgian part of the North Sea (BPNS), over a time span of 13 years (2008–2020). We anticipated that not only the presence of turbines will affect macrobenthic communities, but also climate change. Our 13 years analysis supported what is already generally accepted regarding turbine-related impacts. Higher macrobenthos abundance, species richness and diversity were obtained in sediment with a higher fine-sediment fraction and total organic matter content. It was also possible to confirm the common pattern of higher abundance in gullies between sandbanks. Climate related predictors (SST and AMO) were significant predictors of macrobenthic diversity, abundance, and species richness; however, no clear patterns could be obtained. Therefore, in future studies, it remains important to incorporate local environmental variables (sediment characteristic and organic matter) that are affected by the turbine presence and water depth, alongside climate predictors. Our study further revealed that macrobenthic abundances behaved differently in both OWFs, regarding time since construction, and that no clear stable state (climax stage) has yet been reached, after 13 years of OWF presence in the BPNS. These findings highlight the importance of long-term studies, as more time may be needed to observe clear trends in the response of macrobenthic communities within OWFs.

1. Introduction

With the Royal Decree of 17 May 2004, Belgium delineated a zone of 238 km² in the Belgian part of the North Sea (BPNS) for renewable energy production. The country has already achieved its goal of producing 13% of its electricity from renewable energy sources by 2020, with eight operational offshore windfarms (OWFs) in the BPNS, having a cumulative capacity of 2.26 GW (Rumes *et al*. 2021, 2022; Degraer *et al*. 2022). A new marine spatial plan came into force in March 2020 where a second area of 285 km² is reserved for renewable energy production

(Marine Spatial Plan 2020; Rumes *et al*. 2022), which is expected to provide between 3.15 and 3.5 GW of installed capacity (Degraer *et al*. 2022). With an area of 523 km² reserved for OWFs in the BPNS, ecological impacts are expected, and monitoring is therefore required (Brabant *et al*. 2013; Degraer *et al*. 2022).

The OWFs in the BPNS are generally located in areas with natural soft sediments consisting of medium to coarse sand, with a median grain size between 250 and 500 µm and a relatively low organic matter (OM) content (<1%) (De Maersschalck *et al*. 2006). These types of sediment are usually characterised by macrobenthic assemblages with low density and diversity (Van Hoey *et al*. 2004; De Maersschalck *et al*. 2006; Breine *et al*. 2018).

The construction and presence of OWFs have well-known effects on the ecosystem (Coates *et al*. 2014; Degraer *et al*. 2020; Dannheim *et al*. 2020; Lefaible *et al*. 2023). During the exploration and construction phases, direct removal of substratum and benthos occurs, with slow-moving species being the most affected (Hiscock *et al*. 2002). However, these effects are considered to be short-term, and macrobenthic recovery is observed after two to four years after construction (Van Dalfsen *et al*. 2000; Coates *et al*. 2015). The operational phase (i.e., 20–25 years) involves the implementation of hard-substrate foundations in a naturally sandy environment (Hiscock *et al*. 2002; Dannheim *et al*. 2020). The presence of the turbines leads to local modifications in the hydrodynamical regime (Dannheim *et al*. 2020) and the topography of the seabed (Hiscock *et al*. 2002), and to fining of the sediment in the vicinity of the turbines (Coates *et al* 2014; Lefaible *et al* 2023). This fining can cause a decrease in sediment permeability (Janssen *et al*. 2005; De Backer *et al*. 2014), leading to changes in nutrient cycling in the seabed (Toussaint *et al*. 2021). The turbines and scour protection layers are rapidly colonised by epifaunal organisms (De Mesel *et al*. 2013; Zupan *et al*. 2023). Especially on foundations, the colonising fauna mainly consists of suspension feeders, filtering significant amounts of sea water (Voet *et al*. 2023) for feeding, and also producing large amounts of faecal pellets (Mavraki *et al*. 2022) that are expected to be deposited on the sea floor close to the turbines (Baeye & Fettweis 2015). This in turn can be the reason for the observed higher organic matter content in the seabed around turbines (Coates *et al* 2014), providing additional food availability for macrobenthic communities (Mavraki *et al*. 2022). This leads to an increased species richness and abundance resulting in a shift in community structure of the macrobenthos (Coates *et al*. 2014; Lefaible *et al*. 2023). However, these effects seem dependent on the location of the OWF with respect to the coast and/or the turbine foundation type (Lefaible *et al* 2023), and on environmental factors such as seabed morphology and water depth (Cheng *et al*. 2021; Coolen *et al*. 2022).

In addition, climate change and local weather also affect macrobenthic communities. Wieking & Kröncke (2001) showed that the North Atlantic Oscillation index (NAOi) affects the hydroclimatic state of the North Sea. Negative NAOi values reflect extreme cold winters, which have an important impact on macrobenthic abundance by decreasing the sea surface temperature (SST) (Kröncke *et al*. 2013). Previous studies (Dippner & Kröncke 2003; Kröncke *et al*. 2013) indicated that fluctuations in macrobenthos abundance were related to changes in the winter NAOi. In the North Sea, the water temperature is affected by both natural variability and climate change, which will impact marine species. The southern North Sea and the English Channel are more likely to warm faster due to their shallow depths and proximity to land (Garci-Soto & Pingree 2012; Harris *et al*. 2014). In addition, the NAOi (as mentioned above) and the Atlantic Multidecadal Oscillation (AMO), a natural 60 to 80 year climate cycle that affects SST in the North Atlantic (Kerr 2000), also introduce variability to the marine environment (McLean *et al*. 2018). The

AMO is currently reaching a warming peak; thus, a cooling AMO phase could reverse the observed trends in fish densities (lower), and impact macrobenthos densities as the two biological groups seem to be structured by similar environmental parameters (Buyse *et al*. 2022). In addition, SST is known to affect macrobenthic communities as well (Kröncke *et al*. 1998, 2013), by influencing biological processes (e.g., gene expression, behaviour, phenology, etc.), competitive interactions (Poloczanska *et al*. 2009) and food webs (Philippart *et al*. 2003), which result in a shift in the structure, function, and biodiversity of macrobenthic communities (Kröncke *et al*. 2013; Dippner *et al*. 2014). Cold winters will have a negative impact on species richness, abundance, and biomass of macrobenthos in the North Sea (Kröncke *et al*. 1998). On the other hand, Kröncke *et al*. (1998) found that mild winters are beneficial to macrobenthic communities, resulting in higher biomass, production and reproduction, and reduced mortality.

With the exception of Coolen *et al* (2022), studies on the effect of turbine presence on macrobenthos communities were based on data collected over a relatively short term (Coates *et al*. 2014, 2015; Lefaible *et al* 2023). Long-term studies are needed to understand how macrobenthic communities evolve over decades of OWF presence. When monitoring the long-term impacts of OWF presence on macrobenthic communities, there is a chance that not only the presence of OWFs will have an impact on the communities inhabiting the seabed, but also climate change. Incorporating climate-related aspects in the analysis of long-term data will offer the opportunity to assess the importance of climate change related aspects, and hence will provide a clear view on the effects of the presence of the turbines *per se*.

In this study, we investigated the spatiotemporal variability of macrobenthos communities in two OWFs, Belwind and C-Power, both located in the BPNS, from the time of installation of turbines in 2008 to 2020. More specifically, we investigated how macrobenthos communities (as reflected in biodiversity indices) respond to a variety of (environmental) predictor variables, such as year since construction of OWFs, distance from turbine, fine sediment fraction, total organic matter content, water depth, climate indices (NAOi and AMO) and local weather (with sea surface temperature (SST) as a proxy).

2. Material and methods

2.1. Study area

The C-Power OWF is located on the Thornton Bank at 27 km from the Belgian coastline. In 2008, 6 gravity-based foundations were built and became operational in 2009. In 2011, another 48 jacket foundations were constructed within this OWF (Degraer *et al*. 2013). The six gravity-based turbines have a diameter of 23.5 m and a surrounding scour protection layer with a diameter of 55.5 m. They are located 500 m apart at water depths ranging from 18 to 24 m (Coates *et al*. 2014). The jacket turbines consist of a steel jacket with four legs occupying 18 × 18 m and are located 700 m apart (C-Power. n.d.). The Belwind OWF is situated 46 km off the Belgian Coast on the Bligh Bank. It consists of 55 monopile turbines constructed between 2009 and 2010 and has been operational since 2011. The foundations are located 500–650 m apart at a water depth ranging between 15–40 m (Fig. 1).

2.2. Sample design, collection and treatment

2.2.1. Biotic data

Over a period of 13 years (2008–2020), macrobenthic samples were collected on the Thornton and Bligh Bank during the months of October and November (Autumn period), at different distances from the turbine in both OWFs. In C-Power, samples from 2008 and 2009 were taken in the western (WTA) and eastern concession area (WTB) and in the fringe areas (WTC). In Belwind, samples in

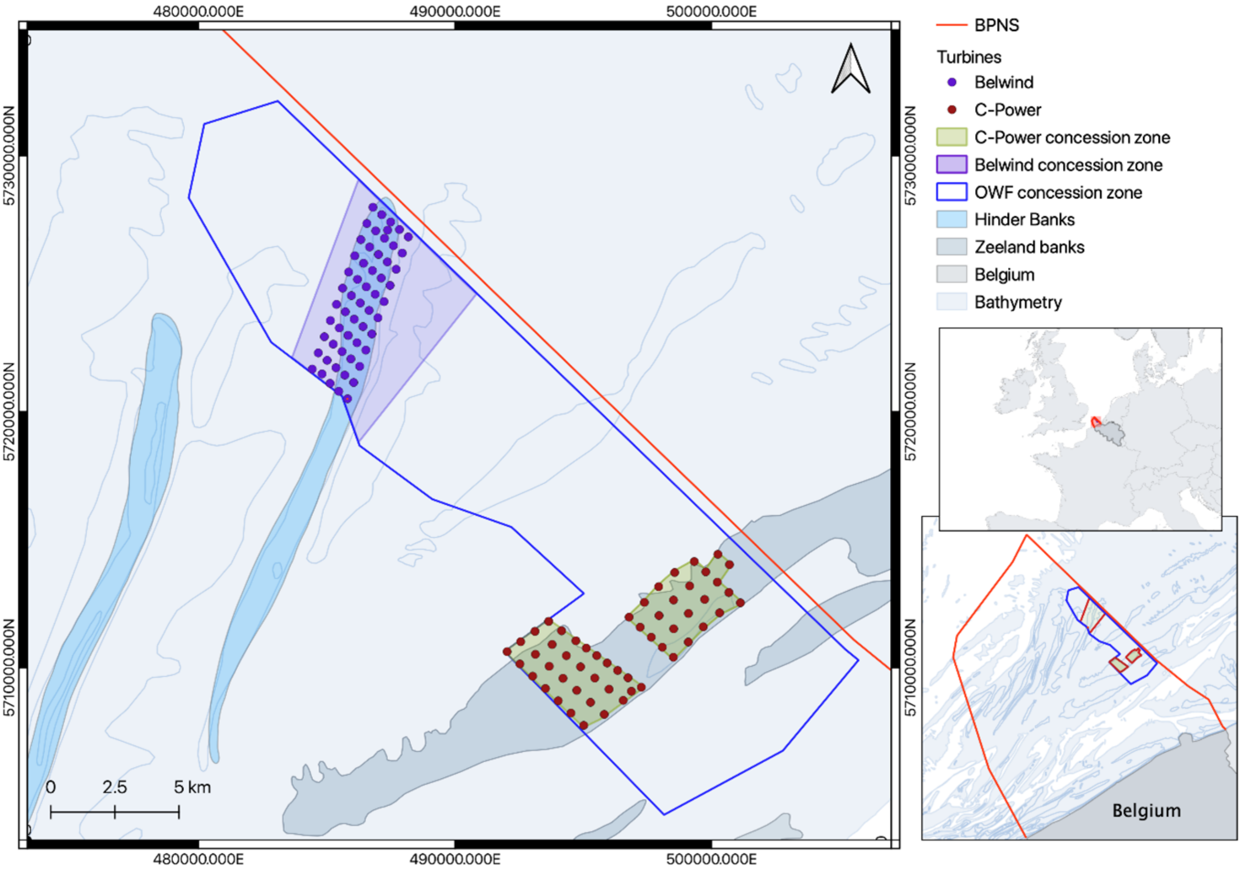

Figure 1. Map of the study area. Turbines in the Belwind and C-Power OWF are represented as dots within each corresponding concession zone (colored rectangles). The entire concession zone is delineated by the blue polygon. Projection: EPSG:32631 – WGS84 / UTM ZONE 31N.

2009 were taken from the impacted (BBI) and the edge area (BBE). From 2010 to 2014, the sampling design was the same in both OWFs as macrobenthos samples were collected in three zones (control, edge, and impact). In addition, from 2010 to 2012, samples were also collected by divers along four gradients (Northwest, Northeast, Southwest and Southeast) at seven different distances (1 m, 7 m, 15 m, 25 m, 50 m, 100 m and 200 m) from the turbine. Between 2015 and 2016, a systematic stratified sampling design was adopted, and samples in both OWFs were collected at two distances from the turbine, close (50 m) and far (350–500 m). From 2017 onwards, the design changed again and samples were collected 'very close' (37.5 m) and far (350–500 m) from the turbines.

Macrobenthos samples were collected by means of a Van Veen (VV) grab or by scientific divers. The sampling surface of the VV grab ranged from 0.0247 to 0.1 m². Scientific divers collected macrobenthos samples with an airlift with mesh bags of 1 mm mesh size, covering a rectangular area of 0.1 m². In early years, until 2014, three replicates were taken from each location. As of 2015, only one sample was taken per location. On board, the samples were sieved over a 1 mm sieve and fixed in 4% formaldehyde-seawater. In the laboratory, the samples were stained with 1% Rose Bengal, and rinsed over a 1 mm sieve. The organisms were counted and identified to the lowest possible taxonomic level and stored in a 4% buffered formaldehyde solution. Biotic data generally included organisms identified to species level.

2.2.2. Environmental data

To collect the environmental data such as the sediment grain size distribution and the total organic matter content (TOM), a subsample was taken from each VV grab sample with a small plexiglass core (3.6 cm diameter). The sediment samples were dried at 60°C for 48h. Grain size distribution was measured using laser diffraction and used to determine 'fine sediment' fraction (% < 250 µm). TOM content in each sample was calculated as the difference between dry weight (drying for 48 h at 60°C) and ash-free dry weight (2 h at 500°C) divided by the dry weight, multiplied by one hundred.

2.2.3. Additional predictor variables

Year since construction was calculated by subtracting the year of construction of the OWF project from the actual sample year. QGIS (QGIS Development Team 2020) was used to extract water depths of the sampling points from a bathymetry layer with a 20-m resolution and to calculate the distances of the sampling points to the center of the nearest turbine. Closest distances of the sampling points from the turbines varied over the years due to the construction of new turbines. Therefore, data was only included from the year in which the minimum distance to a turbine remained fixed. Additionally, samples collected prior to turbine installation or located > 1000 m away were not selected for this analysis, to allow the interpretation of the 'distance to turbine' predictor as within the assumed zone of influence of the turbine. As such, the data are analysed in a gradient design, not in a BACI design.

To investigate the effect of climate and weather variability on top of OWF effects, sea surface temperature (SST), North Atlantic Oscillation index (NAOi) and Atlantic Multidecadal Oscillation index (AMO) were added to the dataset. SST was extracted through the EU Copernicus Marine Environment Monitoring Service (2023; http://marine.copernicus.eu/) for the 0.25º × 0.25º grid cell corresponding to the OWF concession zone (51.75–51.5º N, 2.75–3.0º E). Daily SST measures were averaged by season, and maximum and minimum values were selected for each season to account for extreme events (Fig. 2). The NAO index values were downloaded from the NOAA Climate Prediction Centre website (https://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao.shtml) and AMO index from the NOAA Physical Sciences website (https://www.esrl.noaa.gov/psd/data/timeseries/AMO/). Monthly values were downloaded for both indices and then averaged per season (Fig. 2).

2.2.4. Data quality control

An integrated database was created combining all the data from the 13 year period. Due to an unequal set of replicates over the years, only the first replicate of each location was considered for statistical analysis. Each sampling point included TOM%, fine sediment fraction (fraction of sediment <250 µm) and median grain size (µm), water depth, distance to the nearest turbine, year since construction and climate related variables (SST, AMO, NAOi). Samples that had a fine sediment fraction above 80% and a TOM content above 2% were removed (20 samples in total), since they were considered implausible for the sandy sediments of our study area. In this case, these samples potentially constitute a rare observation of a muddy sediment aggregation in a sandbank environment. The dataset was also checked for inconsistent species identification, and certain species (*Bodotria* sp., *Capitella* spp., *Diastylis* spp., *Eteone* spp., *Glycera* spp., *Pontocrates* spp., *Pseudocuma* spp., *Ophiura* spp., *Polynoidae* spp., *Vaunthompsonlinae* spp.) were therefore lumped to higher taxonomic level. Juveniles were kept in the datasets as species since they can be indicators of change in macrobenthic communities. The dataset exhibited heterogeneity across various variables. The majority of samples were collected using VV Grab during the Autumn season at water depths ranging from 15 to 25 m. Sampling occurred along a gradient of 28 to 1000 m

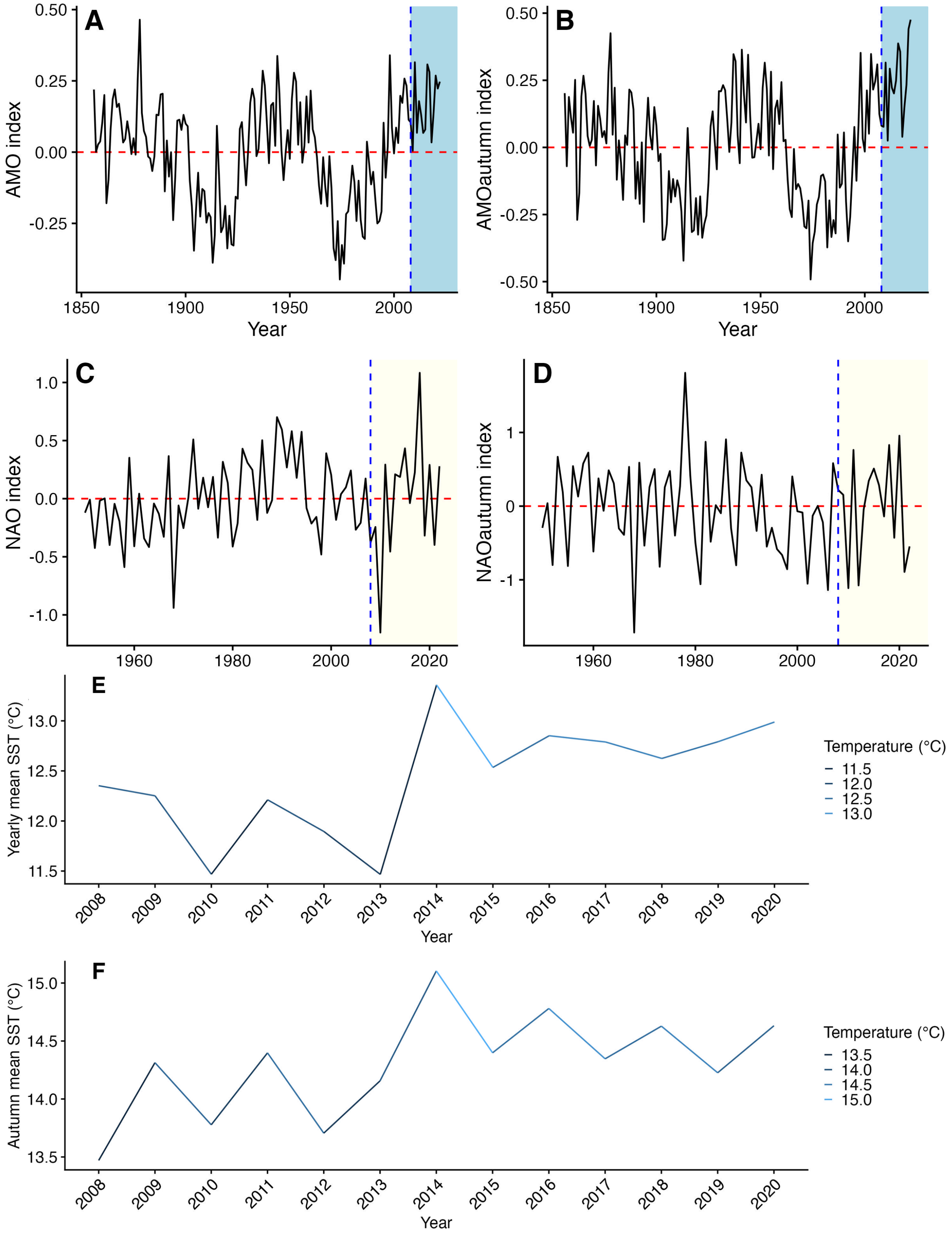

Figure 2. Graphical presentation of climate indices AMO (A–B) and NAOi (C–D) over the years, and evolution of SST throughout the study period (E–F). Blue and yellow squares highlight the period of the study from 2008 to 2020. Autumn-averaged values of AMO (B), NAOi (D) and SST (F) are also shown because the majority of the data is from that period.

Table 1. Sampled stations per year classified by OWF, CP stands for C-Power and BW for Belwind.

	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	Sum
CP	2	8	2	16	26	8	8	48	46	46	44	29	28	311
BW	0	16	3	9	8	9	6	42	37	50	45	28	0	253

from the turbine, with systematic observations at approximately 30–50 m and 400–500 m. A higher frequency of occurrences was recorded in the later years, specifically from 2015 to 2020 (Fig. 3). After the data quality control, a total of 564 samples were kept for analysis: 55% originated from the C-Power OWF and 45% was sampled from the Belwind OWF (Table 1).

2.3. Data analysis

Three community indices were calculated: species richness (S, number of taxa per sample), total abundance (N, number of individuals per sample) and Shannon-Wiener diversity index (H). A data exploration was carried out following the procedures of Zuur *et al.* (2010), where the presence of outliers, collinearity between variables and interactions with location were assessed. High collinearity was found between median grain size and fine-sediment fraction (r=-0.8), and the latter was retained for analysis because of its stronger correlation with macrobenthic community structure (Lefaible *et al.* 2023). Additionally, average SST was highly correlated with SSTmin and SSTmax values (r=0.8). Therefore, the former was excluded from the analysis. No variance inflation factors (VIF) higher than 3 were found between the remaining variables, thus they were all kept

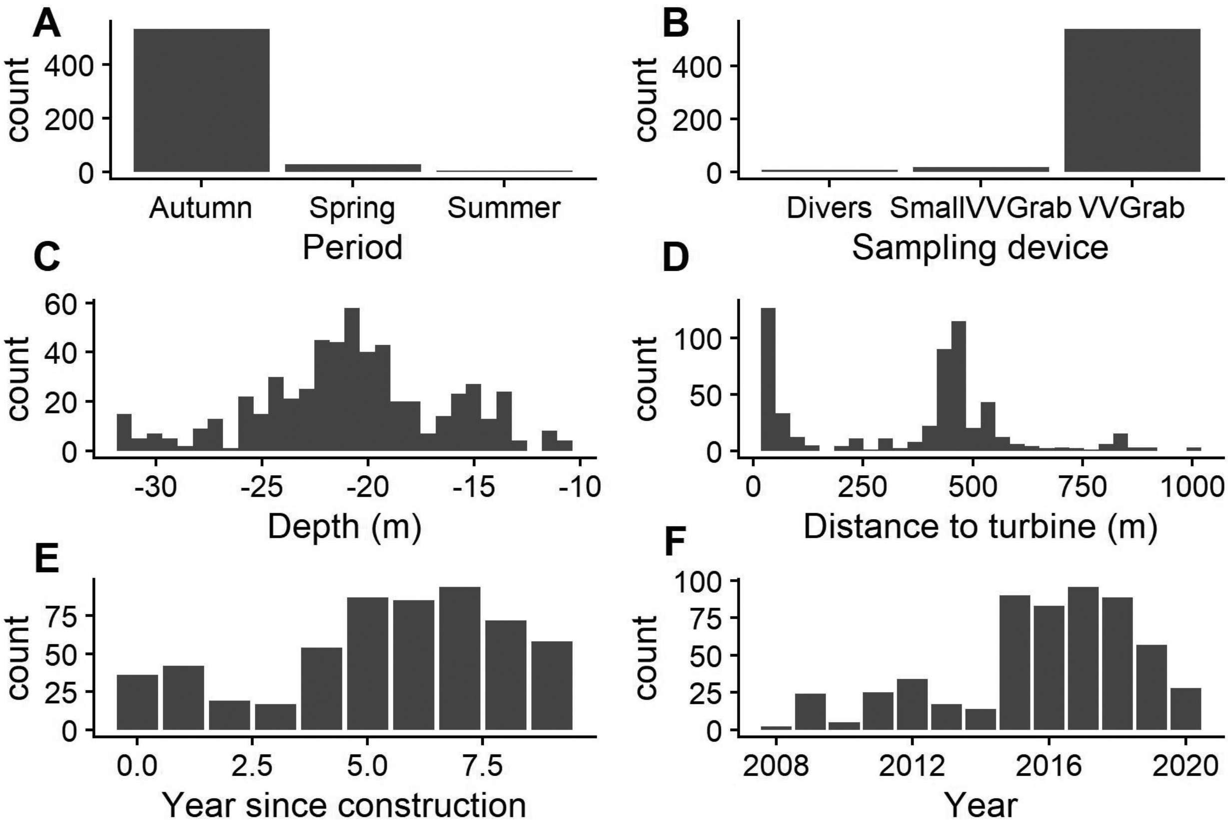

Figure 3. Number of occurrences of variables in the data. Count refers to the number of samples. **A**. Season of sampling. **B**. Sampling device. **C**. Sampling water depth. **D**. Distance to the nearest turbine. **E**. Year since construction of the OWF. **F**. Natural year the sample was taken.

in the analysis. Data exploration showed that sample size was evenly distributed throughout the samples regarding sample surface. To maintain integer count values (necessary for negative binomial distribution, see further), data was not scaled to the smallest sample surface, instead, sampling surface was used as an offset to account for the different sampling surfaces between VV grab (0.1 m^2) and small VV grab (0.0247 m^2), an offset is a model variable with a known or pre-specified coefficient which represents the size of each observational unit. All data operations were carried out in R, version 4.2.2 (R Core Team 2009).

General Additive Models (GAM) were built for species richness, total abundance, and Shannon-Wiener diversity with the "mgcv" package in R (Wood 2006) (Table 2). Models were built using forward selection methodology, starting with the simplest model with one variable and progressively adding new variables. The Akaike Information Criterion (AIC) was used to select the most suitable model, selecting the model with the lowest AIC in every step of the procedure until the AIC value did not decrease anymore or the decrease was less than two points. Effective degrees of freedom (edf) were used to assess the linearity of the predictor variables. If edf was close to 1, the variable was modelled as a linear term. Species richness and total abundance were both modelled using a negative binomial distribution with a log-link function. Shannon Diversity index was modelled using a Gaussian distribution with

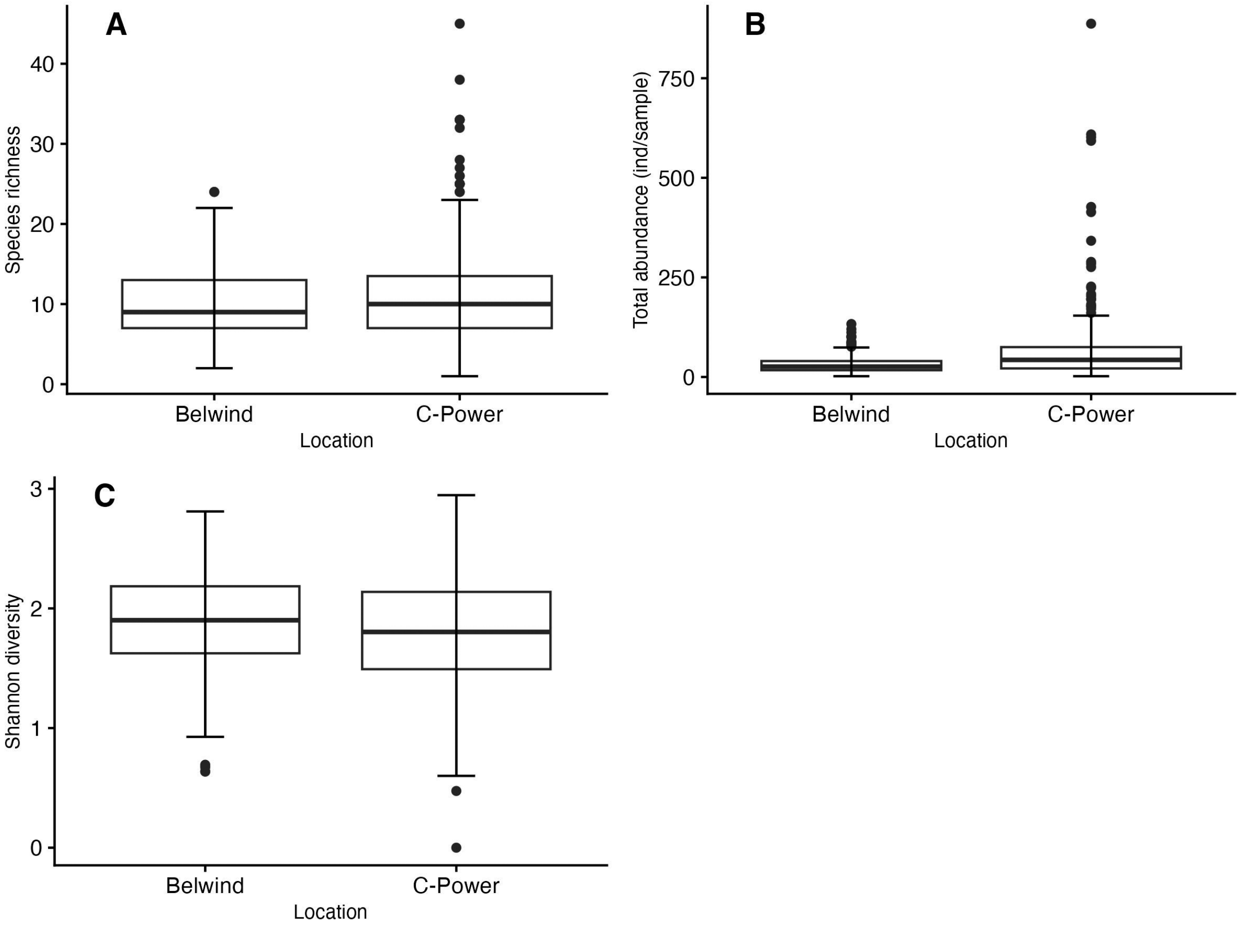

Figure 4. Boxplots representing the values of the three biodiversity indices per location. The line in the middle of the boxplot represents the median, and the lower and upper ends of the box are, respectively, the 25% and 75% quartiles. The lower hinge is the smallest data value, and the upper hinge is the largest data value. Whiskers indicate variability outside the upper and lower quartile. Dots represent outliers.

Table 2. GAM models and predictor variables selected for each response variable. R stands for Response variable (S, N and H); Nb for negative binominal and G for gaussian. Fine (fine sediment fraction, µm), TOM (total organic matter, %), water depth (depth below sea surface, m), distance (meters to the center of the nearest turbine, m), SST max (maximum SST in the season, Celsius), SST min (minimum SST in the season, Celsius), NAOi (North Atlantic Oscillation Index), AMO (Atlantic Multidecadal Oscillation), Location (sand bank where the OWF is located), Period (Autumn, Summer or Spring). ✓indicates inclusion to the model, - indicates exclusion.

	Family	**Link**	**Fine sediment fraction**	**TOM**	**Water Depth**	**Distance to turbine**	**Year since construction**	**SST max**	**SST min**	**NAOi**	**AMO**	**Location**	**Season**
S	Nb	Log	✓	✓	✓	✓	–	–	–	–	✓	–	–
N	Nb	Log	✓	✓	✓	✓	✓	–	✓	–	–	✓	–
H	G	Identity	✓	✓	✓	✓	–	✓	–	–	–	✓	–

an identity-link function. Smoothing functions are indicated with s(). Once the models were selected, residuals were plotted against every covariate (in and out of the model) and were checked visually to confirm assumptions of homogeneity of variance.

Model structure for species richness data:

Species richness = gam (s(fine sediment fraction) + s(water depth) + s(Atlantic Multidecadal Oscillation) + total organic matter + s(distance to turbine, by=Location), with a negative binomial distribution)

Model structure for abundance data:

Total abundance = gam (fine sediment fraction+ s(SST min) + s(water depth) + total organic matter + distance to turbine + Location + s(year since construction, by=Location), with a negative binominal distribution)

Model structure for Shannon-Wiener Diversity index data:

Shannon-Wiener diversity index= gam (s(fine sediment fraction) + water depth + s(SST max) + Location + s(total organic matter) + s(distance to turbine), with a Gaussian distribution)

3. Results

Over the 13 years study period (2008 to 2020), species richness (S) in C-Power and Belwind ranged from 1 to 45 and from 2 to 24 taxa, respectively (Fig. 4A). Abundances in C-Power ranged from 2 to 887 individuals per samples and from 2 to 133 individuals per sample in Belwind. Shannon-Wiener diversity ranged from 0 to 2.95 in C-Power and from 0.63 to 2.81 in Belwind (Fig. 4C).

The predictor variables selected for the species richness GAM model explained 44.2 % of the deviance (Table 3). There was a significant linear increase of S with increasing %TOM. Species richness also showed significant non-linear relationships with the fine sediment fraction, water depth and AMO. Higher species richness was linked with increasing fine-sand fraction and water depth, whereas a fluctuating pattern was found between S and AMO (Fig. 5). Distance to turbine did not significantly affect S at Belwind, while a significant non-linear effect was found at C-Power with increasing values of S at closer distances to the turbine (Fig. 5).

Almost 60% of the variation in total abundance (N) was explained by the final

Table 3. Final models for each response variable. When edf = 1.00 the term is modelled as a linear term, when not, the term has a smoother. : indicates interaction between both variables; location (TB = Thornton Bank; BB = Bligh Bank) is a factor.

Species richness	edf	p-value	Total abundance	edf	p-value	Shannon diversity	edf	p-value
TOM	1.00	5.44e-07	Fine sediment fraction	1.00	< 2e-16	Water depth	1.00	1.73e-06
Fine sediment fraction	1.69	< 2e-16	Distance turbine	1.00	4.07e-09	Location TB	–	0.0017
Water depth	1.72	< 2e-16	Location TB	–	0.0926	Fine sediment fraction	3.38	< 2e-16
AMO	5.85	0.0003	TOM	1.00	3.93e-07	Maximal Sea Surface Temperature	7.82	6.57e-06
Distance to turbine:BB	1.01	0.5332	Minimal Sea Surface Temperature	4.09	< 2e-16	TOM	3.94	0.0025
Distance to turbine:TB	3.15	1.03e-07	Water depth	2.51	< 2e-16	Distance to turbine	4.09	0.0340
Deviance explained: 44.2%			Year since construction:BB	1.00	0.0001	Deviance explained 29.1%		
$R^2 = 0.434$			Year since construction:TB	1.00	0.0030	$R^2 = 0.268$		
			Deviance explained: 59.6%					
			$R^2 = 0.332$					

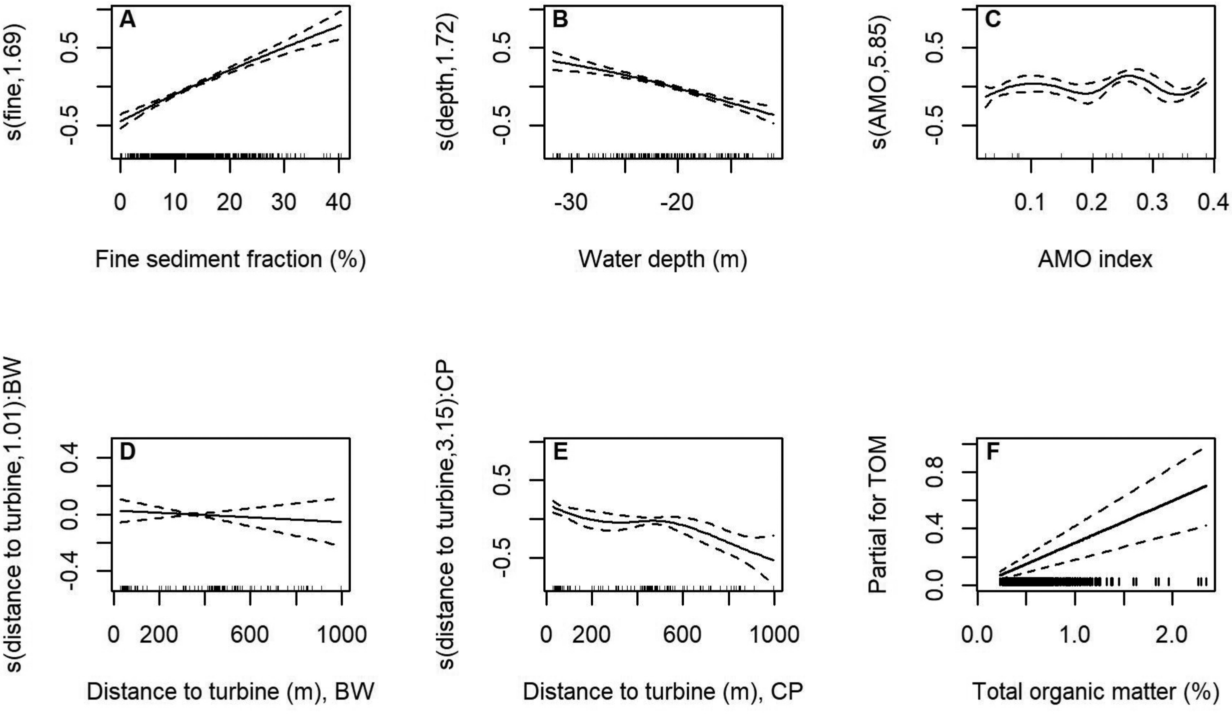

Figure 5. Fitted smoothing curves for the different response variables in the best-fitting GAM, explaining Species Richness patterns in Belwind and in C-Power. Dashed lines: standard errors (SE). N sample = 564. Black lines in X-axis correspond to sample values. In the Y-axis, the smoother for the predictor variable is shown, number between brackets refers to number of basic functions (k).

GAM model (Table 3). There was a significant linear increase of N with increasing fine-sediment fraction and %TOM and with decreasing distance to the turbine for both OWFs. Variation in N throughout years of construction depended on the sandbank: a significant decrease in abundance over the years was observed in Belwind, whereas total abundance increased significantly over time in C-Power. Significant non-linear patterns were found between N and the predictor variables water depth and minimum. N was highest at deeper water depths, while it showed a more fluctuating relationship with minimum SST, with lowest values around 10°C (Fig. 6). Location did not have a significant effect on macrobenthic abundance.

The final GAM model explained about 30% of the deviance in Shannon-Wiener diversity index (H) (Table 3). Whereas H was significantly higher within Belwind compared to C-Power, significant relationships between H and the other predictor variables were rather complex (i.e., non-linear) but comparable between OWFs (irrespective of location). There was an increase of Shannon-Wiener diversity with increasing fine-sediment fraction and water depth. Shannon values increased with increasing %TOM followed by a decline when values were higher than

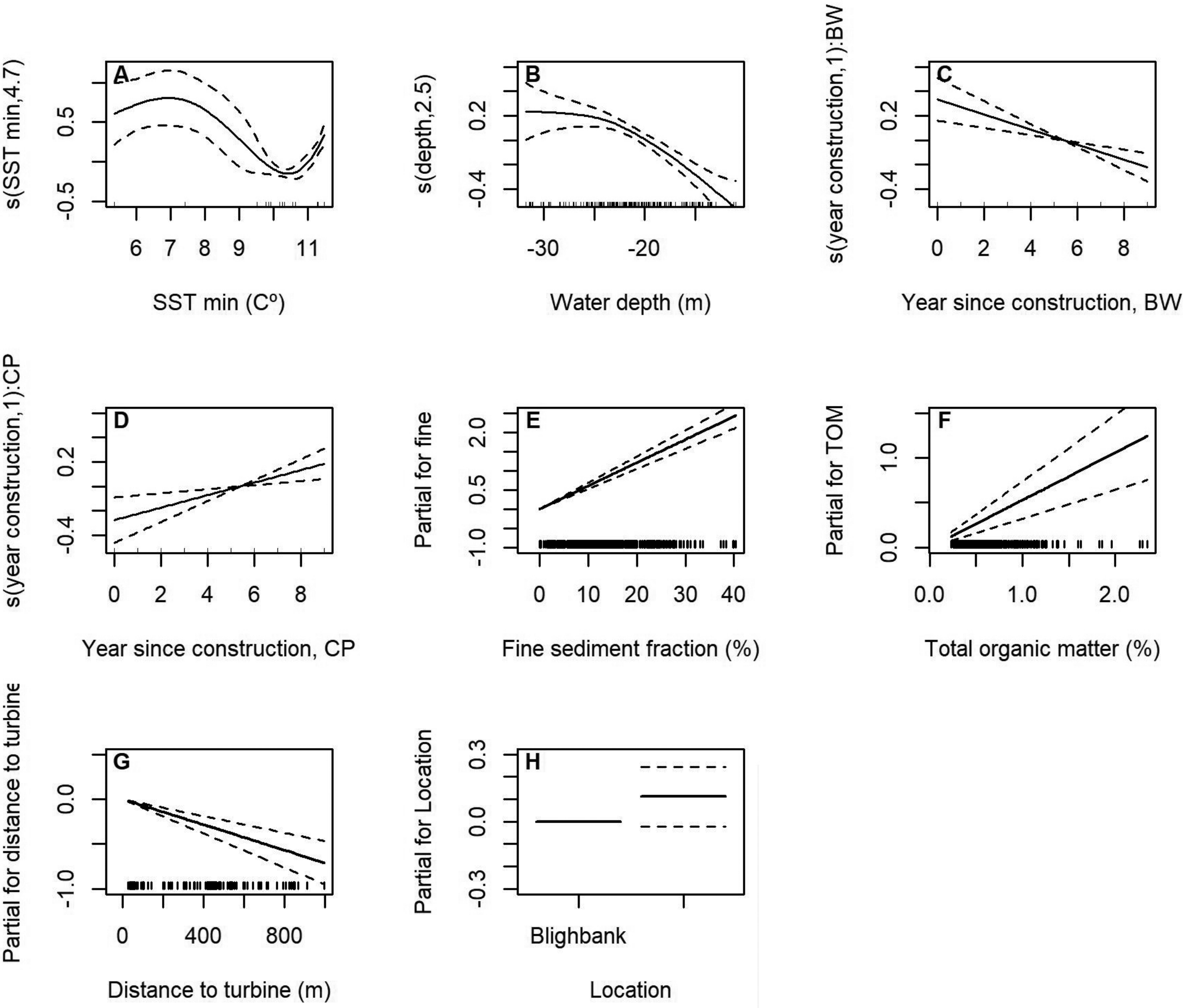

Figure 6. Fitted smoothing curves for the different response variables in the best-fitting GAM, explaining total abundance patterns in Belwind and in C-Power. Dashed lines: standard errors (SE). N sample = 564. Black lines on X-axis correspond to sample values. On the Y-axis the smoother for the predictor variable is shown, number between brackets refers to number of basic functions (k).

1.5%. Diversity remained very similar until distances of 500 m from the turbines, followed by a decrease at further distances. Finally, Shannon-Wiener diversity varied significantly with maximum SST values, with highest values for temperatures between 14 and 16°C, while a clear decrease was seen around a maximum SST of 17-18°C. (Fig. 7).

4. Discussion

Our analysis showed that all predictor variables affected macrobenthic biodiversity of the two OWFs, Belwind and C-Power. Climate or temperature-related effects indeed contributed significantly to our statistical models, but did not surpass the effect of local environmental variables that are influenced by the presence of turbines and water depth. Our analysis further revealed that over the course of 13 years, macrobenthic communities behaved differently in both OWFs regarding time since construction, and that no clear stable state has yet been reached, after 13 years of OWF presence in the BPNS.

4.1. Influence of the turbines

In general, macrobenthos diversity and abundances were higher in samples taken closer to the turbines compared to further distances within both OWFs. This pattern has already been observed, and was explained by sediment fining around the jacket foundations in C-Power (Lefaible *et al.* 2023). Our study confirms that this effect is consistent over the course of 13 years, but the long-term density trends also highlight the complexity of the processes taking place in the OWFs. Indeed, we see that the macrobenthic abundance and diversity increase with the fine sediment fraction, but also with TOM percentage. Coates *et al.* (2014) suggested that the colonization of the turbine foundations by filter-feeding organisms will likely increase the depositional flow of faecal pellets and detritus towards the sediment. These faecal pellets are rich in organic matter (OM). Hence, they bring additional food to the sediment (Maar *et al.* 2009; McKindsey *et al.* 2011; Ysebaert *et al.* 2009). Sediments which undergo fining will have a lower permeability, which facilitates OM retention (Janssen *et al.*

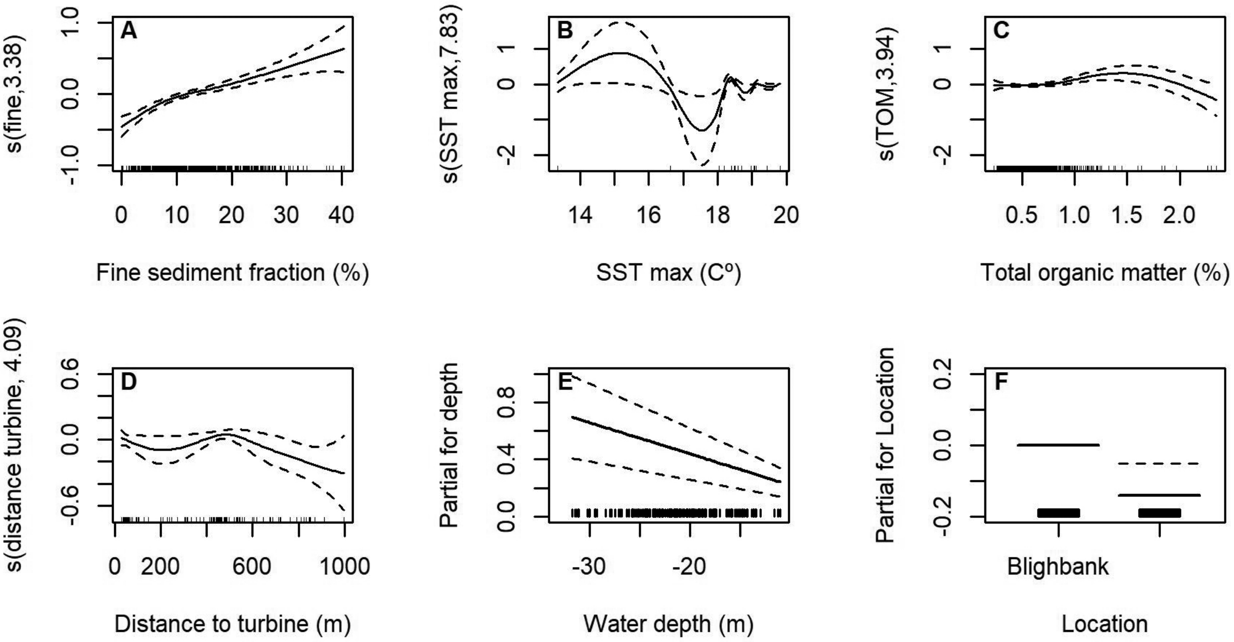

Figure 7. Fitted smoothing curves for the different response variables in the best-fitting GAM, explaining Shannon-Wiener diversity patterns in Belwind and in C-Power. Dashed lines: standard errors (SE). N sample = 564. Black lines on X-axis correspond to sample values. On the Y axis the smoother for the predictor variable is shown, number between brackets refers to number of basic functions (k).

2005). The decreased permeability combined with enrichment in OM particles in the sediment, and the reduced oxygen penetration will in turn affect the sedimentary conditions and the macrobenthos community structure (Maar *et al.* 2009; McKindsey *et al.* 2011, Ysebaert *et al.* 2009). Although a general increase in all three response variables can be observed as fine sediment fraction and TOM percentage increase, the Shannon-Wiener diversity seems to decrease when TOM content exceeds 1.5%. This pattern was also found in the study of Lefaible *et al.* (2023) that used partially the same data, in terms of species richness in nearby sediments around jackets in C-Power. This highlights that we would expect a linear increase in total abundance with TOM, but not necessarily in terms of diversity. One explanation could be that opportunistic species occurring in high densities settle in patches with lots of food (Keeley *et al.* 2013; Johansen *et al.* 2018). In that case, high dominance of a few species will lead to a lower diversity.

Our analysis also revealed that water depth influences macrobenthic biodiversity. The deeper sediments in our study correspond to troughs or gullies in between the sandbanks, which generally consist of fine to very fine sands (Van Lancker *et al.* 2012). In those conditions, macrobenthic communities with a higher abundance and diversity can develop (Breine *et al.* 2018). Consequently, artificial reef (AR) effects could be more pronounced within the troughs and gullies, compared to shallower areas with wave-exposed crests, due to the initial sediment conditions (Lefaible *et al.* 2023; Van Lancker *et al.* 2012). Also, in the 9 years study of Coolen *et al.* (2022) and the 3 years study of Lefaible *et al.* (2023), water depth was a good predictor of macrobenthos abundance and species richness, with abundances increasing with water depths to 30 m, confirming the common pattern in the shallow part of the North Sea (Armonies *et al.* 2014). Similar results were obtained in our study, with all three response variables at their maximal values at 30 m depth, which seems to be the depth with optimal circumstances with respect to several environmental factors such as bottom temperature, tidal currents, and food availability (Künitzer *et al.* 1992). In the Belgian part of the North Sea, there is no thermal stratification (and thus, no decrease in oxygen concentration in deeper waters) and the temperature of bottom waters remains high (> 10°C) (Tomczak & Goedecke 1962). In addition, currents decrease with increasing water depth (Reiss *et al.* 2010), hence macrobenthic communities that inhabit sediments at deeper water depth are less affected by waves and sediment disturbance, allowing more diverse and abundant communities to develop (Armonies *et al.* 2014).

4.2. Influence of weather and climate

Our study also showed that changing SSTs have an impact on macrobenthos. We observed a general increasing trend in SST over the years (Fig. 2E), though with interannual variations. This is also reflected by the positive values obtained for the AMO (Fig. 2A), generally implying that SSTs are becoming warmer over time. Macrobenthos abundance and Shannon-Wiener diversity were both impacted by SST, while AMO was able to predict species richness. However, the trends are not clear due to the lack of a fully developed gradient within the temperature range (few observations between 6–10°C and 14–18°C), which is also related with the choice of sampling times (mostly during autumn). It is hypothesized that rising temperatures will affect primary production, leading to an increase or decrease in the abundance of phytoplankton depending on the specific area (Suikkanen *et al.* 2007; Desmit *et al.* 2020). This, in turn, will influence the flux of organic matter (OM) towards the sediment (Suikkanen *et al.* 2007), which is incorporated in the benthic food web (Lesutiene *et al.* 2014; Karlson *et al.* 2014, 2015). However, it is important to note that increasing temperatures will also enhance pelagic mineralization, potentially limiting the export of OM to the seafloor (Timmermann *et al.* 2012; Wikner & Andersson 2012), and subsequently reducing the availability of food for macrobenthic communities. In the North Sea, phytoplankton appears to be influenced by two main processes: ongoing rising sea surface temperatures and de-eutrophication (Desmit *et al.* 2020). The study by Desmit *et al.* (2020) indicates that a combination of warming temperatures and reduced input of riverine nutrients may lead to a decline in phytoplankton abundance. Previous research (Buchanan *et al.* 1987; Frid *et al.* 2009) has already demonstrated the impact of climate change and OM fluxes on macrobenthic communities. Consequently, the true impact of changing temperatures on macrobenthos communities in Belgian OWFs remains unrevealed.

4.3. OWF - specific patterns

Since the construction of the Belwind and C-Power OWFs, opposite long-term trends were found regarding macrobenthic abundances, with a decrease in Belwind vs an increase in C-Power over time. Frid *et al.* (2009) explained that offshore stations (Belwind) are more influenced by large-scale phenomena such as climate, while stations closer to the coastline (C-Power) will be more influenced by riverine inputs and winter weather. This might explain the contrasting trend in macrobenthos abundance in both OWFs. Furthermore, the difference between Belwind and C-Power in terms of environmental conditions, distance from the shore and foundation types may also contribute to these abundance patterns. Belwind is located in a high-energy system, with strong current velocity (Legrand & Baetens 2021), and is prone to intense resuspension and ephemeral organic enrichment compared to C-Power, as suggested by Lefaible *et al.* (2018). These environmental conditions, combined with the associated impacts of different turbine types (Belwind: monopiles vs C-Power: gravity-based and jackets foundations) will most likely affect the strength and extent of AR-effects on benthic communities. At present, it seems that there are still no direct impacts of OWFs, or none that can be detected, in Belwind (Coates & Vincx 2010). In C-Power, by contrast, the ecological changes linked with the AR-effects are clear (Lefaible *et al.* 2023) and could represent one of the reasons for the opposite abundance trends in both OWFs.

The differences for both OWFs are not only found for macrobenthic abundances, but also for epibenthos and demersal fish (De Backer *et al.* 2019). These differences may be related to different factors, such as site-specific characteristics (hydrography, local weather, distance to shore, etc.), foundation type or possibly other human activities taking place in the area. Moreover, other factors might also be at play when studying the evolution of macrobenthos communities over time. Buchanan *et al.* (1978) and Frid *et al.* (2009) hinted that density-dependent factors, such as food limitation and predation stabilize the community, and are important factors to be considered. Overall, it appears that the macrobenthic communities in both OWFs have not yet reached a climax/stable stage and are still changing after 13 years of OWFs presence.

5. Conclusion

The long-term analysis on the impact of OWFs on macrobenthic communities in the Belwind and C-Power concession areas revealed that all predictor variables had a significant impact on macrobenthic richness, abundance and Shannon diversity. Due to the choice of sampling during a single season, and the fact that climate acts on longer time scale, no clear pattern was observed between weather and climate predictors and the macrobenthic community descriptors. It is therefore suggested to sample during different seasons, to obtain a fully developed gradient of temperatures. Nevertheless, these weather and climate variables cannot be neglected when doing long-term studies, as they are significant predictors in the models. However, it seems that the most important predictors are still local variables such as fine sediment fraction, total organic matter, and water depth. The latter positively impacted macrobenthic communities in both OWFs. Indeed, all three response variables (species richness, total abundance, and Shannon diversity) showed increasing trends with TOM and fine sediment fraction and optimal conditions for macrobenthos were found in the deeper gullies between sandbanks.

Differences in response to distance to turbine and year since construction suggest that impacts can be site-specific and may differ depending on the local conditions and type of turbines within the concession area. Consequently, studies on the impact of the three different types of foundations (jackets, monopiles and gravity-based foundations) in the BPNS are important. Moreover, additional environmental parameters such as primary production should be included in future studies, as phytoplankton abundance can explain a major share of the year-to-year variation in benthic communities (Buchanan 1993).

After 13 years of OWFs presence in the BPNS, it appears that still no climax stage has been reached, or that possibly the climax stage also shows temporal and cyclical variation. This highlights the importance of monitoring and long-term studies, as more time may be needed in order to see a clear stable state, or clear cycles in macrobenthos.

References

Armonies, W., Buschbaum, C. & Hellwig-Armonies, M. 2014. The seaward limit of wave effects on coastal macrobenthos. *Helgoland Marine Research* 68 (1): 1–16.

Baeye, M. & Fettweis, M. 2015. In situ observations of suspended particulate matter plumes at an offshore wind farm, southern North Sea. *Geo-Marine Letters* 35: 247–255.

Bergman, M.J.N. & Hup, M. 1992. Direct effects of beam trawling on macrofauna in a sandy sediment in the Southern North Sea. *ICES Journal of Marine Science* 49: 5–11.

Birchenough, S., Boon, A., Braeckman, U., Brey, T., Brzana, R., Buyse, J., ... & Wilding, T. 2021. Working Group on Marine Benthal and Renewable Energy Developments (WGMBRED). *ICES.*

Brabant, R., Degraer, S. & Rumes, B. 2013. Monitoring offshore wind farms in the Belgian part of the North Sea: Setting the scene. *In*: Degraer, S., Brabant, R. & Rumes, B. (eds) *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Learning from the Past to Optimise Future Monitoring Programmes*. Royal Belgian Institute of Natural Sciences, Operational Directorate Natural Environment, Marine Ecology and Management Section. 239 pp.

Braeckman, U., Lefaible, N., Bruns, E. & Moens, T. 2020. Turbine-related impacts on macrobenthic communities: An analysis of spatial and temporal variability. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds), *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Empirical Evidence Inspiring Priority Monitoring, Research and Management*. Series 'Memoirs on the Marine Environment'. Brussels: Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, 131 p.

Breine, N.T., De Backer, A., Van Colen, C., Moens, T., Hostens, K. & Van Hoey, G. 2018. Structural and functional diversity of soft-bottom macrobenthic communities in the Southern North Sea. *Estuarine, Coastal and Shelf Science* 214: 173–184. https://doi.org/10.1016/j.ecss.2018.09.012

Buchanan, J.B. 1993. Evidence of benthic pelagic coupling at a station off the Northumberland coast. *Journal of Experimental Marine Biology and Ecology* 172 (1–2), 1–10.

Buchanan, J.B. & Moore, J.J. 1987. Long-term studies at a benthic station off the coast of Northumberland. *In*: *Long-Term Changes in Coastal Benthic Communities: Proceedings of a Symposium, held in Brussels, Belgium, December 9–12, 1985*: 121–127. Springer Netherlands.

Buchanan, J., Sheader, M. & Kingston, P. 1978. Sources of Variability in the Benthic Macrofauna off the South Northumberland Coast, 1971–1976. *Journal of the Marine Biological Association of the United Kingdom* 58 (1): 191–209. https://doi.org/10.1017/S0025315400024498

Buyse, J., Hostens, K., Degraer, S. & De Backer, A. 2022. Long-term series demonstrate small-scale differences in trends within fish assemblages explained by climate variability. *Estuarine, Coastal and Shelf Science* 264: e107663. https://doi.org/10.1016/j.ecss.2021.107663

Cheng, C.H., Borsje, B.W., Beauchard, O., O'flynn, S., Ysebaert, T. & Soetaert, K. 2021. Small-scale macrobenthic community structure along asymmetrical sand waves in an underwater seascape. *Marine Ecology* 42 (3): e12657. https://doi.org/10.1111/maec.12657

Climate Prediction Center - Teleconnections: North Atlantic Oscillation.
Available from https://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao.shtml [accessed 2 May 2023].

Coates, D. & Vincx, M. 2010 Monitoring the effects of offshore wind farms on the softsubstratum macrobenthos: Year-1 Bligh Bank and Year-2 Thorntonbank. *In*: Degraer, S., Brabant, R. & Rumes, B. (eds) *Offshore Wind Farms in the Belgian Part of the North Sea: Early Environmental Impact Assessment and Spatio-Temporal Variability*. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine Ecosystem Management Unit. 184 pp. + annexes.

Coates, D.A., Deschutter, Y., Vincx, M. & Vanaverbeke, J. 2014. Enrichment and shifts in macrobenthic assemblages in an offshore wind farm area in the Belgian part of the North Sea. *Marine Environmental Research* 95: 1–12. https://doi.org/10.1016/j.marenvres.2013.12.008

Coates, D.A., Van Hoey, G., Colson, L., Vincx, M. & Vanaverbeke, J. 2015. Rapid macrobenthic recovery after dredging activities in an offshore wind farm in the Belgian part of the North Sea. *Hydrobiologia* 756: 3–18. https://doi.org/10.1007/s10750-014-2103-2

Coolen, J.W., Vanaverbeke, J., Dannheim, J., Garcia, C., Birchenough, S.N., Krone, R. & Beermann, J. 2022. Generalized changes of benthic communities after construction of wind farms in the southern North Sea. *Journal of Environmental Management* 315: e115173.
https://doi.org/10.1016/j.jenvman.2022.115173

C-Power. (n.d.). *Jackets*. Jackets.
Available from https://c-power.be/en/technology/Foundations [retrieved 14 March 2023].

Dannheim, J., Bergström, L., Birchenough, S.N., Brzana, R., Boon, A.R., Coolen, J.W., ... & Degraer, S. 2020. Benthic effects of offshore renewables: identification of knowledge gaps and urgently needed research. *ICES Journal of Marine Science* 77 (3): 1092–1108.
https://doi.org/10.1093/icesjms/fsz018

De Backer, A., Van Hoey, G., Coates, D., Vanaverbeke, J. & Hostens, K. 2014. Similar diversity-disturbance responses to different physical impacts: Three cases of small-scale biodiversity increase in the Belgian part of the North Sea. *Marine Pollution Bulletin* 84 (1): 251–262.
https://doi.org/10.1016/j.marpolbul.2014.05.006

De Backer, A., Polet, H., Sys, K., Vanelslander, B. & Hostens, K. 2019. Fishing activities in and around Belgian offshore wind farms: trends in effort and landings over the period 2006–2017.

In: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds) *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Marking a Decade of Monitoring, Research and Innovation*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 134 p.

Degraer, S., Brabant, R. & Rumes, B. 2011. *Offshore Wind Farms in the Belgian Part of the North Sea Selected Findings from the Baseline and Targeted Monitoring*. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models. Marine ecosystem management unit. 157 pp. + annex.

Degraer, S., Brabant, R., Rumes, B. (eds) 2013. *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Learning from the Past to Optimise Future Monitoring Programmes*. Royal Belgian Institute of Natural Sciences, Operational Directorate Natural Environment, Marine Ecology and Management Section. 239 pp.

Degraer, S., Carey, D.A., Coolen, J.W.P., Hutchison, Z.L., Kerckhof, F., Rumes, B. & Vanaverbeke, J. 2020. Offshore wind farm artificial reefs affect eco- system structure and functioning: A synthesis. *Oceanography* 33 (4):48–57. https://doi.org/10.5670/oceanog.2020.405

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*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 106 pp.

De Maersschalck, V., Hostens, V., Wittoeck, J., Cooreman, K., Vincx, M. & Degraer, S. 2006. *Monitoring van de effecten van het Thornton windmolenpark op de benthische macro invertebraten en de visfauna van zachte substraten-Referentietoestand*.

De Mesel, I., Kerckhof, F., Rumes, B., Norro, A., Houziaux, J.-S., Degraer, S. 2013. Fouling community on the foundations of wind turbines and the surrounding scour protection. *In*: Degraer, S., Brabant, R. & Rumes, B. (eds) *Environmental Impacts of Offshore Windfarms in the Belgian Part of the North Sea: Learning from the Past to Optimise Future Monitoring Programmes*: 123–137. Royal Belgian Institute of Natural Sciences, Operational Directorate Natural Environment, Marine Ecology and Management Section.

Desmit, X., Nohe, A., Borges, A.V., Prins, T., De Cauwer, K., Lagring, R., Van der Zande, D. & Sabbe, K. 2020. Changes in chlorophyll concentration and phenology in the North Sea in relation to de-eutrophication and sea surface warming. *Limnology and Oceanography* 65: 828–847. https://doi.org/10.1002/lno.11351

Dippner, J.W. & Kröncke, I. 2003. Forecast of climate-induced change in macrozoobenthos in the southern North Sea in spring. *Climate Research* 25 (2): 179–182.

Dippner, J.W., Möller, C. & Kröncke, I. 2014. Loss of persistence of the North Atlantic Oscillation and its biological implication. *Frontiers in Ecology and Evolution* 2: 57. https://doi.org/10.3389/fevo.2014.00057

Download Climate Timeseries: AMO SST: NOAA Physical Sciences Laboratory. Available from https://psl.noaa.gov/data/timeseries/AMO/ [accessed 2 May 2023].

Frid, C., Garwood, P. & Robinson, L. 2009. The North Sea benthic system: A 36 year time-series. *Journal of the Marine Biological Association of the United Kingdom* 89 (1): 1–10. https://doi.org/10.1017/S0025315408002956

Garcia-Soto, C. & Pingree, R.D. 2012. Atlantic Multidecadal Oscillation (AMO) and sea surface temperature in the Bay of Biscay and adjacent regions. *Journal of the Marine Biological Association of the United Kingdom* 92 (2): 213–234. https://doi.org/10.1017/S0025315410002134

Harris, V., Edwards, M. & Olhede, S.C. 2014. Multidecadal Atlantic climate variability and its impact on marine pelagic communities. *Journal of Marine Systems* 133: 55–69. https://doi.org/10.1016/j.jmarsys.2013.07.001

Hiscock, K., Tyler-Walters, H. & Jones, H. 2002. High level environment screening study for offshore wind farm developments - marine habitats and species project. *Report from the Marine Biological Association to The Department of Trade and Industry New & Renewable Energy Programme*. (AEA Technology, Environment Contract: W/35/00632/00/00). p. 156

CMEMS 2023. https://marine.copernicus.eu/ [accessed 2 May 2023].

https://c-power.be/ [accessed 2 May 2023].

Janssen, F., Huettel, M. & Witte, U. 2005. Pore-water advection and solute fluxes in permeable marine sediments (II): benthic respiration at three sandy sites with different permeabilities (German Bight, North Sea). *Limnology and Oceanography* 50: 779–792. https://doi.org/10.4319/lo.2005.50.3.0779

Johansen, P.O., Isaksen, T.E., Bye-Ingebrigtsen, E., Haave, M., Dahlgren, T.G., Kvalø, S.E., Greenacre, M., Durand, D. & Rapp, H.T. 2018. Temporal changes in benthic macrofauna on the west coast of Norway resulting from human activities. *Marine Pollution Bulletin* 128: 483–495. https://doi.org/10.1016/j.marpolbul.2018.01.063

Kaiser, M.J. & Spencer, B.E. 1996. The effects of beam-trawl disturbance on infaunal communities in different habitats. *Journal of Animal Ecology 65*, 348-358.

Karlson, A.M.L., Gorokhova, E., Elmgren, R. & Medina, M. 2014. Nitrogen fixed by cyanobacteria is utilized by deposit-feeders. *PLoS One* 9 (8): e104460. https://doi.org/10.1371/journal.pone.0104460

Karlson, A.M.L., Duberg, J., Motwani, N.H., Hogfors, H., Klawonn, I., Ploug, H., Svedén, J.B., Garbaras, A., Sundelin, B., Hajdu, S., Larsson, U., Elmgren, R. & Gorokhova, E. 2015. Nitrogen fixation by cyanobacteria stimulates production in Baltic food webs. *Ambio* 44 (S3): 413–426. https://doi.org/10.1007/s13280-015-0660-x

Keeley, N.B., Forrest, B.M. & Macleod, C.K. 2013. Novel observations of benthic enrichment in contrasting flow regimes with implications for marine farm monitoring and management. *Marine Pollution Bulletin* 66 (1–2): 105–116. https://doi.org/10.1016/j.marpolbul.2012.10.024

Kerr, R. 2000. North Atlantic Climate Pacemaker for the Centuries. *Science* 288 (5473): 1984-5. https://doi.org/10.1126/science.288.5473.1984

Kröncke, I., Dippner, J.W., Heyen, H. & Zeiss, B. 1998. Long-term changes in macrofaunal communities off Norderney (East Frisia, Germany) in relation to climate variability. *Marine Ecology Progress Series* 167: 25–36.

Kröncke, I., Reiss, H. & Dippner, J.W., 2013. Effects of cold winters and regime shifts on macrofauna communities in shallow coastal regions. *Estuarine Coastal and Shelf Science* 119: 79–90. https://doi.org/10.1016/j.ecss.2012.12.024

Künitzer, A., Basford, D., Craeymeersch, J.A., Dewarumez, J.M., Dörjes, J., Duineveld, G.C.A., Eleftheriou, A., Heip, C., Herman, P., Kingston, P., Niermann, U., Rachor, E., Rumohr, H. & De Wilde, P.A.J. 1992. The benthic infauna of the North Sea: species distribution and assemblages. *ICES Journal of Marine Science* 49 (2): 127–143. https://doi.org/10.1093/icesjms/49.2.127

Lefaible, N., Braeckman, U. & Moens, T. 2018. Effects of wind turbine foundations on surrounding macrobenthic communities. *In*: S. Degraer *et al.* (eds) Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Assessing and Managing Effect Spheres of Influence. *Memoirs on the Marine Environment*: 57–77.

Lefaible, N., Braeckman, U., Degraer, S., Vanaverbeke, J. & Moens, T. 2023. A wind of change for soft-sediment infauna within operational offshore windfarms. *Marine Environmental Research* 188: e106009. https://doi.org/10.1016/j.marenvres.2023.106009

Legrand, S. & Baetens, K. 2021. Dataset : Hydrodynamic Forecast for the Belgian Coastal Zone. Royal Belgian Institute of Natural Sciences. Available from https://metadata.naturalsciences.be/geonetwork/srv/eng/catalog.search#/metadata/BCZ_HydroState_V1

Lesutiene, J., Bukaveckas, P.A., Gasiunaite, Z.R., Pilkaityte, R., Razinkovas-Baziukas, A. 2014. Tracing the isotopic signal of a cyanobacteria bloom through the food web of a Baltic Sea coastal lagoon. *Estuarine, Coastal and Shelf Science* 138: 47–56. https://doi.org/10.1016/j.ecss.2013.12.017

Maar, M., Bolding, K., Petersen, J.K., Hansen, J.L.S. & Timmermann, K. 2009. Local effects of blue mussels around turbine foundations in an ecosystem model of Nysted off-shore wind farm, Denmark. *Journal of Sea Research* 62: 159–174. https://doi.org/10.1016/j.seares.2009.01.008

Marine spatial plan 2020. FPS Public Health. Available from https://www.health.belgium.be/en/marine-spatial-plan#article [accessed 24 February 2020).

Mavraki, N., Coolen, J.W., Kapasakali, D.A., Degraer, S., Vanaverbeke, J. & Beermann, J. 2022. Small suspension-feeding amphipods play a pivotal role in carbon dynamics around offshore man-made structures. *Marine Environmental Research* 178: e105664. https://doi.org/10.1016/j.marenvres.2022.105664

McKindsey, C.W., Archambault, P., Callier, M.D. & Olivier, F. 2011. Influence of suspended and off- bottom mussel culture on the sea bottom and benthic habitats: a review. *Canadian Journal of Zoology-Revue Canadienne De Zoologie* 89: 622–646. https://doi.org/10.1139/z11-037

McLean, M., Mouillot, D., Lindegren, M., Engelhard, G., Villéger, S., Marchal, P., Brind'Amour, A. & Auber, A. 2018. A climate-driven functional inversion of connected marine ecosystems. *Current Biology* 28 (22): 3654–3660. https://doi.org/10.1016/j.cub.2018.09.050

Philippart, C.J.M., van Aken, H.M., Beukema, J.J., Bos, O.G., Cadee, G.C. & Dekker, R. 2003. Climate related changes in recruitment of the bivalve *Macoma balthica. Limnology and Oceanography* 48: e2171e2185. https://doi.org/10.4319/lo.2003.48.6.2171

Poloczanska, E.S., Hawkins, S.J., Southward, A.J. & Burrows, M.T. 2008. Modelling the response of populations of competing species to climate change. *Ecology* 89 (11): 3138–3149. https://doi.org/10.1890/07-1169.1

QGIS Development Team. 2020. *QGIS Geographic Information System. Open Source Geospatial Foundation.*

Reiss, H., Degraer, S., Duineveld, G.C., Kröncke, I., Aldridge, J., Craeymeersch, J.A., Eggleton, J.D., Hillewaert, H., Lavaleye, M.S.S., Moll, A., Pohlmann, T., Rachor, E., Robertson, M., Vanden Berghe, E., van Hoey, G. & Rees, H.L. 2010. Spatial patterns of infauna, epifauna, and demersal fish communities in the North Sea. *ICES Journal of Marine Science* 67 (2): 278–293.

Rumes, B. & Brabant, R. 2021. Offshore renewable energy development in the Belgian Part of the North Sea – 2021. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds) *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Attraction, Avoidance and Habitat Use at Various Spatial Scales. Memoirs on the Marine Environment*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 104 pp.

Rumes, B., Brabant, R. & Vigin, L. 2022. Offshore renewable energy in the Belgian Part of the North Sea. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds) *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*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 106 pp.

Suikkanen, S., Laamanen, M. & Huttunen, M. 2007. Long-term changes in summer phytoplankton communities of the open northern Baltic Sea. *Estuarine, Coastal and Shelf Science* 71 (3–4): 580–592. https://doi.org/10.1016/j.ecss.2006.09.004

Team, R.D.C. 2009. *A Language and Environment for Statistical Computing*.

Timmermann, K., Norkko, J., Janas, U., Norkko, A., Gustafsson, B.G. & Bonsdorff, E. 2012. Modelling macrofaunal biomass in relation to hypoxia and nutrient loading. *Journal of Marine Systems* 105–108: 60–69. https://doi.org/10.1016/j.jmarsys.2012.06.001

Tomczak, G. & Goedecke, E. 1962. *Monatskarten der Temperatur der Nordsee: dargestellt für verschiedene Tiefenhorizonte*. Deutsches Hydrographisches Institut.

Toussaint, E., De Borger, E., Braeckman, U., De Backer, A., Soetaert, K. & Vanaverbeke, J. 2021. Faunal and environmental drivers of carbon and nitrogen cycling along a permeability gradient in shallow North Sea sediments. *Science of the Total Environment* 767: e144994. https://doi.org/10.1016/j.scitotenv.2021.144994

Van Dalfsen, J.A., Essink, K., Madsen, H.T., Birklund, J., Romero, J. & Manzanera, M. 2000. Differential response of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. *ICES Journal of Marine Science* 57 (5): 1439–1445. https://doi.org/10.1006/jmsc.2000.0919

Van Hoey, G., Degraer, S. & Vincx, M. 2004. Macrobenthic community structure of soft-bottom sediments at the Belgian Continental Shelf. *Estuarine, Coastal and Shelf Science* 59: 599–613. https://doi.org/10.1016/j.ecss.2003.11.005

Van Lancker, V., Moerkerke, G., Du Four, I., Verfaillie, E., Rabaut, M. & Degraer, S. 2012. Fine-scale geomorphological mapping of sandbank environments for the prediction of macrobenthic occurrences, Belgian part of the North Sea. *In*: *Seafloor Geomorphology as Benthic Habitat*: 251–260. https://doi.org/10.1016/B978-0-12-385140-6.00014-1

Voet, H.E.E., Vlaminck, E., Van Colen, C., Bodé, S., Boeckx, P., Degraer, S., Moens, T., Vanaverbeke, J. & Braeckman, U. 2023. Organic matter processing in a [simulated] offshore wind farm ecosystem in current and future climate and aquaculture scenarios. *Science of the Total Environment* 857: e159285. https://doi.org/10.1016/j.scitotenv.2022.159285

Wieking, G. & Kröncke, I. 2001. Decadal changes in macrofauna communities on the Dogger Bank caused by large-scale climate variability. *Senckenbergiana maritima* 31 (2): 125–141.

Wikner, J. & Andersson, A. 2012. Increased freshwater discharge shifts the trophic balance in the coastal zone of the northern Baltic Sea. *Global Change Biology* 18 (8): 2509–2519. https://doi.org/10.1111/j.1365-2486.2012.02718.x

Wood, S.N. 2006. *Generalized Additive Models; An Introduction with R*.

Ysebaert, T., Hart, M. & Herman, P. M. 2009. Impacts of bottom and suspended cultures of mussels Mytilus spp. on the surrounding sedimentary environment and macrobenthic biodiversity. *Helgoland Marine Research* 63 (1): 59–74.

Zupan, M., Rumes, B., Vanaverbeke, J., Degraer, S. & Kerckhof, F. 2023. Long-Term Succession on Offshore Wind Farms and the Role of Species Interactions. *Diversity* 15 (2): e288. https://doi.org/10.3390/d15020288

Zuur, A.F., Ieno, E.N. & Elphick, C.S. 2010. A protocol for data exploration to avoid common statistical problems. *Methods in ecology and evolution* 1 (1): 3–14. https://doi.org/10.1111/j.2041-210X.2009.00001.x

CHAPTER 3

EFFECTS OF OFFSHORE WIND FARMS ON THE ECOLOGY OF FLATFISH: A CASE STUDY ON PLAICE *PLEURONECTES PLATESSA*

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Abstract

The ecological effects of offshore wind farms (OWFs) on adult plaice were investigated in terms of their spatial distribution, diet, and movements. Spatial distribution was studied on the turbine and wind farm scale using a combination of visual diving transects and beam trawl samples. A trophic analysis study combined gut content analysis with a biomarker approach (fatty acid analysis) to obtain diet information on both short and longer time scales. Condition was studied by calculating several morphometric (i.e., Fulton's K index) and organosomatic condition indices (fullness-, gonadosomatic-, hepatosomatic- and digestive-somatic index) while the movements of plaice were investigated trough an acoustic telemetry study. Results show that plaice is affected by the presence of OWFs, with the artificial hard substrate within the OWFs providing an important habitat for individual plaice by increasing prey availability through the artificial reef effect. Furthermore, our findings suggest that OWFs may act as a refuge for plaice, potentially mitigating direct fishing mortality. We did not find any evidence that the increased prey availability leads to a better condition of plaice, but our sample size might have been too small to detect differences in condition. Results from this study suggest that existing OWFs likely enhance ecological production for plaice. It remains to be investigated whether this translates to spillover effects into adjacent areas where fishing is permitted and how such effects could influence fisheries given the anticipated large-scale expansion of offshore renewable energy.

1. Introduction

The installation of hard substrates in soft sediment environments, which is associated with the development of offshore wind farms (OWFs), can cause changes that may affect local fauna (Inger *et al.* 2009; Langhamer & Wilhelmsson 2009; Lindeboom *et al.* 2011; Raoux *et al.* 2017). In addition, the energy emissions from turbines and cables (such as noise, electromagnetic fields, and light) can impact the behavior of present fauna and potentially reduce their fitness if the effect is significant (Bailey *et al.* 2014; Hutchison *et al.* 2020). However, many fish species and other megafauna are drawn to the scour protection layer (SPL) and turbine foundations in OWFs through the artificial reef effect (Langhamer & Wilhelmsson 2009; Andersson & Öhman 2010; Krone *et al.* 2017; Degraer *et al.* 2020).

The epifauna growing on the hard substrate increases food availability for higher trophic levels, and the rocks and foundations provide a complex habitat that serves as shelter for different organisms. The attraction of certain fish species, such as cod *Gadus morhua*, to artificial structures such as oil and gas platforms, shipwrecks, and OWFs, has been well documented (Reubens *et al.* 2011; Bergström *et al.* 2013; Krone *et al.* 2017; Wright *et al.* 2020). However, the effects of turbine foundations and scour protection on typical soft-sediment fish species (e.g., flatfish) are not yet well understood.

Two distinct hypotheses have been proposed to explain the increased fish abundance around artificial reefs. According to the attraction hypothesis, fish are either attracted to or redistributed around the structures (Bohnsack & Sutherland 1985; Lindberg 1997; Wilson *et al.* 2010; Bergström *et al.* 2013). In contrast, the production hypothesis proposes that the presence of artificial structures increases the carrying capacity of the area, leading to a greater abundance and biomass of fish (i.e., ecological production) within its boundaries. However, few studies have been able to provide evidence of fish production within OWFs (Wilhelmsson *et al.* 2006; Reubens *et al.* 2014; Mavraki *et al.* 2021). The attraction and production hypothesis are not mutually exclusive and are considered to be two extremes on a continuous scale (Osenberg *et al.* 2002). The potential effects of hard substrates within OWFs as artificial reefs likely vary depending on the species and life stage in relation to the SPL material, rock density and surface area, as well as the water depth, location and foundation type (Brickhill *et al.* 2005; Glarou *et al.* 2020).

Additionally, OWFs may also fulfill a similar role as marine protected areas (MPAs) for certain (target) species, as these concession areas are often closed to any commercial fishing activities and can as such be considered as no-take zones (Steins *et al.* 2021). No-take zones can protect fish species, especially those that are targeted by fisheries, and enhance fish biomass, which might even lead to a spillover in nearby fishing areas (Langhamer 2012; Florin *et al.* 2013). Such spillover effects, resulting from the combined artificial reef and refuge effects, have been predicted through modelling approaches (Raoux *et al.* 2017; Halouani *et al.* 2020), but in-situ studies were not able to confirm this yet.

This chapter summarizes the findings of different papers (Buyse *et al.* 2021, 2023; Buyse 2023) that were consolidated within the framework of a PhD study on the effects of OWFs on the ecology of the plaice *Pleuronectes platessa*, a commercial flatfish species, with the following research questions:

- What are the effects of OWFs on the spatial distribution of plaice at the turbine and wind farm scale?
- Which small- and large-scale movements does plaice perform in relation to OWFs?
- What are the effects of OWFs on the diet and condition of plaice?
- Does plaice production occur in OWFs?

Integrating knowledge on the effects of an OWF on plaice' spatial and temporal distribution, diet and condition and spatial (small scale) movements enables us to discuss whether ecological production is occurring within OWFs, which would indirectly imply that OWFs have a protection potential for this species. Fish production is considered likely if we can demonstrate attraction towards and a high association to the hard substrates with a diet consisting mainly of colonizing prey species, a higher food availability (gut fullness), a high residency and increased fish abundances, condition and size within the wind farm area.

2. Material and methods

2.1. Spatial distribution

The spatial distribution of plaice was examined at two different scales, namely the turbine and wind farm scale, in the

C-Power and Belwind offshore wind farms (OWFs) to determine if an attraction effect towards the wind farm and hard structures could be detected (Fig. 1). At the wind farm scale, we utilized beam trawl samples in a Before-After/Control-Impact (BACI) design (Vandendriessche *et al.* 2015; De Backer *et al.* 2022). The samples were collected during the annual autumn monitoring campaigns as part of the WinMon.BE program (period 2004–2019 for C-Power and 2008–2019 for Belwind). The potential attraction of plaice at the turbine scale was investigated through visual diving transects over the scour protection layer (SPL) and the immediate surrounding sand around turbines selected at

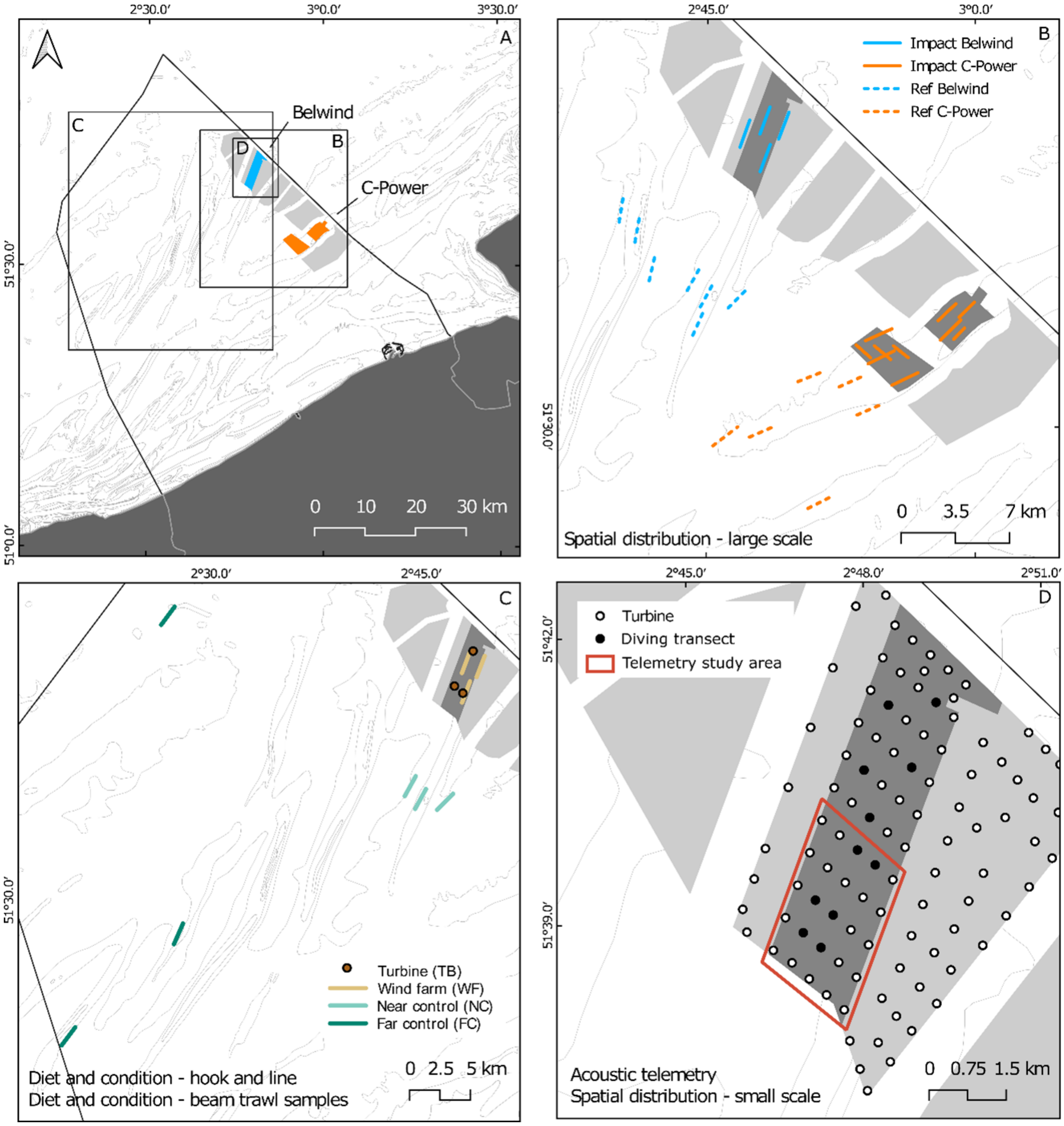

Figure 1. Map showing (**A**) the location of Belwind and C-Power within the Belgian EEZ, (**B**) the locations of the beam trawl samples (Before/After-Control/Impact design) used to study the large scale spatial distribution of plaice, (**C**) the beam trawl and hook-and-line sampling locations for the diet and condition study and (**D**) the turbines where diving transects were carried out for the small-scale spatial distribution study and the area within Belwind where the acoustic telemetry study took place.

random in the Belwind wind farm. For a more detailed explanation of the sampling design and methods, the reader is referred to Buyse *et al.* (2022).

2.2. Diet and condition

The effects of OWFs on the diet and condition of plaice were investigated in four different groups: the turbine group (TB), comprising fish caught by hook-and-line on the SPL of three randomly chosen turbines in Belwind; and the wind farm group (WF), the near control group (NC) and the far control group (FC), comprising fish caught by an 8 m-beam trawl (20 mm mesh size) in between the turbines of Belwind (± 250 m from the turbines), close to the wind farm area, and further away from the wind farm area respectively. A total of 72 fish (18 fish per sample group, six replicates per turbine or haul) were randomly selected and their gut, ovaries or testes, liver, and otoliths were stored for further analyses. The lengths of the fish ranged from 17.5 to 36.5 cm ($\bar{x} = 25.8 \pm 4.3$ SD). Additionally, muscle tissue samples were taken from each fish, and their length and weight before and after evisceration were determined.

The diet composition was studied on a short temporal scale (< 10 h) using gut content analysis, while fatty acid and stable isotope analyses of the muscle samples provided insight into the diet on a longer time scale. A relative abundance method was used to determine the contribution of each prey item to the diet (Amundsen & Sánchez-Hernández 2019), whereby the volume of each prey item for both stomach and gut was visually estimated and expressed as a percentage of a total of 100%. Relative abundances from the stomach and gut were summed to provide a more comprehensive picture of the diet. The stomach fullness index was calculated for each fish based on the weights of the prey relative to the weight of the fish (Mahesh *et al.* 2018). Different morphometric indices (Fulton and Fulton with eviscerated weight) and condition indices (gonadosomatic, digestive-somatic, and hepatosomatic index) were calculated to investigate whether a difference in diet also led to differences in condition or fecundity. Linear mixed models were used, including haul or turbine as random factors. The sampling design, methods, and analyses are thoroughly described in Buyse *et al.* (2023).

2.3. Residency, site fidelity and small-scale movements

In order to examine the presence and spatial movements of plaice within and in relation to OWFs, 31 fish were tagged with acoustic transmitters and subsequently released within a network of acoustic receivers. This study was conducted between May 2020 and August 2021, in the southern region of the Belwind OWF (approx. 3.5–5.5 km^2). Over the course of the study, three different receiver designs were utilized during three consecutive time periods (period 1: 15/05/2020–11/10/2020; period 2: 14/10/2020–22/02/2021; period 3: 25/02/2021–11/07/2021). During the initial period, to obtain a high resolution 2D-positioning, six receivers were arranged in a circular formation around three turbines (B9, C8 and D9), each situated at a distance of 150 meters from the turbines. For the remaining two periods, the receivers were repositioned to cover the largest possible area (approx. 5.5 km^2) whilst still maintaining overlapping detection ranges. The acoustic transmitters emit unique signals at random intervals, which are detected by receivers when the fish swim in their close proximity. Using this presence data, a residency index was computed for the May–October feeding period, reflecting the degree of association between the fish and the OWF area. Triangulation (utilizing the yaps package, available on http://github.com/baktoft/yaps) was employed to estimate the 2D positions of fish that remained in the study area for at least 20 days during the first study period. This facilitated the examination of small-scale movement patterns of plaice in relation to the hard substrates (Baktoft *et al.* 2017, 2019). The estimated positions were utilised to compute the distance between each calculated position and the nearest turbine.

To gain a better understanding of the spatial distribution of fish in the vicinity of the turbines (< 150 m), the number of detections with available positions was calculated per square metre for several distance intervals (0–20 m per 5 m, 20–50 m per 10 m, and 50–150 m per 50 m). In order to investigate the existence of a diurnal pattern in the distance between fish and the hard substrate, information on sunlight times was used to generate a categorical variable *light* consisting of four different levels: dawn, day, dusk, and night. Nautical dusk and dawn times (when the sun is 12° below the horizon) were used to differentiate between dusk/dawn and night, while sunrise and sunset times were used to distinguish between dusk/dawn and day. A linear mixed model (LMM) with a Gaussian distribution was employed to fit the distance of the fish to the nearest turbine as a response variable. The model included light as a fixed effect and fish ID as a random variable to account for the variance between the different fish. The final model that was fitted was: Distance to the turbine ~ *light* + (1 | fish ID).

For a comprehensive overview of the acoustic network, tagging methodology, and data analysis techniques, the reader is referred to Buyse (2023).

3. Results

3.1. Spatial distribution

Over 190 beam trawl samples, 5186 plaice individuals were caught ($\overline{x}=0.18 \pm 0.16$ ind. 100 m^{-2}) across a trawled area of 3.37 km^2 within the Belwind and C-Power offshore wind farms (OWFs). The BACI analysis conducted at the wind farm level yielded contrasting results (Fig. 2). A clear wind farm effect was observed for C-Power

Figure 2. A. Plaice abundance in beam trawl samples (approx. 200 m from the turbines) for C-Power and Belwind OWFs in reference and impact areas (period 2004–2019 and 2008–2019, respectively). The vertical dark grey line indicates the before and after construction period as considered in this study, which also coincides with the moment that fisheries activities were suspended for the second and third phase of C-Power and Belwind (the area around the 6 GBFs in C-Power (first construction phase) was closed in 2008). LOESS-smoothers with a span of 0.7 were fitted to the data to facilitate interpretation. Orange and blue bands indicate the standard error 95% confidence intervals for the impact and reference areas, respectively. **B**. Plaice abundances along the visual diving transects with estimated marginal means for the final GLMM showing the number of plaice per 100 m^2 for both habitat types (SPL = scour protection layer).

($p=0.0008$, $Z=-3.35$), which resulted in plaice abundances almost 4.5 times higher within the impacted OWF area after construction (0.27±0.09 ind. 100 m^{-2}, $CI_{0.95}$[0.13–0.53]) compared to before construction (0.06±0.02 ind. 100 m^{-2}, $CI_{0.95}$[0.03–0.12]). No significant difference in plaice abundance before and after construction was found for reference samples ($p=0.13$, $Z=-1.53$). Additionally, a significant difference in plaice abundances between reference and impact samples after construction was observed ($p=0.03$, $Z=2.22$), while this was not the case in samples taken before construction ($p=0.05$, $Z=-1.93$). On the other hand, no wind farm effects were observed for Belwind ($p=0.25$, $Z=1.16$).

During the 21 visual diving transects, a total of 31 plaice individuals were observed. Among these, 23 were found on the SPL habitat (total searched area = 1028 m^2), while only eight were found on the open sand surrounding the SPL (total searched area = 1436 m^2). As a result, the abundance of plaice was four times higher on the sandy patches between the rocks of the SPL (2.08±0.55 ind. 100 m^{-2}, $CI_{0.95}$[1.24–3.49]) compared to the surrounding sand (0.53±0.20 ind. 100 m^{-2}, $CI_{0.95}$[0.25–1.11]). It was observed that fish were not uniformly distributed on the SPL habitat, and were mainly found where the rock density was relatively low, as opposed to locations where rocks were closely stacked on top of each other.

3.2. Diet and condition

The lengths of the 72 plaice individuals differed significantly among the sample groups, except between the two control groups. The largest fish were found in the TB and WF groups (median length TB: 30.2 cm, WF: 26.2 cm, NC: 23 cm, FC: 23 cm; $F_{3,68}=21.8$, $p<0.001$). The sex ratio also varied across the different groups, with a strong dominance of females in the TB (0.88) and WF group (0.83), while sexes for the control groups were more evenly distributed (NC: 0.67, FC: 0.5). However, the age of the fish did not differ among the sample groups ($\bar{x}=3.42\pm1.77$ SD; $F_{3,68}=0.62$, $p=0.60$).

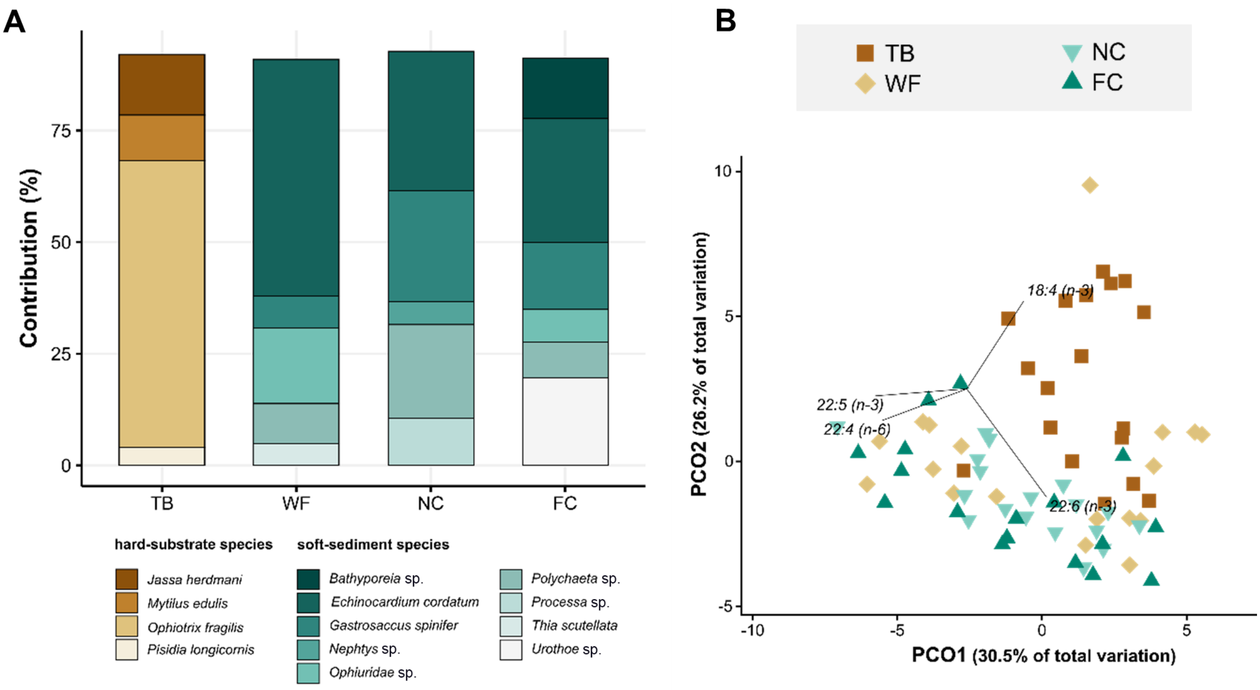

Figure 3. A. Principal coordinates analysis (PCO) results of combined stomach and intestine content using relative abundances of prey items per sample group. The overlay vector shows prey species with a multiple correlation > 0.4. **B**. Contribution of the most abundant prey species (cut-off level 90%) in stomachs and intestines per sample group according to a SIMPER analysis.

The gut content analysis revealed that plaice caught on the SPL had a distinctive diet consisting of typical hard substrate prey species such as *Mytilus edulis*, *Ophiothrix fragilis*, *Jassa herdmani*, and *Pisidia longicornis* ($p_{Permanova}$, pseudo-$F_{3,67}=3.15$, $p=0.005$) (Fig. 3). In contrast, the diet of fish caught on the sand in between the turbines (WF) and outside the Belwind OWF (NC and FC) consisted of prey usually found in soft sediments such as *Echinocardium cordatum* and *Gastrosaccus spinifer*. The fatty acid analysis showed differences in FA profiles between fish caught right next to the turbines (TB) and in between the turbines (WF) versus the control areas (NC and FC).

The fullness index and the Fulton's K index based on the total weight of fish were highest for the turbine group (TB) and decreased as the distance from the hard substrate increased (Fig. 4). However, significant differences were not found between the impact and control groups, likely due to high variability in the data. Additionally, no wind farm effect was observed for the gonadosomatic and hepatosomatic indices, but the digestive-somatic index was significantly higher for fish in the turbine group when compared to the two control groups.

3.3. Residency, site fidelity and small-scale movements

Out of the 31 fish that were tagged, many were found in the Belwind OWF for extended uninterrupted periods after being released during the summer of 2020, and most of them were still present at the beginning of autumn (Fig. 5). Three fish (ID 9257, 9258 and 9262) were only detected for a few days after their release. Eleven individuals (35%) were (re) detected after one year in spring 2021, and seven fish (20%) were detected in the wind farm until the last two weeks of the study. Fewer detections were registered in the study area during the winter months. Several fish (ID 9250, 9255, 9256, 9260, 9269, 9275, 9277, 9280 and 9284) were absent for a long consecutive period, which coincided with the spawning period of plaice (December–March), after which they returned to the study area during the spring of the following year. The residency for fish that were present in the OWF for at least 20 days during the first period (May–October 2020, n = 24) ranged from 0.09–1, with an average residency of 0.78 ± 0.29. Overall, 70% of the fish had a residency index of at least 0.75.

The 21 fish individuals that were present within the study area for at least 20 days during the first monitoring period, were observed at a mean distance of 92 ± 48 m from the turbines with most detections occurring on the sand directly surrounding the SPL (±25 m from the turbine) with a second peak of detections around ±90 m from the turbines (Figs 5–6). The fish positions showed a diurnal pattern in distance from the turbine ($\chi^2=6251.6$, $p<0.001$). Fish were located closer to the hard substrate during the day (84.6 m, $CI_{0.95}$[64.6–105]) and at dawn (80.6 m, $CI_{0.95}$[60.6–101]), compared to dusk (109.8 m, $CI_{0.95}$[89.7–130]) and at night (101.0 m, $CI_{0.95}$[80.9–121]). Most of the detections per m^2 were located on the SPL during daytime, while at night (21:00–01:00 UTC), almost no fish were located close the turbines (0–10 m) (Fig. 6).

4. Discussion

To summarise, the results from the different studies indicate that adult plaice are affected in terms of their spatial distribution, diet and movements by the presence of offshore wind farms (OWFs). The artificial hard substrate structures within the OWFs appear to be an important habitat for individual plaice, providing increased prey availability through the artificial reef effect. Moreover, our findings suggest that OWFs may act as a refuge for plaice, as indicated by the skewed sex ratio and larger fish size, potentially mitigating direct fishing mortality. These collective results suggest that OWFs may enhance ecological production for plaice, but it remains to be investigated whether this translates to spillover effects into adjacent areas where fishing is permitted and if such effects could benefit fisheries.

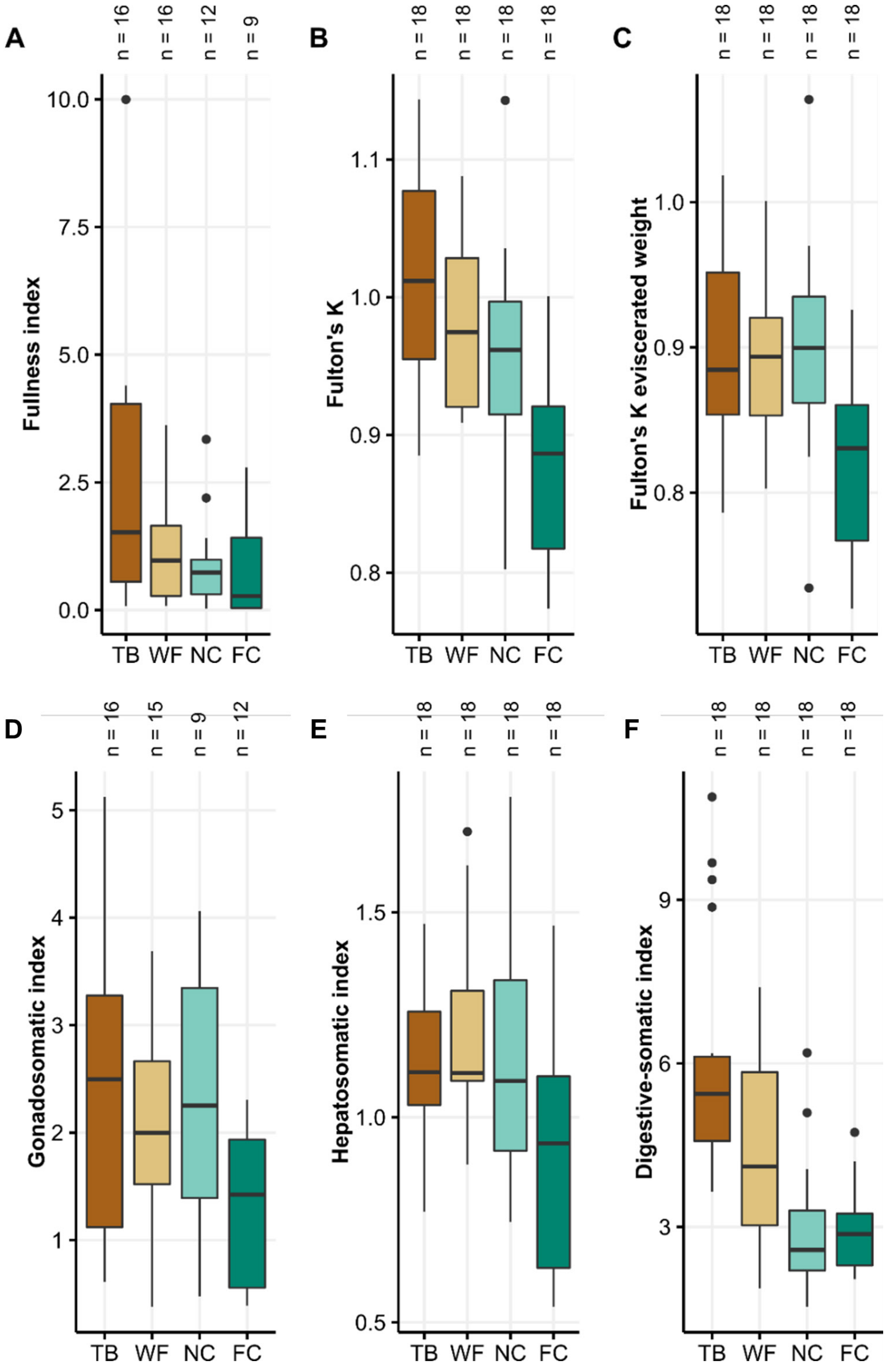

Figure 4. Boxplots of (**A**) fullness index of non-empty stomachs (n = 53), (**B**) Fulton's K condition index calculated with fish total weight, (**C**) Fulton's K condition index calculated with fish eviscerated weight, (**D**) gonadosomatic (GSI), (**E**) hepatosomatic (HSI) and (**F**) digestive-somatic (DSI) index for each sample group with n representing the number of samples used to calculate the index. For the calculation of the GSI, only females were used.

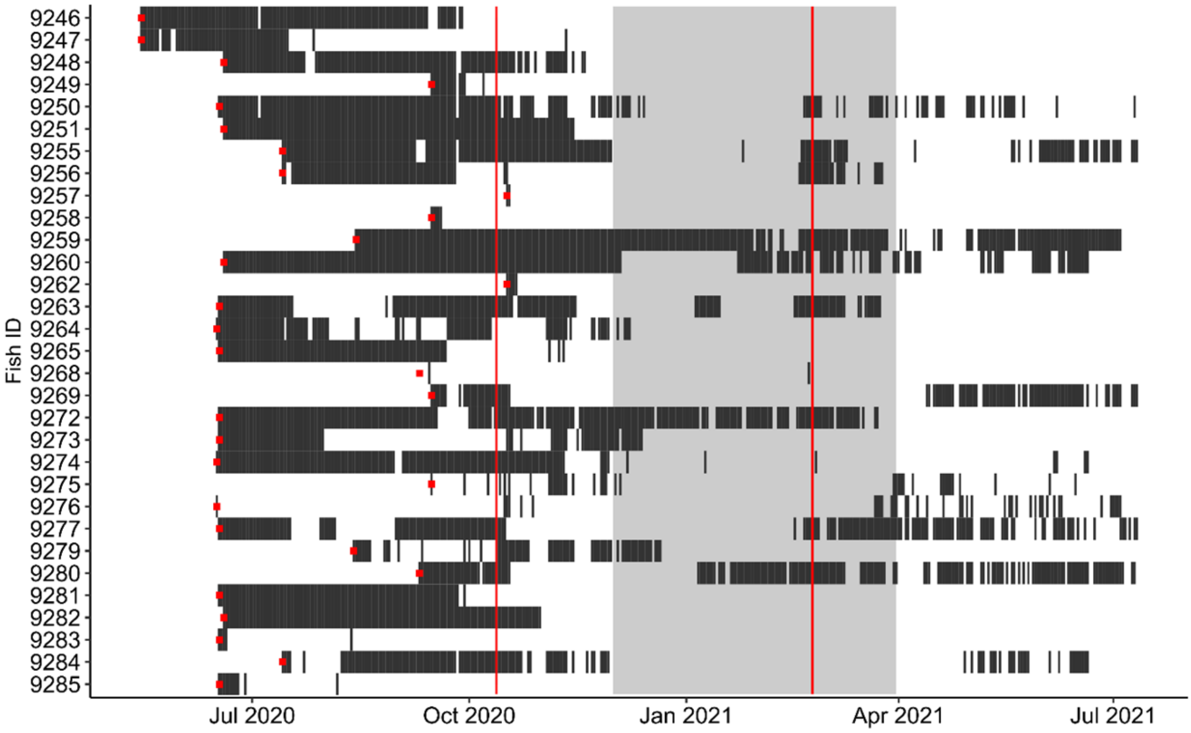

Figure 5. Daily presence of the 31 tagged plaice over the study period (15/05/2020–11/07/2021) in Belwind OWF. Red squares indicate the tagging and release date of the fish, while the red vertical lines show the change in receiver array design., The grey box represents the yearly spawning period for plaice in the southern North Sea (December–March with a peak in January). A fish was considered to be present in the study area if it was detected at least two times on that particular day.

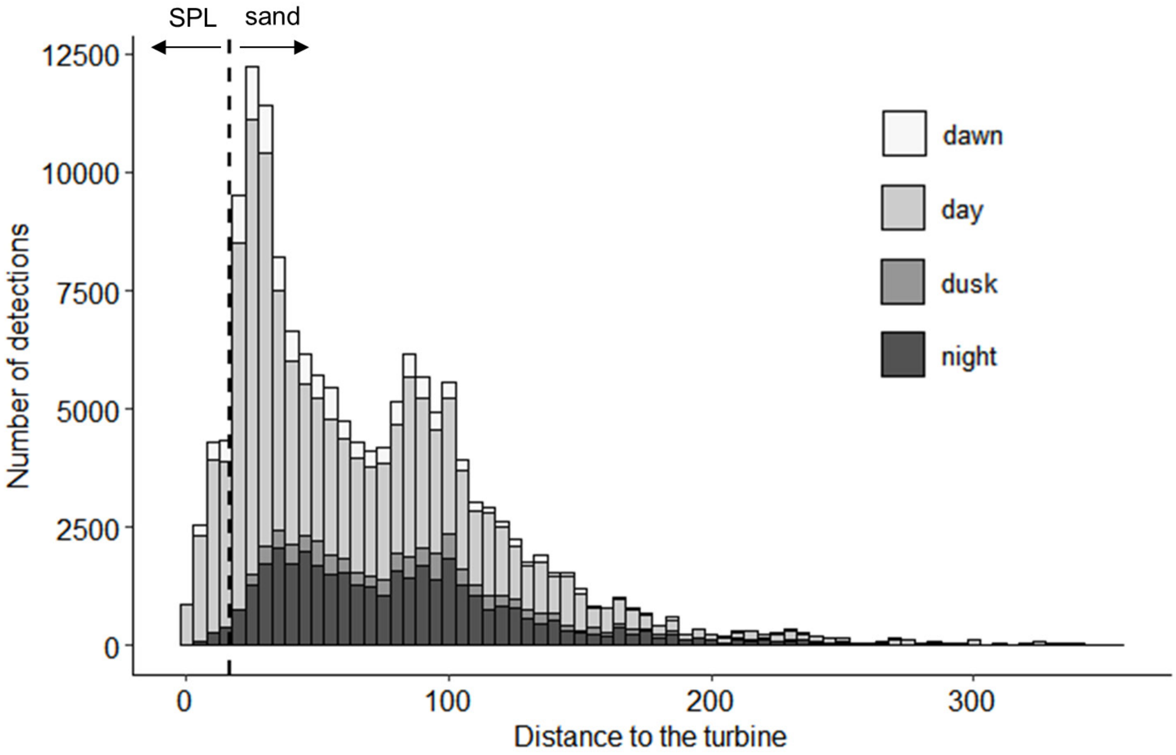

Figure 6. Number of detections over the distance to the nearest turbine for the 21 fish for which positions were estimated. The dashed vertical line indicates the average width (16.5 m from the turbine) of the scour protection layer (SPL) based on the design plans.

Our visual census revealed that plaice is attracted to the habitat formed by the scour protection around turbine foundations. Moreover, the trophic analysis indicated that fish in the vicinity of the turbine (TB) had fuller digestive tracts (higher digestive-somatic index) and consumed primarily hard substrate prey, whereas plaice farther from the turbine consumed mainly soft-sediment prey. This difference was not only observed in

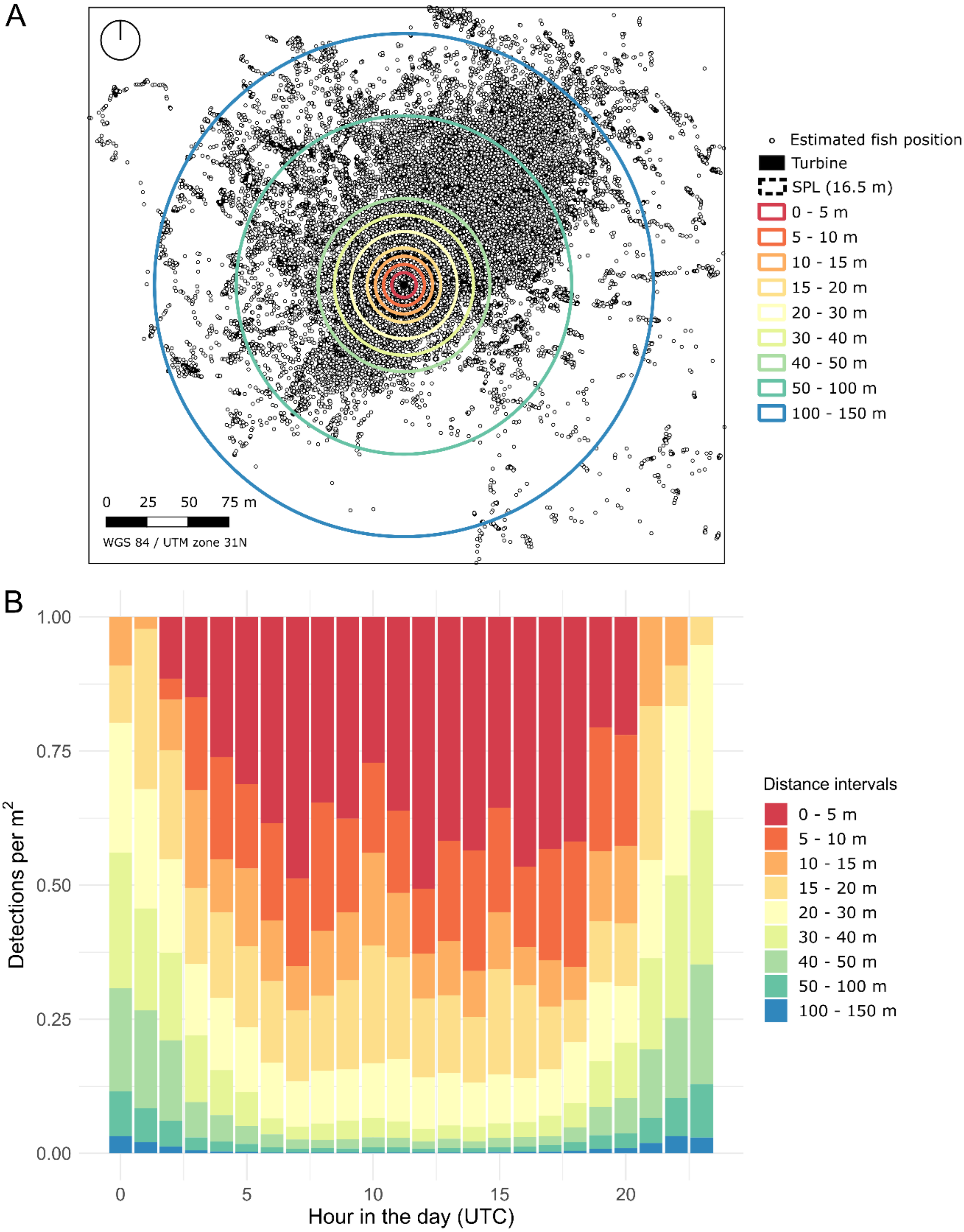

Figure 7. A. Estimated fish positions around turbine B9 in the Belwind OWF from June till October 2020. **B**. Relative number of fish detections per m^2 per hour around turbines B09, C08 and D09 for each distance interval (between 0 and 150 m).

short-term diet analyses but also in assimilated fatty acids, indicating a longer-term diet shift. This suggests that the attraction observed is likely due to the increase in prey availability on and near the hard substrates, known as the artificial reef effect (Bohnsack & Sutherland 1985; Petersen & Malm 2006). While shelter may be another reason for attraction, our tagging study indicated that most detections were located just outside the SPL, making this explanation unlikely for plaice. The tagging study results further support the hypothesis that the observed attraction is linked to feeding behaviour, as a diurnal pattern was found in the distance of the fish to the turbine, with fish being closer to it during the day and at dawn than during night hours and at dusk. During daylight hours, the highest density of detections was even found in the immediate vicinity of the turbine (< 15 m). As plaice is a daytime feeder that relies on its sight to locate and identify prey (Gibson *et al.* 2015), these findings suggest that they undertake 'feeding excursions' towards the SPL and return to the soft sediment to rest. By doing so, they can benefit from the increased prey availability near the hard substrates while still being able to bury themselves into the sand directly surrounding the SPL.

In contrast to our findings, several studies conducted in other offshore wind farms did not report any attraction towards the hard substrates (Krone *et al*. 2017; van Hal *et al.* 2017). During the diving transects, plaice were only seen resting on the sandy patches between the rocks of the SPL, not on top of the rocks. Hence, we speculate that plaice can benefit from the increased prey availability offered by the hard substrates, but only if soft sandy sediment is present in the immediate vicinity. Consequently, we can propose some suggestions for modifying the SPL to improve its ecological role for plaice and flatfish species in general. Since the existence of soft sediment is likely a crucial factor for their attraction, adapting the rock density of the SPL to allow for sand patches between the hard substrate will considerably increase prey accessibility for plaice and potentially other soft sediment fish species.

In addition to an artificial reef effect, our findings suggest that OWFs can also have a refuge effect on plaice. As commercial fishing activities within OWFs are often prohibited, they can be considered as no-take zones, similar to marine protected areas (MPAs) (Ashley *et al.* 2014). Previous studies have reported an increase in fish size, abundance, and total biomass within MPAs compared to adjacent fished areas (Di Franco *et al.* 2009; Florin *et al.* 2013; Guidetti *et al.* 2014; Félix-Hackradt *et al.* 2018). Our study revealed that there is a higher number of plaice present within the C-Power wind farm in between the turbines and that fish within the Belwind wind farm are larger, which indicates the existence of a refuge effect. Additionally, our findings show a higher female-to-male ratio within samples taken in Belwind. Plaice is a sexual dimorphic species, with females growing faster and for a longer time than males. In undisturbed populations, this results in a dominance of male individuals among smaller individuals and a dominance of females among larger individuals, with an even sex ratio for the total population. A higher catchability of larger (and thus more likely female) individuals in fished areas can lead to a decrease in fish size and a higher proportion of males (van Walraven *et al.* 2010; Florin *et al.* 2013). The increased female-to-male ratio of plaice found within Belwind might therefore be a consequence of the cessation of fishing activities. Although the sample size used in the trophic analysis study was sufficient to test differences among groups in diet composition and fatty acid profiles, it was likely too low to study differences in life-history traits and demographic variables (e.g., size, age, sex ratio) due to the inherent large variation within such data. A follow-up study that specifically aims at investigating the existence of refuge effects using larger sample sizes is needed to confirm the findings presented here. In general, monitoring programmes should not only focus on fish abundances alone, as research within MPAs has shown that effects on fish size and age (and thus biomass) are often easier to detect than changes in abundances (Florin *et al.* 2013).

The observed attraction effect is evident, but does it also result in ecological production? There are various proposed mechanisms through which the introduction of artificial hard substrates into soft sediment environments can lead to production (Wilson *et al.* 2001; Mavraki *et al.* 2021). These mechanisms include increasing food availability or feeding efficiency, providing shelter from predation, offering recruitment habitat for settling individuals, and freeing space in the natural habitat for other fish by attracting fish to the artificial reefs (Randall 1963; Stone *et al.* 1979; Bohnsack 1989). We have demonstrated that plaice feeds on the prey typically found on the hard substrates in OWFs, which could result in ecological production for this species. Moreover, prey on rocks or turbine foundations might be more easily accessible to plaice compared to prey buried in the sand, potentially increasing their feeding efficiency. Although we did not observe a higher number of plaice between the turbines in the Belwind wind farm compared to outside, there could still be a higher fish abundance when the entire wind farm area is considered. The tagging study suggests that these fish may be located in the sandy environment closer to the hard substrate. Therefore, the total fish biomass within the OWF area could be higher, without being detected by beam trawling at a distance of approximately 250 m from the turbines. In C-Power, where scour protection is absent around most of the turbines, the aggregation effect near the turbines is likely smaller, and thus, the attraction effect is more visible in the beam trawl samples with higher plaice abundance between the turbines compared to reference areas. These results underscore the importance of considering different spatial scales, e.g., turbine and wind farm, when investigating spatial distribution patterns in OWFs, as they can influence each other. Additionally, they indicate that it is crucial to account for environmental and physical conditions, foundation type, and surrounding fishing pressure when studying OWF effects on fish.

Whether a closed area (i.e., excluding fisheries) can efficiently enhance the biomass of a certain fish species depends largely on its mobility (Shipp 2003). A species that constantly moves in and out of the protected area will profit less from a spatial closure than a relatively stationary species (Miethe *et al.* 2010). The tagging study indicated that plaice has a high residency within the OWF area with most individuals showing a preference for a single turbine during the feeding period. However, plaice also undertakes large-scale spawning migrations during winter, which makes them vulnerable to fishing mortality on the spawning areas and during the migration events itself (Gibson *et al.* 2015). Therefore, any protection effect offered by the OWF areas might equally be nullified at a later stage.

Research has shown that successfully managed MPAs can support (small-scale) fisheries through a reduction of the fishing mortality on commercial species and enhancing their production through an increase in food availability (Friedlander *et al.* 2007; Florin *et al.* 2013; Guidetti *et al.* 2014). In such a case, the combined effect of offering refuge and enhancing production leads to a spillover of adults or juveniles into adjacent fishable areas, thereby increasing fish biomass (Davies *et al.* 2021). It has been estimated that spillover can increase catches outside an OWF with 7% (Halouani *et al.* 2020). However, empirical evidence showing the existence of spillover from OWFs is still lacking. Although BACI designs are effective in identifying impacts, a gradient design that involves taking control samples along a distance gradient from the impacted area may be more suitable for detecting spillover effects (Methratta 2020; De Backer *et al.* 2022). This is because one would expect a gradient in effect size with distance from the OWF.

This study addressed some important knowledge gaps on the ecological effects of OWFs on adult plaice at the scale of individuals and single wind farms and we could demonstrate that ecological production at the adult level on wind farm scale is very likely (Fig. 8). It is clear that individual adult fish profit from the increased prey availability around the hard substrates in an OWF. However, this might not cause any changes on the fish population level, even though the effect size on the individual level is high. For management purposes, it is crucial to further upscale the observed effects to the population level (May *et al.* 2019). To understand how OWFs can impact fish populations, it is crucial to obtain knowledge on how the underlying processes, such as recruitment, growth and mortality, might be affected by the cumulative presence of OWFs (Gill *et al.* 2020). For example, changes in recruitment due to OWFs (e.g., eggs do not reach nursery areas due to increased turbulence) might have more important consequences for the population than changes that mainly affect adults (e.g., food availability on offshore feeding grounds). Therefore, it is important to include different life stages when studying OWF effects on a certain species and to determine which changes in certain processes result in the largest impact at the population level (Gill *et al.* 2020).

5. Conclusions

In this study, we analysed the spatial distribution, trophic ecology and small-scale movements in time and space of the commercial flatfish species plaice *Pleuronectes platessa* at the individual adult level. We conclude that OWFs likely enhance ecological fish production through the existence of an artificial reef effect in combination with a refuge effect. Our findings suggest that plaice is attracted towards the scour protection due to the increased hard substrate prey availability, and that they perform ‘feeding excursions’ from the surrounding sand towards the SPL during daylight hours. Although our results indicate that OWFs likely increase plaice

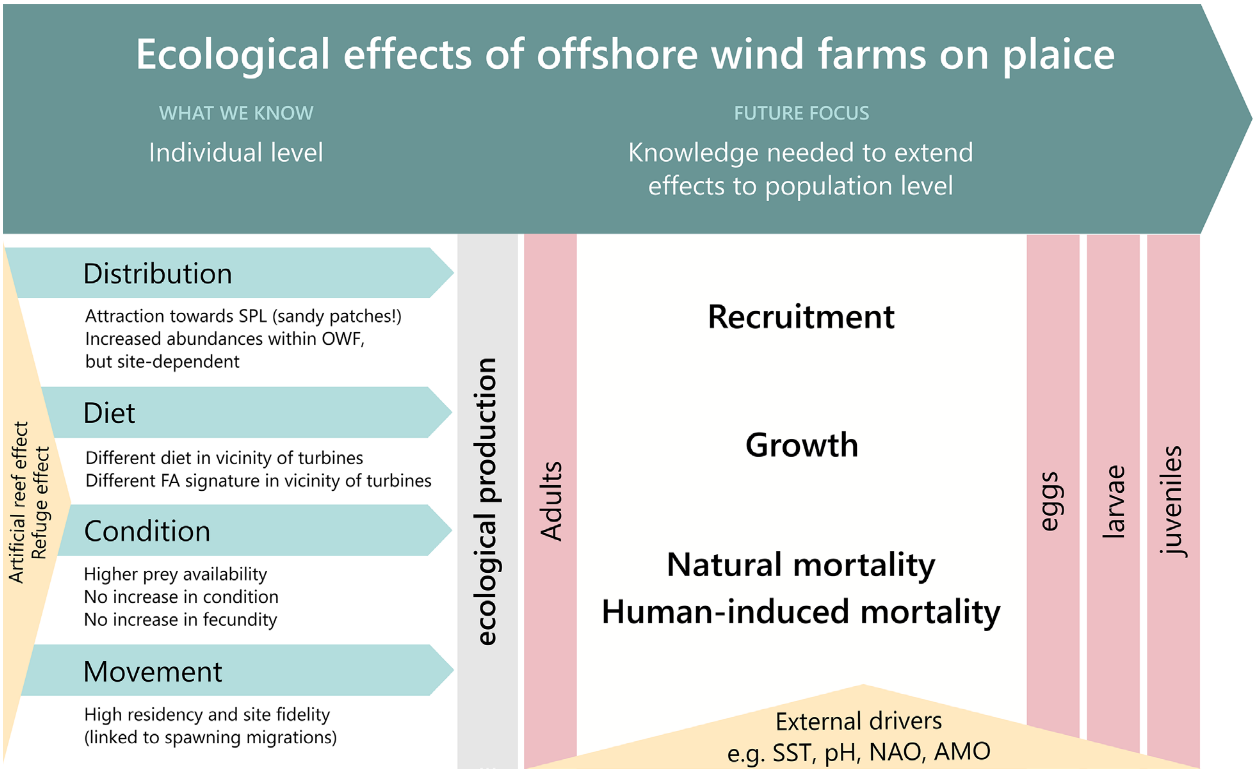

Figure 8. Schematic overview of the most important findings within this PhD study (Buyse 2023) at individual level, and the knowledge gaps for upscaling the observed effects towards the population level.

biomass, it remains unclear whether this could lead to spillover into adjacent areas. Follow-up research should focus on investigating refuge and spillover effects in-situ, and aim at upscaling the effects of OWFs by studying which population-driving processes are most impacted by the large-scale expansion of offshore wind energy developments.

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References

Amundsen, P.A. & Sánchez-Hernández, J. 2019. Feeding studies take guts – critical review and recommendations of methods for stomach contents analysis in fish. *Journal of Fish Biology* 95: 1364–1373. https://doi.org/10.1111/jfb.14151

Andersson, M.H. & Öhman, M.C. 2010. Fish and sessile assemblages associated with wind-turbine constructions in the Baltic Sea. *Marine and Freshwater Research* 61: 642–650. https://doi.org/10.1071/MF09117

Ashley, M.C., Mangi, S.C. & Rodwell, L.D. 2014. The potential of offshore windfarms to act as marine protected areas – A systematic review of current evidence. *Marine Policy* 45: 301–309. https://doi.org/10.1016/j.marpol.2013.09.002

Bailey, H., Brookes, K.L. & Thompson, P.M. 2014, September 14. Assessing environmental impacts of offshore wind farms: Lessons learned and recommendations for the future. *Aquatic Biosystems* 10: e8. https://doi.org/10.1186/2046-9063-10-8

Baktoft, H., Gjelland, K.Ø., Økland, F. & Thygesen, U.H. 2017. Positioning of aquatic animals based on time-of-arrival and random walk models using YAPS (Yet Another Positioning Solver). *Scientific Reports* 7: e14294. https://doi.org/10.1038/s41598-017-14278-z

Baktoft, H., Gjelland, K., Økland, F., Rehage, J., Rodemann, J., Corujo, R.S., Viadero, N. & Thygesen, U. 2019. Opening the black box of high resolution fish tracking using yaps. bioRxiv. https://doi.org/10.1101/2019.12.16.877688

Bergström, L., Sundqvist, F. & Bergström, U. 2013. Effects of an offshore wind farm on temporal and spatial patterns in the demersal fish community. *Marine Ecology Progress Series* 485: 199–210. https://doi.org/10.3354/meps10344

Bohnsack, J.A. 1989. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? *Bulletin of Marine Science* 44 (2): 631–645.

Bohnsack, J.A. & Sutherland, D.L. 1985. Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science* 37: 11–39.

Brickhill, M.J., Lee, S.Y. & Connolly, R.M. 2005. Fishes associated with artificial reefs: attributing changes to attraction or production using novel approaches. *Journal of Fish Biology* 67: 53–71. https://doi.org/10.1111/j.0022-1112.2005.00915.x

Buyse, J. 2023. *Ecological Impacts of Offshore Wind Farms on Flatfish with Emphasis on Plaice Pleuronectes platessa, a Species of Commercial Interest in the Southern North Sea*. PhD thesis, Ghent University. 255 pp.

Buyse, J., De Backer, A. & Hostens, K. 2021. Small-scale distribution patterns of flatfish on artificial hard substrates in a Belgian offshore wind farm. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds) *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea. Attraction, Avoidance and Habitat Use at Various Spatial Scales*: 69–76.

Buyse, J., Hostens, K., Degraer, S. & De Backer, A. 2022. Offshore wind farms affect the spatial distribution pattern of plaice *Pleuronectes platessa* at both the turbine and wind farm scale. *ICES Journal of Marine Science* 79 (6): 1777–1786. https://doi.org/10.1093/icesjms/fsac107

Buyse, J., Hostens, K., Degraer, S., De Troch, M., Wittoeck, J. & De Backer, A. 2023. Increased food availability at offshore wind farms affects trophic ecology of plaice *Pleuronectes platessa*. *Science of the Total Environment* 862: 160730. https://doi.org/10.1016/j.scitotenv.2022.160730

Davies, B.F.R., Holmes, L., Rees, A., Attrill, M.J., Cartwright, A.Y. & Sheehan, E.V. 2021. Ecosystem approach to fisheries management works — how switching from mobile to static fishing gear improves populations of fished and non-fished species inside a marine-protected area. *Journal of Applied Ecology* 58 (11): 2463–2478. https://doi.org/10.1111/1365-2664.13986

De Backer, A., Van Hoey, G., Wittoeck, J. & Hostens, K. 2022. Describing the epibenthos and demersal fish communities in the Belgian part of the North Sea in view of future OWF monitoring. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds) *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea. Memoirs on the Marine Environment*: 19–36.

Degraer, S., Carey, D.A., Coolen, J.W.P., Hutchison, Z.L., Kerckhof, F., Rumes, B. & Vanaverbeke, J. 2020. Offshore wind farm artificial reefs affect ecosystem structure and functioning: a synthesis. *Oceanography* 33 (4): 48–57. https://doi.org/10.5670/oceanog.2020.405

Di Franco, A., Bussotti, S., Navone, A., Panzalis, P. & Guidetti, P. 2009. Evaluating effects of total and partial restrictions to fishing on Mediterranean rocky-reef fish assemblages. *Marine Ecology Progress Series* 387: 275–285. https://doi.org/10.3354/meps08051

Félix-Hackradt, F.C., Hackradt, C.W., Treviño-Otón, J., Pérez-Ruzafa, Á. & García-Charton, J.A. 2018. Effect of marine protected areas on distinct fish life-history stages. *Marine Environmental Research* 140: 200–209. https://doi.org/10.1016/j.marenvres.2018.06.012

Florin, A.-B., Bergström, U., Ustups, D., Lundström, K., Lundström, L. & Jonsson, P.R. 2013. Effects of a large northern European no-take zone on flatfish populations. *Journal of Fish Biology* 83: 939–962. https://doi.org/10.1111/jfb.12097

Friedlander, A.M., Brown, E.K. & Monaco, M.E. 2007. Coupling ecology and GIS to evaluate efficacy of marine protected areas in Hawaii. *Ecological Applications* 17: 715–730.

Gibson, R.N., Nash, R., Geffen, A. & Van der Veer, H. 2015. *Flatfishes: Biology and Exploitation*. Wiley Blackwell, Oxford. 542 pp.

Gill, A.B., Degraer, S., Lipsky, A., Mavraki, N., Methratta, E. & Brabant, R. 2020. Setting the context for offshore wind development effects on fish and fisheries. *Oceanography* 33: 118–127. https://doi.org/10.5670/oceanog.2020.411

Glarou, M., Zrust, M. & Svendsen, J.C. 2020. Using artificial-reef knowledge to enhance the ecological function of offshore wind turbine foundations: Implications for fish abundance and diversity. *Journal of Marine Science and Engineering* 8 (5): e332. https://doi.org/10.3390/jmse8050332

Guidetti, P., Baiata, P., Ballesteros, E., Di Franco, A., Hereu, B., Macpherson, E., Micheli, F., Pais, A., Panzalis, P., Rosenberg, A.A., Zabala, M. & Sala, E. 2014. Large-scale assessment of mediterranean marine protected areas effects on fish assemblages. *PLoS ONE* 9: e91841. https://doi.org/10.1371/journal.pone.0091841

Halouani, G., Villanueva, C.M., Raoux, A., Dauvin, J.C., Ben Rais Lasram, F., Foucher, E., Le Loc'h, F., Safi, G., Araignous, E., Robin, J.P. & Niquil, N. 2020. A spatial food web model to investigate potential spillover effects of a fishery closure in an offshore wind farm. *Journal of Marine Systems* 212: e103434. https://doi.org/10.1016/j.jmarsys.2020.103434

Hutchison, Z.L., Secor, D.H. & Gill, A.B. 2020. The interaction between resource species and electromagnetic fields associated with electricity production by offshore wind farms. *Oceanography* 33: 96–107. https://doi.org/10.5670/oceanog.2020.409

Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., James Grecian, W., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J. & Godley, B.J. 2009. Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology* 46: 1145–1153. https://doi.org/10.1111/j.1365-2664.2009.01697.x

Krone, R., Dederer, G., Kanstinger, P., Krämer, P., Schneider, C. & Schmalenbach, I. 2017. Mobile demersal megafauna at common offshore wind turbine foundations in the German Bight (North Sea) two years after deployment – increased production rate of *Cancer pagurus*. *Marine Environmental Research* 123: 53–61. https://doi.org/10.1016/j.marenvres.2016.11.011

Langhamer, O. 2012. Artificial reef effect in relation to offshore renewable energy conversion: State of the art. *The Scientific World Journal* 2012: e386713. https://doi.org/10.1100/2012/386713

Langhamer, O. & Wilhelmsson, D. 2009. Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes - A field experiment. *Marine Environmental Research* 68: 151–157. https://doi.org/10.1016/j.marenvres.2009.06.003

Lindberg, W.J. 1997. Can science resolve the attraction-production issue? *Fisheries* 22: 10–13.

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., De Haan, D., Dirksen, S., Van Hal, R., Hille Ris Lambers, R., Ter Hofstede, R., Krijgsveld, K.L., Leopold, M. & Scheidat, M. 2011. Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. *Environmental Research Letters* 6: e035101. https://doi.org/10.1088/1748-9326/6/3/035101

Mahesh, V., Ambarish, P.G. & Rekha, J.N. 2018. Stomach content analysis techniques in fishes. *In*: *ICAR Sponsored Winter School on Recent Advances in Fishery Biology techniques for Biodiversity Evaluation and Conservation, 1–21 December 2018, Kochi*: 104–115.

Mavraki, N., Degraer, S. & Vanaverbeke, J. 2021. Offshore wind farms and the attraction – production hypothesis: insights from a combination of stomach content and stable isotope analyses. *Hydrobiologia* 848: 1639–1657. https://doi.org/10.1007/s10750-021-04553-6

May, R., Masden, E.A., Bennet, F. & Perron, M. 2019. Considerations for upscaling individual effects of wind energy development towards population-level impacts on wildlife. *Journal of Environmental Management* 230: 84–93. https://doi.org/10.1016/j.jenvman.2018.09.062

Methratta, E.T. 2020. Monitoring fisheries resources at offshore wind farms: BACI vs. BAG designs. *ICES Journal of Marine Science* 77: 890–900. https://doi.org/10.1093/icesjms/fsaa026

Miethe, T., Dytham, C., Dieckmann, U. & Pitchford, J.W. 2010. Marine reserves and the evolutionary effects of fishing on size at maturation. *ICES Journal of Marine Science* 67 (3): 412–425. https://doi.org/10.1093/icesjms/fsp248

Osenberg, C.W., Mary, C.M.S., Wilson, J.A. & Lindberg, W.J. 2002. A quantitative framework to evaluate the attraction – production controversy. *ICES Journal of Marine Science* 59: S214–S221. https://doi.org/10.1006/jmsc.2002.1222

Petersen, J.K. & Malm, T. 2006. Offshore windmill farms: Threats to or possibilities for the marine environment. *Ambio* 35 (2): 75–80.

Randall, J.E. 1963. An analysis of the fish populations of artificial and natural reefs in the Virgin Islands. *Caribbean Journal of Sciences* 3: 31–47.

Raoux, A., Tecchio, S., Pezy, J.P., Lassalle, G., Degraer, S., Wilhelmsson, D., Cachera, M., Ernande, B., Le Guen, C., Haraldsson, M., Grangeré, K., Le Loc'h, F., Dauvin, J.C. & Niquil, N. 2017. Benthic and fish aggregation inside an offshore wind farm: Which effects on the trophic web functioning? *Ecological Indicators* 72: 33–46. https://doi.org/10.1016/j.ecolind.2016.07.037

Reubens, J., Degraer, S. & Vincx, M. 2011. Aggregation and feeding behaviour of pouting (*Trisopterus luscus*) at wind turbines in the Belgian part of the North Sea. *Fisheries Research* 108: 223–227. https://doi.org/10.1016/j.fishres.2010.11.025

Reubens, J., De Rijcke, M., Degraer, S. & Vincx, M. 2014. Diel variation in feeding and movement patterns of juvenile Atlantic cod at offshore wind farms. *Journal of Sea Research* 85: 214–221. https://doi.org/10.1016/j.seares.2013.05.005

Shipp, R.L. 2003. A Perspective on Marine Reserves as a Management Tool. *Fisheries* 28: 10–21. https://doi.org/10.1577/1548-8446(2003)28[10:APOMRA]2.0.CO;2

Steins, N.A., Veraart, J.A., Klostermann, J.E.M. & Poelman, M. 2021. Combining offshore wind farms, nature conservation and seafood: Lessons from a Dutch community of practice. *Marine Policy* 126: 104371. https://doi.org/10.1016/j.marpol.2020.104371

Stone, R.B., Pratt, H.L., Parker, R.O. & Davis, G.E. 1979. A comparison of fish populations on an artificial and natural reef in the Florida Keys. *Marine Fisheries Review* 41: 1–11.

van Hal, R., Griffioen, A.B. & van Keeken, O.A. 2017. Changes in fish communities on a small spatial scale, an effect of increased habitat complexity by an offshore wind farm. *Marine Environmental Research* 126: 26–36. https://doi.org/10.1016/j.marenvres.2017.01.009

van Walraven, L., Mollet, F.M., van Damme, C.J.G. & Rijnsdorp, A.D. 2010. Fisheries-induced evolution in growth, maturation and reproductive investment of the sexually dimorphic North Sea plaice (*Pleuronectes platessa* L.). *Journal of Sea Research* 64: 85–93. https://doi.org/10.1016/j.seares.2009.07.003

Vandendriessche, S., Derweduwen, J. & Hostens, K. 2015. Equivocal effects of offshore wind farms in Belgium on soft substrate epibenthos and fish assemblages. *Hydrobiologia* 756: 19–35. https://doi.org/10.1007/s10750-014-1997-z

Wilhelmsson, D., Malm, T. & Öhman, M.C. 2006. The influence of offshore windpower on demersal fish. *ICES Journal of Marine Science* 63 (5): 775–784. https://doi.org/10.1016/j.icesjms.2006.02.001

Wilson, J., Osenberg, C.W., Mary, C.M.S., Watson, C.A. & Lindberg, W.J. 2001. Artificial reefs, the attraction-production issue, and density dependence in marine ornamental fishes. *Aquarium Sciences and Conservation* 3: 95–105. https://doi.org/10.1023/A:1011343312031

Wilson, J.C., Elliott, M., Cutts, N.D., Mander, L., Mendão, V., Perez-Dominguez, R. & Phelps, A. 2010. Coastal and offshore wind energy generation: Is it environmentally benign? *Energies* 3: 1383–1422. https://doi.org/10.3390/en3071383

Wright, S.R., Lynam, C.P., Righton, D.A., Metcalfe, J., Hunter, E., Riley, A., Garcia, L., Posen, P. & Hyder, K. 2020. Structure in a sea of sand: Fish abundance in relation to man-made structures in the North Sea. *ICES Journal of Marine Science* 77 (3): 1206–1218. https://doi.org/10.1093/icesjms/fsy142

CHAPTER 4

SEASONAL DISTRIBUTION OF HARBOUR PORPOISES (*PHOCOENA PHOCOENA*) AND RESPONSE TO OPERATIONAL OFFSHORE WIND FARMS IN THE BELGIAN NORTH SEA

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Abstract

Human activities at sea, such as shipping, fisheries, mariculture, and offshore renewable energy developments, potentially influence habitat use of marine mammals. In the Belgian part of the North Sea (BPNS), the most common marine mammal species is the harbour porpoise (*Phocoena phocoena*). In this study, we update the occurrence and seasonal distribution of this species in the BPNS and investigate the potential effect of operational offshore wind farms (OWFs). To do so, we used aerial survey data collected between 2009 and 2022 and analysed the spatio-temporal distribution of the harbour porpoise as a function of a selection of environmental drivers and anthropogenic stressors. The species' distribution followed a consistent seasonal pattern, with the highest densities in spring, but with a high interannual variability in abundance, with peaks in 2011, 2014 and 2018. Porpoise distribution was explained by latitude and longitude, with the species preferring the western part of the BPNS, revealing a strong overlap with the Vlaamse Banken Special Area of Conservation (SAC). The distribution was also significantly negatively correlated with marine traffic intensity and distance to the closest OWF, but caution is needed in order not to overinterpret these correlations. Further studies are recommended to support or confute the findings of this study, and to better understand the interaction between natural factors, such as prey availability, and anthropogenic stressors driving the species distribution. The results of such studies may influence the management of future activities at sea and assist in conservation efforts.

1. Introduction

The rapid acceleration of human activities in, and exploitation of continental shelf areas has effects on marine mammals worldwide (e.g., Hawkins *et al.* 2017; Avila *et al.* 2018). For many species, essential habitats, including migration routes, feeding grounds and breeding areas, overlap with areas of intensive human activities, leading to a range of effects (Bearzi *et al.* 2019; Minton *et al.* 2021). The North Sea is a hotspot of anthropogenic activities, including shipping, fisheries, sand extraction, port development and rapidly increasing renewable energy production (Peschko *et al.* 2016; Nachtsheim *et al.* 2021). All of these activities have an effect on cetaceans, and potentially lead to habitat degradation or loss (e.g., Gilles *et al.* 2009), fisheries bycatch (e.g., Brownell *et al.* 2019), collisions with vessels (e.g., Schoeman *et al.* 2020) and disturbance due to noise pollution caused by marine traffic and offshore wind farm (OWF) development (e.g., Haelters *et al.* 2014; Verfuss *et al.* 2016; Wisniewska *et al.* 2018).

In the southern North Sea, including the Belgian part of the North Sea (BPNS), the most common marine mammal species is the harbour porpoise (*Phocoena phocoena*) (Haelters *et al.* 2011; Bouveroux *et al.* 2020; OSPAR 2023). Harbour porpoises are wide-ranging, highly mobile and energetically demanding small odontocetes that feed on a range of fish, such as sandeels, clupeids, gadoids, gobies and flatfish, and cephalopods (Haelters *et al.* 2012; Ransijn *et al.* 2019; Nachtsheim *et al.* 2021). After a strong decline in the 1960s, numbers of harbour porpoises have steadily increased in this region, likely as a result of a southward shift in distribution (Camphuysen 2011; Geelhoed *et al.* 2013; IJsseldijk *et al.* 2020). In the BPNS, animals can now be observed year-round, but their occurrence displays strong yearly fluctuations. There is a seasonal pattern with a peak from February to April (Haelters *et al.* 2013; Van Nieuwenhove *et al.* 2023); a similar seasonal pattern is observed in adjacent waters in the Netherlands and in the English Channel (Scheidat *et al.* 2012; Geelhoed *et al.* 2013; Gilles *et al.* 2016; Bouveroux *et al.* 2020). Within the BPNS, the density of animals was described as the highest in the south-western/western part, and as higher further offshore vs inshore (Haelters *et al.* 2013). While the reasons driving a small-scale spatio-temporal distribution of harbour porpoises remain unclear, seasonal patterns may be driven by local prey availability (Haelters *et al.* 2011).

As a vulnerable species, the harbour porpoise is listed in Annexes II and IV of the European Union (EU) Habitats Directive (Council Directive 1992/43/EEC). As such, EU member states have the obligation to ensure its conservation through the implementation of, where needed, protection measures, supported by the necessary research and monitoring activities. Under the Marine Strategy Framework Directive (Directive 2008/56/EC), national and international indicators and targets are developed, including for marine mammals. Threats and mitigation measures are also discussed in the framework of the regional agreement ASCOBANS (Agreement on the Conservation of Small Cetaceans of the Baltic, North East Atlantic, Irish and North Seas, 2009), concluded under the auspices of the Convention on the Conservation of Migratory Species of Wild Animals (CMS or Bonn Convention).

With the expansion of OWFs in the last decades in the North Sea (Xu *et al.* 2020; Degraer *et al.* 2022, 2021) and the planned ones for the next decade (Degraer *et al.* 2022, 2023), the investigation of responses of harbour porpoises to OWFs during their construction and operational phases is of high importance in the frame of reaching conservation objectives. Studies using aerial survey and passive acoustic monitoring (PAM) have shown that OWF construction (and more in particular, pile driving of foundations) causes disturbance and large-scale (temporal) displacement and avoidance reactions (e.g., Carstensen *et al.* 2006; Dähne *et al.* 2013; Thompson *et al.* 2013; Haelters *et al.* 2014; Brandt *et al.* 2018). Sound mitigation measures have been developed, tested and used in practice with positive results: a temporal ban on piling limits the number of animals exposed where seasonally differences occur in animals present, and measures such as single and double bubble curtains and ramp-up procedures reduce the number of animals exposed to loud noise, the risk of physical damage to animals and the spatial extent of disturbance (Lucke *et al.* 2011; Dähne *et al.* 2017; Rumes & Degraer 2020; Rumes & Zupan 2021).

The potential negative or positive effects of operational OWFs on harbour porpoises have, in comparison to acute effects due to piling, received less attention. Studies have investigated if harbour porpoises were attracted to operational windfarms, were indifferent to them or if they avoided them, but contrasting behavioural responses lead to the question remaining largely unanswered (e.g., Blew *et al.* 2006; Tougaard *et al.* 2006a, 2006b; Scheidat *et al.* 2009, 2011; van Polanen Petel *et al.* 2012; Teilmann & Carstensen 2012; Dähne *et al.* 2014; Vallejo *et al.* 2017; Collier *et al.* 2022). Behavioural responses may be site-specific, as the interplay between positive effects (i.e., high habitat quality, artificial reef effect for prey species, sheltering effect, effect of a diminished ship traffic) and negative effects (i.e., low habitat quality, noise disturbance) would yield different outcomes, depending on the underlying ecological features (Tougaard *et al.* 2005; Scheidat *et al.* 2011; Haelters *et al.* 2013). Potential responses could also be masked by a natural distribution, independent of the presence of offshore wind turbines.

In the BPNS, eight OWFs, totalling 399 turbines, became operational over the course of 15 years (Rumes *et al.* 2022) and an additional zone for offshore renewable energy has been designated in the national marine spatial plan (MSP 2020–2026). However, no assessment specifically aimed at elucidating potential effects of operational OWFs on harbour porpoise distribution and abundance, has been undertaken. Also, given future developments, it is useful to update information on the species' presence in the BPNS. The aim of this study is to analyse the spatio-temporal distribution of harbour porpoises in Belgian waters as a function of a selection of environmental drivers and anthropogenic stressors using aerial survey data. Specifically, this study aimed at analysing distribution patterns with special attention to the influence of operational OWFs.

2. Material and methods

2.1. Study area

The BPNS is located in the southwestern part of the North Sea basin (Figure 1); it has a surface of 3454 km^2. The area is physically, geologically, and ecologically heterogeneous, consisting of a shallow sandbank system that classifies as Habitat 1110 ("sandbanks permanently covered with seawater") under the European Habitats Directive. The habitat hosts a rich and highly productive benthic ecosystem (Pecceu *et al.* 2021). Offshore, predominantly in the northwestern part of the area, gravel beds occur that formerly hosted oyster beds (Habitats directive Habitat 1170; "reefs"). Within the soft sediment, aggregations of the polychaete worm *Lanice conchilega* are widely dispersed; also, these aggregations were classified under habitat type 1170. The habitats present act as nursery areas for fish, cephalopods and crustaceans (Houziaux *et al.* 2008). The presence of habitat types 1110 and 1170 are the background for the delimitation, in 2012, of a Special Area of Conservation (SAC) of approximately 1.112 km^2 (*the Vlaamse Banken*) in the western part of Belgian waters (Pecceu *et al.* 2016).

In the eastern part of the BPNS, close to the border with Dutch waters, an area of 238 km^2 was allocated to the production of renewable energy. Between 2009 and 2022, eight OWF were constructed, totalling 399 turbines with a total capacity of 2.26 GW (timeline and current status described in Rumes *et al.* 2022). The area is located between two major shipping lanes in the Southern North Sea (Figure 1). Due to its proximity to the English Channel and to large ports, such as those of Antwerp and Rotterdam, the BPNS and its surrounding waters have one of the busiest shipping traffic worldwide (Schallier & Van Roy 2015; Putland *et al.* 2022).

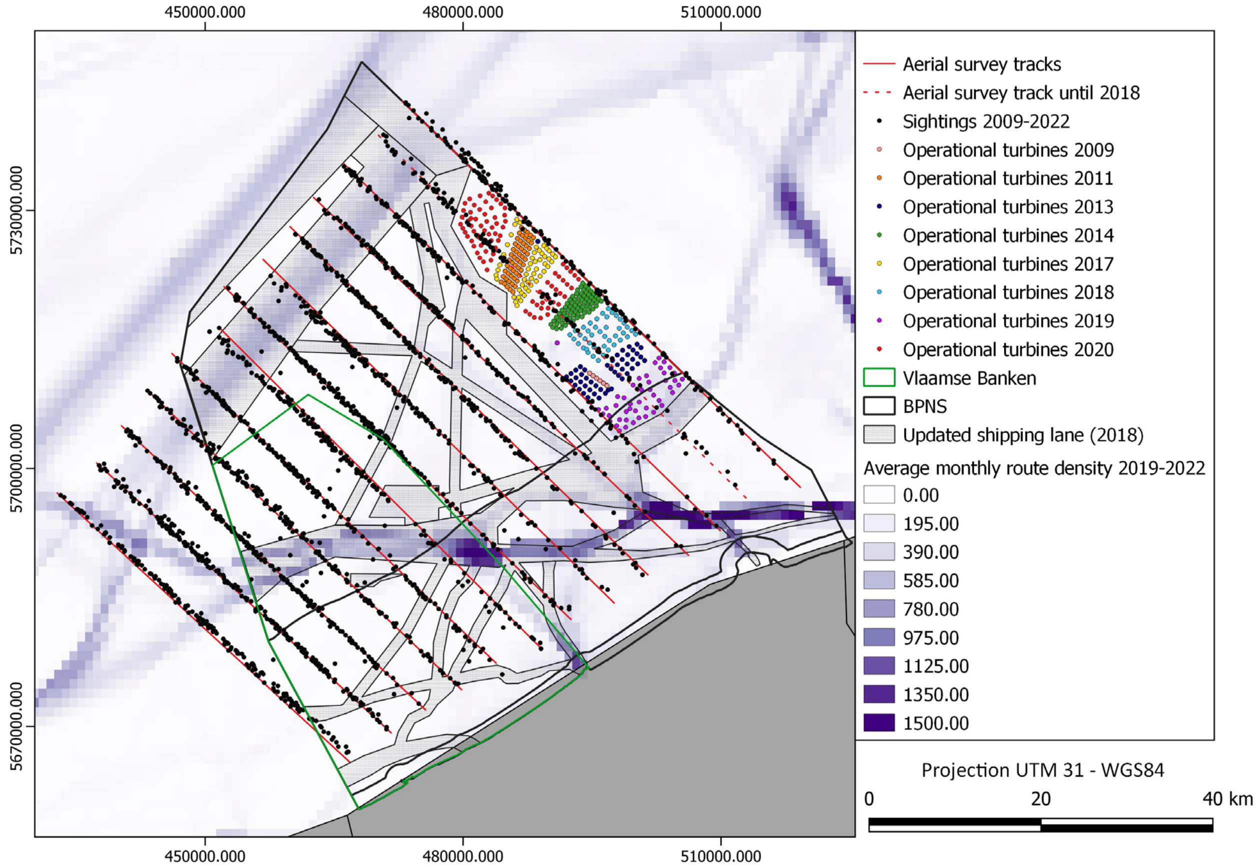

Figure 1. Overview of all harbour porpoise sightings during dedicated aerial surveys (2009-2022) outside of periods with piling operations in Belgian waters. Red tracks represent the line transects as planned. OWFs are colour coded based on the year they became operational. An average monthly route density map (number of vessels detected by AIS in a grid cell of 1 × 1 km each month) is shown underneath, on top of which is the latest shipping lanes plan as revised in the MSP (2020–2026).

2.2. Aerial surveys

Highly standardized and dedicated aerial surveys were carried out following the line-transect sampling strategy (Buckland *et al.* 2001). Surveys followed predefined track lines 5 km apart and perpendicular to the coastline to follow an onshore-offshore gradient (Figure 1). For practical reasons, part of the westerly survey tracks is located in French waters. A detailed description of the survey design and data collection are given by Haelters (2009) and Haelters *et al.* (2013). During the flight, sightings were recorded at non-predefined distances from the track line. Group sizes and presence of calves were noted. To calculate the perpendicular distance of each animal from the track (x), the altitude (h) was recorded, together with the angle (θ) between the horizon and the perpendicular line from the track to the animal, using a hand-held Suunto clinometer PM-5/360PC. The distance of the animal from the trackline was calculated with the Eq. 1: $\chi = h * \tan(90° - \theta)$.

The aircraft used was a Norman Britten Islander equipped with two bubble windows, accommodating two observers. Flight altitude was kept constant at 600 feet (183 m) and groundspeed was 100 knots (185 km/h). Data on ground speed, altitude, time, GPS-position and heading were recorded with a high temporal frequency (every second). Given the high availability of the aircraft, flights were only performed during good observation conditions (sea state ≤ 2 and visibility > 2 km). Preferably, surveys were completed within one day, but if this was not possible, the tracks were completed in a subsequent flight, in most cases less than a few days later, thus assuming a similar species abundance and distribution in both flights. Survey flights were always combined with regular coastguard tasks: tracks were temporarily interrupted to record and document detections of e.g., oil slicks or shipping navigation violations.

Analyses of the data were carried out using Distance 7.5 Release 1 (Thomas *et al.* 2010). Given the highly standardized nature of the surveys, all observations could be pooled to obtain one detection function. A half-normal detection function with cosine adjustments was selected on the basis of the lowest Akaike Information Criterion (AIC; Thomas *et al.* 2010).

From the detection model, an effective half strip width (E(1/2)SW) of 147.97 m (137.93-158.74) could be estimated, using 2926 observations of in total 3552 animals during 40 surveys. As not all animals were seen by the observers (perception bias), and as some animals were not visible at or near the surface, given that they were too deep (availability bias), a g(0) was applied as estimated for similar surveys (g(0): 0.364; Hammond *et al.* 2021). Using different values of g(0) would influence the absolute value of density and abundance estimates but would not change the relative distribution or encounter rate (animals observed/effort). Hence, as the aim of this study was to investigate variability in abundance and density distributions, the use of partially corrected observations is as informative (as seen in Vallejo *et al.* 2017).

In this study, 40 aerial surveys performed between 2009 and 2022 were considered. Surveys with a deviating track or a different technical setup (1 bubble window instead of 2) were excluded from the analysis (3 surveys) and in some cases consecutive surveys that were carried out over a very short period were combined and considered as one survey, with some or all tracks being flown more than once. This resulted in 31 full coverage surveys. Of these, surveys that were carried out during or very shortly (≤ 48 hrs) after piling operations in Belgian or Dutch waters (e.g., Borssele) were excluded from the analyses, reducing the number of surveys considered to a total of 21. The surveys were analysed for the purpose of assessing factors that could influence harbour porpoise distribution and abundance, and especially with a focus on possible effects of operational OWFs.

2.3. Data processing in QGIS

2.3.1. Seasonal maps of observed estimated densities

A squared grid of resolution 5x5 km was created to cover the entire surveyed area. The grid was aligned as much as possible with the surveyed transects to maximize the evenness of the survey effort across grid cells. For each survey, the total length of the flight track and the total number of observed individuals in each grid cell were calculated. To only retain representatively surveyed grid cells for each survey, grid cells with a surveyed effort smaller than 3.75 km were excluded from the dataset, corresponding to a threshold of 75% coverage of the grid cell dimension (length of 5 km). The survey effort (hereafter called "observed km^2") was calculated as the length of the flight track in the grid cell (hereafter called "transect length") multiplied by the total effective strip width (295.94 m). Grid cells which were not representatively surveyed, or which fell outside the surveyed area, were assigned a N/A value. For each representatively surveyed grid cell, the encounter rate was calculated as the number of harbour porpoises observed per km surveyed (ind/km). The estimated density (D) was calculated as the number of individuals observed per observed km^2, the latter multiplied by g(0) (0.364) (Eq. 2):

$$D_{grid\ cell} = \frac{No.\ of\ ind_{grid\ cell}}{Transect\ length_{grid\ cell} * ESW * g(0)}$$

Estimated density distribution maps were produced for each survey. After visual scrutiny, the estimated density distributions were averaged at grid cell-level for each season to obtain seasonal distribution maps, except for winter where surveys were too scarce in number. In winter, given that observation conditions are usually unfavourable, mostly due to short days and a low hanging sun leading to a lot of glare, only two surveys were performed. Seasonal distribution maps were preferred over an overall map as strong seasonal patterns are known for the species in Belgian waters (Haelters *et al.* 2011, 2013) and in the North Sea in general (Gilles *et al.* 2009, 2016). The averaging exercise followed the assumption of a spatial symmetric distribution in different surveyed years but during the same season, similarly to what was done by Gilles *et al.* (2009). Furthermore, the individual survey detection curves were verified to be similar, as done in Scheidat *et al.* (2008). To quantify and visualize the variability in total number of observations in the same grid cell across years, estimated density standard deviation (SD) maps were computed for each season. This choice was justified by the scope of the study, which aimed at understanding and visualizing relative abundances and spatial distribution of harbour porpoises in Belgian waters rather than obtaining exact absolute numbers and density values at a relatively small spatial scale for this highly mobile species.

2.3.2. Calculation of explanatory variables

Several environmental and anthropogenic factors were considered for the investigation of potential drivers influencing the relative distribution of harbour porpoises in the BPNS. Information on each observation of position (latitude, longitude, corrected for distance to the aircraft), season and year were available from the survey data. A bathymetry map with a resolution of 115 × 115 m was downloaded from the open-source Map Viewer of the European Marine Observation and Data Network (EMODnet; https://emodnet.ec.europa.eu/geoviewer/). Depth values were extracted at each corrected observation position using the plugin 'point sampling tool'. Monthly route density maps for the period 2019-2022 (i.e., maximum time interval available) with 1 × 1 km resolution were downloaded from EMODnet Map Viewer. Monthly route density maps represent the total number of vessels of all types detected in each grid cell in a given month using the Automatic Identification System (AIC), and account as a proxy of marine traffic. All monthly maps were averaged to obtain an overall proxy map for marine traffic and shipping intensity (Figure 1). Shipping intensity values were extracted at each observation location using the plugin 'point sampling tool'. To quantify the potential attraction or avoidance effect of OWFs on harbour porpoises, the distance of each observed individual from the closest OWF (i.e., the closest turbine) was calculated using the function 'distance to the nearest hub'. As different OWFs became operational in different years, distances of observations made in a specific year were calculated exclusively with respect to the turbines present at the time, following the development timeline presented in Rumes *et al.* (2022; Figure 2). Finally, the underlying seafloor habitat type was considered as a proxy for other ecological factors driving the species distribution. The seafloor habitat classification in Pecceu *et al.* (2021) was used as it considered sandbanks, the probability of the occurrence of aggregations of *Lanice* and the occurrence of gravel beds (Habitats 1110 and 1170 in the Habitat Directive). It further subdivided sandbank habitats into five types of macrobenthic communities, and Habitat type 1170 into gravel beds and the probability of the occurrence of aggregations of *Lanice conchilega* (for a total of seven benthic community types, hereafter called 'habitat type'). To obtain a habitat type for each grid cell, grid cells were assigned to the classification with the highest coverage percentage. Grid cells where the habitat type was not available (e.g., for the parts of the tracks over French waters) were classified as 'unknown'. For the grid cells where no individuals were observed during a specific survey, all explanatory variables were calculated from the grid cell centroid coordinates.

2.4. Statistical analysis

Due to the spatial nature of the data, two separate sets of statistical analyses were done:

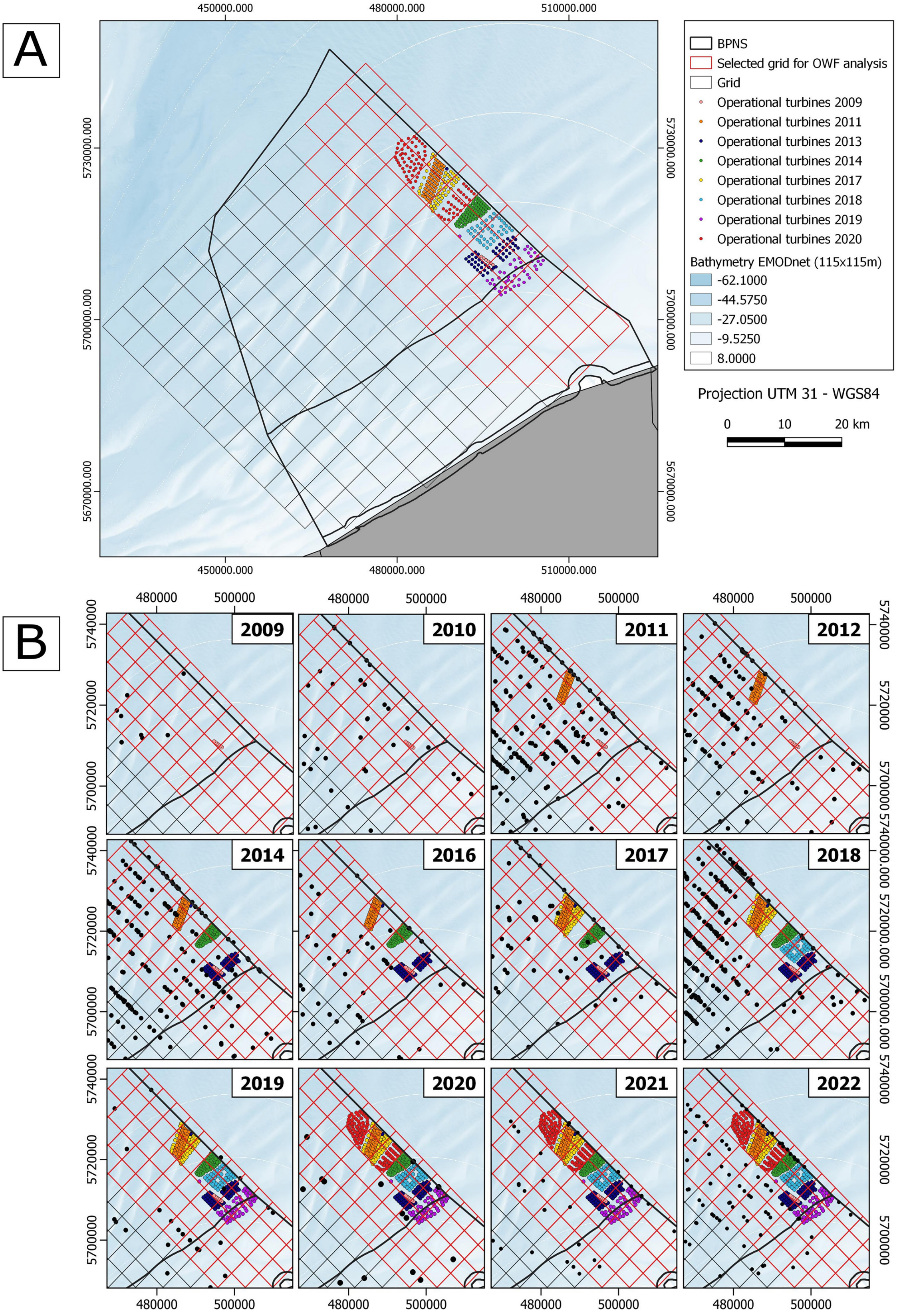

Figure 2. A. Grid area (red) of 1475 km^2 up to 15 km from all turbines in surveyed Belgian waters selected for the spatio-temporal analysis of the potential influence of operational OWFs on the distribution of harbour porpoises. **B**. Timeline of the sequence of OWFs becoming operational in the BPNS between 2009 and 2022 with associated yearly harbour porpoise sightings made during aerial surveys, and underlying bathymetry map. From 2019 onwards, one track directly over the OWF was not flown anymore for safety reasons (see Fig. 1).

(1) a spatio-temporal analysis of the potential influence of environmental and anthropogenic factors on the overall distribution of harbour porpoises in the BPNS; (2) a spatio-temporal analysis of the potential influence of operational OWFs on the distribution of harbour porpoises in the area surrounding, and including, the OWF area.

Operational OWFs in the BPNS are highly clustered and localized at the eastern part of the BPNS, along the border with the Dutch EEZ and neighbouring the most westerly Dutch OWFs. As such, they have a strong spatial correlation with other environmental features and gradients such as latitude, longitude and depth when considered at the overall BPNS level (Figure 1). Moreover, wind farms are located in the eastern part of Belgian waters, while porpoises naturally occur in higher densities in more westerly waters, as documented by previous literature (Haelters *et al.* 2011, 2013), with results indicating a higher sighting rate in more westerly waters within the BPNS (Figure 1). To reduce such spatial correlation and to produce a meaningful assessment of the potential effect of operational OWFs on harbour porpoise distribution, the area of interest in the latter statistical analysis was reduced to 1475 km^2 (59 of 255 grid cells), as such covering a surface defined by a 15 km radius surrounding the OWF area (Figure 2). The area east of the OWF area was not selected as it covered Dutch waters with no survey effort.

All statistical analyses were performed in Rstudio (ver. 4.1.1; Rstudio Team 2020). Both data from response and exploratory variables were inspected for correlation, outliers, normality and homoscedasticity prior to the modelling exercise following the protocol from Zuur *et al.* (2010). The response variable used in both analyses was the sighting rate (ind/nm) in each grid cell per survey (transformed into integer counts for the modelling exercise). As it is often the case with species distribution count data (Dénes *et al.* 2015), especially when derived from visual surveys (Zipkin *et al.* 2014; Vallejo *et al.* 2017), the data were zero-inflated. Zero-inflation occurs when the number of zeros is excessive compared to the integer counts and influences the modelling of a Poisson regression causing overdispersion (Yang *et al.* 2017). Zeros divide into true zeros (i.e., the animal is absent) and false zeros (i.e., due to observed error, sampling error, or wrong survey design; Zuur *et al.* 2009). Therefore, zero-inflated (ZINB) and zero-altered negative binomial regression (ZANB) models were used and tested. The difference stands in how the zeros are handled, but both apply two different distributions to the data: a logistic distribution to the zeros and a negative binomial distribution to the counts (Zuur *et al.* 2009). ZINB models were built with the function *zeroinfl* (package *pscl*), while ZANB models were built with the function *hurdle* (package *pscl*). Backward stepwise model selection was done separately for both models based on the AIC. Model selection between the best ZINB and ZANB models was based on AIC scores. Models were validated by assessing the residuals' normality, the residuals versus fitted values, and the residuals versus each covariate. For all analyses, an alpha threshold of $p=0.05$ was used for statistical significance.

3. Results

3.1. Seasonal distribution

A total of 2738 harbour porpoises were observed during the 21 aerial surveys considered for this analysis (Table 1). The average sighting rate (number of animals observed per 100 nautical miles surveyed) was much higher in spring (54.93 ± 87.14) than in summer (16.58 ± 33.61) or autumn (10.60 ± 26.58) (mean ± SD) (Wilcoxon Rank Sum Test p-value < 0.0001 for both pairs). The sighting of (presumed) mother-calf pairs was, as can be expected giving the calving season (May–July), highest in summer (54 out of 84 calves observed). The observed average group size was 1.12 in spring, 1.30 in summer, and 1.29 in autumn.

Table 1. Overview of aerial surveys used in this study with associated features, including the survey year, the season, total on task effort in nautical miles, the total number of harbour porpoises observed, the number of calves, the average group size and the average sighting rate expressed as observed individuals per 100 nautical miles surveyed (mean ± SD). Surveys with asterisks (*) indicate surveys that are the result of a combination of surveys undertaken within short timeframes.

Year	Season	Survey effort (nm flown)	No. of individuals	No. of calves	Average group size	Sighting rate (ind/100 nm)
2009	Spring	275.35	13	0	1.08	4.32
2010	Winter	347.04	51	0	1.30	14.74
2010	Spring*	344.50	59	0	1.04	15.60
2010	Summer	346.29	38	2	1.28	10.13
2011	Spring*	1118.09	646	0	1.24	56.64
2011	Winter	327.89	100	2	1.79	29.84
2012	Spring	348.02	196	1	1.14	53.42
2012	Autumn	344.32	40	0	1.28	11.15
2014	Spring	333.71	331	0	1.09	94.48
2014	Autumn*	728.15	64	0	1.32	8.13
2016	Spring	333.30	104	0	1.05	29.86
2017	Summer	357.78	116	21	1.25	30.49
2017	Autumn	302.23	21	2	1.07	6.98
2018	Spring	359.81	404	0	1.15	108.62
2018	Summer	287.46	41	6	1.28	12.73
2019	Summer*	706.64	93	12	1.23	12.38
2020	Autumn	325.37	37	0	1.19	9.94
2021	Summer	315.49	52	10	1.49	16.31
2021	Autumn	323.96	52	7	1.58	13.47
2022	Spring	334.66	235	21	1.19	64.94
2022	Autumn	284.66	45	0	1.32	14.51
Total (n = 21)/average		**8444.76**	**2738**	**84**	**1.26**	**30.33**

The strong seasonal difference in estimated density is associated with a strong seasonal spatial distribution pattern (Figure 3). Although the total number of observations made in each grid cell varied across years in the same season, the spatial pattern of the observations did not differ, and seasonal maps of mean densities could be achieved. In spring, both observations and estimated density distribution followed a clear pattern, with a relatively high density offshore and in the western part of Belgian waters, continuing into adjacent French waters. In this area, estimated densities reached average values of 8 individuals per km^2. Observations and estimated densities were low in coastal waters within the first 12 nautical miles, except for the coastal area off Nieuwpoort. In summer, the distribution gradients were less defined and homogeneous, but still revealed higher estimated densities in offshore waters, especially in the western part of the BPNS. Estimated densities per grid cell ranged from 0 to 4.1 individuals/km^2. Most mother-calf pairs were observed in the western part of the BPNS. Several grid cells showed a higher variability in density over different years. In autumn, observations and estimated densities were spread across the BPNS without a clear distributional pattern. Most mother-calf pairs were seen offshore, and relatively many animals were observed in waters close to shore. In autumn, the highest estimated density per grid cell was 2.7 individuals/km^2.

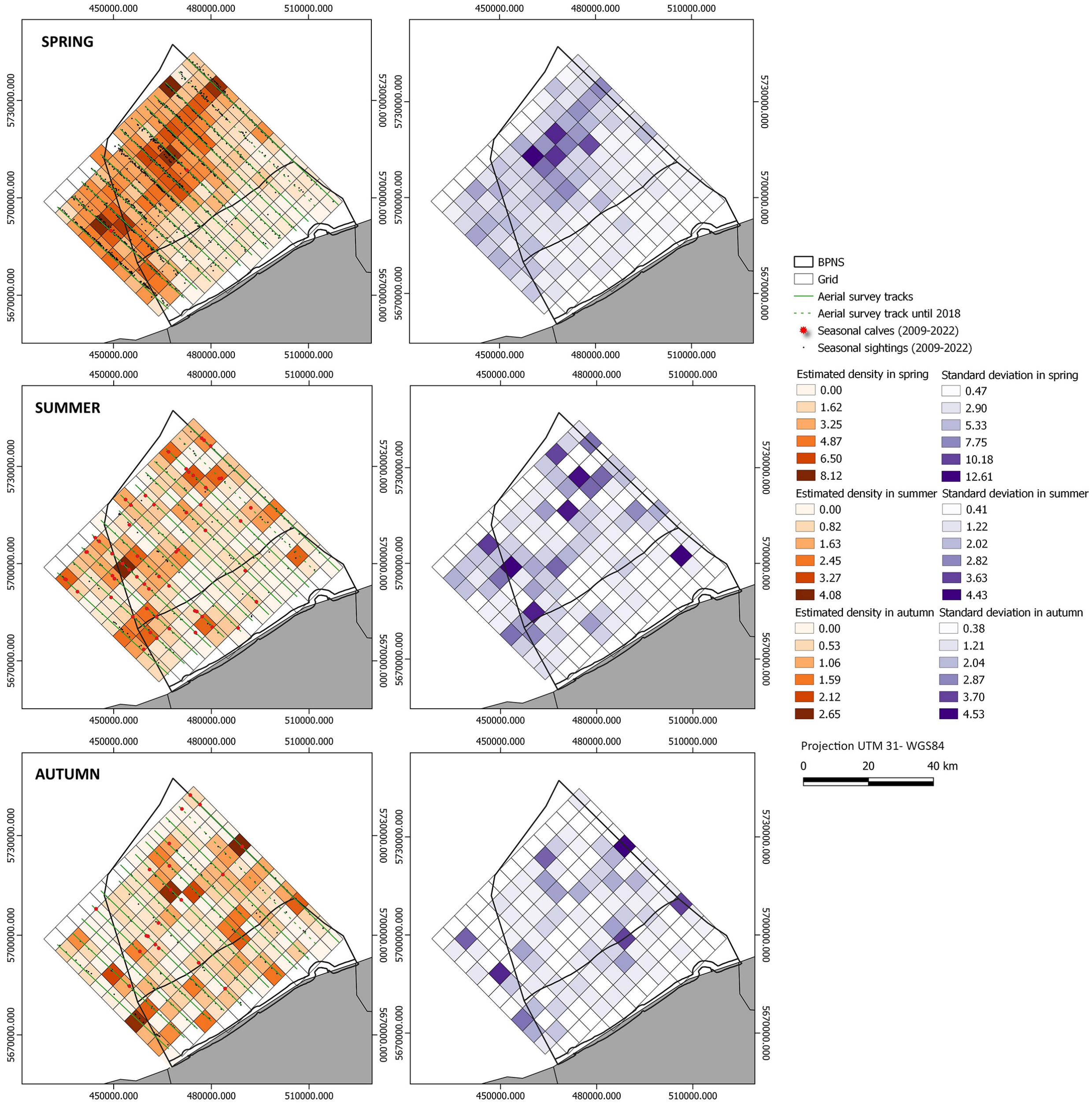

Figure 3. Seasonal density distribution maps of harbour porpoises (ind/km^2) (left) and associated variability (right) in the survey area in spring (March–May), summer (June–August), and autumn (September–November), calculated as mean between 2009 and 2022. Grid cell size: 5 × 5 km. Black dots indicate every observation; red stars indicate calves.

3.2. Factors influencing overall harbour porpoise distribution in the BPNS

The variability in the overall harbour porpoise distribution in the BPNS, represented by sighting rate (ind/nm) per grid cell, was statistically significantly explained by the season, the year, the latitudinal and longitudinal coordinates and their interaction, depth and marine traffic (Table 2). Predicted values in function of scaled explanatory variables and predicted regression curves are shown in Figure 4. Sighting rate (ind/nm) showed significant variability in function of years (overall p-value < 0.0001), with higher rates in 2011, 2014 and 2018. Significant variability was also found in function of season, with spring significantly different (p-value < 0.0001) from summer and autumn, which did not differ from each other.

Table 2. ZINB regression model with best performance used to describe overall distribution of sighting rates (ind/nm) in the BPNS as a function of broad-scale environmental and anthropogenic factors. Model estimated coefficients for the count model part are presented as mean ± SE. Chi-Square values and p-values are shown for each explanatory variable selected by the model.

		Negative binomial part (counts)			**Zero-inflated part (zeros)**		
Response variable	**Explanatory variable**	**Est. coeff. (mean ± SE)**	**Chi-Square**	**p-value**	**Est. coeff. (mean ± SE)**	**Chi-Square**	**p-value**
Sighting rate (ind/nm)	Shipping intensity	-0.06 ± 0.02	7.88	**0.005**	0.13 ± 0.05	6.91	**<0.0001**
	Latitude	0.10 ± 0.02	21.86	**<0.0001**	-0.17 ± 0.05	15.10	**<0.0001**
	Longitude	-0.20 ± 0.02	72.75	**<0.0001**	0.39 ± 0.05	70.20	**<0.0001**
	Interaction Lat*Long	0.07 ± 0.03	4.00	0.045	-0.16 ± 0.07	4.84	**0.028**
	Season-Spring	0.73 ± 0.07	113.81	**<0.0001**	-1.85 ± 0.14	181.00	**<0.0001**
	Season-Summer	0.05 ± 0.09		0.597	-0.77 ± 0.16		**<0.0001**
	Year	0.78 ± 0.23	188.28	**<0.0001**	-2.80 ± 0.40	234.41	**<0.0001**

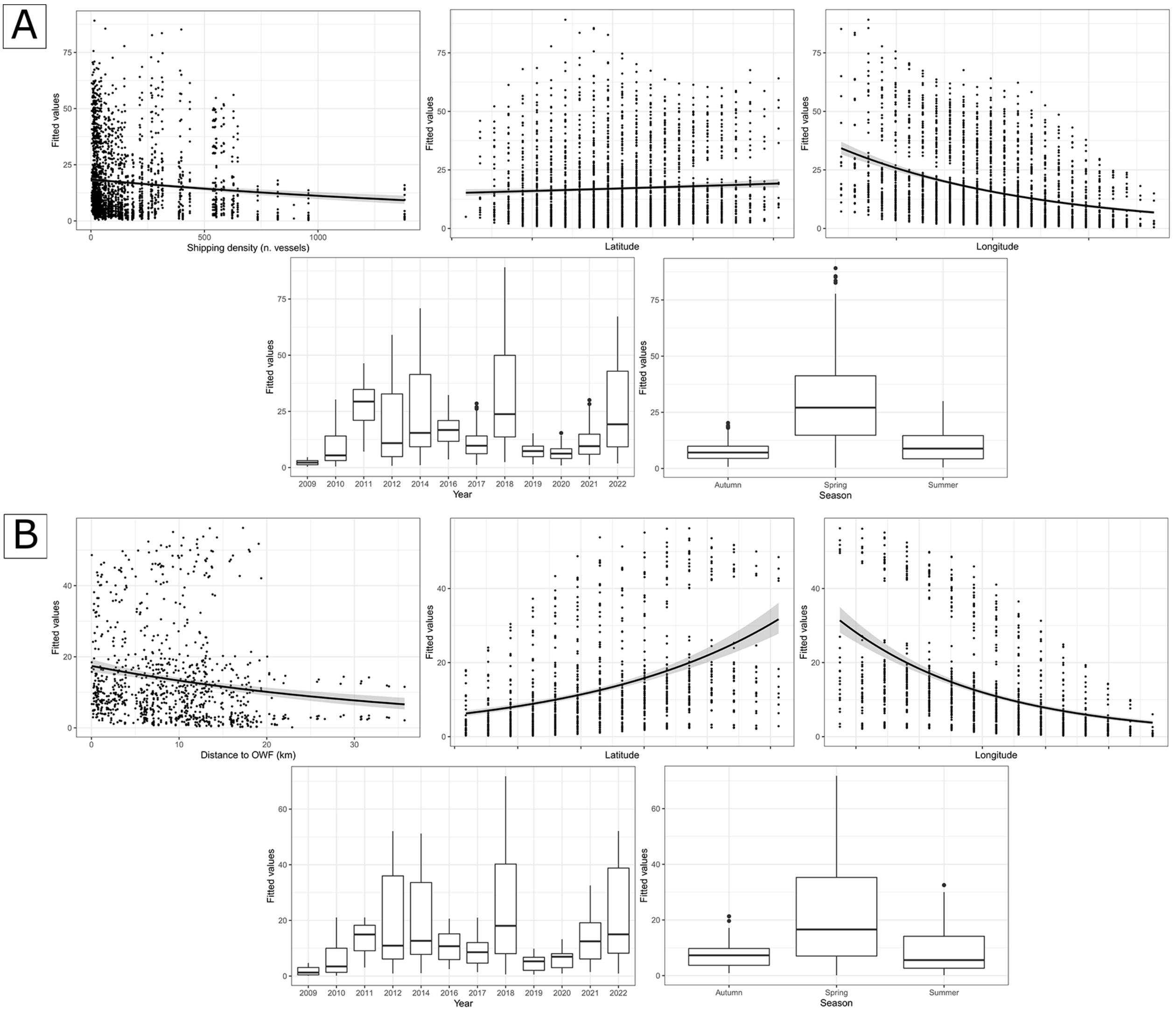

Figure 4. Negative binomial regression lines fitted by the best selected ZINB model between predicted sighting rates (fitted values) and each statistically significant explanatory variable for A) the analysis of the overall distribution of harbour porpoises in the BPNS, and B) the analysis of the distribution of harbour porpoises in the surrounding of the OWF concession areas as a function of distance to operational turbines, and other environmental drivers.

Table 3. ZINB regression model with best performance used to describe distribution of sighting rates (ind/nm) as a function of their distance to operational OWF and other local environmental factors, in the selected area of 2125 km^2 surrounding and including all OWF concession areas. Model estimated coefficients for the count model part are presented as mean ± SE. Chi-Square values and p-values are shown for each explanatory variable selected by the model.

		Negative binomial part (counts)			**Zero-inflated part (zeros)**		
Response variable	**Explanatory variable**	**Est. coeff. (mean ± SE)**	**Chi-Square**	**p-value**	**Est. coeff. (mean ± SE)**	**Chi-Square**	**p-value**
Sighting rate (ind/nm)	Distance to OWF	-0.18 ± 0.11	2.47	0.116	–	–	–
	Latitude	–	–	–	1.04 ± 0.21	44.97	**<0.0001**
	Longitude	-0.36 ± 0.05	34.44	**<0.0001**	0.47 ± 0.19	41.59	**<0.0001**
	Interaction Lat*Long	–	–	–	-1.36 ± 0.22	38.06	**<0.0001**
	Season-Spring	0.75 ± 0.11	91.84	**<0.0001**	-1.60 ± 0.24	52.05	**<0.0001**
	Season-Summer	0.27 ± 0.15		0.068	-0.25 ± 0.28		0.356
	Year	0.88 ± 0.65	97.97	**<0.0001**	-3.54 ± 0.66	124.63	**<0.0001**

Spatially, the overall distribution was significantly explained by latitude, longitude and their interaction. Sighting rate (ind/nm) increased with increasing latitude (p-value = 0.004) and decreasing longitude (p-value < 0.0001). Habitat type was highly correlated with depth and latitudinal and longitudinal coordinates, and therefore could not be tested in the same model. Habitat type was tested in an alternative competing model, but it did not significantly explain variability in density (data not shown). Finally, marine traffic significantly affected the overall distribution (p-value < 0.0001), with species density being higher with lower traffic intensity.

3.3. Effect of operational offshore wind farms on harbour porpoise distribution

The distribution of harbour porpoises in the vicinity of the OWFs in the BPNS was represented by sighting rates (ind/nm) per grid cell in a selected area of 1475 km^2 comprising and extending beyond the OWF concession areas. The spatio-temporal distribution could be explained by distance to the closest OWF, as well as by season, year, latitudinal and longitudinal coordinates, and their interaction (Table 3, Figure 4). The distance of each observation from the closest OWF (i.e., turbine) did not significantly explain part of the variability in the observed distribution despite the variable was retained in the best model (p-value = 0.116), with sighting rates marginally decreasing with increasing distance to the OWF. Sighting rates in the selected area varied from 0 ind/nm to 5.56 ind/nm per grid cell, and distances ranged from 42 m to 35 km. The remaining variability was significantly explained by the environmental factors, in line with the results of the analyses over the entire survey area. Sighting rate was significantly different among years (overall p-value < 0.0001), with higher rates in 2011, 2014 and 2018. Significant variability was found in function of season, where sighting rate was higher in spring (p-value = 0.0001) but similar in summer and autumn. Finally, the distribution was significantly explained by longitude and by the interaction between latitude and longitude. The sighting rate increased with decreasing longitude (p-value < 0.0001).

4. Discussion

Data were collected in a highly standardised way, as such allowing for analyses on a larger spatial scale, such as the southern and central North Sea (Gilles *et al.* 2016) and even the whole of the North Sea and adjacent Atlantic Ocean (Waggit *et al.* 2019). While the surveys conducted here were conducted on a relatively small spatial scale, they were conducted, compared to large-scale surveys such as SCANS surveys (Hammond *et al.* 2021), with a relatively high spatio-temporal resolution, as such allowing for, for instance, finer-scale temporal and spatial analyses.

4.1. Seasonal and interannual variability

Dedicated marine mammal surveys conducted between 2009 and 2022 revealed a clear seasonal and interannual variability in the abundance of harbour porpoises. Porpoises could be observed year-round, but the highest numbers were observed in spring. In spring, mean sighting rate was 54.9 ind/100 nm and mean estimated density was 2.78 ind/km^2. In summer, abundances were lower, but sightings of calves were most common, as females give birth in late spring or early summer (Gilles *et al.* 2009). Animals were mostly sighted alone or in small groups, leading to a small average group size (1.26 individuals). Observations are in line with those of Bouveroux *et al.* (2020) who recorded the highest numbers in the eastern English Channel in winter, and Haelters *et al*. (2013) and Scheidat *et al.* (2012), who recorded the highest abundances in winter and early spring in Belgium and the Netherlands. Due to reasons explained above, very few surveys were conducted in winter, but acoustic monitoring between 2010 and 2018 in the BPNS confirm this trend (Haelters *et al.* 2016; Augustijns 2018). Seasonal trends in relative spatial distribution were consistent across all surveys, but the number of sightings varied interannually, as also reported by Haelters *et al*. (2013). These variations are potentially the consequence of the small spatial scale of this analysis and the highly mobile nature of the species, but they may also be caused by large-scale natural variations in distribution, possibly instigated by changes in the distribution and abundance of the most important prey species (Hammond *et al.* 2013; Dähne *et al.* 2014; Geelhoed & Scheidat 2018).

4.2. Patterns in distribution

The seasonal distribution maps (Figure 3) display the standard deviation in sighting rate recorded in each grid cell across years. As it is based on a large number of data and as the resulting density distribution shows a similar pattern throughout the years, it is probable that this distributional pattern is the consequence of a combination of environmental and anthropogenic factors. Instead, the summer and autumn maps should be treated with more caution, and they could partly be the result of animals passing through the area, with an ad hoc location that is influenced to a lesser extent by local environmental conditions or effects of anthropogenic activities. Nevertheless, the observed spatial distribution in this study is in accordance with what is already known for the species in Belgian waters (Haelters *et al.* 2011, 2013). Harbour porpoises could be observed throughout the surveyed area. Coastal waters, within the 12 nautical mile zone, had the lowest sighting rate during the study period. A strong longitudinal gradient in distribution was confirmed in this study, with relatively high sighting rates in the north-western/western part of the study area, near and beyond the border with the French EEZ. This distribution was especially apparent in late winter and spring. In summer however, and more noticeably in autumn, animals were more evenly distributed across the BPNS.

The harbour porpoise is a highly mobile species with an extensive range within the North Sea. It feeds opportunistically on a large number of prey species. Therefore, the array of ecological and anthropogenic factors and their interactions driving the species' spatio-temporal distribution are hard to unravel (Gilles *et al.* 2016). Large-scale studies performed in one season and with a low temporal resolution, such as the SCANS surveys (Hammond *et al.* 2013, 2017) and small-scale surveys performed with a higher frequency, such as the ones described here, both have their value in unravelling spatio-temporal patterns, as a first step towards understanding the drivers of the patterns. For the harbour porpoise, a small, warm-blooded mammal that lives in a relatively cold environment, the availability of food is key to its survival (Kastelein *et al.* 1997;

IJsseldijk, 2021). Therefore, one should be able to explain, at least partly, its occurrence and distribution by the distribution and availability of its preferred prey (Lambert *et al.* 2016; Ransijn *et al.* 2019; Nachtsheim *et al*. 2021). In turn, prey distribution is influenced by several underlying ecological factors (Skov & Thomsen 2008; Ransijn *et al.* 2019). Given the frequently very high densities recorded locally in the survey area in this study, this area should be considered as forming part of a highly valuable area for the species, with also in adjacent French and nearby Dutch waters frequent records of high densities of porpoises (Geelhoed & Scheidat 2018; Bouveroux *et al.* 2020).

In this study, statistical analyses revealed that latitude, longitude and their interaction accounted for a large part of the distribution variability of harbour porpoises. These factors act as proxies for underlying ecological gradients that are not directly accounted for in the analysis (IAMMWG 2015). Harbour porpoises inhabit dynamic shallow waters of continental shelves which host suitable habitat conditions for feeding (Skov & Thomsen 2008; Lambert *et al.* 2016). Water current speed was not included in this study, although in studies it was reported to be a significant driver for the presence of porpoises, as stronger currents can promote primary productivity and prey abundance (Bouveroux *et al.* 2020). In the BPNS, prey distribution may be influenced by the underlying benthic habitat type. Habitat type was considered in the analyses but did not significantly explain variability in harbour porpoise distribution, potentially because of the very different spatial resolution by which habitat type and porpoise distribution are described, and/or because of the fact that part of the diet of porpoises consists of fish that are pelagic, or at least partially pelagic (e.g., sandeels), as such independent of the habitat type used in the analysis. However, as part of the diet of porpoises consists of pelagic fish species, with an occurrence partly independent of local benthic habitat type, local concentrations and seasonal movement patterns of porpoises may be the consequence of the presence and migration of these prey species.

Independently of the considerations mentioned above, it is clear that at least in late winter and spring porpoises occurred in a relatively high density in the western and northwestern part of the study area, with a strong overlap with the Vlaamse Banken SAC (MSP 2020–2026). This SAC was established in 2012 to protect an ecosystem of sandbanks that included Habitat 1110 and 1170, and to conserve some of the richest remnants of gravel beds in Belgian waters known to occur in their gullies (the Hinder Banks) (Houziaux *et al.* 2008; Pecceu *et al.* 2016, 2021; Montereale-Gavazzi *et al.* 2023). The area is known to sustain a complex food web that includes species of commercial interest, and it is therefore subject of a sustained exploitation by fisheries (Pecceu *et al.* 2021).

4.3. Influence of shipping

Marine traffic is the primary source of underwater noise pollution worldwide and it is known to cause behavioural responses in cetaceans (e.g., Gomez *et al.* 2016; Avila *et al.* 2018; Pirotta *et al.* 2018). Harbour porpoises have been seen to fluke, dive and interrupt foraging and even echolocation when encountering noisy vessels (Wisniewska *et al.* 2018). Noise avoidance and behavioural changes in foraging may particularly be affecting the species' fitness and survival as it extensively uses echolocation for its incessant feeding pace (Wisniewska *et al.* 2018).

In this study, shipping intensity was found to significantly explain the distribution of porpoises, with a decline in sighting rate with increasing shipping traffic. The highest densities of porpoises were found within and around the Vlaamse banken SAC which is void of shipping lanes, and therefore has lower traffic intensities compared to other areas.

However, such results should be considered carefully, as other factors could affect this relationship. For instance, as the data on shipping relies on AIS detections, the anchorage area in front of Ostend is overrepresented. Moreover, the area around the port of Zeebrugge has a very high shipping density, and a low harbour porpoise density, but the latter could also be due to a less suitable habitat, with for instance a higher turbidity and/or a lower density of suitable prey species.

4.4. Influence of operational OWFs

Underwater noise generated by operating OWF is another source of anthropogenic noise of which the effect on marine mammal behaviour is still unclear. While marine mammals may be attracted to operational OWFs because of a higher food availability due to artificial reef effects and the absence of fishing, they may also avoid the area because of the increased underwater noise from the turbines and the vessel traffic in their surroundings (Tougaard *et al.* 2005; Scheidat *et al.* 2011; Haelters *et al.* 2013). In the Netherlands, an increase in the number of harbour porpoises was detected by acoustic devices inside the operational OWF Egmond aan Zee (OWEZ; Scheidat *et al*. 2009, 2011). The increase in food availability and/or a sheltering effect from fisheries disturbance were proposed as a possible explanation for this. In contrast, no differences were detected inside vs outside the operational OWF Prinses Amalia windfarm (van Polanen Petel *et al.* 2012). Similar results were obtained with aerial surveys over the Borssele OWF, where no conclusive support for either avoidance or attraction was found (Collier *et al.* 2022). In Denmark, no difference in number of animals was detected by acoustic devices inside and outside the OWF Horns Reef, with a complete recovery to baseline levels observed within one year of the operational phase (Tougaard *et al.* 2006b; Blew *et al.* 2006). In contrast, a long-term negative effect from the construction extending into the operational phase on the species' occurrence was suspected for the OWF Nysted: acoustic detections had returned to baseline levels in the nearby reference area after two years, but within the OWF itself they had not recovered after 10 years (Tougaard *et al.* 2006a; Teilmann & Carstensen 2012). In the UK, no difference in the number of harbour porpoises between the preconstruction and the operational phase was observed during ship surveys at the Robin Rigg OWF in UK (Vallejo *et al.* 2017). The quick return of animals after the construction phase ended was hypothetically linked to habitat quality: animals may display a shortened avoidance behaviour if the habitat is of high quality for feeding. In Germany, the operational OWF Alpha Ventus did not seem to affect harbour porpoise distribution, which was apparently driven by a large-scale natural variation (Dähne *et al.* 2014). Overall, contrasting results have been potentially linked to site-specificities such as differences in OWF features, or underlying ecological aspects driving the harbour porpoise response (Scheidat *et al.* 2009; van Polanen Petel *et al.* 2012), underlining the importance of region-specific investigations.

As OWFs are densely clustered and localised in the eastern part of the BPNS, at the border with the Dutch EEZ, they have a strong spatial correlation with other environmental gradients in the BPNS such as latitude, longitude and depth. The results of our analysis revealed that variability in harbour porpoise distribution was not significantly explained by the distance of each observation to the closest OWF, despite the variable being retained in the best model with marginally higher sighting rates at decreasing distances from the turbines. A similar density of harbour porpoises outside and within the OWF area could, in theory, be due to the trade-offs between the introduced underwater noise of operational wind turbines and the availability of suitable prey. However, the results presented here may have been confounded by presumptions and analytical constraints. Although the analysis was performed in a selected area to avoid the covariates of spatial distribution affecting the results, the natural gradient in the distribution of harbour porpoises across the BPNS may still have influenced the analysis. As discussed above, porpoises were generally distributed according to an east-west gradient, and perhaps favour more westerly waters, independently of the presence of OWFs. Furthermore, the survey track directly above the OWFs could not be flown from 2019 onwards. The lower coverage of the OWF area may have influenced the analysis. All in all, this study cannot come to conclusion about the effect of operational OWFs on the occurrence of harbour porpoises in Belgian waters. This may be due to the method being not very suitable. Visual aerial surveys typically generate data in a low temporal and spatial resolution, but over a wide area, and are as such considered suitable to assess distribution and abundance patterns of highly mobile species. However, as operational OWFs generate relatively low underwater noise levels, and as underwater noise levels are also generated through other activities, it is likely that effects play at a smaller spatial scale than can be detected through aerial surveys (which remain useful for assessing activities potentially generating wider-ranging effects, such as piling). Using passive acoustic monitoring with a sufficiently dense distribution of sensors within and outside OWFs, tagging animals or performing digital aerial surveys in a higher temporal and spatial resolution (Williamson *et al.* 2016; Collier *et al.* 2022) may be more suitable methods to reveal effects.

Despite the limitations of the analyses, the results presented in this study contribute valuable information to the discussions on the potential implications of current and future OWF development and other human activities at sea for the harbour porpoise wellbeing.

5. Conclusions

The aerial survey data collected between 2009 and 2022 revealed a seasonal pattern in the presence of harbour porpoises in the survey area, with the highest densities recorded during late winter and spring. They also revealed a high temporal variability, with years with very high and years with much lower densities. The results of the analyses, although conducted on a small scale considering the high mobility of the species and its wide dispersal, still clearly showed that in spring harbour porpoises were most common in the northern and western part of the survey area, with especially in the northwestern part frequently very high densities. The study also suggests that shipping intensity was a factor negatively influencing densities on a local scale.

The density of harbour porpoises near operational wind turbines was relatively low. The background for this could have been natural, while it could also have been partly caused by the presence of wind turbines and the related activities in and near the wind farm. It was concluded that the use of aerial surveys, in the way they were conducted, is probably not the best method to reveal possible small-scale changes in porpoise distribution due to the presence of offshore wind turbines. Changes in distribution outside and inside an OWF area may be hard to distinguish from larger scale spatio-temporal variability driven by larger scale environmental gradients.

The results of this study are useful for informing the management of current and future activities in Belgian waters, such as fisheries and renewable energy development, and provide a basis for appropriate measures needed in light of the ever-increasing human presence at sea.

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References

ASCOBANS 2009. *Conservation Plan for Harbour Porpoises (Phocoena phocoena) in the North Sea*. ASCOBANS MOP6/Doc.7-02 (AC).

Augustijns, T. 2018. *Harbour Porpoises in the Belgian Part of the North Sea: Using Passive Acoustic Monitoring to Determine Spatio-temporal Patterns in Distribution and Feeding Behaviour*. MSc Thesis. Faculty of Sciences, Biology Department, Research Group Marine Biology: Gent. 47 pp.

Avila, I.C., Kaschner, K. & Dormann, C.F. 2018. Current global risks to marine mammals: Taking stock of the threats. *Biological Conservation* 221: 44–58. https://doi.org/10.1016/j.biocon.2018.02.021

Bearzi, G., Piwetz, S. & Reeves, R.R. 2019. Odontocete adaptations to human impact and vice versa. *Ethology and Behavioral Ecology of Marine Mammals* 211–235. https://doi.org/10.1007/978-3-030-16663-2_10

Blew, J., Diederichs, A., Grünkorn, T., Hoffmann, M. & Nehls, G. 2006. *Investigations of the Bird Collision Risk and the Responses of Harbour Porpoises in the Offshore Wind Farms at Horns Rev, North Sea and Nysted, Baltic Sea, in Denmark*. Status report 2005 to the Environmental Group. Hamburg, BioConsult SH.

Bouveroux, T., Waggitt, J.J., Belhadjer, A., Cazenave, P.W., Evans, P.G.H. & Kiszka, J.J. 2020. Modelling fine-scale distribution and relative abundance of harbour porpoises in the Southern Bight of the North Sea using platform-of-opportunity data. *Journal of the Marine Biological Association of the United Kingdom* 100 (3): 1–9. https://doi.org/10.1017/s0025315420000326

Brandt, M., Dragon, A., Diederichs, A., Bellman, M.A., Wahl, V., Piper, W., Nabe-Nielsen, J., Nehls, G. 2018. Disturbance of harbour porpoises during construction of the first seven offshore wind farms in Germany. *Marine Ecology Progress Series* 596: 213–232. https://doi.org/10.3354/meps12560

Brownell, Jr, R.L., Reeves, R.R., Read, A.J., Smith, B.D. *et al*. 2019. Bycatch in gillnet fisheries threatens Critically Endangered small cetaceans and other aquatic megafauna. *Endangered Species Research* 40: 285–296. https://doi.org/10.3354/esr00994

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. & Thomas, L. 2001. *Introduction to Distance Sampling: Estimating Abundance of Biological Populations*. Oxford University Press, London.

Camphuysen, K.C.J. 2011. Recent trends and spatial patterns in nearshore sightings of harbour porpoises (*Phocoena phocoena*) in the Netherlands (Southern Bight, North Sea), 1990–2010. *Lutra* 54: 39–47.

Carstensen, J., Henriksen, O.D. & Teilmann, J. 2006. Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). *Marine Ecology Progress Series* 321: 295–308.

Collier, M.P., Middelveld, R.P., van Bemmelen. R.S.A., Weiß, F., Irwin, C.G. & Fijn, R.C. 2022. High-definition bird and marine mammal aerial survey image collection in Borssele: First-year report. *Waardenburg Ecology Report* 22-272. Waardenburg Ecology, Culemborg.

Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora, CONSIL, 206 OJ L (1992). http://data.europa.eu/eli/dir/1992/43/oj/eng

Dähne, M., Gilles, A., Lucke, K., Peschko, V., Adler, S., Krügel, K., Sundermeyer, J. & Siebert, U. 2013. Effects of pile-driving on harbour porpoises (*Phocoena phocoena*) at the first offshore wind farm in Germany. *Environmental Research Letters* 8 (2): e025002. https://doi.org/10.1088/1748-9326/8/2/025002

Dähne, M., Peschko, V., Gilles, A., Lucke, K., Adler, S., Ronnenberg, K. & Siebert, U. 2014. Marine mammals and windfarms: Effects of alpha ventus on harbour porpoises. *Ecological Research at the Offshore Windfarm Alpha Ventus*: 133–149. https://doi.org/10.1007/978-3-658-02462-8_13

Dähne, M., Tougaard, J., Carstensen, J., Rose, A. & Nabe-Nielsen, J. 2017. Bubble curtains attenuate noise from offshore wind farm construction and reduce temporary habitat loss for harbour porpoises. *Marine Ecology Progress Series* 580: 221–237. https://doi.org/10.3354/meps12257

Dénes, F.V., Silveira, L.F. & Beissinger, S.R. 2015. Estimating abundance of unmarked animal populations: accounting for imperfect detection and other sources of zero inflation. *Methods in Ecology and Evolution* 6 (5): 543–556. https://doi.org/10.1111/2041-210x.12333

Degraer, S., Brabant, R. & Vanaverbeke, J. (eds). 2023. EDEN 2000 – Exploring options for a nature-proof development of offshore wind farms inside a Natura 2000 area. *Memoirs on the Marine Environment*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 440 pp.

Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds). 2021. Environmental impacts of offshore wind farms in the Belgian Part of the North Sea: Attraction, avoidance and habitat use at various spatial scales. *Memoirs on the Marine Environment*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 104 pp.

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*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 106 pp.

Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive) (Text with EEA relevance), EP, CONSIL, 164 OJ L (2008). http://data.europa.eu/eli/dir/2008/56/oj/eng

Geelhoed, S.C.V. & Scheidat, M. 2018. Abundance of harbour porpoises (*Phocoena phocoena*) on the Dutch Continental Shelf, aerial surveys 2012–2017. *Lutra* 61 (1): 127–136.

Geelhoed, S.C.V., Scheidat, M., van Bemmelen, R.S.A. & Aarts, G. 2013. Abundance of harbour porpoises (*Phocoena phocoena*) on the Dutch Continental Shelf, aerial surveys in July 2010–March 2011. *Lutra* 56 (1): 45–57.

Gilles, A., Scheidat, M. & Siebert, U. 2009. Seasonal distribution of harbour porpoises and possible interference of offshore wind farms in the German North Sea. *Marine Ecology Progress Series* 383: 295–307. https://doi.org/10.3354/meps08020

Gilles, A., Viquerat, S., Becker, E.A., Forney, K.A., Geelhoed, S.C.V., Haelters, J. Nabe-Nielsen, J., Scheidat, M., Siebert, U., Sveegaard, S., van Beest, F.M., van Bemmelen, R. & Aarts, G. 2016. Seasonal habitat-based density models for a marine top predator, the harbour porpoise, in a dynamic environment. *Ecosphere* 7 (6): e01367. https://doi.org/10.1002/ecs2.1367

Gomez, C., Lawson, J.W., Wright, A.J., Buren, A.D., Tollit, D. & Lesage, V. 2016. A systematic review on the behavioural responses of wild marine mammals to noise: the disparity between science and policy. *Canadian Journal of Zoology* 94 (12): 801–819. https://doi.org/10.1139/cjz-2016-0098

Haelters, J., 2009. Monitoring of marine mammals in the framework of the construction and exploitation of offshore windfarms in Belgian marine waters. *In*: Degraer, S. & Brabant, R. (eds), *Offshore Windfarms in the Belgian Part of the North Sea: State of the Art After Two Years of Environmental Monitoring*: 237–266. Royal Belgian Institute of Natural Sciences, Brussels, Belgium.

Haelters, J., Kerckhof, F., Jacques, T.G. & Degraer, S. 2011. The harbour porpoise *Phocoena phocoena* in the Belgian part of the North Sea: trends in abundance and distribution. *Belgian Journal of Zoology* 141: 75–84.

Haelters, J., Kerckhof, F., Toussaint, E., Jauniaux, T. & Degraer, S. 2012. *The Diet of Harbour Porpoises Bycaught or Washed Ashore in Belgium, and Relationship with Relevant Data from the Strandings Database*. Royal Belgian Institute of Natural Sciences (RBINS/MUMM) and Federal Public Service for Health, Food Chain Safety and Environment (Marine Environment), Brussels, 45 p.

Haelters, J., Vigin, L. & Degraer, S. 2013. Attraction of harbour porpoises to offshore wind farms: what can be expected? *In*: Degraer, S., Brabant, R. & Rumes, B. (eds), *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Learning from the Past to Optimize Future Monitoring Programmes*. Royal Belgian Institute of Natural Sciences, Operational Directorate Natural Environment, Marine Ecology and Management Section, 239 pp.

Haelters, J., Dulière, V., Vigin, L. & Degraer, S. 2014. Towards a numerical model to simulate the observed displacement of harbour porpoises *Phocoena phocoena* due to pile driving in Belgian waters. *Hydrobiologia* 756 (1): 105–116. https://doi.org/10.1007/s10750-014-2138-4

Haelters, J., Rumes, B., Vanaverbeke, J. & Degraer, S. 2016. Seasonal and interannual patterns in the presence of harbour porpoises (*Phocoena phocoena*) in Belgian waters from 2010 to 2015 as derived from passive acoustic monitoring. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds), *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Environmental Impact Monitoring Reloaded*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management Section, 287 pp.

Hammond, P.S., Macleod, K., Berggren, P., Borchers, D.L., *et al*. 2013. Cetacean abundance and distribution in European Atlantic shelf waters to inform conservation and management. *Biological Conservation* 164: 107–122. https://doi.org/10.1016/j.biocon.2013.04.010

Hammond, P.S., Lacey, C., Gille, A., Viquerat, S., Börjesson, P., Macleod, K., Ridoux, V., Santos, M.B., Scheidat, M., Teilmann, J., *et al.* 2021. *Estimates of Cetacean Abundance in European Atlantic Waters in Summer 2016 from the SCANS-III Aerial and Shipboard Surveys*. Final Report 2021, 42 pp.

Hawkins, E.R., Harcourt, R., Bejder, L., Brooks, L.O., Grech, A., Christiansen, F., Marsh, H., Harrison, P. L. 2017. Best practice framework and principles for monitoring the effect of coastal development on marine mammals. *Frontiers in Marine Science* 4: e00059. https://doi.org/10.3389/fmars.2017.00059

Houziaux J.-S., Kerckhof, F., Degrendele, K., Roche, M. & Norro, A. 2008. *The Hinder Banks: yet an Important Region for Belgian Marine Biodiversity?* Final report HINDERS. Belgian Science Policy Office, 249 pp.

IAMMWG 2015. *The Use of Harbour Porpoise Sightings Data to Inform the Development of Special Areas of Conservation in UK Waters*. JNCC Report No. 565, JNCC Peterborough.

IJsseldijk, L.L. 2021. *Living on a Knife-edge. Unravelling Harbour Porpoise Health through Multidisciplinary and Cross-border Approaches*. PhD Thesis, Utrecht University.

IJsseldijk, L.L., ten Doeschate, M., Brownlow, A., Davison, N., Deaville, R., Galatius, A., Gilles, A., Haelters, J., Jepson, P., Keijl, G., Kinze, K., Olsen, M.T., Siebert, U., Thøstesen, C., van den Broek, J., Gröne, A. & Heesterbeek, H. 2020. Spatiotemporal mortality and demographic trends in a small cetacean: Strandings to inform conservation management. *Biological Conservation* 249: e108733. https://doi.org/10.1016/j.biocon.2020.108733

Kastelein, R.A., Hardeman, J. & Boer, H. 1997. Food consumption and body weight of harbour porpoises (*Phocoena phocoena*). *In*: Read, A.J., Wiepkema, P.R. & Nachtigall, P.E. (eds), *The Biology of the Harbour Porpoise*: 217–233. De Spil Publishers, Woerden, The Netherlands.

Lambert, C., Pettex, E., Dorémus, G., Laran, S., Stéphan, E., Van Canneyt, O. & Ridoux, V. 2016. How does ocean seasonality drive habitat preferences of highly mobile top predators? Part II: the eastern North-Atlantic. *Deep-Sea Research Part II* 141: 133–154. https://doi.org/10.1016/j.dsr2.2016.06.011

Lucke, K., Lepper, P.A., Blanchet, M-A. & Siebert, U. 2011. The use of an air bubble curtain to reduce the received sound levels for harbor porpoises (*Phocoena phocoena*). *The Journal of the Acoustical Society of America* 130 (5): 3406–3412. https://doi.org/10.1121/1.3626123

Marine Spatial Planning 2020–2026. *Something is Moving at Sea*.

Minton, G., Folegot, T., Cosandey-Godin, A., Jacob, T., Lancaster, M. & Ushio, M. 2021. *Shipping and Cetaceans: A Review of Impacts and Mitigation Options for Policy Makers and Other Stakeholders*. WWF Report. 72 pages.

Montereale-Gavazzi G., Degraer, S. & Van Lancker, V. 2023. Predictive modelling of seafloor surficial gravel distribution within a new wind farm concession area: Belgian part of the North Sea. *In*: Degraer, S., Brabant, R. & Vanaverbeke, J. (eds), *EDEN 2000 – Exploring Options for a Nature-proof Development of Offshore Wind Farms inside a Natura 2000 Area. Memoirs on the Marine Environment*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 440 pp.

Nachtsheim, D.A., Viquerat, S., Ramírez-Martínez, N.C., Unger, B., Siebert, U., Gilles, A. 2021. Small cetacean in a human high-use area: Trends in harbor porpoise abundance in the North Sea over two decades. *Frontiers in Marine Sciences* 7: e606609. https://doi.org/10.3389/fmars.2020.606609

OSPAR, 2023. *OSPAR Quality Status Report 2023*. OSPAR Commission.

Pecceu, E., Hostens, K. & Maes, F. 2016. Governance analysis of MPAs in the Belgian part of the North Sea. *Marine Policy* 71: 265–274. https://doi.org/10.1016/j.marpol.2015.12.017

Pecceu, E., Paoletti, S., Van Hoey, G., Vanelslander, B., Verlé, K., Degraer, S., Van Lancker, V., Hostens, K. & Polet, H. 2021. *Scientific Background Report in Preparation of Fisheries Measures to Protect the Bottom Integrity and the Different Habitats within the Belgian Part of* *the North Sea*. Report in preparation for the Federal Public Service Health, Food Chain Safety and Environment.

Peschko, V., Ronnenberg, K., Siebert, U. & Gilles, A. 2016. Trends of harbour porpoise (*Phocoena phocoena*) density in the southern North Sea. *Ecological Indicators* 60: 174–183. https://doi.org/10.1016/j.ecolind.2015.06.030

Pirotta, V., Grech, A., Jonsen, I.D., Laurance, W.F. & Harcourt, R.G. 2018. Consequences of global shipping traffic for marine giants. *Frontiers in Ecology and the Environment*. https://doi.org/10.1002/fee.1987

Putland, R.L., de Jong, C.A.F., Binnerts, B., Farcas, A., Merchant, N.D. 2022. Multi-site validation of shipping noise maps using field measurements. *Marine Pollution Bulletin* 179: e113733. https://doi.org/10.1016/j.marpolbul.2022.113733

Ransijn, J.M., Booth, C. & Smout, S.C. 2019. *A Calorific Map of Harbour Porpoise Prey in the North Sea*. JNCC Report No. 633. JNCC, Peterborough, ISSN 0963 8091.

RStudio Team 2020. *RStudio: Integrated Development for R*. RStudio, PBC, Boston.

Rumes, B. & Degraer, S. 2020. Fit for porpoise? Assessing the effectiveness of underwater sound mitigation measures. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds), *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Empirical Evidence Inspiring Priority Monitoring, Research and Management. Memoirs on the Marine Environment*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 131 p

Rumes, B. & Zupan, M. 2021. Effects of the use of noise-mitigation during offshore pile drive on harbour porpoise (*Phocoena phocoena*). *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds), *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Attraction, avoidance and habitat use at various spatial scales. Memoirs on the Marine Environment*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 104 pp.

Rumes, B., Brabant, R. & Vigin, L. 2022. Offshore renewable energy in the Belgian Part of the North Sea. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds), *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*. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels, 106 pp.

Schallier, R. & Van Roy, W. 2014. Oil pollution in and around the waters of Belgium. *Oil Pollution in the North Sea* 93–115. https://doi.org/10.1007/698_2014_330

Scheidat, M., Aarts, G., Bakker, A., Brasseur, S., Carstensen, J., van Leeuwen, P.W., Leopold, M., van Polanen Petel, T., Reijnders, P., Teilmann, J., Tougaard, J., Verdaat, H. 2009. *Assessment of the Effects of the Offshore Wind Farm Egmond aan Zee (OWEZ) for Harbour Porpoise (comparison T0 and T1)*. Final Report, pp. 45.

Scheidat, M., Tougaard, J., Brasseur, S., Carstensen, J., van Polanen Petel, T., Teilmann, J. & Reijnders, P. 2011. Harbour porpoises (*Phocoena phocoena*) and wind farms: a case study in the Dutch North Sea. *Environmental Research Letters* 6 (2): e025102. https://doi.org/10.1088/1748-9326/6/2/025102

Scheidat, M., Verdaat, H. & Aarts, G. 2012. Using aerial surveys to estimate density and distribution of harbour porpoises in Dutch waters. *Journal of Sea Research* 69: 1–7. https://doi.org/10.1016/j.seares.2011.12.004

Schoeman, R.P., Patterson-Abrolat, C. & Plön, S. 2020. A Global Review of Vessel Collisions With Marine Animals. *Frontiers in Marine Science* 7: e292. https://doi.org/10.3389/fmars.2020.00292

Skov, H. & Thomsen, F. 2008. Resolving fine-scale spatio-temporal dynamics in the harbour porpoise *Phocoena phocoena*. *Marine Ecology Progress Series* 373: 173–186. https://doi.org/10.3354/meps07666

Teilmann, J. & Carstensen, J. 2012. Negative long term effects on harbour porpoises from a large scale offshore wind farm in the Baltic—evidence of slow recovery. *Environmental Research Letters* 7 (4): e045101. https://doi.org/10.1088/1748-9326/7/4/045101

Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R.B., Marques T.A. & Burnham, K.P. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology* 47: 5–14. https://doi.org/10.1111/j.1365-2664.2009.01737.x

Thompson, P.M., Brookes, K.L., Graham, I.M., Barton, T.R., Needham, K., Bradbury, G. & Merchant, N.D. 2013. Short-term disturbance by a commercial two-dimensional seismic survey does not lead to long-term displacement of harbour porpoises. Pr*oceedings of the Royal Society B: Biological Sciences* 280 (1771): e20132001. https://doi.org/10.1098/rspb.2013.2001

Tougaard, J., Carstensen, J., Teilmann, J. & Bech, N.I. 2005. *Effects of the Nysted Offshore Wind Farm on Harbour Porpoises*. Technical report to Energi E2 A/S. Roskilde, Denmark, NERI.

Tougaard, J., Carstensen, J., Bech, N.I. & Teilmann, J. 2006a. *Final Report on the Effect of Nysted Offshore Wind Farm on Harbour Porpoises*. Annual report to Energi E2 A/S. Roskilde, Denmark, NERI.

Tougaard, J., Carstensen, J., Wisz, M.S., Jespersen, M., Teilmann, J. & Bech, N.I. 2006b. *Harbour Porpoises on Horns Reef Effects of the Horns Reef Wind Farm*. Final report to Vattenfall A/S. Roskilde, Denmark, NERI.

Vallejo, G.C., Grellier, K., Nelson, E.J., McGregor, R.M., Canning, S.J., Caryl, F.M. & McLean, N. 2017. Responses of two marine top predators to an offshore wind farm. *Ecology and Evolution* 7 (21): 8698–8708. https://doi.org/10.1002/ece3.3389

Van Nieuwenhove, A., Van Roozendael, B., Scheldeman, K., Merveille, J.-B., Haelters, J., Van Roy, W. & Schallier, R. 2023. *30 Years of Belgian North Sea Aerial Surveillance – Evolution, Trends, and Developments*. Royal Belgian Institute of Natural Sciences, Operational Directorate Natural Environment, Brussels, 120 pp.

van Polanen Petel, T., Geelhoed, S. & Meesters, E. 2012. *Harbour Porpoise Occurrence in Relation to the Prinses Amaliawindpark*. IMARES Wageningen UR. Report Number 177/10.

Verfuss, U.K., Sparling, C.E., Arnot, C., Judd, A. & Coyle, M. 2016. Review of offshore wind farm impact monitoring and mitigation with regard to marine mammals. *Advances in Experimental Medicine and Biology*: 1175–1182. https://doi.org/10.1007/978-1-4939-2981-8_147

Waggit, J., Evans, P.G.H., Andrade, J., Banks, A., *et al*. 2019. Distribution maps of cetacean and seabird populations in the North-East Atlantic. *Journal of Applied Ecology* 57 (2): 253–269. https://doi.org/10.1111/1365-2664.13525

Williamson, L.D., Brookes, K.L., Scott, B.E., Graham, I.M., Bradbury, G., Hammond, P.S. & Thompson, P.M. 2016. Echolocation detections and digital video surveys provide reliable estimates of the relative density of harbour porpoises. *Methods in Ecology and Evolution* 7 (7): 762–769. https://doi.org/10.1111/2041-210x.12538

Wisniewska, D.M., Johnson, M., Teilmann, J., Siebert, U., Galatius, A., Dietz, R. & Madsen, P.T. 2018. High rates of vessel noise disrupt foraging in wild harbour porpoises (*Phocoena phocoena*). *Proceedings of the Royal Society B: Biological Sciences* 285 (1872): e20172314. https://doi.org/10.1098/rspb.2017.2314

Xu, W., Liu, Y., Wu, W., Dong, Y., Lu, W., Liu, Y., Zhao, B., Li, H., Yang, R. 2020. Proliferation of offshore wind farms in the North Sea and surrounding waters revealed by satellite image time series. *Renewable and Sustainable Energy Reviews* 133: e110167. https://doi.org/10.1016/j.rser.2020.110167

Yang, S., Harlow, L.I., Puggioni, G. & Redding, C.A. 2017. A comparison of different methods of zero-inflated data analysis and an application in health surveys. *Journal of Modern Applied Statistical Methods* 16 (1): 518–543. https://doi.org/10.22237/jmasm/1493598600

Zipkin, E.F., Leirness, J.B., Kinlan, B.P., O'Connell, A.F. & Silverman, E.D. 2014. Fitting statistical distributions to sea duck count data: Implications for survey design and abundance estimation. *Statistical Methodology* 17: 67–81. https://doi.org/10.1016/j.stamet.2012.10.002

Zuur, A.F., Ieno, E.N., Walker, N., Saveliev, A.A. & Smith, G.M. 2009. *Mixed Effects Models and Extensions in Ecology with R. Statistics for Biology and Health*. https://doi.org/10.1007/978-0-387-87458-6

Zuur, A.F., Ieno, E.N. & Elphick, C.S. 2010. A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution* 1 (1): 3–14. https://doi.org/10.1111/j.2041-210x.2009.00001.x

CHAPTER 5

SEABIRDS AND OFFSHORE WIND FARMS – DISPLACEMENT MONITORING 2.0

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Abstract

This report sets out the count results collected in the period February 2021 to April 2023, following a revised monitoring design. The results presented at this stage need to be considered as indicative since more data and advanced spatial modelling are needed to detect seabird avoidance or attraction effects with sufficient confidence. Nevertheless, making use of a limited dataset and mean values only, it is interesting to see that the results are often in line with what has been found before and/or elsewhere, such as indications of attraction effects for great black-backed gull and great cormorant, and of avoidance by northern gannet. On the other hand, our results no longer seem to indicate (strong) avoidance of common guillemots and even increased numbers of razorbills between the turbines. It is yet unclear whether the results for auks indicate habituation rather than a specific habitat preference. The new monitoring strategy not only aims to detect displacement responses and is also designed to detect disturbance distances (with regard to migration corridors) and the effect of turbine density on seabird displacement levels. Ultimately, considering the huge expansion of offshore wind farm development in the North Sea, this monitoring approach hopes to fill important knowledge gaps and to inform future planning decisions regarding wind farm configuration and mitigation of impact on seabirds.

1. Introduction

Since the end of 2020, the Belgian offshore wind farm (OWF) concession zone is fully operational and now holds 399 turbines. As this is a very different situation compared to the isolated clusters of turbines that were present in the period 2009–2016, a new seabird monitoring program was initiated in February 2021. While continuing to assess species-specific displacement effects, we will also look for temporal trends and spatial patterns in wind farm impact on seabirds. The potential habituation to OWFs, the mitigating effect of migration corridors or the correlation between seabird displacement levels and wind farm configuration characteristics, to name just a few, are highly relevant knowledge gaps in the light of future planning and (cumulative) impact assessments. The new seabird monitoring program aims to add some pieces to this puzzle. As a first step towards these analyses, this report provides an overview of the count results collected during six seabird monitoring campaigns between February 2021 and April 2023.

2. Methods

The new seabird monitoring program encompasses eight SE–NW oriented tracks across the full extent of the OWF concession zone as well as through an area southwest of and adjacent to the wind farms (Fig. 1), the latter serving as the control area. The monitoring can be completed in two days, and is intended to be carried out 5 times per year (in February, April, August, October and December).

The seabird counts were carried out from a research vessel, following a standardised and internationally applied method, combining a *transect count* for birds in contact with the water and repeated *snapshot counts* for flying birds (Tasker *et al*. 1984). The focus is on a 300 m wide transect along one side of the ship's track, and while steaming at a speed of about 10 knots all birds in touch with the water (swimming, dipping, diving) within this transect are counted (i.e., the *transect count*). Applying four distance categories (A = 0–50 m; B = 50–100 m; C = 100–200 m; D = 200–300 m), the distance to each observed bird (group) is estimated, allowing to correct for decreasing detectability with increasing distance afterwards. Counting all flying birds encountered inside this transect, however, would be measuring bird flux rather than bird density (Tasker *et al*. 1984). The density of flying birds is therefore assessed through one-minute interval counts of all birds flying within a quadrant of 300 by 300 m inside the transect (the so-called *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'. Birds observed outside the transect and

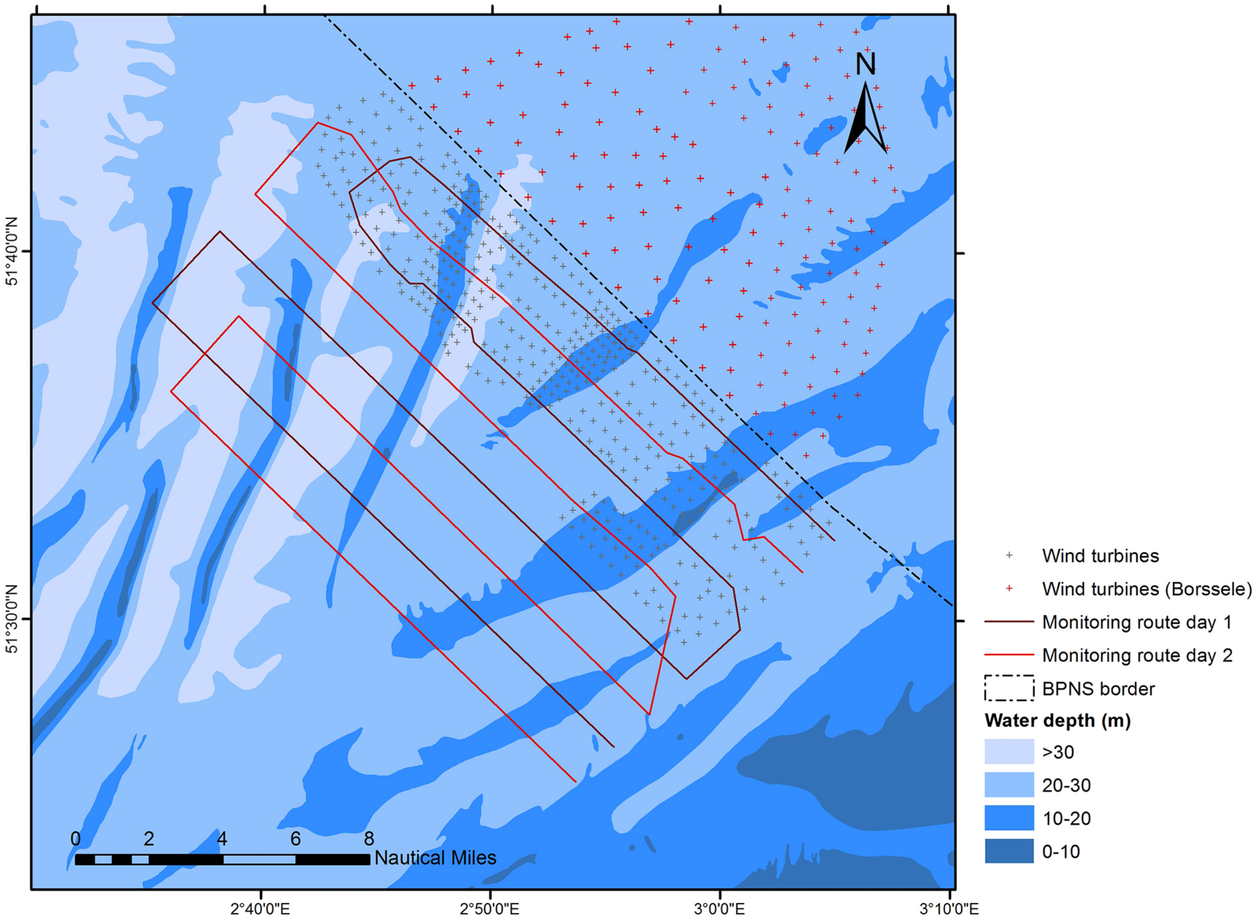

Figure 1. Seabird displacement monitoring strategy since February 2021.

Table 1. Overview of the surveys executed within the new seabird displacement monitoring program.

Campaign	Date	Remarks
February 2021	23/02/2021	Counts performed from the 'old' *Belgica*
	24/02/2021	Counts performed from the 'old' *Belgica*
August 2022	21/08/2022	
	22/08/2022	
October 2022	10/10/2022	
	13/10/2022	
December 2022	14/12/2022	
	15/12/2022	
February 2023	15/02/2023	
	17/02/2023	The transects crossing the wind farms could not be sailed
April 2023	19/04/2023	
	20/04/2023	

snapshot counts are noted down as well, yet cannot be included in the calculation of seabird densities.

Between February 2021 and April 2023, six monitoring campaigns were carried out (Table 1). Only during the first campaign, counts were performed from the 'old' RV Belgica, while all other campaigns were executed with the new vessel. The campaign in February 2023 was only partly executed as we were not allowed to enter the wind farms during the second monitoring day due to adverse weather conditions with wind speeds exceeding 25 knots.

In total we collected 2404 counts within the study area, the effort per count varying between 0.05 and 0.24 km² (calculated by multiplying the sailed track length with the transect width of 300 m). In the results section we present both the density (N/km²) as well as the number observed per km (N/km) for each of two zones, i.e., the concession zone (the area built with turbines) and the control area outside the wind farms (>1 km away from the nearest turbine). For some species, the number of individuals observed per km (including those outside the transect) is a more representative measure for their occurrence, especially in case of scarcer species generating few data and also for species that tend to concentrate around the wind turbines. The latter is due to the fact that the turbine foundations are often well outside the 300 m wide count transect that is used for density calculations. In the Results section (§3), the number observed per km is further used to illustrate species distribution across the study area.

3. Results

In total we observed 46 species of birds, with a total number of 11 585 individuals counted (see Table 2 in Annex). The most abundant (positively identified) species was lesser black-backed gull (*Larus fuscus*) (N = 2763). We further counted 3219 unidentified large gulls (*Larus* sp.), generally birds associated with fishing trawlers and observed from a long distance. Other common species were northern gannet (*Morus bassanus*), great cormorant (*Phalacrocorax carbo*), little gull (*Hydrocoloeus minutus*), common gull (*Larus canus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), black-legged kittiwake (*Rissa tridactyla*), Sandwich tern (*Thalasseus sandvicensis*), common guillemot (*Uria aalge*) and razorbill (*Alca torda*). Each of these species will be discussed in more detail in the paragraphs below.

3.1. Northern gannet

In total we observed 567 northern gannets in the study area. Outside the wind farms we observed 0.46 birds per km, compared to 0.26 birds per km between the turbines. The difference is even more pronounced when considering densities, with 0.40 birds per km² outside compared to 0.10 birds per km² inside the wind farms. The species occurred quite homogenously distributed in the control area, with the highest numbers encountered in far offshore waters, as opposed to a more limited and scattered presence inside the concession zone (Fig. 3). The results for northern gannet thus seem to point towards wind farm avoidance. Only at the Norther wind farm, in the SE corner of the concession zone, presence seems to reflect background numbers. Interestingly this wind farm is characterised by wide spacing between the turbines, and is also outside the 'shadow' of the Borssele wind farm.

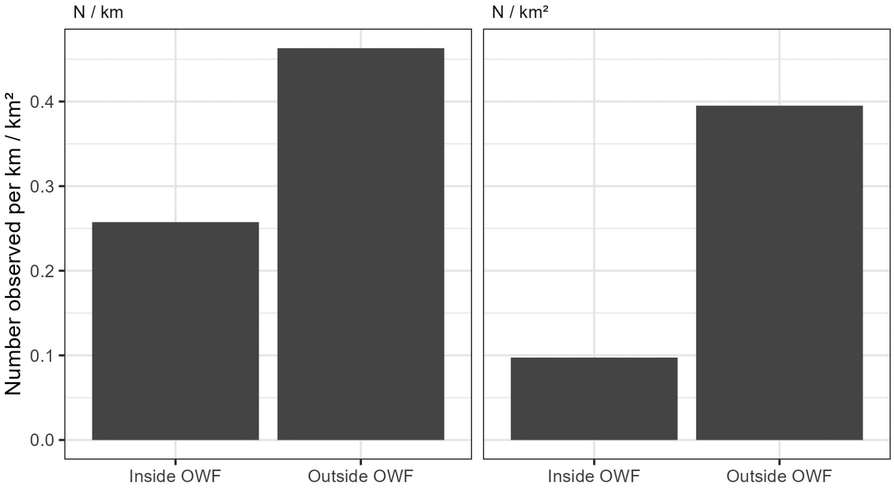

Figure 2. Occurrence of northern gannet inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

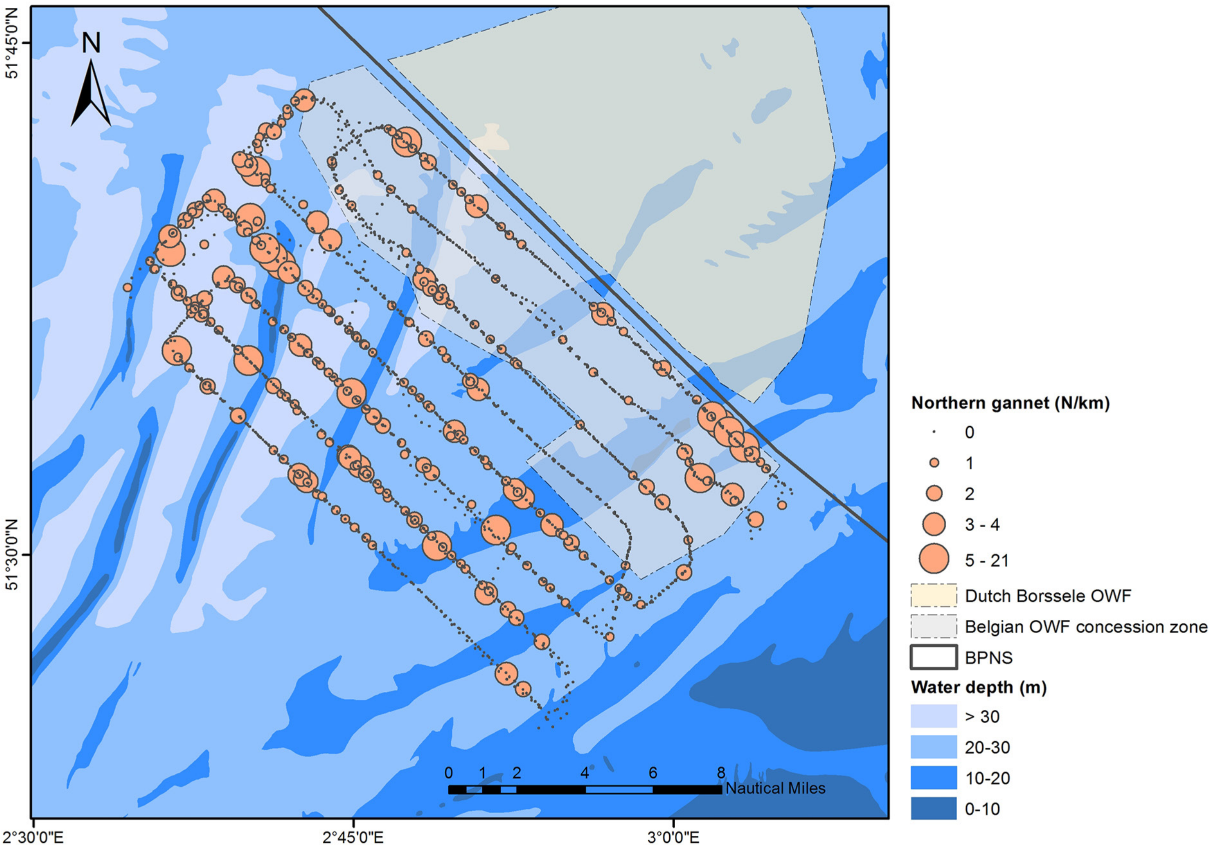

Figure 3. Northern gannet observations (N/km) in the study area.

3.2. Great cormorant

With only 99 individuals counted, great cormorant was the least common of the species discussed in this report. The major part of these birds (N = 61) was associated with the turbine foundations, with a clear preference to the jacket foundations in the C-Power wind farm. As the turbines are generally located (just) outside our 300 m wide count transect, the number observed per km is a more representative parameter to describe the species' presence compared to their measured density. As such, we observed 0.18 birds per km inside the wind farms, compared to only 0.001 birds per km outside the concession zone, suggesting a strong attraction effect. Note that the species' preference to rest on (and concentrate near) turbine foundations also explains the very low densities shown in the right panel of Fig. 4.

3.3. Little gull

Little gulls were encountered relatively often during the campaigns of December 2022 (N = 174) and April 2023 (N = 240), during which we observed the highest numbers outside the wind farms. This difference is most pronounced when considering densities, with 0.30 birds per km² inside compared to 0.76 birds per km² outside the OWF concession zone. Figure 6 further shows a distinct onshore-offshore gradient in the species' distribution across the study area, with the major part of the observations located within 30 km away from the coast, and no more little gulls over 40 km offshore.

3.4. Common gull

The results for common gull suggest attraction to the OWFs. The numbers observed per km as well as the encountered densities were about three times higher inside compared to outside the OWF concession zone. Interestingly, highest numbers occurred along the outer transect next to the Dutch border (Fig. 8). This should, however, not be mistaken for an edge effect as the Dutch Borssele wind farm is located right across the border, with no actual corridor in between the Belgian and Dutch turbines.

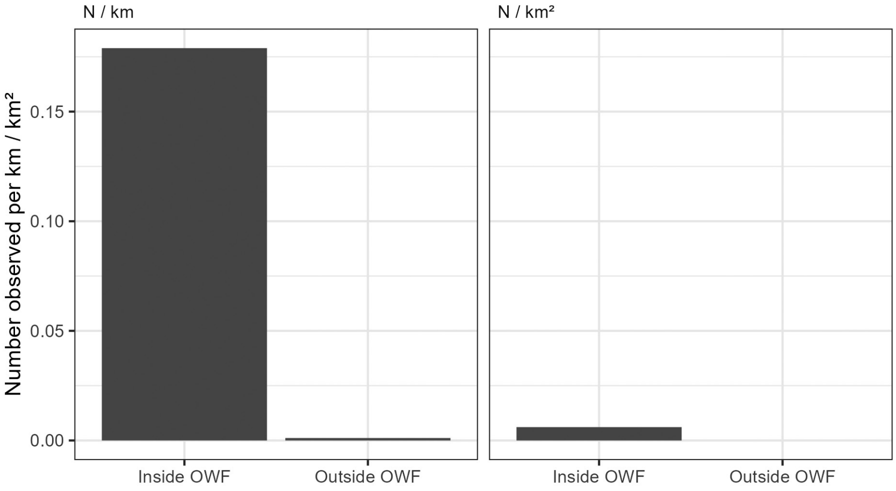

Figure 4. Occurrence of great cormorant inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

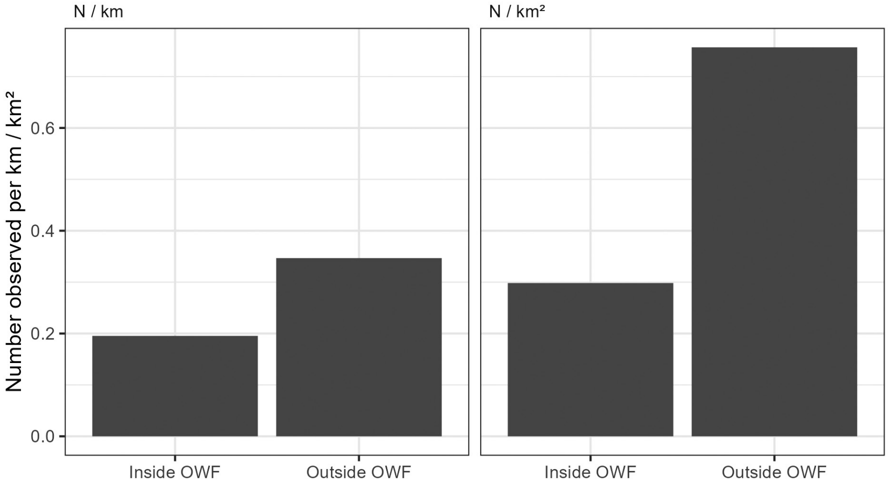

Figure 5. Occurrence of little gull inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

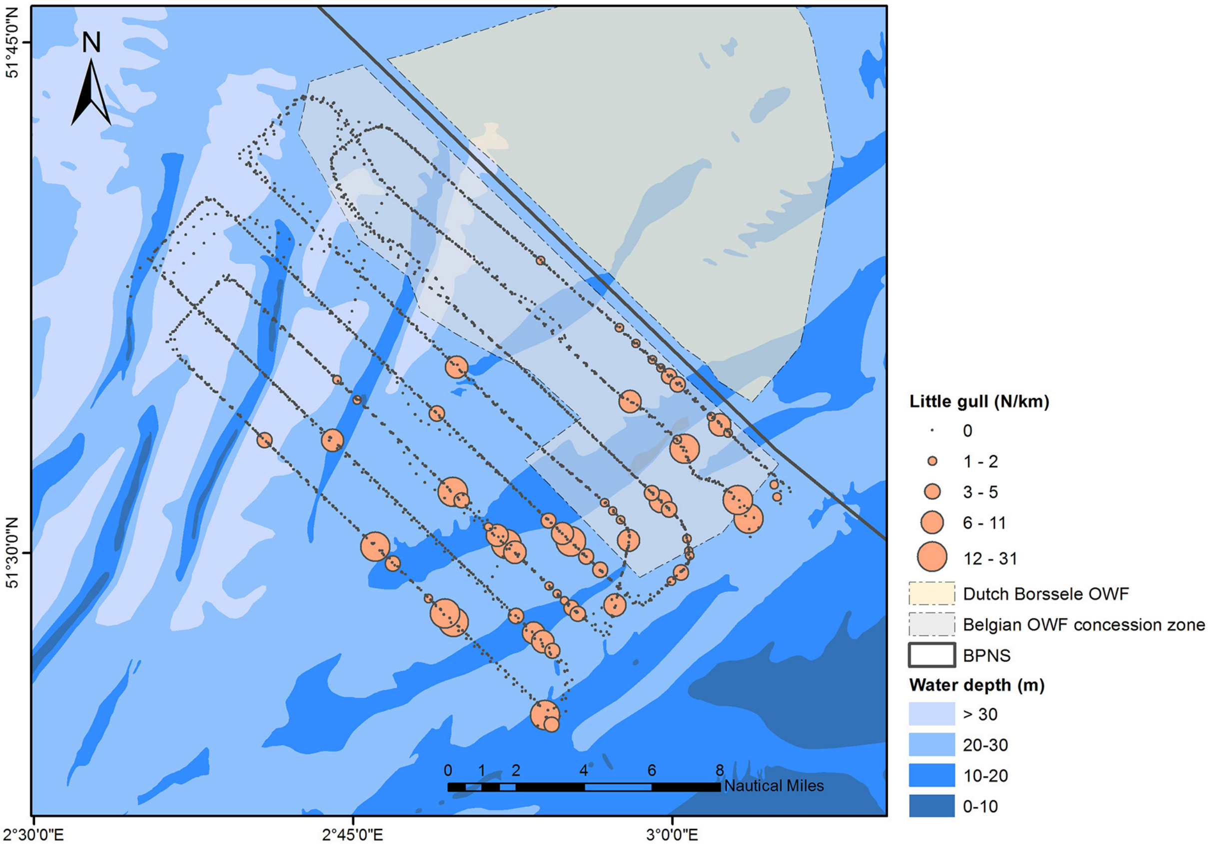

Figure 6. Little gull observations (N/km) in the study area.

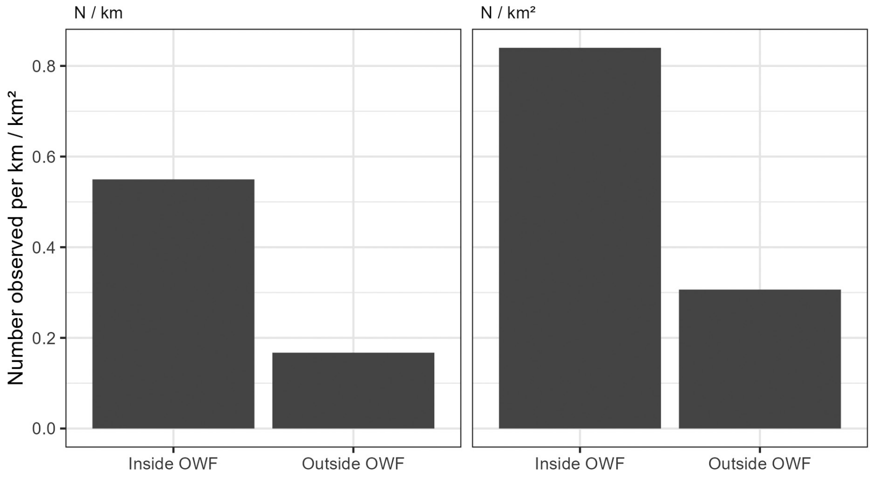

Figure 7. Occurrence of common gull inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

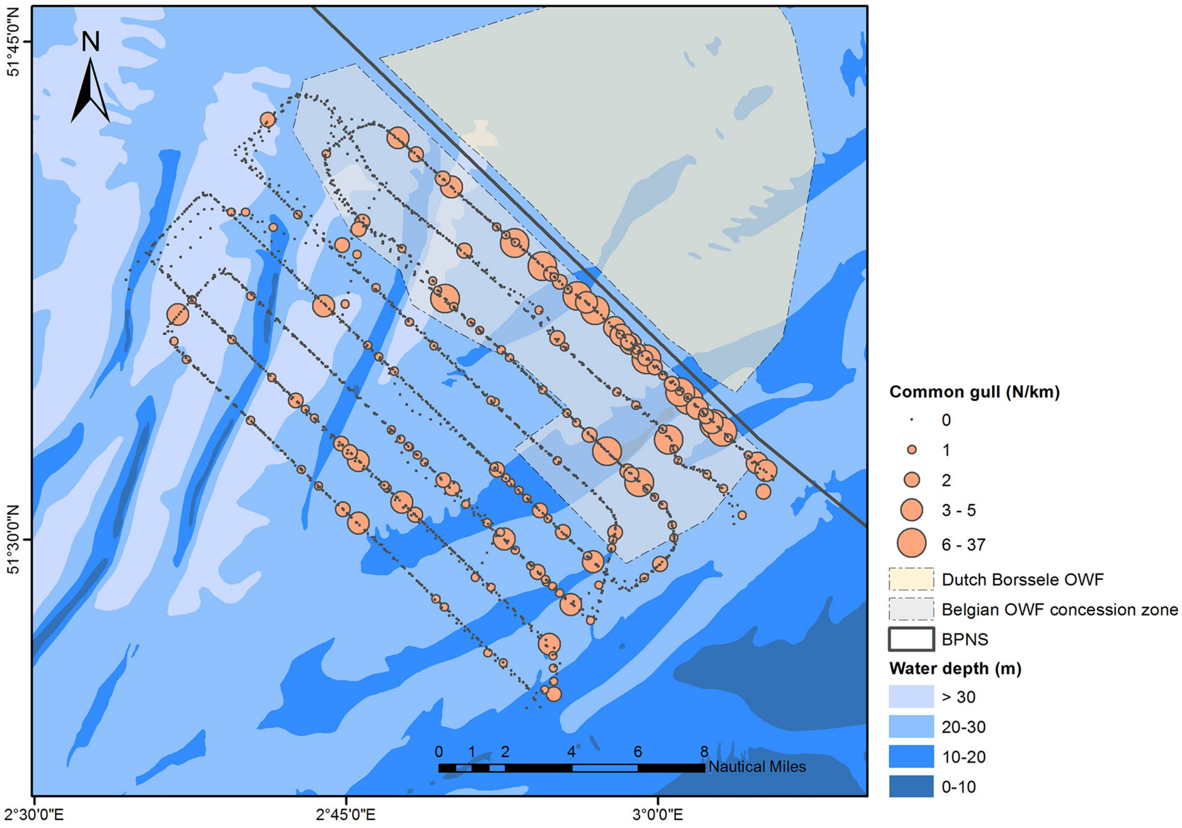

Figure 8. Common gull observations (N/km) in the study area.

3.5. Lesser black-backed gull

With 921 observations of 2763 individuals, lesser black-backed gull was the most common species observed in the study area. Only a small minority (7%) of the birds observed in the wind farms was associated with the turbines. The species' distribution across the study area shows somewhat lower presence in the more offshore part of the control area, and based upon this pattern one could suspect an attraction effect (Fig. 10). The observed density of 4.7 birds per km² outside as opposed to only 1.3 birds per km² inside the concession zone, however, rather indicates avoidance of the wind farms. Note that the high densities of lesser black-backed

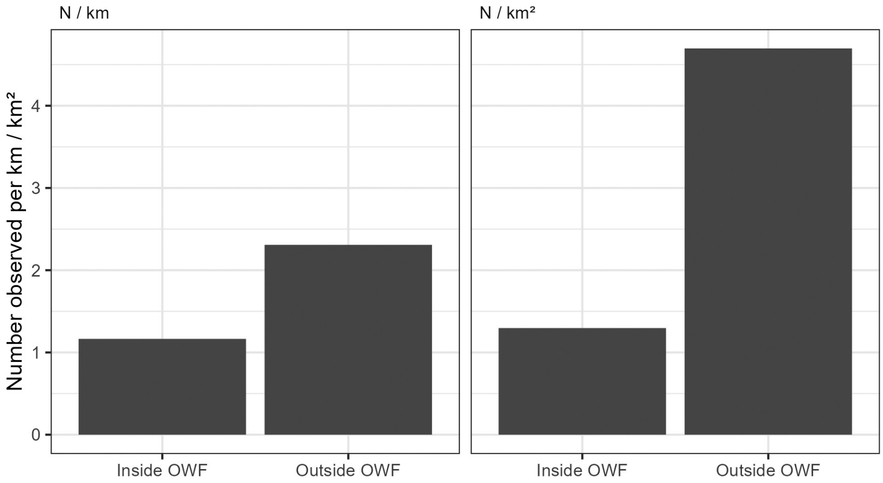

Figure 9. Occurrence of lesser black-backed gull inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

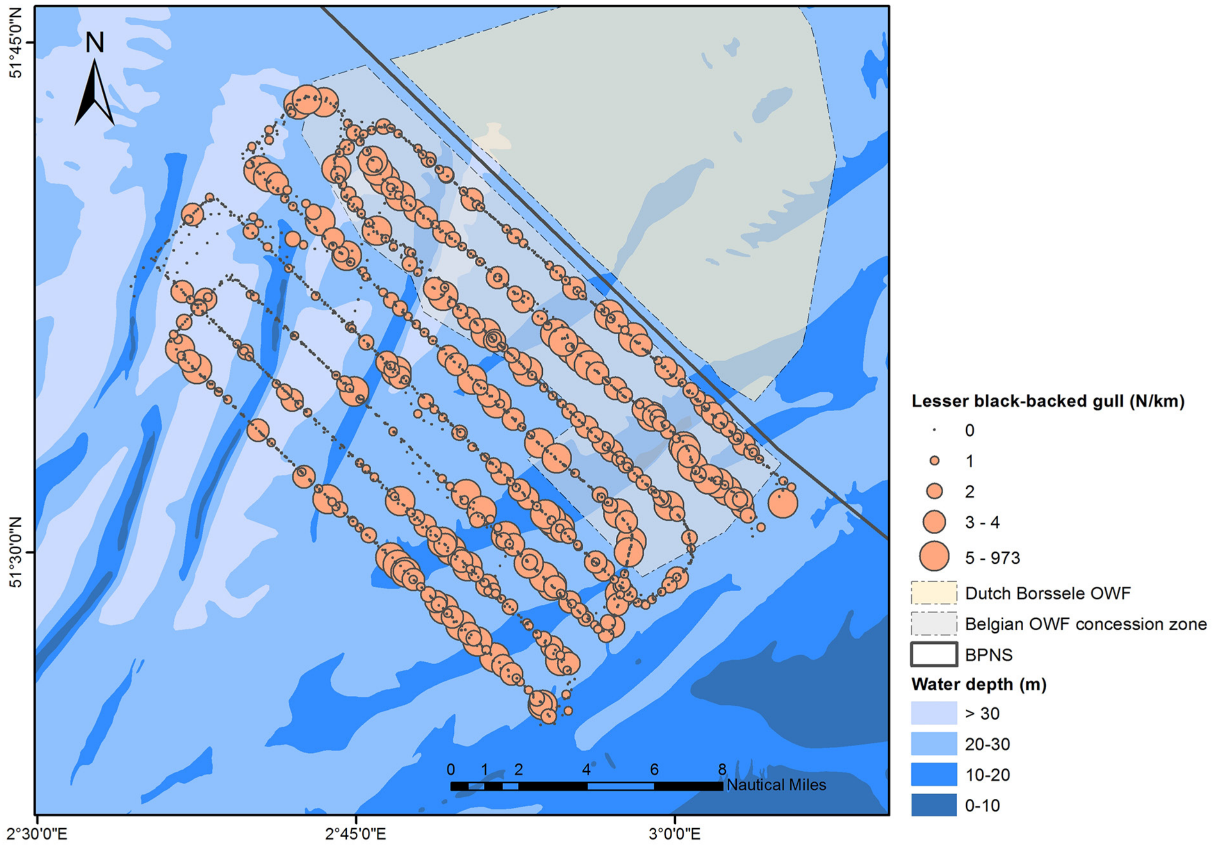

Figure 10. Lesser black-backed gull observations (N/km) in the study area.

gull in the southern part of the control area could not be linked to fishery activities, at least not directly. Though we did encounter large numbers of *Larus* gulls associated with trawlers in the study area, these were mostly observed from a large distance and were not determined to species level (and thus not included in the results).

3.6. Herring gull

About one third of the herring gulls observed in the concession zone was associated with the turbines, showing a preference to jacket foundations. This implies an underestimation of the actual densities inside the wind farms, considering the methodological constraints of a 300 m wide transect. At the same time, this explains the large difference in measured densities between the control and impact area (Fig. 11). Looking at the numbers observed per km indeed shows that only slightly more herring gulls were seen outside compared to inside the OWFs (0.17 versus 0.14 birds per km respectively).

3.7. Great black-backed gull

Great black-backed gulls clearly concentrated inside the OWF concession zone (Fig. 13). At the same time, nearly 60% of the birds observed inside the wind farms was associated with the turbine foundations, implying that the number observed per km is the most reliable measure to assess the species' occurrence in the impact area. As such, the number observed per km inside the wind farms was 5 times higher inside compared to outside the OWFs (0.31 versus 0.06 birds per km), suggesting a strong attraction effect.

3.8. Black-legged kittiwake

With 936 individuals observed, black-legged kittiwake was one of the most common species in the study area. Inside the wind farm concession zone, the species' density measured 1.3 birds per km², opposed to a considerably lower density of 0.82 birds per km² outside the wind farms. The difference, however, is less pronounced when considering the number of birds observed per km. Despite these results suggesting an attraction effect, there was no clear pattern in the distribution of black-legged kittiwakes across the study area (Fig. 15).

3.9. Sandwich tern

Sandwich tern densities encountered outside the wind farm concession zone measured more than twice the densities inside the wind farms (0.11 versus 0.05 birds per km²). The difference is even more pronounced when considering the number of birds observed per km. Interestingly, most observations occurred in the extreme southeastern end of the study area, reflecting a strong onshore-offshore gradient in the species' distribution (Fig. 17). For this reason, it seems doubtful that the difference in densities as shown in Fig. 16 reflects an actual avoidance response.

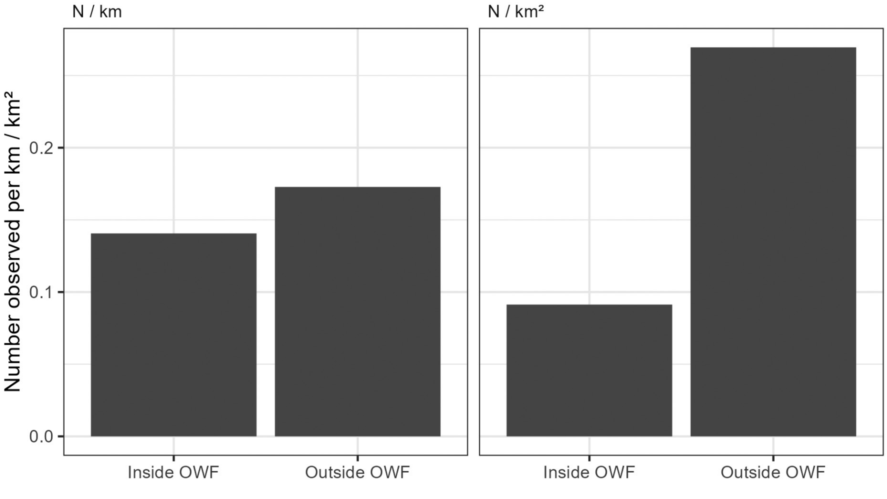

Figure 11. Occurrence of herring gull inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

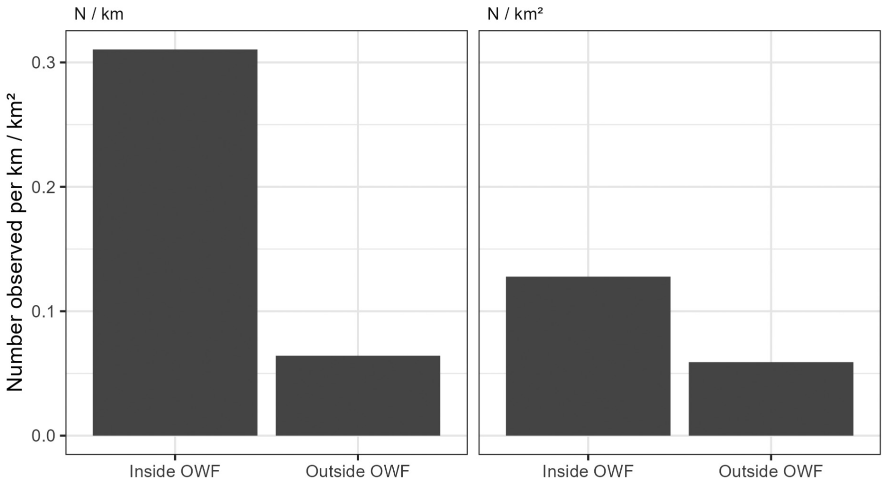

Figure 12. Occurrence of great black-backed gull inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

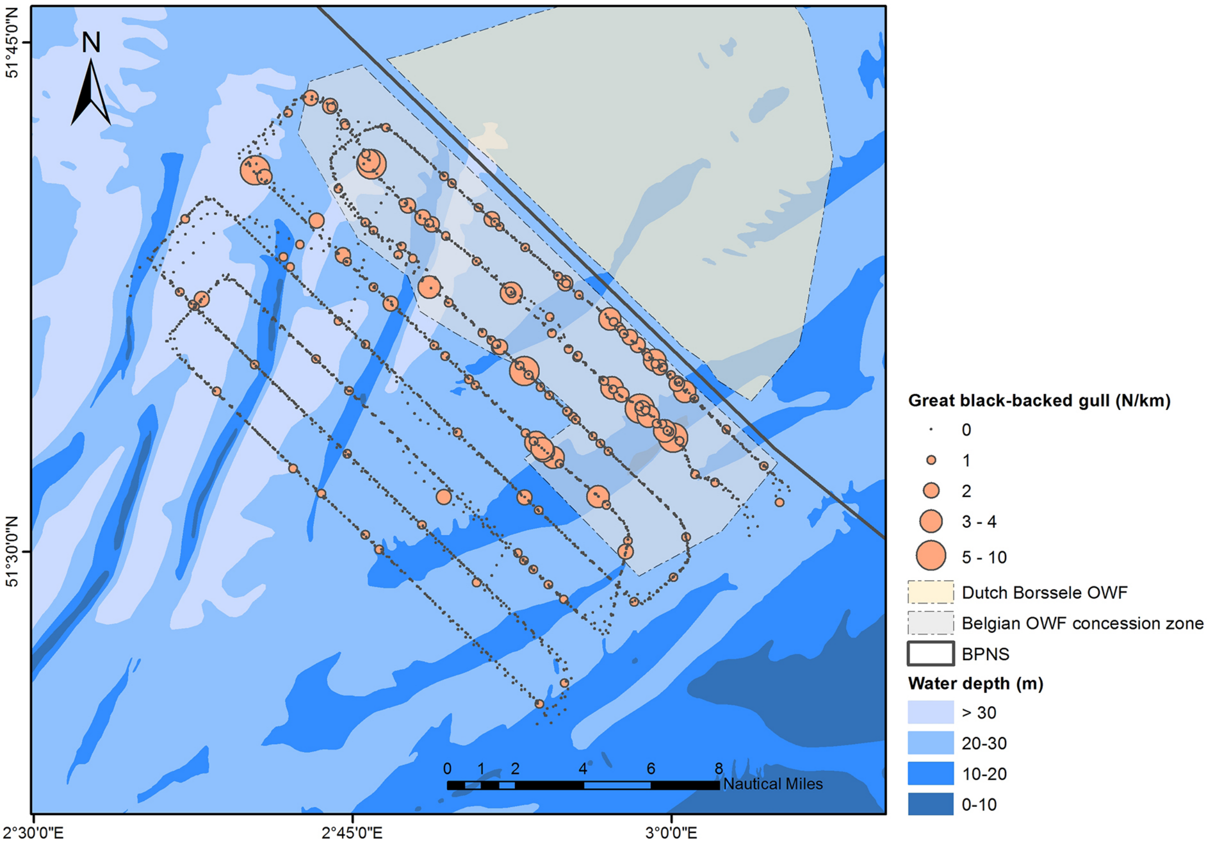

Figure 13. Great black-backed gull observations (N/km) in the study area.

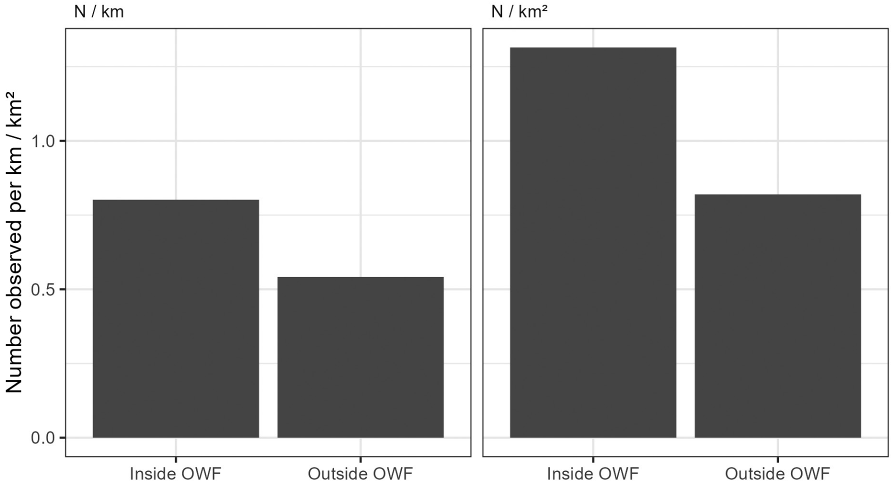

Figure 14. Occurrence of black-legged kittiwake inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

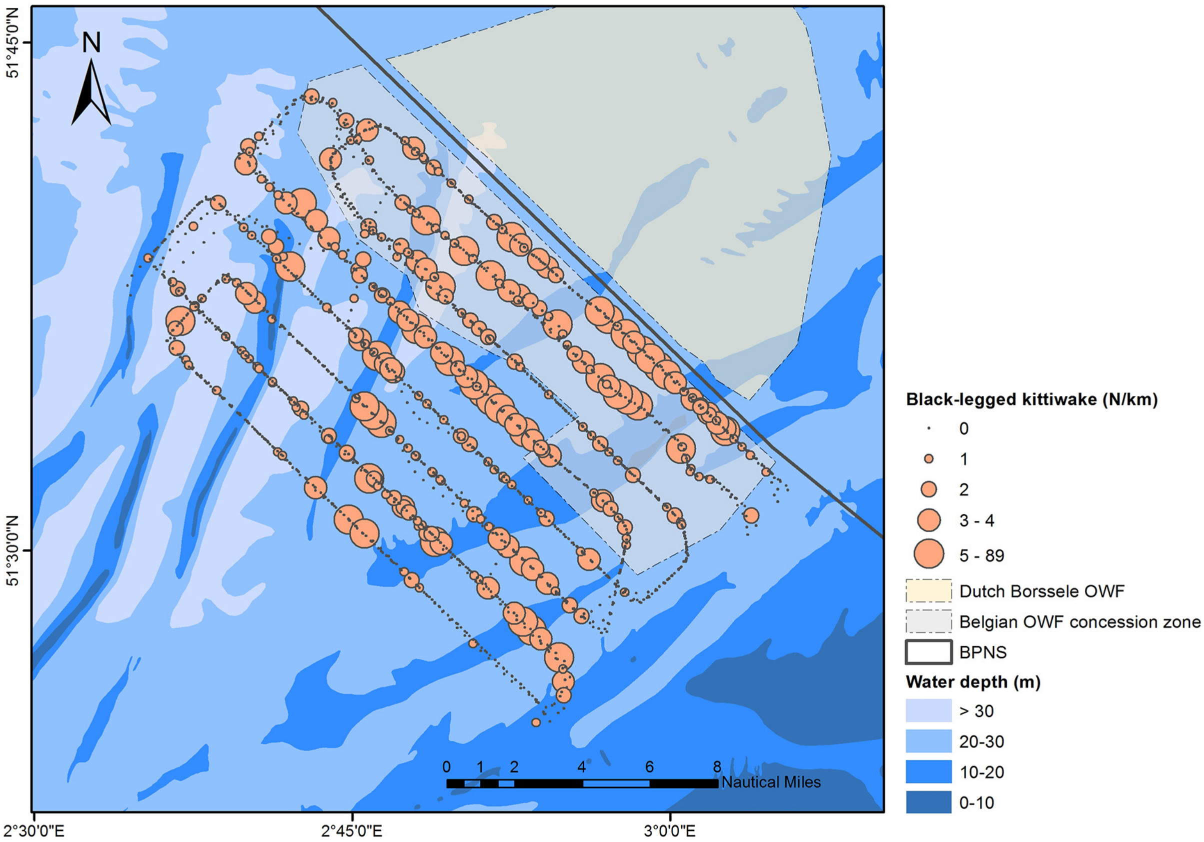

Figure 15. Black-legged kittiwake observations (N/km) in the study area.

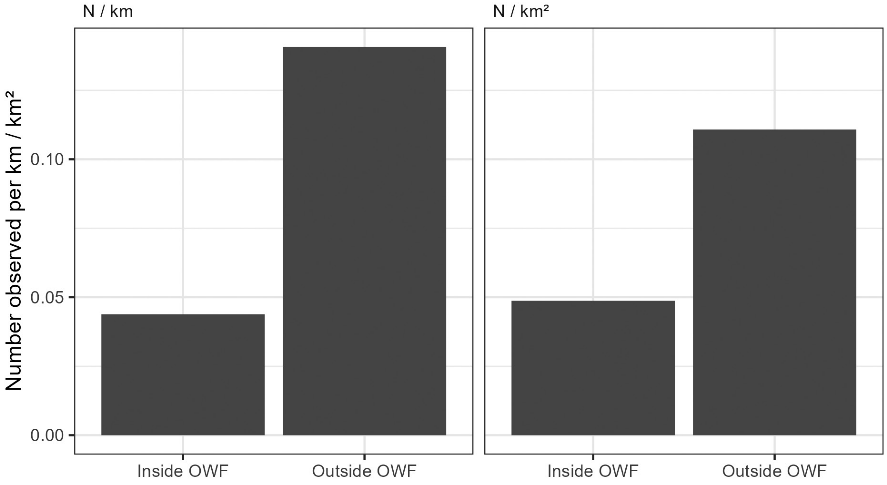

Figure 16. Occurrence of Sandwich tern inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

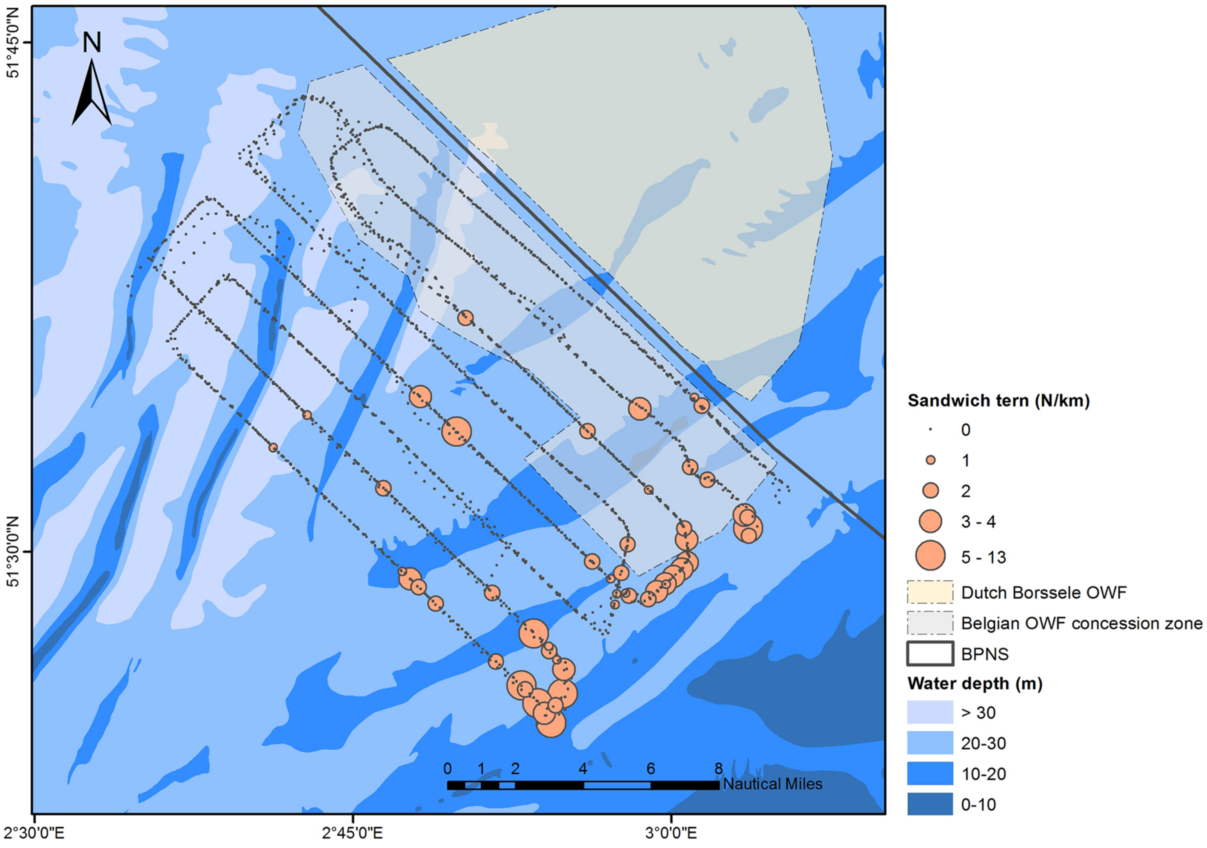

Figure 17. Sandwich tern observations (N/km) in the study area.

3.10. Common guillemot

Common guillemots occurred homogenously spread across the study area, with no clear distributional pattern. Accordingly, densities encountered in the impact and control area differed only slightly, with 0.35 birds per km² encountered inside the OWF concession zone, opposed to 0.45 birds per km² outside this area.

3.11. Razorbill

Interestingly, razorbills (N=628) were far more numerous in the study area compared to common guillemots (N=288), though in general the latter is much more abundant at the Belgian part of the North Sea. Furthermore, razorbill densities inside the wind farm concession zone outreached those outside, with 1.25 and 0.83 birds per km² respectively. This difference is less pronounced when considering numbers observed per km. The species does not display a clear distributional pattern across the study area, apart from a concentration of observations in the southern corner of the study area (Fig. 20).

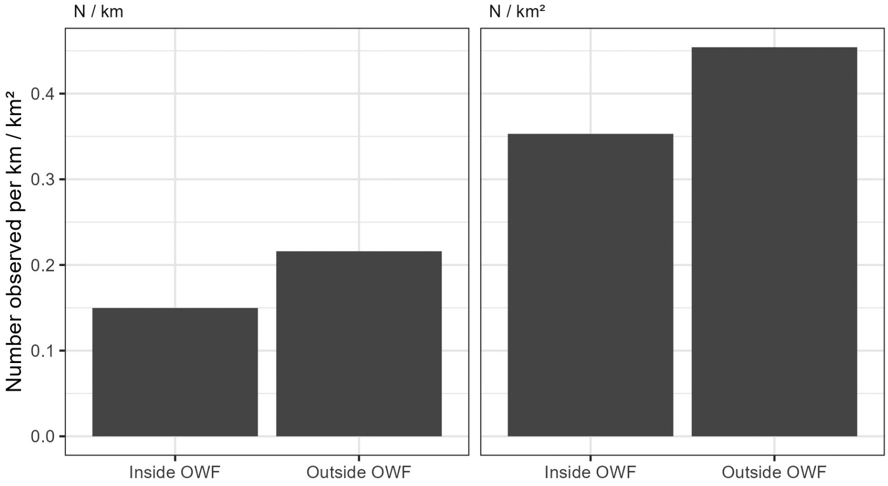

Figure 18. Occurrence of common guillemot inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

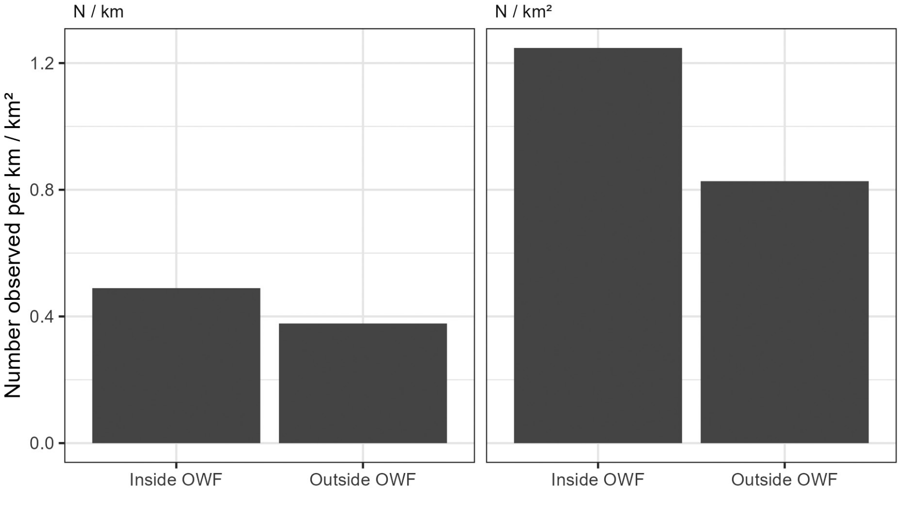

Figure 19. Occurrence of razorbill inside and outside the OWF concession zone, expressed as the number observed per km on the left and the number per km² on the right.

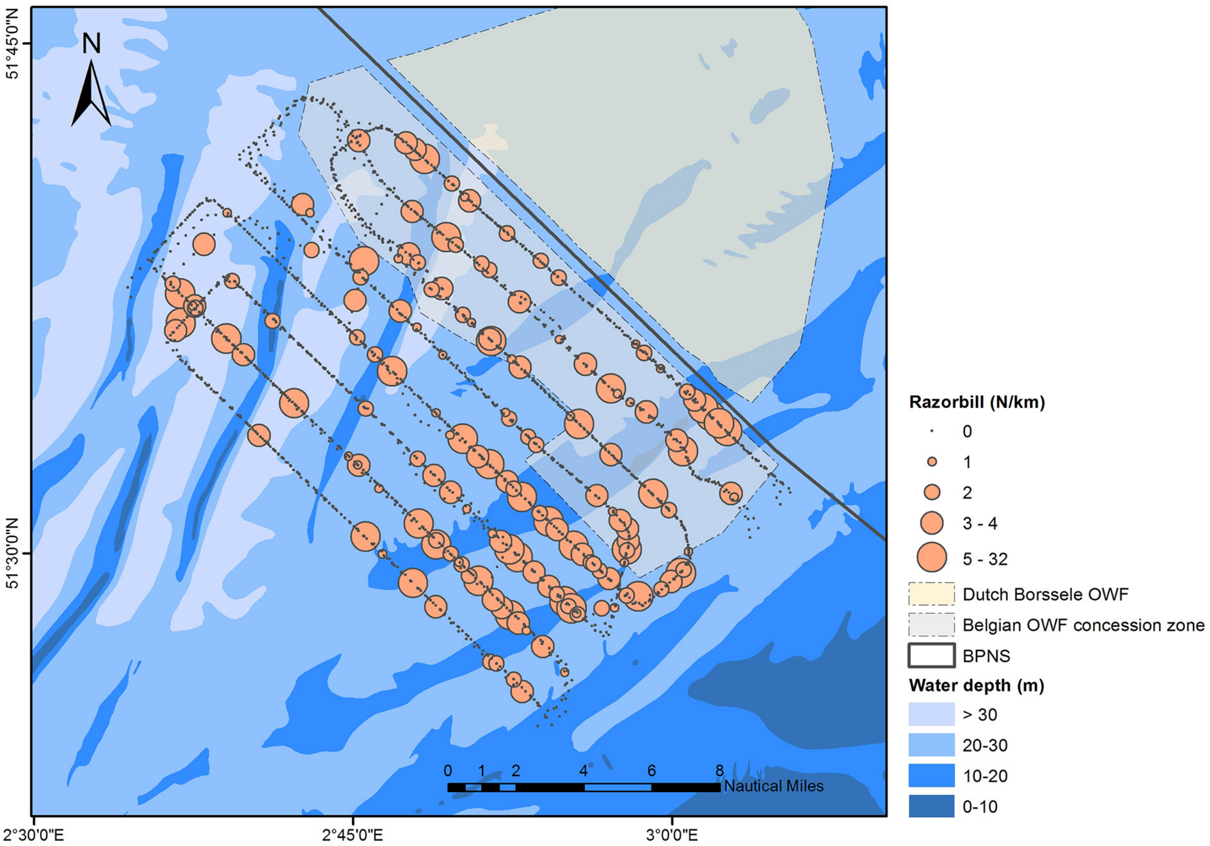

Figure 20. Razorbill observations (N/km) in the study area.

4. Discussion

This report sets out the count results collected over the period February 2021 to April 2023, following a revised monitoring design across the full extent of the Belgian OWF concession zone. It is important to highlight that the results presented here are provisional, and need to be considered as indicative. More data need to be collected before we will be able to detect seabird avoidance or attraction effects with sufficient confidence, by means of the intended spatial modelling. Considering the strong dynamics characterising marine environments and the high natural variability in seabird abundance, it can be statistically challenging to detect (potentially small) displacement effects (Vanermen *et al*. 2015; Cuttat & Skov 2020). As such, taking account of key habitat features and their effect on seabird distribution may be necessary to reliably detect seabird displacement, and is only possible when using monitoring data with high spatial resolution. Also note that this monitoring design does not include seabird densities present prior to construction, further stressing the importance of including habitat features in the modelling process.

Nevertheless, though making use of a limited dataset and mean values only, it is interesting to see that the results so far are often in line with what has been found before and/or elsewhere (Vanermen & Stienen 2019), such as indications of attraction in great black-backed gull and great cormorant, and of avoidance by northern gannet. On the other hand, current results no longer seem to indicate (strong) avoidance of common guillemots and even increased numbers of razorbills between the turbines. Spatial modelling will tell whether displacement effects on auks have actually decreased over time, providing proof for habituation, or whether the observed numbers in the wind farms result from specific habitat features inside the concession zone.

The new monitoring strategy not only aims to detect displacement responses. It is also designed to detect disturbance distances (with regard to migration corridors) and the effect of turbine density on seabird displacement levels. Regarding the latter, Leopold *et al*. (2013) found stronger negative responses of gannets and auks towards the PAWP wind farm compared to the OWEZ wind farm, which was hypothesised to result from the higher turbine density at the former. In another study including a third wind farm, Heinänen & Skov (2018) too found a decreasing impact on both auk species comparing the PAWP, Luchterduinen and OWEZ wind farms, reflecting the decreasing density of turbines in the respective wind farms. Note that the distance between the turbines at OWEZ ranges between 650 and 1000 m, which is considerably less than the distance between turbines at more recently built OWFs (see Fig. 1 to compare the configuration of the Borssele wind farm - built after 2020 - with the configuration of wind farms in the Belgian concession zone for example). With advancing technology, wind turbines tend to grow larger each year, and since there is a clear correlation between the necessary spacing between wind turbines and their rotor diameter, turbine density is expected to decrease even further in the future.

Taking account of all actually planned OWFs, wind farm capacity in the North Sea will soon increase from 26 to 61 GW. Actual ambitions reach even further, and aim for 117 GW by 2030 (RHDHV 2022). Clearly, the need to achieve a rapid transition from fossil to renewable energies is high, but unfortunately the precautionary principle regarding marine biodiversity impact seems to be abandoned. Politics now aim to achieve biodiversity goals by mitigating (rather than avoiding) the effects of large-scale wind exploitation. As such, granting procedures now often incorporate the demand for installing effective mitigating measures. However, wide knowledge gaps still persist regarding the latter and with this monitoring approach we hope to be able to inform future planning decisions regarding wind farm configuration and mitigation of impact on seabirds.

References

Cuttat, F. & Skov, H. 2020. *SEANSE – Cumulative Displacement Impacts on Seabirds*. DHI.

Heinänen, S. & Skov, H. 2018. *Offshore Windfarm Eneco Luchterduinen Ecological Monitoring of Seabirds*. T3 (Final) report. DHI.

Leopold, M.F., van Bemmelen, R.S.A. & Zuur, A. 2013. *Responses of Local Birds to the Offshore Wind Farms PAWP and OWEZ off the Dutch Mainland Coast*. Report C151/12. IMARES.

RHDHV 2022. *Spatial Study North Seas 2030 – Offshore Wind Development*. RHDHV-BI4271. HaskoningDHV Nederland B.V.

Tasker, M.L., Jones, P.H., Dixon, T.J. & Blake, B.F. 1984. Counting seabirds at sea from ships: a review of methods employed and a suggestion for a standardised approach. *Auk* 101: 567–577. https://doi.org/10.1093/auk/101.3.567

Vanermen, N., Onkelinx, T., Verschelde, P., Courtens, W., Van de walle, M., Verstraete, H. & Stienen, E.W.M. 2015. Assessing seabird displacement at offshore wind farms: power ranges of a monitoring and data handling protocol. *Hydrobiologia* 756: 155–167. https://doi.org/10.1007/s10750-014-2156-2

Vanermen, N. & Stienen, E. W.M. 2019. Seabird displacement. *In*: Perrow, M.R. (ed.) *Wildlife and Wind farms, Conflicts and Solutions – Offshore: Potential Effects*: 174–205. Pelagic Publishing.

Annex

Table 2. List of all bird species recorded during seabird monitoring campaigns in the period 2021–2023.

Species (scientific name)	Species (English name)	Number of observations	Sum
Gavia stellata	Red-throated diver	6	**6**
Gavia sp.	Unidentified diver	2	**5**
Podiceps grisegena	Red-necked grebe	1	**2**
Fulmarus glacialis	Northern fulmar	1	1
Hydrobates pelagicus	European storm petrel	1	**1**
Morus bassanus	Northern gannet	424	**567**
Phalacrocorax carbo	Great cormorant	35	**99**
Phalacrocorax aristotelis	European shag	3	**4**
Phalacrocorax sp.	Unidentified cormorant	1	1
Anser / Branta sp.	Unidentified goose	1	**16**
Anser anser	Greylag goose	1	**18**
Branta bernicla	Brent goose	3	**71**
Mareca penelope	Eurasian wigeon	2	**12**
Anas acuta	Northern pintail	1	20
Melanitta nigra	Common scoter	6	**29**
–	Unidentified duck	3	**170**
Accipiter nisus	Eurasian sparrowhawk	1	**1**
Falco tinnunculus	Common kestrel	2	**2**
Falco columbarius	Merlin	1	1
Pluvialis squatarola	Grey plover	1	**1**
Gallinago gallinago	Common snipe	1	**1**
Limosa lapponica	Bar-tailed godwit	3	**32**
Numenius phaeopus	Eurasian whimbrel	1	**8**
Arenaria interpres	Ruddy turnstone	1	1
Stercorarius parasiticus	Arctic skua	3	**3**
Stercorarius skua	Great skua	1	**1**
Larus melanocephalus	Mediterranean gull	1	**1**
Hydrocoloeus minutus	Little gull	115	**425**
Larus ridibundus	Black-headed gull	22	35
Larus canus	Common gull	278	**464**
–	Unidentified small gull	1	**15**

Species (scientific name)	Species (English name)	Number of observations	Sum
Larus fuscus	Lesser Black-backed gull	921	**2763**
Larus argentatus	European Herring gull	197	**240**
Larus michahellis	Yellow-legged gull	45	50
Larus cachinnans	Caspian gull	10	**10**
Larus marinus	Great Black-backed gull	194	**251**
Larus sp.	Unidentified *Larus* gull	37	**3219**
Rissa tridactyla	Black-legged kittiwake	544	**936**
–	Unidentified gull	5	505
Sterna sandvicensis	Sandwich tern	71	**157**
Sterna hirundo	Common tern	10	**39**
Sterna hirundo / paradisaea	Common / Arctic tern	1	**2**
Sterna paradisaea	Arctic tern	2	**2**
Chlidonias niger	Black tern	1	1
Uria aalge	Common guillemot	240	**288**
Uria aalge / Alca torda	Common guillemot / Razorbill	103	**362**
Alca torda	Razorbill	256	**628**
Columba livia domestica	Feral dove	2	**7**
Streptopelia decaocto	Eurasian collared dove	1	1
Alauda arvensis	Eurasian skylark	2	**7**
Anthus pratensis	Meadow pipit	4	**6**
Motacilla alba	White wagtail	1	**1**
Turdus pilaris	Fieldfare	1	**1**
Turdus sp.	Unidentified thrush	1	**1**
Sturnus vulgaris	Common starling	14	**93**
Fringilla coelebs	Common chaffinch	1	**1**
Passeriformes	Unidentified passerine	1	**1**

CHAPTER 6

OFFSHORE WIND TURBINE CURTAILMENT STRATEGIES IN NORTH SEA COUNTRIES TO REDUCE BIRD COLLISIONS

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Abstract

The Southern North Sea is part of one of the main migration flyways in Europe. The highest flight intensities at sea are recorded at night during spring and autumn migration and are mostly migrating passerines. These songbirds migrate at high altitudes, up to several kilometres but a portion of these birds flies at rotor height and is thus at risk of collision. Temporarily stopping the turbine operation during high collision risk events for songbirds, such as adverse weather conditions during migration bringing large numbers of passerines into the range of turbine rotors, may prevent a large number of collision victims. However, curtailing turbines at sea to reduce the collision risk is not yet being applied on a large scale. To support the ongoing discussions on this topic, this report aims to present an overview of curtailment strategies in wind farms at sea in North Sea countries. From the collected information it is clear that the Netherlands is pioneering in implementing curtailment measures in wind farms at sea, but also Germany and France are starting to perform tests. Other countries are open for discussions on the topic. Although temporary turbine shutdowns could be highly effective for reducing collision mortalities in certain scenarios, there is still a need for sound, site-specific monitoring programs to assess the effectiveness and to finetune the implemented measures. A regional approach to the implementation of curtailment strategies, could maximise the efficiency and ecological benefits of such policy measures. It is important to note that wind turbine curtailment is only one aspect to mitigate the impact of wind farms on bird populations. Responsible development further entails proper site selection, pre-construction environmental assessments, and post-construction monitoring.

1. Introduction

Birds and bats are directly affected by wind turbines through the risk of collision with structures (Drewitt & Langston 2006; Fox *et al*. 2006; Voigt *et al*. 2015; Thaxter *et al*. 2017). This conflict between renewable energy production and nature conservation is referred to as a ‘green-green’ dilemma (Voigt *et al*. 2019). To minimize the collision risk for birds and bats it is important to use optimal siting strategies for wind farms, for example by avoiding sensitive habitats (Marques *et al.* 2014; Harwood & Perrow 2019) or migratory pathways. Additionally, regulating the operation of wind turbines can further mitigate collision mortality for birds and bats (Cook *et al.* 2011; May 2017). This implies that the operation of some or all wind turbines of a wind farm is intentionally reduced or stopped during specific times when the risk of collisions is high. The reduced rotor speed increases the visibility of the turbine blades, and it reduces the probability of a bird or a bat flying through the rotor swept zone of being hit by a blade (Harwood & Perrow 2019).

One way to do this is 'shutdown on demand'. This means that a turbine or some turbines in a wind farm are stopped when the collision risk is high, e.g., when a bird is flying close to the rotor swept zone of a turbine (Marques *et al.* 2014). This has been successfully applied in wind farms located at bottlenecks for migratory soaring birds like some raptor species, storks, cranes, etc. (Smallwood & Karas 2009; de Lucas *et al.* 2012; Tomé *et al.* 2017a, 2017b). In most cases, the shutdown is initiated by human observers, with or without technological aids like radar, cameras, etc.

Secondly, turbine operations can be restricted in a wind farm as a whole, during massive migration events or during certain weather conditions when the collision risk is high (Marques *et al.* 2014). In this overview, this approach is further referred to as curtailment. Such general curtailment measure is often applied to reduce bat fatalities by increasing the cut-in wind speed (defined as the lowest wind speed at which turbines generate power) to those critical wind speeds at which bats reduce their activity (Arnett *et al.* 2010; Adams *et al.* 2021; Behr *et al.* 2018). Arnett *et al.* (2010) showed that reducing turbine operation during periods of low wind speeds reduced bat mortality with 44% to 93% and marginal annual power loss (< 1% of total annual output). For birds, restricting wind farm operation could be implemented when there is an identified, anticipated high collision risk. For example, wind farms on migratory routes could be shut down at nights of poor weather conditions to reduce the collision risk for nocturnally migrating birds (Marques *et al.* 2014).

Also at sea, significant collision risks exist for local and migrating seabirds and migrating terrestrial birds. The Belgian part of the North Sea (BPNS) is part of one of the main European migration flyways. Because of its shape, the Southern North Sea acts as a migration bottleneck, concentrating seabirds during migration. An estimated number of no less than 1.0 to 1.3 million seabirds migrate through this area on an annual basis (Stienen *et al.* 2007). Also, large numbers of non-seabirds are known to migrate at sea (Buurma 1987; Alerstam 1990; Lensink *et al.* 2002; Bradarić *et al.* 2020; Manola *et al.* 2020). Estimates of the number of birds seasonally travelling through the Southern North Sea vary from 85 million (Lensink *et al.* 2002) up to several hundreds of millions (estimates of Helgoland mentioned in Hüppop *et al.* 2006), of which the vast majority are terrestrial birds (Bradarić *et al.* 2020). While migratory ducks, geese and waders are restricted to coastal areas, songbirds migrate along a broad front across the North Sea (Vanermen *et al.* 2006). The highest flight intensities are recorded at night during spring and autumn migration and mainly consist of migrating passerines, especially Blackbird *Turdus merula*, Song Thrush *Turdus philomelos*, Redwing *Turdus iliacus* and Robin *Erithacus rubecula* (Krijgsveld *et al.* 2011; Fijn *et al.* 2015). Aside from barrier effects, the development of offshore wind farms (OWFs) in the North Sea might also directly impact these migrating birds through the risk of collision with the turbines, which results in an increased mortality rate.

Songbirds, migrate at altitudes up to several kilometres (Lensink *et al.* 2002; Krijgsveld *et al*. 2011; Brabant *et al*. 2021). According to the radar study by Krijgsveld *et al*. (2011) in the Dutch OWF OWEZ, an average of 330,000 (groups) birds/km crossed the wind farm zone between sea level and 1,385 m altitude (the maximum height of the radar image) each autumn, over 30% of these birds flew at rotor height. The flight altitude is, however, influenced by weather conditions. A general phenomenon is that birds fly at higher altitude with tailwind and that they fly at a lower altitude with headwind (Buurma 1987; Lensink *et al.* 2002).

Peaks of intense songbird migration in the North Sea occur during good weather with favourable, supporting wind conditions (Bradarić *et al*. 2020). Weather conditions can, however, change en route and therefore, crossing the North Sea basin is a risk for terrestrial birds as they cannot rest and refuel if weather conditions become adverse (Bradarić *et al.* 2020; Manola *et al.* 2020). When weather conditions deteriorate, birds will lower their flight altitude which results in large numbers flying at rotor height and thus at risk of collision. Lensink *et al.* (1999) reported three of these "falls" in the period from 1978 until 1990, but concluded, based on limited data at sea, that these events must occur yearly in the Southern North Sea. Fijn *et al.* (2015) also reported regular occurrence of intense bird migration at rotor height in the Dutch part of the North Sea.

At the research platform FINO 1 in the German Bight, a total of 767 dead birds were found during 160 visits between October 2003 and December 2007, distributed over 45 visits and 34 species (Hüppop *et al.* 2016). The most commonly found species were thrushes (76%), followed by starlings (9%) and other songbirds (10%). Collision was the main cause of death (75%) and more than half of the casualties occurred on only three autumn nights. The majority of these casualties occurred during very specific and difficult to predict conditions, characterized by favourable conditions in the areas of departure and rapidly deteriorating weather conditions over the sea during the following night, such as increasing cloudiness, fog, rain and changing wind conditions (Hüppop *et al.* 2006, 2016).

Temporarily stopping the turbine operation during such high collision risk events for songbirds can reduce the number of collision victims. However, curtailing turbines at sea to reduce the collision risk for birds is not yet being applied on a large scale. On 13 May 2023, the wind turbines at the Dutch offshore wind farms Borssele and Egmond aan Zee were stopped for four hours, during a massive songbird migration event (https://www.offshorewind.biz/2023/05/17/dutch-stop-offshore-wind-turbines-to-protect-migratory-birds-in-international-first/). This was an international first for wind farms at sea and was part of a pilot phase of the implementation of such procedure in the Netherlands. To support the ongoing discussions on this topic, this report aims (1) to present an overview of curtailment strategies and procedures to reduce bird collisions in wind farms at sea in North Sea countries and (2) to assess the possibilities to implement such mitigation measure in Belgian OWFs.

2. Methods

To collect information about current or planned curtailment measures in North Sea countries, governmental agencies, research institutes and wind farm developers from the Netherlands, Germany, Denmark, the United Kingdom, France, Norway and Belgium were contacted (Table 1). We inquired if there were any curtailment measures imposed on current and/or planned wind farms at sea or, if this was not the case, whether it is being actively discussed for potential future implementation.

3. Results

3.1. The Netherlands

As mentioned in the introduction, the wind turbines at the Dutch offshore wind farms Borssele and Egmond aan Zee were stopped for four hours, during a massive bird migration event on 13 May 2023 as a test. The rotation speed of the wind turbines was reduced to a

Table 1. List of persons that were contacted to gather information on the application of curtailment in wind farms at sea.

Name	Country	Organisation
Jos de Visser	The Netherlands	Rijkswaterstaat Zee en Delta
Karen Krijgsveld	The Netherlands	Wageningen Environmental Research
Marie Dahmen	Germany	Bundesamt für Seeschifffahrt und Hydrographie (BSH) - Federal Maritime and Hydrographic Agency
Benedikt Holtmann	Germany	Bundesamt für Seeschifffahrt und Hydrographie (BSH) - Federal Maritime and Hydrographic Agency
Freerk Nanninga	Germany	Skyborn Renewables offshore solutions GmbH
Helmut Wendeln	Germany	IBL Umweltplanung GmbH
Søren Keller	Denmark	Danish Energy Agency
Alex Banks	United Kingdom	Natural England
Julie Black	United Kingdom	Joint Nature Conservation Committee (JNCC)
Yann Planque	France	France energies marines
Etienne Berille	France	EDF Renouvelables
Emma Gouze	France	EDF Renouvelables
Roel May	Norway	Norwegian Institute for Nature Research
Steven Vandenborre	Belgium	Federal Public Service Health, Food chain safety and Environment

maximum of two rotations per minute during the predicted night-time peak migration to reduce the collision risk. Egmond aan Zee is the Netherlands’ first offshore wind farm, consisting of 36 wind turbines located 10 to 18 kilometres off the Dutch coast. The two wind farms in Borssele are located at more than 20 kilometres off the coast of the province of Zeeland and comprise 94 and 77 wind turbines.

The shutdown of the wind farms Egmond aan Zee and Borssele during bird migration, was an international first for wind farms at sea and was part of a pilot phase. The goal of the Dutch government is to make this the standard for all operational and future wind farms (Table 2). Rijkswaterstaat is implementing a curtailment procedure for offshore windfarms in the Dutch part of the North Sea on behalf of the Ministry of Economic Affairs and Climate Policy (Van Bemmelen *et al.* 2022). The reasoning of the Dutch government to install a stand-still procedure is based on the Nature Protection Act, which prohibits intentional killing of birds. Article 3.1 of the Nature Protection Act states that it is prohibited to deliberately kill or capture birds naturally living in the wild in the Netherlands of species referred to in Article 1 of the Birds Directive. In addition to that, the duty of care under the Nature Protection Act also requires that ‘damage to all wild animals and plants has to be prevented as far as reasonably practicable’. The competent authority has ruled that the ban on killing birds applies to offshore wind farms in the Netherlands, but that considering the implementation of mitigation measures to reduce bird collisions, an exemption can be granted.

The shutdown procedure relies on a bird migration prediction model, developed by the University of Amsterdam (Bradarić *et al.* in prep.). The model is based on bird migration data from a bird radar installed at sea and meteorological data. It predicts bird migration intensity up to 48 hours in advance. If a certain threshold is exceeded, then the intention is to shut down the turbines. The application

Table 2. Summary of the curtailment measures to reduce the collision risk for migrating birds in offshore wind farms in North Sea countries.

Country	Curtailment implemented in the North Sea	Curtailment in other sea basins	Status of implementation	Criteria to start curtailment
The Netherlands	**yes**	NA	Implementation in pilot phase. Procedure based on predictive bird migration model.	If predefined bird intensity threshold is exceeded, curtailment protocol is initiated.
Germany	no	**yes**	In OWFs in bird migration corridor in the Baltic Sea.	If predefined bird intensity threshold is exceeded.
United Kingdom	no	no	NA	NA
Denmark	no	no	NA	NA
France	no	**yes**	Curtailment implemented in an OWF test site in the Mediterranean.	Curtailment initiated during predefined periods in spring and autumn migration season.
Norway	no	no	NA	NA
Belgium	no	NA	NA	NA

of the standstill provision is concretized in a protocol in consultation with various stakeholders (https://www.noordzeeloket.nl/functies-gebruik/windenergie/start-stop/). Shutting down turbines based on real time bird radar is not possible because this would cause instability in the electricity network. Predicting the intense migration events well in advance with this model, allows grid operator TenneT enough time to maintain the stability of the high-voltage grid, and if necessary, purchase natural gas to guarantee the energy supply during the shutdown. As a validation of the model, a team of bird migration experts also provides a prediction of the bird migration intensity based on their expertise, on predicted weather conditions at the departure locations of migrating birds, next to observations of birds at migration counting sites. They then assess and compare with the prediction model outcome. The final decision to stop the wind farms at sea is with the ministry for Economic Affairs and Climate. This is then communicated to the wind farm operators with an indication of the date and time that turbines need to be stopped.

At this point, the prediction model is based on a limited dataset of bird radar data. The model will be improved by adding more bird radar data to train the model. In order to make the prediction model as accurate as possible and, in time, possibly differentiate between different wind energy areas in the North Sea, it will have to be based on multi-year data from bird radar systems on site. Wind farms will therefore be equipped with such bird radars at the expense of the Dutch government. This research can also reduce knowledge gaps on migratory bird species and thus contribute to future decision-making on offshore wind farms (RVO 2023a). This may also lead to a future adjustment of the shutdown threshold and a differentiation in the threshold value between regions in the North Sea.

The current model is developed for the wind energy areas along the Zeeland and Dutch Coast. Later, a separate model will be developed for the wind area North of the Wadden Islands, and the development of a third model is foreseen for the areas further offshore.

The condition to shut down the turbines during events of heavy bird migration is included in the ‘parcel decision document’ – a parcel is a designated area at sea where wind farms can be constructed. This condition states that the rotation speed of wind turbines needs to be reduced to less than two rotations per minute when the bird density exceeds a certain threshold.

Following on the parcel decision document, a tender is published and wind farm developers can prepare a bid to develop a project in that parcel. The bids are being scored based on criteria that are described in the tender document. In the latest tender, many non-price criteria are included (RVO 2023b). One of these is to develop a shutdown on demand measure which is a local curtailment strategy that partially shuts down the wind farm when target species are at risk of collision (e.g., little gull *Hydrocoloeus minutus*, northern gannet *Morus bassanus*, lesser black-backed gull *Larus fuscus*, black legged kittiwake *Rissa tridactyla*, greater black backed gull *Larus marinus*, herring gull *Larus argentatus*). This will be on top of the general shutdown procedure during heavy songbird migration, as described above.

3.2. Germany

There are curtailment measures for wind farms in the German part of the Baltic Sea but not (yet) for the German North Sea area. The Federal Maritime and Hydrographic Agency of Germany (BSH) is establishing a broad monitoring program on collision risk for the North Sea OWFs, which is described in the latest draft site development plan (BSH 2023a). Wind farm developers will need to develop a monitoring plan to continuously register bird flights in the wind farms. With that knowledge, a curtailment strategy will then be developed that might be applied in the future in the German part of the North Sea.

In the Baltic Sea, a curtailment procedure to reduce collision risk during mass bird migration events will be mandatory for all OWFs within a migration corridor that was described by the German Federal Agency for Nature Conservation (BfN) between the southern tip of Sweden and the coast of Mecklenburg-Western Pomerania (BfN 2020; Table 2). This has been identified as an area of particular importance for bird migration and should therefore be given special consideration in planning (BSH 2023b). In the environmental license of the latest OWF in the Baltic Sea, the Baltic Eagle project, there are specific conditions on bird migration monitoring and curtailment (BSH 2023b). For the first three years the wind farm developer needs to implement monitoring infrastructure consisting of a bird radar system and minimum five camera systems to continuously monitor the bird flux in the wind farm up to an altitude of 1000 m. If the estimated collision mortality is more than 1% of that total number of birds, the operation of the turbines needs to be stopped. The findings from the monitoring during the operational phase will be used to develop and implement site-specific mitigation measures considering varying collision risks at different weather conditions. The wind farm developer needs to propose a plan to BSH on how this curtailment procedure will be applied, but at this point (October 2023) no details are known. This same approach will be applied to all offshore wind farms that are in the Baltic bird migration route.

Aside from the general policy in German waters, there is a particular case where, as a result of a lawsuit by two environmental NGOs, turbine curtailment for nocturnal migration was implemented in the nearshore wind farm Nordergründe, in the inner part of the German Bight close to the Wadden Sea. The Nordergründe offshore wind farm consists of 18 turbines totalling 111 MW. It is located 17 kilometres off the coast of Lower Saxony and was Commissioned in 2017, in shallow coastal waters of the German Bight, in the vicinity of two nature conservation sites. It is 560 m away from the Site of Community Importance (SCI) and Special Protection Area (SPA) 'Nationalpark Niedersächsisches Wattenmeer' (DE 2306-301, DE 2210-401) and around 14 km away from the Site of Community Importance (SCI) 'Hamburgisches Wattenmeer' (DE 2016-301). Additionally, there are two areas of bird protection interest at ca. 1–5 km to the North and to the West of the project site, respectively named Roter Sand and Küstenmeer vor den Ostfriesischen Inseln.

Normally, the construction of an offshore wind farm would not be allowed there because of the possible negative effects on the nearby marine protected areas, but the Nordergründe site was exceptionally included in the spatial development plan as a testing location to gain experience before the development of offshore wind farms at a larger distance from the German coast. Two German NGOs, Bund für Umwelt und Naturschutz Deutschland (BUND) and WWF Germany, launched legal appeals against the project as they considered the permit a breach of nature conservation laws because of the related risks to birds. The project developer, in return, appealed against certain permit conditions. In March 2011, an agreement was concluded between the state of Lower Saxony, the project developer, and the two NGOs, BUND and WWF. The settlement defined a four-year research program to study the bird migration on site and the implementation of a stand-still procedure at the developer's expense. The latter obliged the project developer to stop the turbines during major bird migration events to reduce the collision risks for migrating birds. The stand-still procedure was implemented with the use of a vertical bird radar installed on site. During the migration periods the radar screen was monitored in real time each day for four hours starting at dusk. Radar echoes, assumed to be birds, where counted by the radar operator from sea level up to 300 m altitude on the radar images that were generated every three minutes. When a threshold of 20 radar echoes per radar image or 120 echoes in 20 minutes was exceeded, the turbines were shut down. In the settlement, it was agreed not to have more than 10 shutdowns per year and not to shut down when wind speeds were very high (>7 Bft, as bird migration was expected to be low) or very low (<3 Bft, as turbines are in idling mode). During five years the threshold was exceeded for 14 times. In 11 of those cases, the turbines were shut down. In the other three cases the wind speed was lower than 3 Bft and turbines were already in idling mode. After five years, the state ministry of the environment of Lower Saxony decided that the procedure could be stopped since the results gained from the research program on bird migration (Hill *et al.* 2022) did not provide evidence for a need to continue the obligation to shut down the turbines.

3.3. Denmark

Denmark has no curtailment procedures in place in existing offshore windfarms. The Danish Energy agency is confident that the wind farms have been located outside of important bird migration routes during the planning phase, and that, therefore, there is no reason for curtailment strategies.

In 2026, the wind farm “Aflandshage” will be constructed near “Øresund”, in the Baltic Sea. At that site a curtailment procedure will probably be implemented due to the collision risk for bats.

3.4. United Kingdom

The contacted persons at Natural England and JNCC confirm that at this moment no curtailment measures are applied anywhere in UK waters, and that there are no plans to implement curtailment in the near future. Although it is not being actively discussed at this point, it should also not be ruled out as a future measure being put in place in UK waters and it may be something that is discussed/considered more in the future.

3.5. France

The development of offshore wind farms in France recently started. The first wind farm in French waters, at Saint-Nazaire, was commissioned in 2022. Other projects are currently under construction. A curtailment procedure is not included in the environmental measures of the first French OWF. However, this type of procedure is increasingly being discussed in the context of future wind farms in France, although these discussions are still at an early stage.

In the test wind farm Provence grand large (PGL) in the French Mediterranean, consisting of three floating turbines that will be commissioned in the first quarter of 2024, a curtailment measure will be implemented during the spring and autumn migration periods (Table 2). In April, the turbines need to be stopped during six consecutive nights, starting one hour before sunset. In September, this will be the case during seven consecutive nights. This measure is imposed by the environmental permit of the project and these periods were chosen based on a bird radar study at the coastline, during which the highest fluxes were detected in April and September. In 2024, a bird radar will be installed on the platform of one of the turbines and the aim is to improve the curtailment measure based on insights from that study.

3.6. Norway

Only last year, a first offshore wind farm, consisting of 11 floating turbines, was installed in Norwegian waters. A research programme to study bird migration in the area, including the use of weather radar data, a bird radar on site and citizen science has recently been financed. However, mitigating measures such as curtailment are not implemented yet.

3.7. Belgium

In Belgium, a first round of wind farms is constructed and fully operational since 2020. A second area for wind farm development – the Princess Elisabeth Zone (PEZ) – has been designated in the marine spatial plan (Chapter 1). Applications for environmental permits for wind farms in the PEZ have recently been submitted and are now subject to a licensing procedure. Conditions "to avoid, prevent or limit and, if possible, compensate significant adverse effects on the environment" (art.21 §2 – Law for the Protection of the Marine Environment and for the Organization of Marine Spatial Planning in the Belgian maritime areas December 11, 2022) can be imposed in the environmental permit. Standstill procedures to reduce the collision risk are not implemented in the environmental permits of the first round of Belgian offshore wind farms. However, the legislation foresees an option to change or add conditions to existing environmental permits, if there is a necessity to do so. Up until now, this has not been the case. But, from a juridical perspective there is a possibility to implement a curtailment procedure for existing and future wind farms.

4. Discussion

From this assessment, it is clear that in the different North Sea countries, the Netherlands is pioneering with the implementation of the curtailment measures in wind farms at sea, but also Germany and France are initiating tests with curtailment. The Netherlands have developed an approach based on a predictive bird migration model that allows all stakeholders (e.g., wind farm operators, grid operators) to prepare for a planned shutdown of the turbines when a high collision risk is expected. By gradually improving the bird migration model and developing it for different areas within the Dutch part of the North Sea, it will be possible to apply this approach on all wind farms at sea in the Netherlands. Other countries are also taking steps towards the implementation of some form of curtailment strategies or are open for discussions on the topic.

To ensure the effectiveness of such measures, it is essential to correctly assess events with a high risk of collisions. This is crucial not only for minimizing energy production losses but also for maximizing the prevention of collisions.

The earlier statements by for example Cook *et al.* 2011 and Marques *et al.* 2014 regarding the effectiveness of curtailment measures are confirmed by recent studies. A study by Klop & Brenninkmeijer (2020) in the Eemshaven wind warm on land shows that applying a standstill measure during nocturnal migration peaks is an effective measure to prevent collision victims. Not a single collision victim was found underneath

10 turbines which were shut down for 10 nights of relatively intensive bird migration, while the number of victims found beneath the operating turbines was nearly three times higher than the average found on all other nights. Bradarić *et al.* (2023) claim that in the Dutch North Sea, curtailments should be performed during only 18 hours in spring and 26 in autumn, to minimize collision risk for 50% of birds migrating through the area. This represent 2.5% and 5.5% of the migration period, respectively and in that case, the yearly amount of energy lost due to curtailments would be 0.05% in spring and 0.07% in autumn. Hill *et al.* (2022) assessed the effectiveness of the shutdowns of the Nordergründe wind farm in the German Baltic Sea by calculating how many additional collisions continued operation of the wind turbines on the shutdown nights could theoretically have led to. They concluded that the number of collisions per year would have increased only insignificantly if no shutdowns had been carried out and the wind farm had continued to operate on the shutdown nights.

This demonstrates the need for sound, site-specific monitoring programs to assess the effectiveness of the implemented measures and to further finetune them. As such, the science-base to validate that shutting down turbines during events of mass migration actually prevents collisions of birds will be enlarged. One of the most important knowledge gaps remains the direct measurements of bird collisions at offshore wind facilities (Potiek *et al.* 2022). Such monitoring programs are being developed in several North Sea countries, e.g., Germany (BSH 2023a) and the Netherlands (RVO 2023a). These will gain valuable insights into the patterns of bird migration behaviour at sea and contribute to the development of more targeted mitigation and conservation strategies.

As curtailment measures can result in significant power generation reductions, wind farm operators are a key stakeholder for the effective implementation of curtailment strategies. To anticipate on the possible obligation to implement curtailment measures in the future, more and more industry parties are also investing in tools and technologies to improve curtailment strategies for birds and bats. For example, Ørsted recently invested in a "deep-tech" start-up that is developing an artificial intelligence system to monitor and track birdlife at offshore wind farms. The Vestas Bat Protection System (VBPS) is a newly developed software module intended to incorporate additional weather variables into curtailment decisions and increase power generation while maintaining conservation benefits (Whitby *et al.* 2023).

With the rapid growth in the number of wind farms in the North Sea, it is extremely important that this is done in the most ecologically responsible way possible. One of the conclusions of a stakeholder involvement process done by Voigt *et al.* (2019) was that concerns about bird collisions and other environmental impacts are common among stakeholders, including local communities, environmental organizations, and policymakers, and that proactive measures are necessary to make wind energy production ecologically sustainable. Temporary turbine shutdowns could be highly effective for reducing collision mortalities during certain scenarios, such as poor weather conditions during migration, bringing large numbers of passerines into the range of turbine rotors (Harwood & Perrow 2019). It is however important to note that wind turbine curtailment is just one aspect of a comprehensive approach to mitigate the impact of wind farms on bird populations. Responsible development further entails proper site selection, pre-construction environmental assessments, and post-construction monitoring. A regional approach to the implementation of curtailment strategies, including all possible stakeholders, could maximise the efficiency and ecological benefits of such policy measures.

References

Adams, E.M., Gulka, J., & Williams, K.A. 2021. A review of the effectiveness of operational curtailment for reducing bat fatalities at terrestrial wind farms in North America. *PLoS One* 16 (11): e0256382. https://doi.org/10.1371/journal.pone.0256382

Alerstam T. 1990. *Bird Migration*. Cambridge University Press, Cambridge, pp. 420.

Arnett, E.B., Huso, M.M.P., Schirmacher, M.R. & Hayes, J.P. 2010. Altering turbine speed reduces bat mortality at wind-energy facilities. *Frontiers in Ecology and the Environment* 9: 209–214. https://doi.org/10.1890/100103

Behr, O., Brinkmann, R., Hochradel, K., Mages,J., Korner-Nievergelt, F., Reinhard, H., Simon, R., Stiller, F., Weber, N. & Nagy, M. 2018. *Bestimmung des Kollisionsrisikos von Fledermäusen an Onshore-Windenergieanlagen in der Planungspraxis*. Final report of the project financed by Federal Ministry for Economics and Energy (Förderkennzeichen 0327638E).

BfN, 2020. *Naturschutzfachlicher Planungsbeitrag des Bundesamtes für Naturschutz zur Fortschreibung der Raumordnungspläne für die deutsche Ausschließliche Wirtschaftszone in der Nord- und Ostsee*. 77 pp.

Brabant, R., Rumes, B. & Degraer, S. 2021. Occurrence of intense bird migration events at rotor height in Belgian offshore wind farms and curtailment as possible mitigation to reduce collision risk. *In*: Degraer, S., Brabant, R., Rumes, B. & Vigin, L. (eds) *Environmental Impacts of Offshore Wind Farms in the Belgian Part of the North Sea: Attraction, Avoidance and Habitat Use at Various Spatial Scales*. Memoirs on the Marine Environment: 47–55. Royal Belgian Institute of Natural Sciences, OD Natural Environment, Marine Ecology and Management, Brussels.

Bradarić, M., Bouten, W., Fijn, R.C., Krijgsveld, K.L. & Shamoun-Baranes, J. 2020. Winds at departure shape seasonal patterns of nocturnal bird migration over the North Sea. *Journal of Avian Biology* 51 (10): e02562. https://doi.org/10.1111/jav.02562

Bradarić, M., Kranstauber, B., Bouten, W. & Shamoun-Baranes, J. 2023. *The North Sea Wind Turbine Curtailments Informed by Near-term Forecasts*. CWW 2023 Book of Abstracts.

Bradarić, M., Kranstauber, B., Bouten, W. & Shamoun-Baranes, J., in prep. Forecasting nocturnal bird migration to mitigate collisions with offshore wind turbines in the southern North Sea. *In*: *On the Radar: Weather, Bird Migration and Aeroconservation over the North Sea*: 95-115.

BSH, 2023a. *Vorentwurf Flächenentwicklungsplan*. 80 pp.

BSH, 2023b. *Planfeststellungsbeschluss Offshore-Windenergiepark "Baltic Eagle" Aktenzeichen: 111/Baltic Eagle/PFV*. 415 pp.

Buurma, L.S. 1987. Patronen van hoge vogeltrek boven het Noordzeegebied in oktober. *Limosa* 60: 63–74.

Cook, A.S.C.P., Ross-Smith, V.H, Roos, S., Burton, N.H.K., Beale, N., Coleman, C., Daniel, H., Fitzpatrick, S., Rankin, E., Norman, K. & Martin, G. 2011. *Identifying a Range of Options to Prevent or Reduce Avian Collision with Offshore Wind Farms Using a UK-Based Case Study*. BTO Research Report No. 580. British Trust for Ornithology, The Nunnery, Thetford, Norfolk, IP24 2PU UK, 197 p.

de Lucas, M., Ferrer, M., Bechard, M.J. & Muñoz, A.R. 2012. Griffon vulture mortality at wind farms in southern Spain: distribution of fatalities and active mitigation measures. *Biological Conservation* 147: 184–189. https://doi.org/10.1016/j.biocon.2011.12.029

Drewitt, A.L. & Langston, R.H.W. 2006. Assessing the impact of wind farms on birds. *Ibis* 148: 29–42.

Fijn R.C., Krijgsveld K.L., Poot M.J.M. and Dirksen S. 2015. Bird movements at rotor heights measured continuously with vertical radar at a Dutch offshore wind farm. *Ibis* 157 (3): 558–566.

Fox, A.D., Desholm, M., Kahlert, J., Christensen, T.K. & Petersen, I.K. 2006. Information needs to support environmental impact assessment of the effects of European marine offshore wind farms on birds. *Ibis* 148: 129–144.

Harwood, A.J.P. & Perrow, M.R. 2019. Mitigation for birds with implications for bats. *In*: Perrow, M.R. (ed.) *Wildlife and Wind Farms-Conflicts and Solutions: Offshore: Monitoring and Mitigation*: 242–280. Pelagic Publishing Ltd.

Hill, R., Rebke, M. & Aumüller, R. 2022. *Bericht zur Durchführung des Forschungsvorhabens Vogelzug am Offshore-Windpark Nordergründe. Ergebnisse der vierjährigen Untersuchungen 03/2018 – 02/2022 und zusätzlich separat des vierten Untersuchungsjahres 2021*. Avitec Research GbR. 167 p.

Hüppop, O., Dierschke J., Exo K.-M., Fredrich E. & Hill R. 2006. Bird migration studies and potential collision risk with offshore wind turbines. *Ibis* 148 (s1): 90–109.

Hüppop, O., Hüppop, K., Dierschke, J. & Hill, R. 2016. Bird collisions at an offshore platform in the North Sea. *Bird Study* 63: 73–82.

Klop, E. & Brenninkmeijer, A. 2020. *Aanvaringsslachtoffers Windpark Eemshaven najaar 2018 & voorjaar 2019*. A&W-rapport 3189, Altenburg & Wymenga ecologisch onderzoek, Feanwâlden.

Krijgsveld, K., Fijn, R., Japink, M., van Horssen, P., Heunks, C., Collier, M., Poot, M., Beuker, D. & Dirksen, S. 2011. *Effect Studies Offshore Wind Farm Egmond aan Zee - Final Report on Fluxes, Flight Altitudes and Behaviour of Flying Birds*. Report No. OWEZ_R_231_T1_20111114_flux&fligh, Report by Bureau Waardenburg bv, pp. 334.

Lensink, R., Camphuysen, C.J., Jonkers, D.A., Leopold, M.F., Schekkerman, H. & Dirksen, S. 1999. *Falls of Migrant Birds, an Analysis of Current Knowledge*. Report No. 99.55, Bureau Waardenburg bv, Culemborg, Netherlands.

Lensink, R., van Gasteren, H., Hustings, F., Buurma, L., van Duin, G., Linnartz, L., Vogelzang, F. & Witkamp, C. 2002. *Vogeltrek over Nederland 1978–1993*. Schuyt & Co., Haarlem.

Manola, I., Bradarić, M., Groenland, R., Fijn, R., Bouten, W. & Shamoun-Baranes, J. 2020. Associations of synoptic weather conditions with nocturnal bird migration over the North Sea. *Frontiers in Ecology and Evolution* 8: e542438. https://doi.org/10.3389/fevo.2020.542438

Marques, A.T., Batalha H., Rodrigues S., Costa H., Ramos Pereira M.J., Fonseca C., Mascarenhas M. & Bernardino, J. 2014. Understanding bird collisions at wind farms: an updated review on the causes and possible mitigation strategies. *Biological Conservation* 179: 40–52. https://doi.org/10.1016/j.biocon.2014.08.017

May, R.F. 2017. Mitigation for birds. In M. R. Perrow (ed.) *Wildlife and windfarms, conflicts and solutions. Volume 2: Monitoring and Mitigation*, Pelagic Publishing Ltd. 124-144.

Offshorewind.biz. 2023. Dutch shut down offshore wind turbines to save birds in 'international first'. https://www.offshorewind.biz/2023/05/17/dutch-stop-offshore-wind-turbines-to-protect-migratory-birds-in-international-first/ [accessed 12 September 2023].

Potiek, A., Leemans, J.J., Middelveld, R.P. & Gyimesi, A. 2022. *Cumulative Impact Assessment of Collisions with Existing and Planned Offshore Wind Turbines in the Southern North Sea. Analysis of Aadditional Mortality Using Rate modelling and Impact Assessment Based on Population Modelling for the KEC 4.0*. Report 21-205. Bureau Waardenburg, Culemborg.

RVO, 2023a. *Ontwerpkavelbesluit kavel Alpha windenergiegebied IJmuiden Ver.* 119 pp.

RVO, 2023b. *Regeling vergunningverlening kavel Alpha in windenergiegebied IJmuiden Ver*. 41 pp.

Smallwood K.S. & Karas B. 2009. Avian and bat fatality rates at old-generation and repowered wind turbines in California. *Journal of Wildlife Management* 73 (7): 1062–1071. https://doi.org/10.2193/2008-464

Stienen, E.W.M., Van Waeyenberge, J., Kuijken, E. & Seys, J. 2007. Trapped within the corridor of the Southern North Sea: the potential impact of offshore wind farms on seabirds. *In*: de Lucas, M., Janss, G.F.E. & Ferrer, M. (eds). *Birds and Wind Farms - Risk Assessment and Mitigation*: 71–80. Quercus, Madrid.

Thaxter, C.B., Buchanan, G.M., Carr, J., Butchart, S.H.M., Newbold, T., Green, R.E., Tobias, J.A., Foden, W.B., O'Brien, S. & Pearce-Higgins, J.W. 2017. Bird and bat species' global vulnerability to collision mortality at wind farms revealed through a trait-based assessment. *Proceedings of the Royal Society of London B: Biological Sciences* 284 (1862): e20170829. https://doi.org/10.1098/rspb.2017.0829

Tomé, R., Leitão, A.H., Pires, N. & Canário, F. 2017a. Inter-and intra-specific variation in avoidance behaviour at different scales in migratory soaring birds. *In*: *Book of abstracts. Conference on Wind Energy and Wildlife Impacts, Estoril, Portugal*: 61–62. Available from https://tethys.pnnl.gov/publications/2017-conference-wind-energy-wildlife-impacts-book-abstracts [accessed 8 April 2019].

Tomé, R., Canário, F., Leitão, A.H., Pires, N. & Repas, M. 2017b. Radar assisted shutdown on demand ensures zero soaring bird mortality at a wind farm located in a migratory flyway. *In*: Köppel, J. (ed.) *Conference on Wind Energy and Wildlife Interactions Presentations from the CWW2015 conference. Cham*: 119–133. Springer International Publishing.

Van Bemmelen, R.S.A., de Groeve, J. & Potiek, A. 2022. *Potential Curtailment Regimes for Offshore Wind Farms: Exploring the Relation Between Wind Speed, Power Yield and Bird Migration Intensity. Spatial variation in migration intensity and optimization of curtailment threshold*. Bureau Waardenburg Report 22-171. Bureau Waardenburg, Culemborg.

Vanermen, N., Stienen, E.W.M., Courtens, W. & Van de walle, M. 2006. *Referentiestudie van de avifauna van de Thorntonbank*. Rapport INBO.A.2006.22. Instituut voor Natuur- en Bosonderzoek, Brussel.

Voigt, C.C., Lehnert, L.S., Petersons, G., Adorf, F. & Bach, L. 2015. Wildlife and renewable energy: German politics cross migratory bats. *European Journal of Wildlife Research* 61: 213–219. https://doi.org/10.1007/s10344-015-0903-y

Voigt, C.C., Straka, T.M., & Fritze, M. 2019. Producing wind energy at the cost of biodiversity: A stakeholder view on a green-green dilemma. *Journal of Renewable and Sustainable Energy* 11 (6): e063303. https://doi.org/10.1063/1.5118784

Whitby, M., Gottlieb, I., Donovan, C., New, L., Leckband, J. & Allison, T. 2023. *Developing and Evaluating a Smart Curtailment Strategy Integrated with a Wind Turbine Manufacturer Platform*. CWW 2023 Book of Abstracts.

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