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Assessing the Impact of the Westdiep Sea Farm on Avifauna

Evaluation of the Monitoring Approach through Power Analysis and Recommendations for Future Monitoring

Robin Daelemans, Nicolas Vanermen, Wouter Courtens, Marc Van de walle, Hilbran Verstraete, Eric Stienen

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Authors:
Robin Daelemans, Nicolas Vanermen, Wouter Courtens, Marc Van de walle, Hilbran Verstraete, Eric Stienen

Research Institute for Nature and Forest (INBO)

Reviewers:
Robin Brabant
Hilde Eggermont

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Location:
Herman Teirlinckgebouw
INBO Brussel
Havenlaan 88 bus 73, 1000 Brussels, Belgium
vlaanderen.be/inbo

e-mail:
robin.daelemans@inbo.be

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ASSESSING THE IMPACT OF THE WESTDIEP SEA FARM ON AVIFAUNA

Evaluation of the Monitoring Approach through Power Analysis and Recommendations for Future Monitoring

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Robin Brabant (RBINS)

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Abstract

In 2020, an environmental permit was granted for the development of the Westdiep Sea Farm, a longline aquaculture project located in the nearshore waters of the Belgian North Sea (BNS), approximately 4.5 km offshore Nieuwpoort. The 4.54 km² project area is designated under the Belgian Marine Spatial Plan for extractive species such as mussels, oysters, and seaweed. The area lies entirely within SPA-B1, a Special Protection Area under the EU Birds Directive that is recognized for its ecological importance as a foraging and resting habitat for species including Sandwich tern (*Thalasseus sandvicensis*), great crested grebe (*Podiceps cristatus*), and red-throated diver (*Gavia stellata*). The presence of aquaculture structures and increased vessel traffic associated with the sea farm may lead to avoidance in disturbance-sensitive seabird species, while other species might be attracted due to enhanced food and resting opportunities.

To comply with permit conditions, a seabird monitoring program was established to detect such responses, following a Before-After Control-Impact (BACI) design with four ship-based surveys annually over a five-year period. However, the first year of baseline monitoring revealed that seabird densities within the project area were generally low and highly variable, raising concerns about the program's ability to detect statistically significant changes in seabird densities over time. To further investigate these concerns, a simulation-based power analysis was conducted. Based on species- and location-specific data characteristics derived from the INBO seabird dataset (2001-2023) and the initial four baseline surveys, a large set of randomised datasets was generated. In these datasets, various increases and decreases in seabird densities were simulated under different monitoring strategies, namely seasonal surveys, peak-density surveys, and monthly surveys. Statistical tests were then applied to assess whether these changes in seabird densities would be detected as significant, enabling us to estimate the probability of detecting a true change in seabird density under the proposed monitoring design—i.e. the statistical power.

Results from the power analysis indicate that the current monitoring design (16 post-impact surveys) is unlikely to achieve the targeted statistical power of 80% to detect a 50% change in seabird numbers for most species. Even under optimized scenarios that align surveys with month of peak seabird densities, statistical power remained limited, although notable improvements were observed for species such as great cormorant and Sandwich tern. Increasing the sampling effort, for example by conducting monthly surveys, did not lead to proportional gains in power and is not feasible within the available resources.

Therefore, we recommend adjusting the timing of the 16 planned surveys to coincide with periods of peak abundance of key species in the area. Scheduling surveys between January and April is expected to maximize impact detection potential for red-throated diver, great crested grebe, and common scoter. In addition, these targeted surveys will be supplemented by observations from INBO's long-term seabird monitoring campaigns and anecdotal sightings reported by the crew of the sea farm. This revised monitoring design is expected to generate valuable insights, not only about the ecological impact of the current project but also for building the scientific and policy frameworks that will guide future offshore aquaculture developments.

Samenvatting

In 2020 werd een milieuvergunning afgeleverd voor de ontwikkeling van de Westdiep Sea Farm, een longline-aquacultuurproject in de kustwateren van de Belgische Noordzee (BNZ), op ongeveer 4,5 km voor de kust van Nieuwpoort. De projectzone van 4,54 km² is volgens het Belgisch Marien Ruimtelijk Plan bestemd voor de kweek van extractieve soorten zoals mosselen, oesters en zeewier. De zone ligt volledig binnen SBZ-V1, een Speciale Beschermingszone onder de Europese Vogelrichtlijn die erkend is als belangrijk foerageer- en rustgebied voor onder meer de grote stern (*Thalasseus sandvicensis*), fuut (*Podiceps cristatus*) en roodkeelduiker (*Gavia stellata*). De aanwezigheid van aquacultuurinfrastructuur en het toegenomen scheepsverkeer van en naar de zeeboerderij kunnen verstoringsgevoelige zeevogels mogelijk verdrijven, terwijl andere soorten net aangetrokken kunnen worden door extra foerageer- en rustmogelijkheden.

Om aan de vergunningsvoorwaarden te voldoen, werd een monitoringsprogramma voor zeevogels opgezet, gebaseerd op een Before-After Control-Impact (BACI) design, met vier scheepstellingen per jaar over een periode van vijf jaar. Tijdens het eerste referentiejaar bleken de zeevogeldensiteiten in het studiegebied laag en sterk variabel, wat vragen opriep over de mogelijkheid om statistisch significante veranderingen in densiteiten te detecteren. Daarom werd een poweranalyse uitgevoerd, waarbij op basis van soortspecifieke en locatiegebonden datakarakteristieken uit de INBO-zeevogeldatabank (2001–2023) en de vier referentietellingen verschillende datasets werden gegenereerd. In deze datasets werden verschillende toe- of afnames in zeevogeldichtheden gesimuleerd onder uiteenlopende monitoringsstrategieën, zijnde seizoensgebonden tellingen, tellingen in piekmaanden en maandelijkse telingen. Aan de hand van statistische testen werd nagegaan of deze gesimuleerde veranderingen in zeevogelaantallen significant zouden worden bevonden, waarna de statistische power werd berekend, dit is de kans dat een werkelijke verandering zou worden opgemerkt.

De resultaten tonen aan dat de huidige monitoringstrategie (16 post-impact tellingen) voor de meeste soorten waarschijnlijk onvoldoende statistische power zal genereren om een verandering van 50% in zeevogelaantallen aan te tonen met een beoogde power van 80%. Zelfs in geoptimaliseerde scenario's die focussen op de maanden met de hoogste dichtheden bleef de power beperkt, al was er bij soorten als aalscholver en grote stern wel een verbetering. Het verhogen van de telfrequentie, bijvoorbeeld tot maandelijkse tellingen, leidde niet tot proportioneel hogere power en is bovendien niet haalbaar binnen de beschikbare middelen.

Op basis van deze resultaten wordt aanbevolen om de 16 geplande tellingen uit te voeren in de maanden waarin de grootste aantallen worden verwacht van enkele belangrijke soorten in het gebied. Tellingen tussen januari en april bieden het meeste potentieel voor het detecteren van veranderingen bij roodkeelduiker, fuut en zwarte zee-eend. Deze gerichte tellingen zullen bovendien worden aangevuld met waarnemingen uit het langlopende zeevogelmonitoringsprogramma van het INBO, alsook met anekdotische observaties van de bemanning van de zeeboerderij. Deze aangepaste monitoringsstrategie zal belangrijke inzichten opleveren, zowel over de ecologische impact van dit project als voor het uitwerken van wetenschappelijke en beleidskaders voor toekomstige offshore aquacultuurprojecten.

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1 INTRODUCTION

In December 2020, 'Westdiep Sea Farm' was granted an environmental permit for the installation and operation of a nearshore longline aquaculture project in 'Zone C' (Figure 1A). This zone is located 4.5 km offshore from Nieuwpoort and is one of the commercial and industrial areas designated under the Marine Spatial Plan for the Belgian part of the North Sea (BNS) for the period 2020-2026 (Royal Decree MSP 2020). Under the current legal framework, only the cultivation of extractive species is permitted – i.e. species that do not require external nutrient input. The project area covers a total of 4.54 km^2, of which 67% is allocated for mussel cultivation, 6% for oyster farming, 6% for seaweed production and 22% is reserved for navigational passage. The project is being implemented in phases. The first 50 longlines were installed in 2022, followed by an additional 96 longlines in 2024. As of 2025, approximately 1 km^2 (ca. 25%) of zone C has been developed, all dedicated to mussel cultivation, concluding phase I of the project. The project aims to fully develop the sea farm by 2028 (IMDC 2020).

The entire project area lies within Special Protection Area for birds SPA-B1, a 110 km^2 area with a high ecological value as a foraging and resting habitat for seabirds. It was designated as an SPA under the European Birds directive (79/409/EEC) due to its importance for Sandwich tern (*Thalasseus sandvicensis*) and great crested grebe (*Podiceps cristatus*), but regularly also supports significant numbers of red-throated diver (*Gavia stellata*), common scoter (*Melanitta nigra*), little gull (*Hydrocoloeus minitus*), lesser black-backed gull (*Larus fuscus*) and great black-backed gull (*Larus marinus*) (Degraer et al., 2020). During the operational phase of the project, several potential impacts on seabirds can be anticipated. Operation of the sea farm will involve approximately 600 vessel movements per year to and from the site, potentially causing disturbance to disturbance-sensitive species such as red-throated diver and common scoter (Garthe & Hüppop, 2004). In addition, the project will result in a direct but limited habitat loss for these species. Conversely, certain species – particularly great cormorants and gulls – may be attracted to the area due to the increased availability of resting structures. Also an increased food availability potentially attracts seabirds. Common scoters may for instance be drawn to the area due to the presence of mussels (Richman, 2013), although this is not very likely given the high disturbance-sensitivity of this species. The tidal currents around the structures combined with increased prey availability can possibly attract great crested grebes, terns, gulls and other seabirds (Lieber et al., 2019; Price et al., 2017).

The environmental permit associated with the project requires the implementation of a monitoring programme to assess its effects on the marine environment, including seabirds. In particular, the monitoring must evaluate potential attraction or avoidance responses of seabirds to both the physical presence of the aquaculture structures and the increased vessel traffic associated with operational activities. The monitoring strategy serves a dual purpose. First, it must enable the *a posteriori* detection and quantification of project-related effects, allowing for the proposal of site-specific mitigation measures in the event of significant or irreversible impacts. Second, it aims to enhance understanding of the nature and underlying drivers of such effects, so that the acquired knowledge can be used to *a priori* inform the further development of the current project and guide the design of future similar initiatives—thereby helping to prevent negative impacts beyond the specific site (BMM, 2020). Consequently, as part of the environmental impact assessment, a monitoring plan was imposed to assess the attraction and avoidance behaviour of seabirds in response to the sea farm. According to Annex F of the advisory document (BMM, 2020) this monitoring should consist of four ship-based surveys per year for a five-year period to estimate seabird densities in both the project area and a designated reference area. The planned timeline for this monitoring consists of one year during Phase I, one year during Phase II, and three years during Phase III of the project.

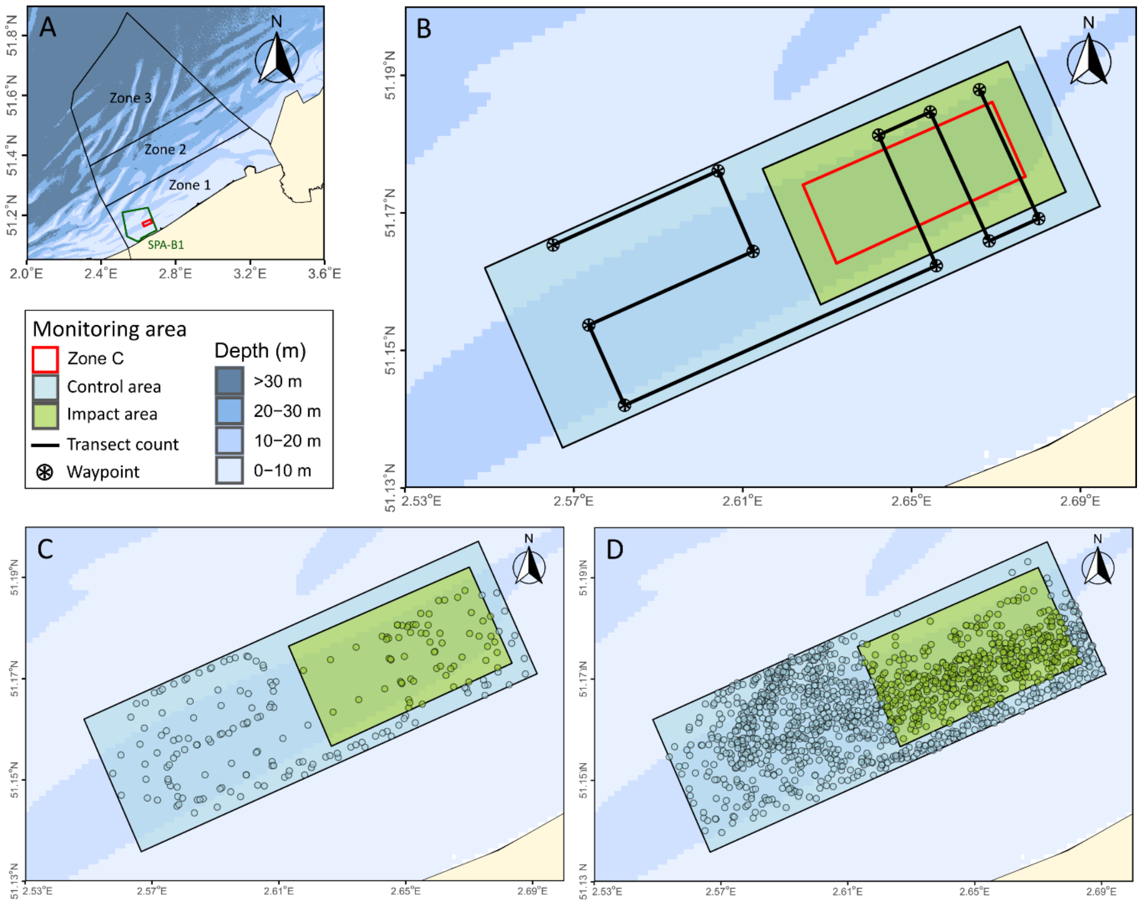

Figure 1. Overview of the study area. (A) Project area 'Zone C' (red rectangle) located within the Special Protection Area for Birds 'SPA-B1'. The three larger zones (Zone 1-3) indicate the nearshore, midshore and offshore parts of the Belgian North Sea. (B) The impact area (green, Zone C plus a 500 m buffer), and reference area (blue) used for the BACI analyses. The black line and waypoints indicate the monitoring transect. (C) Bird observations *(circles) recorded during the four targeted monitoring campaigns. (D) Bird observations (circles) from the targeted monitoring campaigns supplemented with observations from the INBO seabird dataset collected between 2001 and 2023.*

In 2023, results were reported from the first four ship-based seabird surveys, covering both the project and reference areas. These initial surveys revealed that the current monitoring approach generated insufficient data to reliably characterise the seasonal occurrence of different seabird species. This was primarily due to the limited number of surveys, unexpectedly low seabird densities during the survey periods, and the relatively small size of the study area. To establish a more robust baseline, these data were supplemented with records from INBO's seabird dataset collected between 2001 and 2023. However, the scarcity of data resulting from the initial four surveys raised concerns about whether the planned post-impact monitoring strategy - comprising 16 surveys - would be sufficient to detect potential impacts of the sea farm on seabirds with adequate statistical confidence. As a result, it was proposed to formally evaluate the current monitoring strategy using a power analysis. Such analysis calculates the statistical power, which is the probability of correctly detecting a true effect when it is present. Statistical power is primarily influenced by the size of the effect being tested, the sample size, the variability within the data, and the chosen significance level (Underwood & Chapman, 2003). Based on the outcomes of this analysis, adjustments to the monitoring strategy or alternative approaches may be recommended to improve the ability to detect potential impacts with sufficient statistical confidence.

This report presents the results of the power analysis. Using species- and location-specific data characteristics derived from the INBO seabird dataset and the results of the first four surveys, a large number of randomised datasets were generated. These datasets were used to simulate both decreases and increases in seabird densities - taking into account the currently planned monitoring strategy - followed by statistical testing to determine whether such changes would be detected as statistically significant. This allowed us to estimate the likelihood that a true change in seabird density would be detected under the proposed monitoring programme—i.e. the statistical power. Our objective was to achieve a power of at least 80% for detecting a 50% increase or decrease in the most relevant seabird species. Based on the outcomes of this analysis, we provide recommendations on how to adapt the monitoring strategy for the future.

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2 MATERIALS AND METHODS

2.1 STUDY DESIGN AND MONITORING SETUP

To assess the potential impact of the Westdiep sea farm on seabirds, an impact area and a reference area were delineated around the project zone (Figure 1B). The impact area includes the project site 'Zone C' and a 500-meter buffer, corresponding to the safety zone defined during the initial project phase, where effects of the sea farm are expected. Adjacent to this impact area, a reference area was established with similar seabird densities, coastal proximity and water depth to ensure comparability between both areas (Vanermen et al., 2010, 2015). To monitor seabird presence within and around the project zone, a transect crossing both areas was established, along which ship-based seabird counts were conducted following the standard ESAS (European Seabirds at Sea) methodology (Figure 1B). This method combines transect counts for birds resting on the water with snapshot counts for birds in flight, using a standardized transect width of 300 m (Tasker et al., 1984). All surveys were conducted on days with wind speeds below 6 Bft, and wave heights not exceeding 2 m. Seabird data were recorded either in 2-minute or 10-minute intervals, but aggregated by area (impact vs. control) and by monitoring day to reduce potential autocorrelation between consecutive observations and to minimize overall variance (Vanermen et al., 2015). Daily totals for each area were subsequently calculated and, after adjusting for the distance covered, converted into seabird densities.

Using this method to estimate seabird densities in both the control and impact area allows to perform a Before-After Control-Impact (BACI) analysis, where bird counts collected prior to the development of the sea farm can be compared to those collected afterward. The use of a reference area unaffected by the project makes it possible to separate changes in seabird presence caused by the sea farm from natural temporal variations in seabird numbers.

2.2 DATA SELECTION, DATA SIMULATION AND POWER ANALYSIS

As previously mentioned, data from the four targeted monitoring campaigns were supplemented with seabird observations collected by INBO between 2001 and 2023. This substantially increased the dataset from 203 to 1421 observations within the study area (Figure 1C, D), allowing for a more robust estimation of baseline seabird densities prior to the construction of the sea farm (Vanermen et al., 2023). This reference dataset was subsequently used for the power analyses, following the approach outlined in Vanermen et al. (2015).

In a first step, species-specific data characteristics such as over-dispersion, seasonal variation, and zero inflation were modelled using the reference dataset. The models were fitted using either a negative binomial (NB) or a zero-inflated negative binomial (ZINB) distribution. Negative binomial distributions are generally preferred over Poisson or quasi-Poisson models in cases of strong overdispersion, which is common in seabird data due to their aggregated distribution, leading to a high proportion of zero counts and a variance that exceeds the mean (Zuur et al., 2009). When the number of zero counts in the data exceeded what could be explained by a standard NB model, a ZINB model was applied. This model accounts for excess zeros by combining two components: one that models actual bird counts and another that estimates the probability of structural zeros. The response variable was the total number of birds observed per survey. Explanatory variables included area type (control vs. impact, categorical) and seasonality (continuous) to account for temporal trends in seabird densities. Seasonality was modelled either by including month as a factor or by fitting a linear combination of sine and cosine terms to represent cyclical patterns (Table A1; Vanermen et al., 2015). To correct for variation in survey effort, the surveyed area (in km²) was included as an offset variable in the models.

Such model was constructed for each bird species by testing for zero inflation, selecting the most appropriate seasonal structure, and assessing whether the inclusion of the area variable (control vs. impact) improved model fit. Model selection was based on Akaike Information Criterion (AIC) values, and the resulting parameter estimates were used to simulate artificial Before-After Control-Impact (BACI) datasets. For each species, nine scenarios were simulated, based on different monitoring designs. These included: (1) seasonal surveys conducted four times per year, (2) surveys restricted to the four months in which the species reached its highest densities, and (3) monthly surveys (12 per year). For each monitoring design, simulated decreases in species numbers of 25%, 50%, and 75% were applied, resulting in a total of nine scenarios per species. It is worth noting that simulating an increase in species numbers would yield equivalent results due to the symmetrical nature of the power analysis, i.e. the power to detect for instance a 50% decrease in numbers is the same as to detect a 50% increase in numbers The monitoring period was defined as 22 years prior to impact for the reference data (2001–2023) and 15 years after impact for the simulated data. For great cormorant, only 17 years prior to impact were considered, reflecting their population recovery that began in the late 1980s after several decades of near absence, with numbers stabilizing from 2006 onward. A total of 500 datasets were simulated for each scenario.

Finally, to assess statistical power, impact models were fitted to each of the simulated datasets to estimate the likelihood of statistically detecting seabird responses to the sea farm under the different monitoring scenarios. These models tested for the effect of the sea farm by either adding an interaction term between time (BA, before vs. after impact) and area (CI, control vs. impact) to the data simulation model, or - when the area variable was not included in the original model - by adding a binary factor indicating farm presence (F). Statistical power was then estimated as the percentage of simulations in which the model detected a significant effect ($P < 0.10$) for either the interaction term (BA:CI) or the farm presence variable (F), thus reflecting the likelihood of identifying a true effect of the sea farm on seabird densities under the given scenario. A significance level of 0.10 was used instead of the conventional 0.05 to increase sensitivity to potential impacts, which is considered appropriate in early warning contexts where detecting subtle but ecologically relevant changes is critical (Vanermen et al., 2015).

All statistical analyses were performed in R v.4.2.2 (R Core Team, 2024), making use of the following packages: dplyr and tidyr for data wrangling (Wickham et al., 2023), ggplot2 for data visualisation (Wickham, 2016), lmtest for regression diagnostics (Zeileis & Hothorn, 2002), and foreign for importing data from external statistical software formats (R Core Team, 2024). Spatial data were processed and visualised using sf (Pebesma, 2018), and lubridate was used to simplify the handling of date-time objects (Spinu et al., 2023).

2.3 SPECIES SELECTION AND DESCRIPTION

In the reference period between 2000 and 2023, a total of 59 bird species were observed in and around the project area. Most of these (n = 44) were recorded fewer than 100 times. For this study, we focus on the 15 most commonly observed species: red-throated diver (*Gavia stellata*), great crested grebe (*Podiceps cristatus*), northern gannet (*Morus bassanus*), great cormorant (*Phalacrocorax carbo*), common scoter (*Melanitta nigra*), little gull (*Hydrocoloeus minutus*),

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common gull (*Larus canus*), lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), black-legged kittiwake (*Rissa tridactyla*), Sandwich tern (*Thalasseus sandvicensis*), common tern (*Sterna hirundo*), common guillemot (*Uria aalge*), and razorbill (*Alca torda*). Except for great cormorant, all of these species are subject to conservation objectives under either the Natura 2000 framework or the Marine Strategy Framework Directive (MSFD). The following paragraphs provide a brief profile for each species, including information on habitat, feeding ecology, and vulnerability. Based on reference data collected by INBO between 2001 and 2023, distribution maps were made to visualise the spatial occurrence of each species in the Belgian part of the North Sea. In addition, we briefly revisit the results of the analyses presented in Vanermen et al. (2015), in which seabird densities within the project area (including a 1-nautical-mile buffer) were compared to those in the nearshore waters of the Belgian part of the North Sea (nearshore Zone 1 in Figure 1A). These comparisons provide context for the observed densities and help evaluate the relative importance of the project area for each species. Figure 2 shows the distribution of each species at the BNS during the season when it reaches its highest density.

Red-throated diver (*Gavia stellata*) The red-throated diver is mainly observed in the BNS from November to March. It is a migratory and wintering species for which SPA-B1 is considered as 'very important' (Degraer et al., 2010). This species is a pursuit diver, feeding largely on small fish which it captures by diving to depths of up to 15 m (maximum 25 m). Divers (*Gavia sp.)* are particularly sensitive to disturbance and often fly off at considerable distance when approached by vessels, which sometimes makes species-level identification difficult (Table A2; Degraer et al., 2010). However, since more than 97% of all identified divers in the INBO seabird database are red-throated divers (Degraer et al., 2010), all records of *Gavia sp.* are treated as red-throated divers in this study for simplicity. Distribution maps show a preference for nearshore waters, particularly between 5–15 km offshore (Figure 2). Important concentration areas include the Oostende and Middelkerke Banks and the area around the Vlakte van de Raan. Winter densities are slightly lower in the project area (0.42 birds/km²) than those in the broader nearshore zone (0.57 birds/km²) (Figure A1).

Great crested grebe (*Podiceps cristatus*) The great crested grebe is present in the BNS mainly during the winter and early spring months, with peak numbers partly depending on the severity of winter inland. It primarily inhabits the more turbid nearshore waters, particularly along the western coast from De Panne to Ostend but also around the Wenduinebank and the Vlakte van de Raan (Figure 2). Great crested grebes are pursuit divers that feed primarily on small fish, but they also consume crustaceans and other aquatic invertebrates. They are moderately sensitive to ship disturbance (Table A2; Degraer et al., 2010). The Special Protection Area SPA-B1, in which the project area is situated, is considered essential for this species as it hosts over 15% of the BNS population (Degraer et al., 2010). In winter, bird densities in the project area (2.79 birds/km²) exceed those in the broader nearshore zone (1.97 birds/km²), and a similar pattern is observed in spring (1.73 birds/km² vs. 0.17 birds/km²), underscoring the project area's relative importance (Figure A1).

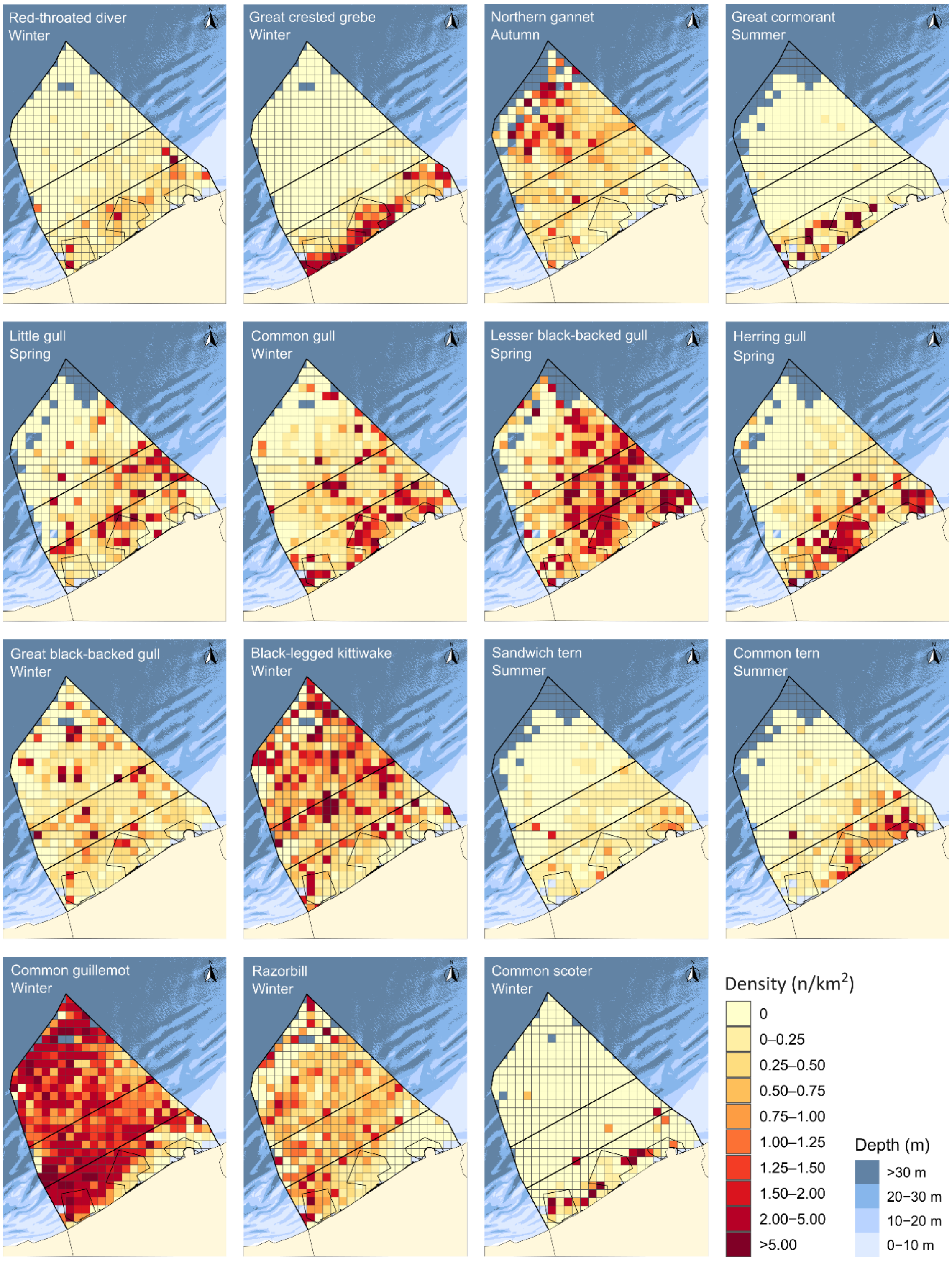

Figure 2. Spatial occurrence of the selected seabird species in the Belgian part of the North Sea, based on standardized ship-based surveys conducted by INBO between 2001 and 2023. For each species, only data from the season in which it reaches its highest density are shown. Mean density (individuals/km^2) is indicated in 3 km^2 raster cells; cells with no available data are left transparent. Black lines delineate the nearshore, midshore and offshore zones.

Northern gannet (*Morus bassanus*) Northern gannets are primarily observed in the BNS during autumn. It is a pelagic species feeding on larger fish caught by plunge-diving or foraging behind fishing vessels. It is generally less sensitive to vessel disturbance compared to divers or sea ducks (Table A2; Degraer et al., 2010). Densities in the project area are low (up to 0.33 birds/km²), as its core distribution lies further offshore (Figure 2). Although densities are a little bit higher in the project area compared to the entire nearshore zone (0.27 birds/km²), the project area is not of particular importance for this species (Vanermen et al., 2023).

Great cormorant (*Phalacrocorax carbo*) Great cormorants are present year-round in the project area and are more abundant there than in the nearshore zone overall. They reach their highest densities in summer, with densities over 0.95 birds/km² in the project area and 0.37 birds/km² in the entire nearshore zone (Figure A1). This is likely due to proximity to the breeding colony in the Hannecartbos (Oostduinkerke), with the project area serving as an important foraging ground. They primarily feed on larger fish, diving from the surface and pursuing prey underwater. They are moderately sensitive to disturbance but often forage close to shore and human activity (Table A2; Degraer et al., 2010)

Little gull (*Hydrocoloeus minutus*) Little gulls are mainly observed during spring and autumn migration in the BNS. They feed on small fish and aquatic invertebrates, foraging by surface-dipping and picking prey from the water. Little gulls show low sensitivity to disturbance from vessels during the day, but may be more sensitive to nocturnal disturbance (Table A2; Degraer et al., 2010). The species occurs primarily in nearshore waters but is much more frequently recorded in the wider coastal zone than in the project area (Figure 2). In spring, densities average only 0.18 birds/km² in the project area, compared to 1.28 birds/km² across the entire nearshore zone (Figure A1). Although SPA-B1 is designated as 'very important' for this species, the project area itself holds relatively few individuals and is not considered of particular importance.

Common gull (*Larus canus*) Common gulls occur in the BNS year-round, with higher densities typically observed in winter. They are generalist feeders, consuming fish, invertebrates, and discards from fishing vessels. Their spatial distribution is influenced by fishing activities, and they often aggregate near active fisheries. Densities in the project area in winter are lower than those in the broader coastal waters (2.49 birds/km² vs. 3.17 birds/km², respectively), with no indication of the area being of specific importance (Figure A1).

Lesser black-backed gull (*Larus fuscus*), herring gull (*Larus argentatus*) and great black-backed gull (*Larus marinus*) These large gull species are present year-round in the BNS and exhibit overlapping ecological traits. They are generalist feeders and opportunistic scavengers, feeding on fish, invertebrates, and discards from human activities, particularly fisheries. Their distribution is strongly influenced by the location and intensity of fishing activities, which can cause considerable spatial and temporal variation in observed densities. Consequently, they all show low sensitive to vessel disturbance (Table A2). During the breeding season, lesser black-backed gulls and herring gulls are linked to colonies in Zeebrugge and Oostende, where their numbers tend to peak (Figure 2). Great black-backed gulls typically occur in lower numbers and are more solitary, but also rely heavily on food sources derived from human activity. In all three cases, seasonal patterns in the project area closely reflect those in the broader nearshore zone, and the area is not of particular importance for any of these species (Figure A2; Vanermen et al., 2023).

Black-legged kittiwake (*Rissa tridactyla*) Black-legged kittiwakes are most common in the BNS during winter and autumn. In winter, they are significantly more abundant in the nearshore waters (0.87 birds/km^2) than in the project area (0.22 birds/km^2), whereas autumn densities are comparable between both zones (around 0.50 birds/km^2; Figure A2). As a more pelagic gull species, its core distribution lies further offshore (Figure 2). Kittiwakes feed on small pelagic fish, forage by surface seizing, and are moderately sensitive to vessel disturbance (Table A2; Degraer et al., 2010). Given its wider offshore range, the project area is not considered of particular importance.

Sandwich tern (*Thalasseus sandvicensis*) Sandwich terns exhibit similar seasonal patterns in both the project area and the nearshore zone, with highest densities recorded in spring and summer (around 0.4 birds/km^2; Figure A2). Sandwich terns predominantly prey on small fish and invertebrates, which are captured through shallow plunge-diving (to depths of up to 1.5 metres) or by surface picking (Degraer et al., 2010). Unlike common terns, they undertake longer foraging flights and are less dependent on colony proximity. The nearest breeding colonies are located in Platier d'Oye (France) and Zeebrugge (Belgium), although the number of breeding pairs at these sites can vary considerably between years. Sandwich terns show low sensitivity to vessel disturbance (Table A2). The Special Protection Area SPA-B1 is designated as 'very important' for this species due to concentrated occurrence during the breeding season (Degraer et al., 2010).

Common tern (*Sterna hirundo*) Common terns are present from spring through summer and show the highest densities in the breeding season. They feed on small fish, caught by plunge-diving, and their distribution is strongly influenced by the presence of breeding colonies in Zeebrugge and Oostende. Densities in the project area are markedly lower than in the nearshore zone as a whole (around 0.48 birds/km^2 vs. 1.16 birds/km^2), reflecting its peripheral role in the species' foraging range (Figure A2; Vanermen et al., 2023). Common terns are generally not sensitive to vessel disturbance (Table A2). The project area is not of particular importance for this species.

Common guillemot (*Uria aalge*) and razorbill (*Alca torda*) Common guillemot and razorbill are mainly present in the BNS during the winter months, with razorbill also showing higher densities in autumn. They feed on small fish, which they catch by pursuit diving. While their general distribution lies further offshore, relatively high winter densities have been recorded in the project area, with densities of common guillemot and razorbill reaching 4.69 and 1.44 birds/km^2, respectively, compared to 3.52 and 0.87 birds/km^2 in the broader nearshore zone (Figure A3). This may reflect a seasonal preference for the clearer nearshore waters along the western coast (Figure 2). Nevertheless, considering their widespread and primarily offshore distribution, the project area is not considered of particular importance for either species. Both species are considered moderately sensitive to disturbance from vessels (Table A2; Degraer et al., 2010).

Common scoter (*Melanitta nigra*) The common scoter is considered highly important in the context of SPA-B1 (Degraer et al., 2010). Ship-based surveys show high spring densities in the project area (nearly 3 birds/km^2) compared to around 2 birds/km^2 in the broader nearshore waters (Vanermen et al., 2023). However, this peak resulted from a single observation, so it may not reflect regular use of the site. Although they are typically present in winter and early spring, some larger groups have also been present in summer and early autumn in recent years. As scoters are highly sensitive to disturbance, aircraft surveys are preferred for monitoring (Table A2; Degraer et al., 2010). These have shown few records in or near the project area between

February and March (Figure 3). Recurrent concentrations are mostly found south and southwest of the area. Few common scoters were observed between the project area and Nieuwpoort, which is the most likely navigation route for maintenance vessels. Scoters feed mainly on benthic invertebrates, such as molluscs and crustaceans, which they dive for in shallow coastal waters.

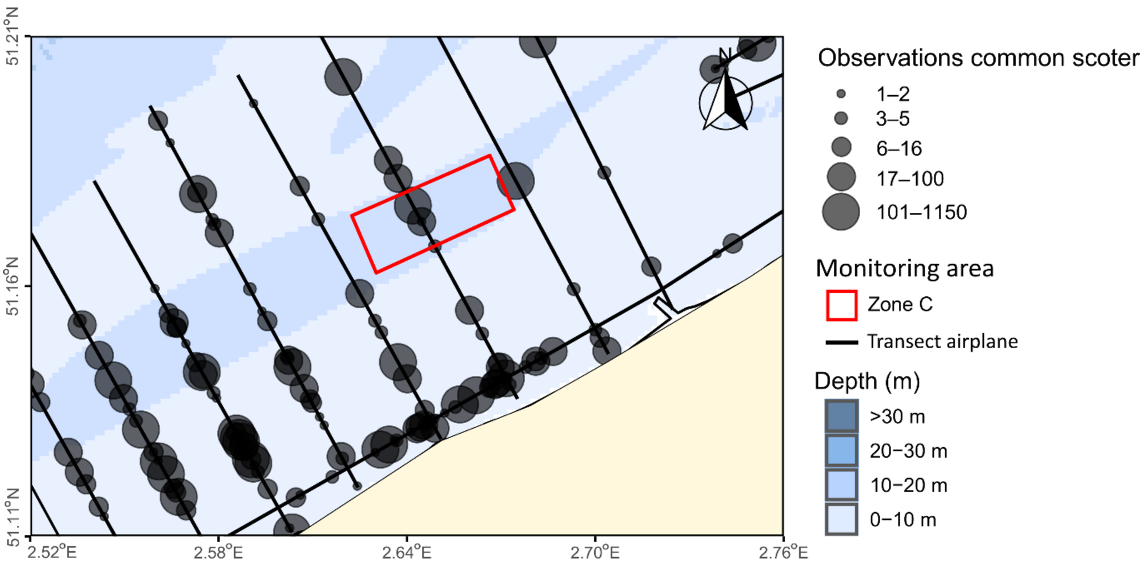

Figure 3. Spatial occurrence of common scoters around the project area in February and March based on yearly airplane counts between 2001 and 2023.

3 RESULTS

The modelled seasonal patterns for each species are shown in figure 4 and illustrate how the predicted seabird densities within the study area vary throughout the year. For 13 out of 15 species, model selection supported a seasonal structure best captured by a double sine function with periodicities of 12 and 6 months (seasonality type S3; see Table A1). For great cormorant, seasonal variation was best described by a single sine curve with a period of 12 months (seasonality type S2, see Table A1). Also northern gannet showed a different seasonal pattern, with the best model fit achieved by a double sine curve with periods of 12 and 4 months (seasonality type S4, see Table A1). For most species, the predicted seasonal patterns in the study area were consistent with those previously described for the broader BNS by Degraer et al. (2010). Some deviations were observed, however. For red-throated diver, the highest densities are indeed predicted in the winter months, but with peak densities in the study area in November, which is earlier than the January peak reported by Degraer et al. (2010). Great cormorant, which is not discussed in detail in Degraer et al. (2010), was predicted to be present year-round, with a marked peak in August after the breeding season. Lesser black-backed gull and herring gull are also not discussed in Degraer et al. (2010). The highest densities for lesser black-backed gull were predicted in spring and early autumn, with slightly lower densities in summer. For herring gull there was a clear peak in spring and a second lower peak in late summer to early autumn. Great black-backed gull showed highest predicted densities in autumn and early winter, in line with Degrear et al. (2010), but our models also suggested a second peak in spring.

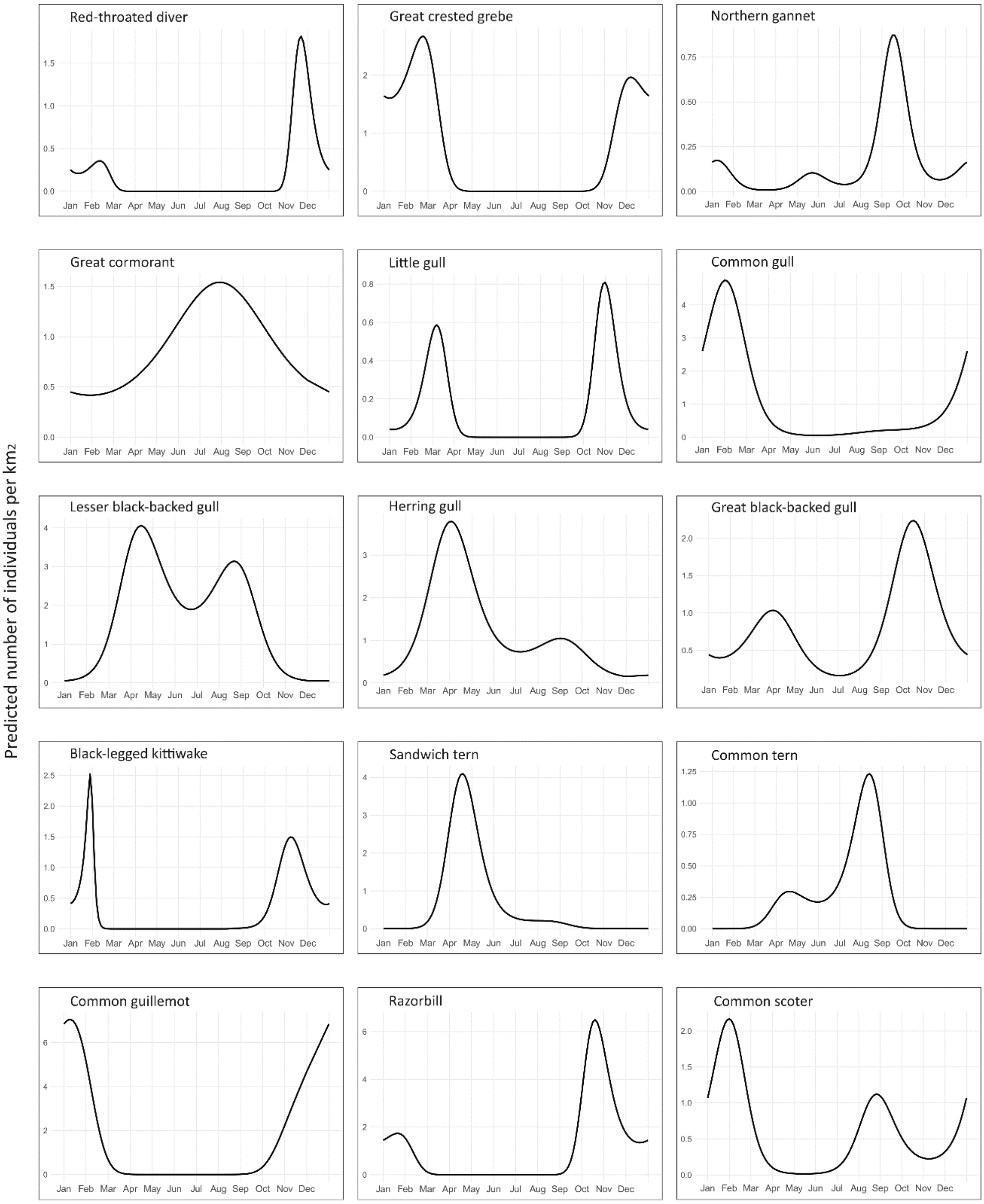

Figure 4. Seasonal patterns of the selected seabird species in the study area (impact and control zones combined), based on reference data collected between 2001 and 2023 (2006–2023 for great cormorant). Seasonality was modelled by fitting a linear combination of sine and cosine functions to capture cyclical fluctuations in species occurrence.

Common tern was predicted to reach highest densities between spring and late summer, but with a peak in August, contrasting with the May peak reported for the BPNS as a whole. This may reflect the fact that the project area lies outside the typical foraging range of nearby breeding colonies during the breeding season. Finally, for common scoter, peak densities were predicted in winter and early spring, consistent with Degraer et al. (2010), though an additional peak in September was observed in our model results. For each species, the four months with the highest predicted densities in the study area were selected to define the second monitoring scenario, which includes four annual surveys during these peak months.

Next, all species-specific models were used to simulate Before-After Control-Impact (BACI) datasets for each of the three monitoring scenarios: monthly surveys, seasonal surveys, and surveys during the four months with peak densities. For each scenario, simulated changes in species numbers of 75%, 50%, and 25% (both increases and decreases) were applied within the impact area. Power analyses were then performed by fitting impact models to these simulated datasets to assess the likelihood of statistically detecting a seabird response to the sea farm under each combination of monitoring strategy and effect size. The results of these analyses are presented in Figures 5 to 7.

In monitoring scenario 1, which assumes one survey per season (four surveys per year), none of the 15 studied species achieved the target power of 80% to detect a 50% change in numbers within the planned 16 post-impact surveys (i.e., after 4 years of monitoring). The highest statistical power for this scenario was observed for great cormorant, reaching 48%. For seven species, power ranged between 20% and 48%, while the remaining seven species showed a power of less than 20%. Even when extending the monitoring period with an additional 11 years, only great cormorant, lesser black-backed gull, and Sandwich tern were predicted to reach the target power of 80% to detect a 50% change in numbers. When considering a larger effect size — a 75% change in numbers — only great cormorant reached the 80% power threshold within the initially planned monitoring period of 16 post-impact surveys. Changes of only 25% in numbers would not be detectable within 15 years for any of the species. Model selection indicated significant zero-inflation for 8 of the 15 species, including northern gannet, little gull, herring gull, Sandwich tern, common tern, common guillemot, razorbill, and common scoter. This suggests that for these species, the number of zero counts (i.e. surveys with no individuals observed) was predicted to be higher than expected based on a standard negative binomial distribution, potentially reflecting their patchy distribution, generally low occurrence, or complete absence during certain seasons within the study area.

In monitoring scenario 2, which focuses monitoring efforts on the four months with the highest predicted species densities, the statistical power to detect changes generally improved compared to the fixed seasonal surveys of scenario 1. Under this design, great cormorant and Sandwich tern were the only species to reach the 80% power threshold for detecting a 50% change in numbers within the planned four years of post-impact monitoring. Extending the monitoring period to 15 years allowed this threshold to also be reached for lesser black-backed gull and razorbill. For a larger effect size of 75%, sufficient power within four years was achieved for great cormorant and Sandwich tern, and within 15 years for red-throated diver, great crested grebe, little gull, lesser black-backed gull, herring gull, great black-backed gull, common guillemot, and razorbill. In contrast, detecting more subtle changes of 25% remained difficult: only Sandwich tern reached 80% power within four years, and only great cormorant within 15 years of monitoring.

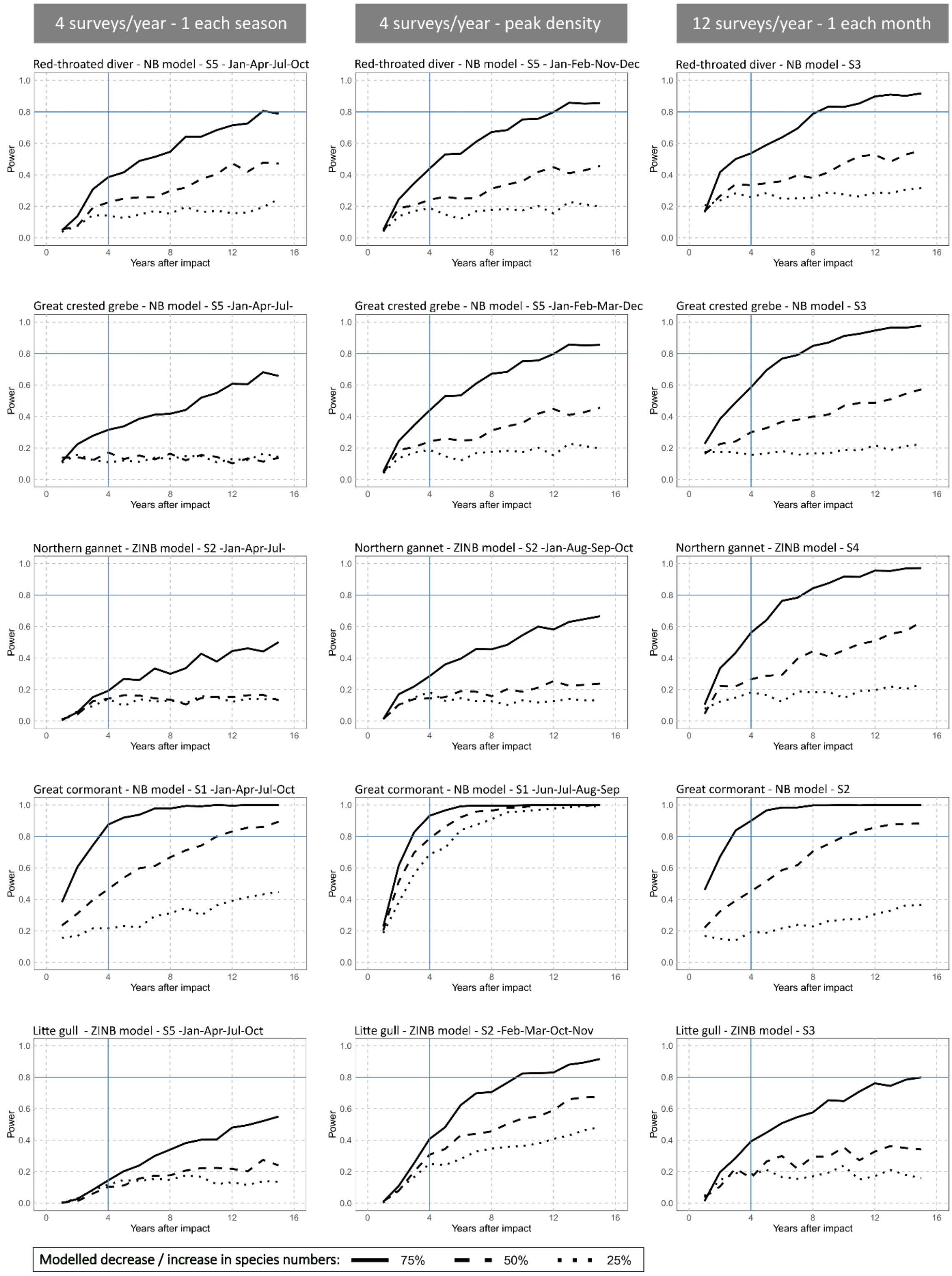

Figure 5. Power analysis results for red-throated diver, great crested grebe, northern gannet, great cormorant and little gull. For each species, three scenarios are shown: one survey each season (left), four surveys in the months with peak density (middle), and monthly surveys (right). Survey months, model type (NB: negative binomial; ZINB: zero-inflated negative binomial) and seasonality structure (S1-S5, see table A1 for details) are indicated above each graph. The horizontal blue line marks the targeted power of 80% and the vertical blue line indicates the planned four years of post-impact monitoring.

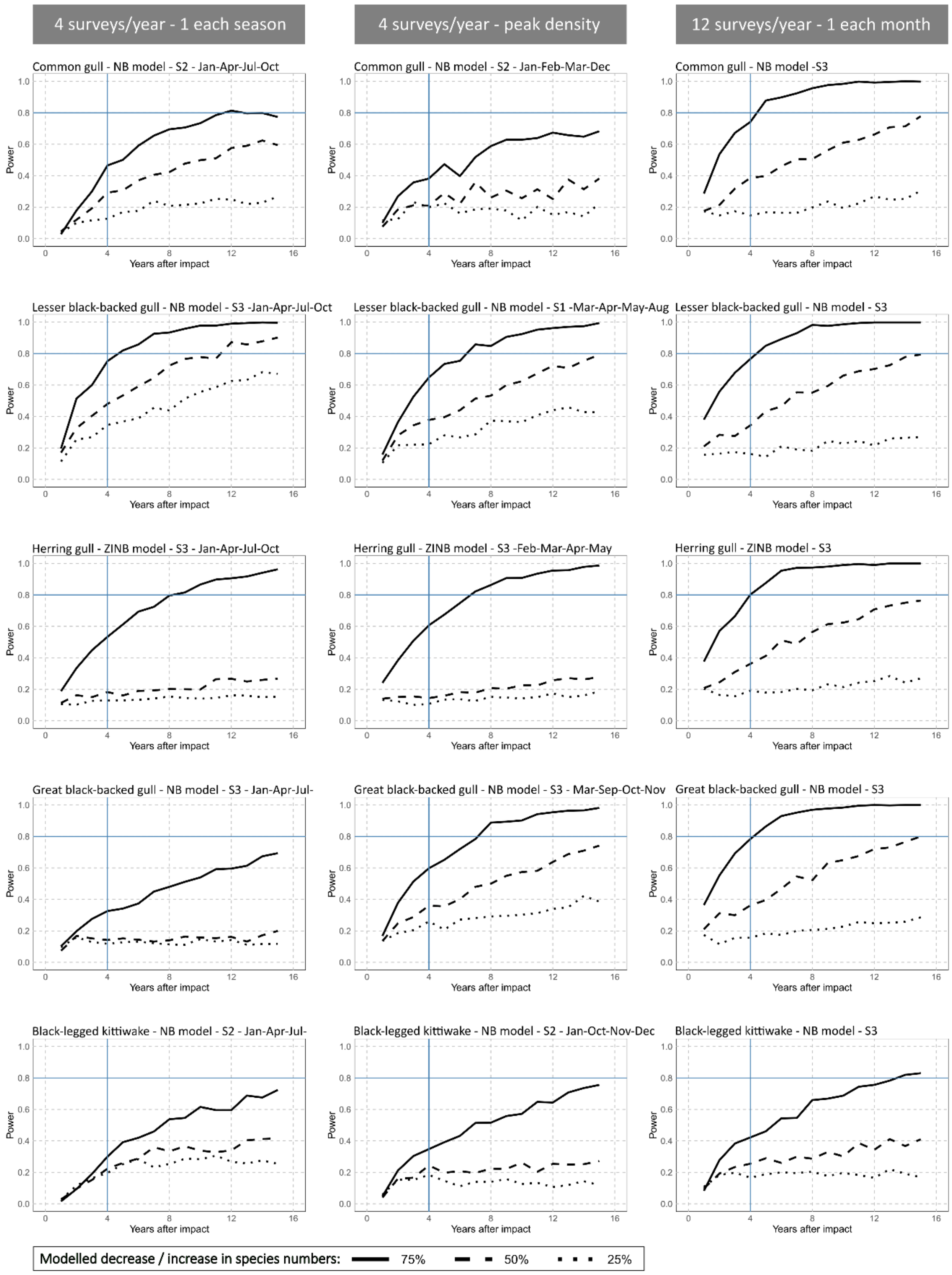

Figure 6. Results of the power analyses for common gull, lesser black-backed gull, herring gull, great black-backed gull and black-legged kittiwake. For each species, three scenarios are shown: one survey each season (left), four surveys in the months with peak density (middle), and monthly surveys (right). Survey months, model type (NB: negative binomial; ZINB: zero-inflated negative binomial) and seasonality structure (S1-S5, see table A1 for details) are indicated above each graph. The horizontal blue line marks the targeted power of 80% and the vertical blue line indicates the planned four years of post-impact monitoring.

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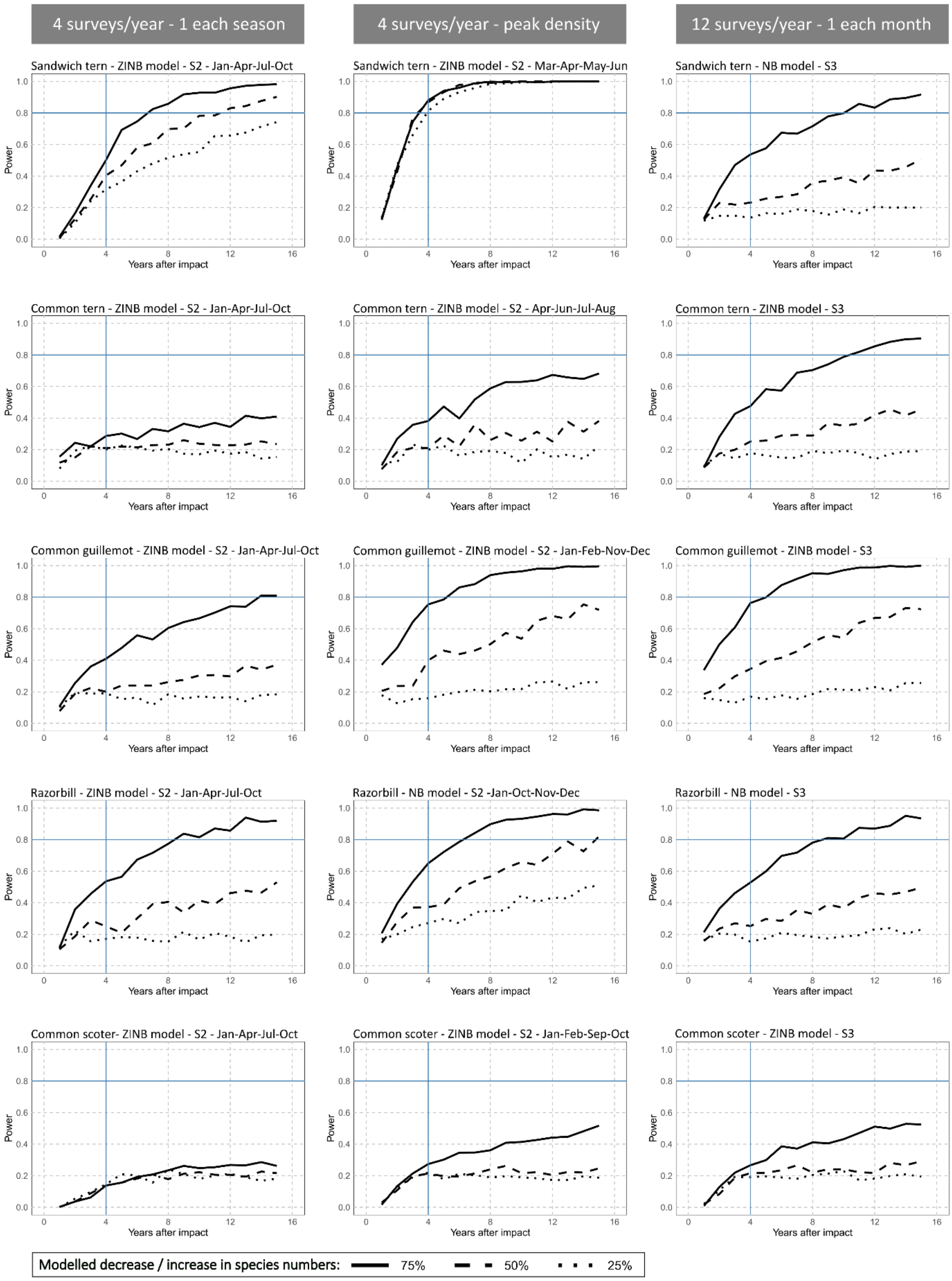

Figure 7. Results of the power analyses for Sandwich tern, common tern, common guillemot, razorbill and common scoter. For each species, three scenarios are shown: one survey each season (left), four surveys in the months with peak density (middle), and monthly surveys (right). Survey months, model type (NB: negative binomial; ZINB: zero-inflated negative binomial) and seasonality structure (S1-S5, see table A1 for details) are indicated above each graph. The horizontal blue line marks the targeted power of 80% and the vertical blue line indicates the planned four years of post-impact monitoring.

Scenario 3 explored the potential of a more intensive monitoring strategy, with monthly surveys yielding 48 observations over a 4-year post-impact period - three times more than in the standard design. This increased survey frequency generally improved statistical power across species, although the gains were not consistent. Surprisingly, for great cormorant and Sandwich tern, the model performance was poorer than in scenario 2, and none of the species achieved the 80% power threshold to detect a 50% change in numbers within the initial 4-year monitoring period. Over a longer timeframe of 15 years, this threshold was only reached for great cormorant, common gull, lesser black-backed gull, and great black-backed gull. For a more pronounced effect size of 75%, sufficient power was achieved within 4 years for great cormorant and herring gull, and within 15 years for all species except common scoter. Detecting subtle changes of 25% remained challenging, with no species reaching more than 40% power even after 15 years of monthly monitoring.

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4 DISCUSSION

4.1 POWER ANALYSIS AND EVALUATION OF THE PLANNED MONITORING PROGRAM

To evaluate the potential impact of the Westdiep sea farm on seabirds, a BACI monitoring program was designed consisting of four ship-based surveys per year over a five-year period. The first four surveys had as goal to establish a baseline for seabird densities within both an impact area and a control area prior to the development of the farm. The remaining 16 surveys are planned to be distributed across phase II (one year) and phase III (three years) of the project implementation, meaning they can all be considered post-impact surveys. An earlier analysis of the four reference surveys by Vanermen et al. (2023) concluded that they yielded few seabird observations and many zero counts, resulting in a dataset of limited usefulness for significance testing. This was attributed to the low number of surveys, the relatively low seabird densities recorded during those surveys, and the small surface area of the study zones. To evaluate whether the planned monitoring strategy consisting of 16 post-impact surveys will be sufficient to detect an effect of the sea farm on seabirds with adequate statistical power, a power analysis was conducted in this study.

Our analysis revealed that, overall, the reference dataset collected within the study area - despite being supplemented with addition data gathered by INBO between 2001 and 2023 - was of limited statistical quality. This was primarily due to high levels of overdispersion and/or zero inflation in the data, both of which reduce the ability to detect significant changes in seabird densities. When using the standard seasonal monitoring approach, none of the 15 studied species reached the desired statistical power of 80% to detect a 50% change in numbers within the planned 16 post-impact surveys. Targeted surveys conducted during the months when each species typically reaches its peak abundance did improve statistical power slightly, allowing great cormorant and Sandwich tern to reach the 80% threshold. However, for most species, power remained insufficient to detect changes in species numbers with sufficient statistical power. The improvement observed during peak-month surveys likely reflects the more consistent and concentrated presence of seabirds during those months, leading to fewer zero counts and reduced variability. Surprisingly, increasing survey frequency to monthly intervals—tripling the survey effort—did not yield proportional improvements in power. Although the statistical power to detect changes generally showed small increases, no species reached the 80% threshold for detecting a 50% change after four years of monthly monitoring. While our modelling approach accounted for both seasonality and zero inflation, these adjustments were not sufficient to overcome the underlying challenges of highly variable and zero-dominated count data. This suggests that, for many species, increasing survey frequency outside their main period of occurrence adds relatively little statistical information and may instead introduce additional noise, limiting overall gains in power.

The low statistical power to detect changes in seabird numbers in our study is in line with other studies. Vanermen et al. (2015) applied a BACI design and simulation-based power analysis to assess seabird responses to the Thorntonbank offshore wind farm in Belgian waters. Due to high overdispersion and zero inflation, statistical power to detect a 50% decline in numbers was generally low, with only northern gannet and common guillemot reaching 90% power within ten years of post-impact monitoring. Maclean et al. (2013) also reported low statistical power when analysing long-term seabird monitoring data collected through aerial surveys in the UK. Even with 12 surveys per year, the power to detect a 50% change in bird numbers after four years remained below 80% for all studied taxa, despite applying a more relaxed significance level of 0.20. The low power was mainly attributed to high natural variability in seabird numbers across locations, with the authors suggesting that incorporating explanatory variables such as hydrodynamic conditions may help improve power. Similarly, in our study, high variability in count data limited statistical power, despite methodological efforts to reduce it. Only surveys that covered both the impact and control areas on the same day were included, in order to minimize day-to-day variation in seabird abundance and observation conditions. Seasonal trends were accounted for using cyclic sine terms, and count data were aggregated into daily totals to avoid autocorrelation between consecutive transect segments. Still, unmodelled sources of variance - such as interannual fluctuations, short-term variation (e.g. day-to-day), tidal influence, or weather-related detectability - likely contributed to remaining noise in the data (Schwemmer et al., 2009; Cox et al., 2013).

Furthermore, the very small spatial extent of our study area likely played a key role, offering certain advantages but also clear limitations in terms of statistical power. While Vanermen et al. (2015) already described their study area - comprising impact and control zones of approximately 140 km² each - as small relative to other research programmes in Danish, Dutch, and UK waters, the impact and control zones in our study were each only about 10 km². This makes our monitoring set-up extremely limited in spatial scope. On the one hand, this allowed for a closely matched control zone in terms of environmental conditions and expected seabird densities, likely helping to minimize between-area variability. On the other hand, such small areas are inherently more susceptible to zero counts, particularly for species with patchy distributions or low local densities, as is often the case for seabirds (Maclean et al., 2013). This increases the likelihood of zero-inflation in the data, which in turn reduces statistical power. Additionally, patterns observed at such a small scale may not reflect broader regional trends, limiting both the robustness of the BACI design and the generalizability of the results. In larger or more complex systems, broader monitoring approaches combined with spatially explicit modelling may provide more reliable alternatives (Mackenzie et al., 2013).

As mentioned before, statistical power is not only influenced by data variability but also by sample size, the chosen significance threshold, and the magnitude of the effect being tested. Increasing sample size - either by surveying more frequently or covering larger areas - can improve power, although this is not always feasible. In our case, expanding the survey area is not possible due to the small size and the restricted access to the operational zone, but increasing the number of surveys may be an option. However, additional surveys are only likely to be effective if they are strategically timed to coincide with periods when target species reach relatively high densities. As shown in our results, monitoring during periods of low presence may lead to more zero counts and increased data variability, which can counteract the potential gains in power. Lowering the significance threshold is another potential route to increase power; however, we already applied a relatively relaxed threshold of 0.10, chosen to increase sensitivity to potential impacts. While this value is still considered appropriate in early warning contexts, further lowering the threshold would compromise the reliability of statistical conclusions. Lastly, as demonstrated by our results, the magnitude of the simulated effects has a strong influence on statistical power. While we tested changes in species densities ranging from 25% to 75%, larger effect sizes could have been used to improve detectability. However, the chosen range already reflects changes that can have significant ecological consequences for seabird populations, particularly if such changes occur over large spatial scales or persist over extended periods.

4.2 HOW TO PROCEED? RECOMMENDATIONS FOR FUTURE MONITORING

The results of our power analysis indicate that statistically significant effects of the Westdiep sea farm on seabirds are unlikely to be detected under the current monitoring setup. However, it is important to recognize that a lack of statistical significance does not imply the absence of any effects. Even non-significant findings can yield valuable ecological insights and should not be disregarded (Nakagawa & Cuthill, 2007). Non-significant trends may, for instance, still align or contrast with patterns observed in other regions or similar projects, and contribute to the growing body of knowledge on the potential impacts of offshore aquaculture developments. This is especially important as similar sea farm projects may be developed in the future, making it possible to combine data across sites and years to draw more robust conclusions at larger spatial scales. One possible solution to improve statistical power would be to significantly increase the monitoring effort; however, this would result in a substantial additional cost. Therefore, we aim to explore alternative approaches that optimize data collection and analysis within the available resources.

Since the standardized ESAS monitoring protocol to count seabirds requires observers to be positioned at approximately 10 meters above sea level, relatively large research vessels are typically used for seabird surveys. However, these vessels cannot safely navigate within the operational sections of project Zone C due to manoeuvrability constraints. Although some parts of Zone C will remain accessible during the continued development of the sea farm, access will eventually be limited to the safety perimeter surrounding the installation. Consequently, it will no longer be feasible to follow the original monitoring transect or to strictly adhere to the ESAS protocol, which assumes that birds are counted within a 300-meter zone around the vessel. To maximize coverage of the impact area under these new constraints, we recommend adapting the monitoring protocol to include visual counts at greater distances when conditions allow. Observers can estimate distances using the ship's position in combination with the sea farm's buoys as reference points, and photograph the area during each survey to support species detection and identification. Despite these adjustments, efforts should be made to maintain the core principles of the standardized ESAS methodology, including the use of both transect and snapshot counts. Surveys in the control area, which remains fully accessible, can continue to follow the planned transect and original ESAS protocol without modification.

Given the expected reduction in species detectability due to increased observation distances, seabirds will be grouped into morphotypes when necessary. These morphotype groups include large gulls (lesser black-backed gull, herring gull, and great black-backed gull), small gulls (common gull, black-legged kittiwake, and little gull), auks (common guillemot and razorbill), and terns (common tern and Sandwich tern). Species more easily identified at a distance (e.g. divers, northern gannet, great crested grebe, great cormorant, and common scoter) will be recorded separately. Grouping species has the additional benefit of reducing the proportion of zero counts, which may improve statistical power.

As such, we recommend proceeding with the 16 planned post-impact surveys, maintaining the original transect and monitoring method for as long as access allows. As the project area becomes increasingly operational, the protocol will be adapted to allow surveys from the surrounding safety zone. In addition, INBO's long-term seabird monitoring transects frequently pass near the project area. Although they only briefly pass the site and at a greater distance —

preventing standardized seabird counts within the farm—we propose collecting photographs during these transects to document the presence and approximate abundance of seabird morphotypes. While these observations cannot replace the post-impact surveys, they may serve as valuable supplementary data. As these long-terms surveys occur approximately once a month, they can provide additional insights into temporal dynamics and support the interpretation of dedicated survey data.

Regarding the timing of the monitoring surveys, our results indicate that distributing four surveys equally across the seasons is suboptimal. Instead, survey timing should be adjusted to coincide with the months in which key species reach peak densities within the study area. The most relevant species in the project area, which are red-throated diver, great crested grebe, and common scoter, exhibit peak densities during winter and early spring. Therefore, we recommend scheduling the four annual surveys between January and April. For Sandwich tern, another important species, peak densities occur slightly later, from March to June. Although only two of the planned surveys (March and April) will fall within this peak period, they can be complemented with incidental data collected during INBO’s long-term monitoring campaigns that pass the area in May and June. Given that power analyses for Sandwich tern showed high statistical power when surveys are conducted during these months, this combined approach should still provide valuable insights into potential effects of the sea farm on this species' abundance.

Given the novelty of this type of offshore infrastructure in the Belgian part of the North Sea, anecdotal observations can also offer valuable ecological insights. For example, crew working at the Westdiep sea farm have reported observing Sandwich terns resting on buoys and engaging in courtship behaviour in spring, between mid-March and mid-April. Such observations are particularly noteworthy, as little is currently known about the behaviour and activity of this species in and around offshore installations (but see Stienen et al. 2024). In addition, reports from crew members of the sea farm, especially during periods of high operational activity such as harvesting, may help identify potential correlations between seabird presence and specific farm-related activities. These observations could be used as exploratory variables in future modelling efforts. To support this, a seabird observation reporting form will be developed and distributed to the crew.

Finally, if feasible within the available budget, ship-based surveys could potentially be complemented by drone flights. Drones offer the advantage of providing full visual coverage of the sea farm and, if operated from land, may represent a cost- and time-efficient alternative survey method. However, current airspace restrictions near the Raversijde military base only allow drone flights after 5 p.m. and during weekends. Unless a solution to these restrictions is found, the use of drones remains impractical. In addition, a suitable drone would be required - one capable of operating over long distances and in offshore conditions. INBO does not currently have access to such equipment, although the Colruyt Group reportedly owns an appropriate drone, and the Federal Public Service Mobility and Transport may also have drones available for offshore use. The use of stationary PTZ cameras with remote control was also considered as a potential monitoring alternative. However, this option is not feasible because there are no stable structures within the sea farm on which cameras could be mounted. As a result, excessive movement would severely compromise image quality. Moreover, since no elements of the sea farm extend significantly above the water surface, it would be impossible to obtain a sufficiently comprehensive overview of the area.

The revised monitoring design, consisting of targeted seasonal surveys complemented by observations from other long-term survey and sightings reported by crew members, will provide a unique dataset on seabird presence and dynamics in relation to the Westdiep sea farm. Beyond assessing potential ecological impacts of the current project, these data will also be crucial to identify current knowledge gaps, both on species-specific responses to offshore aquaculture and on monitoring design. By testing and refining methodological approaches under practical constraints, the monitoring program contributes directly to the optimization of survey designs for future projects. In this way, the current project serves as a pilot case whose outcomes extend well beyond the scope of the current project, establishing a scientific framework and supporting evidence-based policies for future aquaculture developments.

5 CONCLUSION

The power analysis demonstrated that the 16 planned post-impact surveys will likely be insufficient to detect the impact of the Westdiep sea farm on seabirds with high statistical confidence. This limited power is primarily due to the small study area and the inherently dynamic and sparse distribution of seabirds, leading to overdispersed and zero-inflated data.

Nevertheless, the insights resulting from the surveys - whether statistically significant or not - remain ecologically important. Additionally, by refining the survey design and integrating supplementary data, we aim to further increase the robustness of the findings and provide a stronger foundation for future evaluations.

As our power analysis shows increased statistical power when monitoring is aligned with periods of peak seabird density, we propose conducting the four annual surveys during the months of January, February, March, and April. These months coincide with the highest expected densities of key species in the area, including red-throated diver, great crested grebe, and common scoter.

The data collected through these surveys will be further complemented in two ways: first, by additional observations based on photographs taken during INBO's ongoing long-term seabird monitoring campaigns, and second, by incorporating anecdotal sightings reported by the crew of the sea farm via dedicated seabird observation forms. In addition, as the operational area of the sea farm is expected to expand in the coming years, the monitoring transect and methodology will be adjusted accordingly to ensure continued data collection.

Taken together, these efforts will not only improve our understanding of seabird dynamics in and around the Westdiep sea farm and help assess the potential impacts of this specific project, but will also generate knowledge that is critical for optimizing monitoring designs and informing the development of policy frameworks for future offshore aquaculture projects.

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Appendix

Figure A1. Seasonal densities of red-throated diver, great crested grebe, northern gannet, great cormorant, little gull and common gull in the nearshore waters of the BNS (Zone 1) and in the project area (Zone C, including a buffer of 1 nautical mile), based on ship-based surveys conducted between 2001 and 2023.

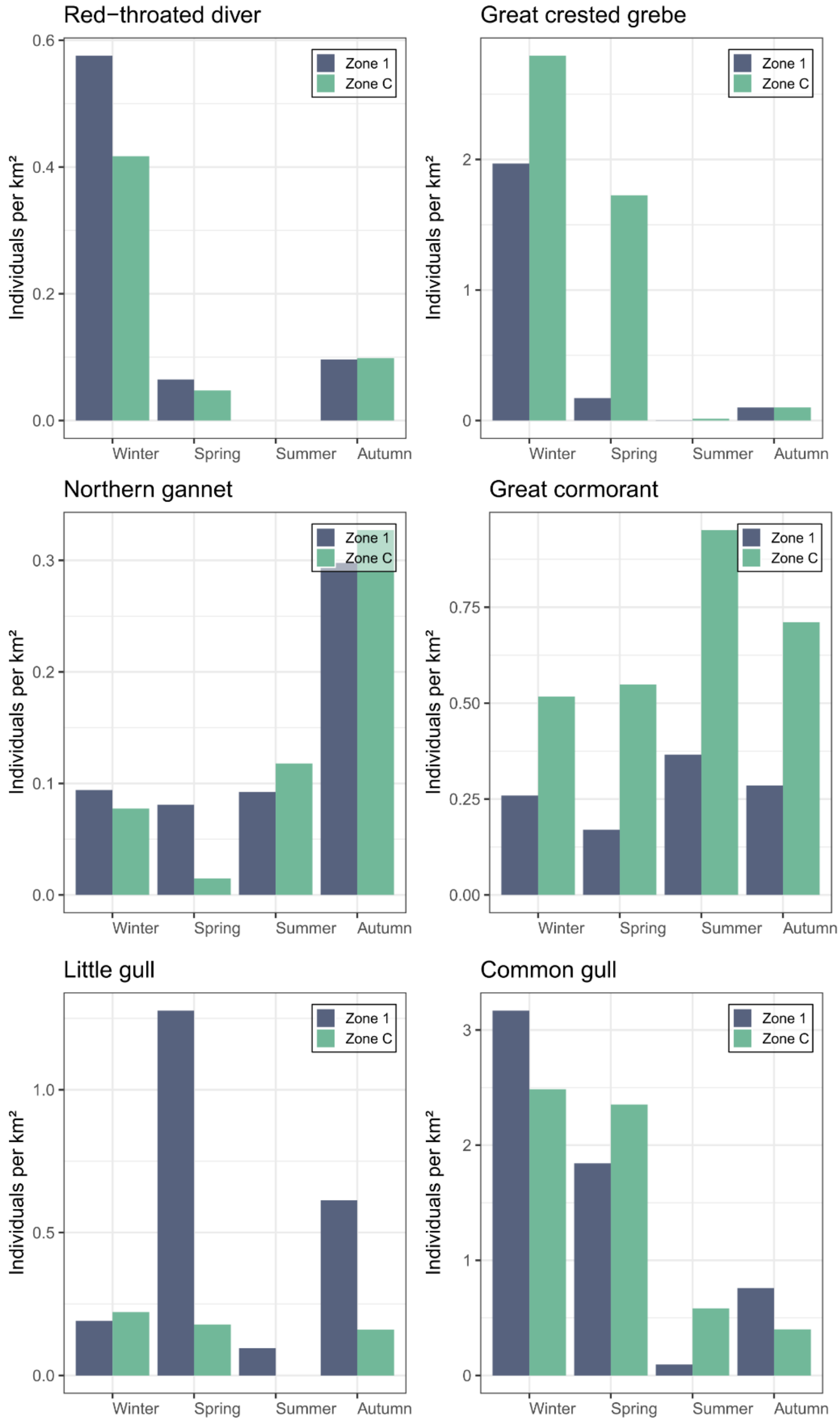

Figure A2. Seasonal densities of lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern and common tern in the nearshore waters of the BNS (Zone 1) and in the project area (Zone C, including a buffer of 1 nautical mile), based on ship-based surveys conducted between 2001 and 2023.

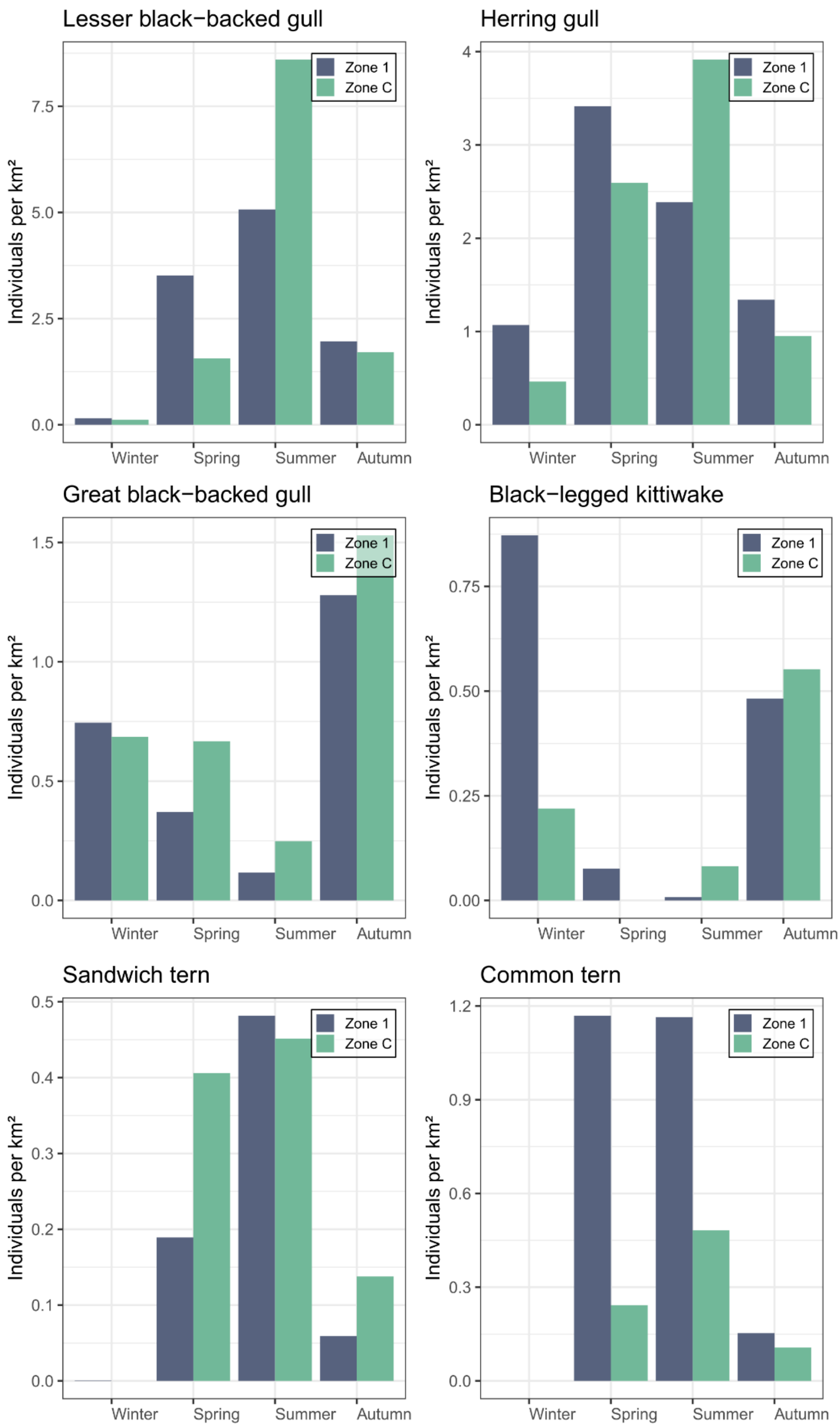

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Figure A3. Seasonal densities of common guillemot and razorbill in the nearshore waters of the BNS (Zone 1) and in the project area (Zone C, including a buffer of 1 nautical mile), based on ship-based surveys conducted between 2001 and 2023.

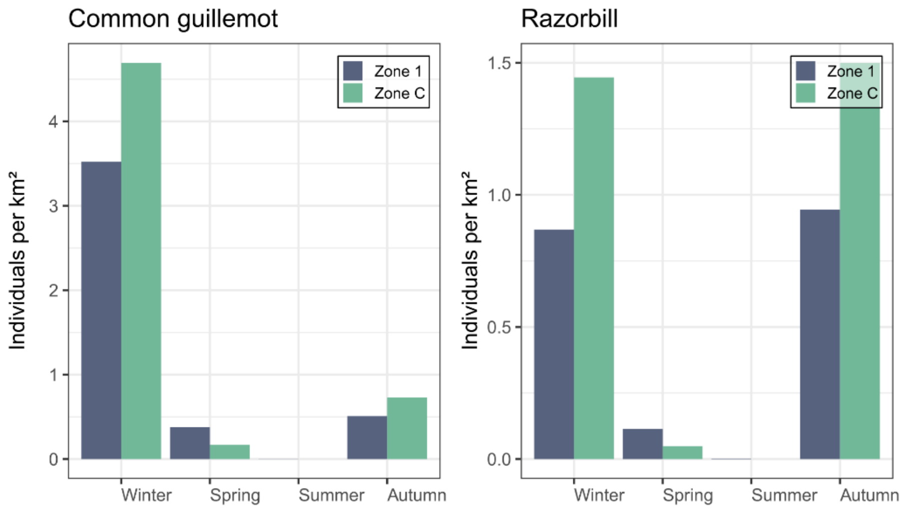

Table A1. Overview of the possible model formulas with different ways of modelling seasonality in seabird occurrences. Y = the total number of birds observed per survey in either the control or impact area. m = month, either used as a continuous variable in a (sum of) sine terms or included as a categorical fixed effect. The seasonality terms are highlighted in bold.

	4 surveys per year	12 surveys per year
S1 - No seasonality	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + a_2 \cdot CI$	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + a_2 \cdot CI$
S2 – Single seasonality	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot sin(2\pi m/4) + a_3 \cdot cos(2\pi m/4)} + a_4 \cdot CI$	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot sin(2\pi m/12) + a_3 \cdot cos(2\pi m/12)} + a_4 \cdot CI$
S3 – Double seasonality	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot sin(2\pi m/4) + a_3 \cdot cos(2\pi m/4) + a_4 \cdot sin(2\pi m/2) + a_5 \cdot cos(2\pi m/2)} + a_6 \cdot CI$	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot sin(2\pi m/12) + a_3 \cdot cos(2\pi m/12) + a_4 \cdot sin(2\pi m/6) + a_5 \cdot cos(2\pi m/6)} + a_6 \cdot CI$
S4 – Double seasonality (2)	/	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot sin(2\pi m/12) + a_3 \cdot cos(2\pi m/12) + a_4 \cdot sin(2\pi m/4) + a_5 \cdot cos(2\pi m/4)} + a_6 \cdot CI$
S5 – Month as fixed factor	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot m}$	$\log(Y) = \text{offset}(\log(km^2)) + a_1 + \mathbf{a_2 \cdot m}$

Table A2. Disturbance Vulnerability Index (DVI) scores for the study species (except for common guillemot and common gull), reflecting sensitivity to ship traffic based on species' shyness, escape costs and cost at the population level as reported by Fliessback et al. (2019).

Species	**Disturbance Vulnerability Index (DVI)**
Red-throated diver	77.8
Razorbill	51.3
Common scoter	43.3
Great cormorant	24.4
Great crested grebe	21.7
Northern gannet	15.6
Little gull	12.0
Great black-backed gull	11.0
Black-legged kittiwake	9.3
Sandwich tern	6.7
Lesser black-backed gull	6.7
Herring gull	6.2
Common tern	3.9