Population level effects of displacement of marine birds due to offshore wind energy developments, KEC 5

Author(s): Soudijn, F.H., Hin, V., Melis, E., Chen, C., van Donk, S., Benden, D., Poot, M.J.M.

Wageningen University &
Research report: C094/24 A

Population level effects of displacement of marine birds due to offshore wind energy developments, KEC 5

Author(s): Soudijn, F.H., Hin, V., Melis, E., Chen, C., van Donk, S., Benden, D., Poot, M.J.M.

Wageningen Marine Research

Wageningen Marine Research
IJmuiden, februari 2025

Wageningen Marine Research rapport C094/24 A

Keywords: habitat loss, seabirds, KEC, population effects, ALI, offshore wind farms

Client		Rijkswaterstraat Water, Verkeer & Leefomgeving
T.a.v. Martine Graafland, Meik Verdonk
Griffioenlaan 2
3626 LA Utrecht

Wageningen Marine Research is ISO 9001:2015 certified.

Photo cover: Steve Geelhoed

Wageningen Marine Research, an institute within the legal entity Stichting Wageningen Research (a foundation under Dutch private law) represented by Drs.ir. M.T. van Manen, Director Operations

KvK nr. 09098104,
WMR BTW nr. NL 8113.83.696.B16.
Code BIC/SWIFT address: RABONL2U
IBAN code: NL 73 RABO 0373599285

A_4_3_2 V33 (2023)

Contents

Publiekssamenvatting

De Nederlandse overheid heeft tot en met 2031 nieuwe gebieden op zee aangewezen voor het ontwikkelen van windenergie. In deze studie werd een inschatting gemaakt van de effecten van habitatverlies op zeevogels door de potentiële nieuwe windparken binnen deze gebieden. De effecten zijn bepaald voor de vogelsoorten: jan-van-gent, grote stern, zeekoet en alk. Deze soorten zijn bekende 'vermijders' van gebieden op zee waar windturbines staan. Op de een of andere manier worden ze afgeschrokken door de aanwezigheid van de turbines, de geluiden die gepaard gaan met operationele windturbines, of de activiteit van onderhoudsschepen.

De inschattingen van de effecten van habitatverlies zijn gedaan voor vier verschillende scenario's.

- Een scenario omvat de al bestaande windparken in Nederland ('Basic scenario').
- Een tweede scenario omvat de vergunde windparken in Nederland tot en met 2024 ('Basic plus scenario').
- Een derde scenario omvat alle geplande windparken tot en met 2031 in Nederland ('Total scenario').
- Het laatste scenario omvat alle geplande windparken tot en met 2031 in Nederland en de omliggende delen van de zuidelijke Noordzee, internationaal ('International scenario').

De inschatting van de slachtofferaantallen van habitatverlies is gebaseerd op de overlap tussen vogeldichtheidskaarten en de windenergie (zoek-)gebieden. Een deel van de vogels die normaal in een gebied met windturbines zouden voorkomen wordt verwacht het gebied te gaan vermijden, en van dit deel, wordt verwacht dat, op termijn, weer een deel sterft door deze verdrijving. Sterfte kan bijvoorbeeld optreden omdat er niet langer toegang is tot belangrijke foerageergebieden, of doordat lokaal verhoogde dichtheden ontstaan waarbij door een verhoogde concurrentie niet voor alle vogels voldoende voedsel te vinden is. Hoeveel van de vogels sterven door hun aanpassing aan de aanwezige windturbines, is onbekend. We rekenden daarom voor verschillende sterftekansen voor de verdreven individuen uit wat de verdrijvingssterfte zou zijn per scenario.

De effecten van het geschatte aantal slachtoffers op de populatie ontwikkeling zijn berekend met populatiemodellen, die gebaseerd zijn op de meest recente veldgegevens van reproductie en overleving van de betreffende soorten zeevogels. Deze effecten werden vervolgens, voor de Nederlandse scenario's, vergeleken met vooraf vastgestelde acceptabele niveaus van impact (ALIs). Het ministerie van LVVN stelt vast tot welke mate de vogelpopulatie ontwikkeling in de toekomst mag worden geremd door de ontwikkeling van windenergie op zee. In dit rapport is gewerkt met normen zoals het ministerie van LVVN deze medio 2024 heeft aangeleverd.

Ten opzichte van KEC 4, werden enkele kleine aanpassingen gedaan in de gebruikte parameter waardes. Zo werden er nieuwe waardes gekozen voor de populatieparameters en de vermijdingsparameters, gebaseerd op recent gepubliceerde literatuur. Daarnaast werden er nu per soort afstanden van de windturbines gebruikt waarover vogels mogelijk vermijding vertonen en werd de verdrijvingssterfte door windturbines per levensstadium anders gewogen. In de methodiek zelf werden ook aanpassingen gedaan. Er werd gebruik gemaakt van nieuwe dichtheidskaarten, en een zogenaamde 'displacement matrix' voor de slachtofferberekening in plaats van de 'Bradbury methode', die werd toegepast in KEC 4. Ook werden de uitkomsten van de modellen getoetst met de in begin 2024 aangepaste ALI methodiek. Het ministerie van LVVN heeft gebaseerd op deze nieuwe methodiek medio 2024 normen voor de ALIs aangeleverd. Tenslotte, werd een nieuwe methode voor de onzekerheidsanalyse ontwikkeld die onzekerheden in alle modelparameters en de dichtheidskaarten meeneemt, om de onzekerheid in de uitkomsten goed mee te kunnen nemen in de effectvoorspellingen.

Naast bovengenoemde verbeteringen is gewerkt aan een nieuwe methode voor de slachtofferberekeningen, gebaseerd op voedselinname en energieverbruik op het niveau van individuele vogels. Deze methode kon echter nog niet worden opgenomen in de effect-inschatting omdat deze nog niet voldoende ontwikkeld was. De komende jaren zal hier verder aan gewerkt worden.

De geschatte populatieontwikkeling leek nauwelijks te worden beïnvloed door windmolenparken voor de grote stern en de alk. Geen van de ALIs werden overschreden voor deze soorten. Voor de jan-van-gent en de zeekoet werd een langzamere populatiegroei geschat met windturbines dan zonder windturbines. Vanwege de grote onzekerheden over sterfte door verdrijving rekenden we zowel voor 1, 2, 5 en 10% sterfte van verdreven individuen uit wat de totale sterfte zou zijn per scenario. Voor het scenario met alle geplande windparken tot en met 2031 in Nederland, en een sterfte van verdreven individuen van 10%, werd de gemiddelde populatieomvang na 40 jaar van de jan-van-gent 4,9% lager geschat dan zonder windturbines. Voor de zeekoet werd deze populatieomvang na 40 jaar zelfs 31% lager geschat bij de aanname van 10% sterfte onder de verdreven vogels. Voor de jan-van-gent werd een ALI overschrijding gevonden voor het scenario met al vergunde parken in 2024 in Nederland en voor het scenario met alle geplande windparken tot en met 2031 in Nederland bij een sterfte van verdreven individuen van 10%, maar niet onder de aanname dat de sterfte lager is (5%, 2% of 1%). Voor de zeekoet werden ALI overschrijdingen gevonden voor alle Nederlandse scenario's bij een sterfte van verdreven individuen van 10%. De ALI werd ook overschreden voor het scenario met alle geplande windparken tot en met 2031 en voor het scenario met de al toegestane parken in 2024 in Nederland bij een sterfte van verdreven individuen van 5%, maar niet onder de aanname dat de sterfte lager is (2% of 1%).

De ALI werd niet getest voor het internationale scenario, maar in alle gevallen zijn de geschatte effecten op de populatie-ontwikkeling sterker voor de internationale scenario's dan voor de nationale. Voor het internationale scenario werd gemiddeld een hoger percentage overlap gevonden tussen aantallen vogels en de windparken dan voor de nationale scenario's. Waarschijnlijk omdat voor zeekoet, alk en jan-van-gent de dichtheden vogels gemiddeld het hoogst zijn in internationale wateren door de nabije ligging van de grote broedkolonies langs de UK oostkust.

Het sterftepercentage dat werd aangenomen had veel effect op de uiteindelijke uitkomsten. Het is zeer lastig een realistisch sterftepercentage aan te geven voor de verdreven individuen. Er zijn op zee duidelijke effecten waargenomen van windturbines op de ruimtelijke verdelingen van zeevogels. Maar om inschattingen te kunnen doen van sterfte door dit soort aanpassingen in ruimtegebruik, zouden individuen voor langere tijd gevolgd moeten worden (over meerdere jaren) en zouden regelmatig conditie metingen gedaan moeten worden. Daarnaast zouden gegevens over de voedselbeschikbaarheid nodig zijn op gedetailleerde ruimtelijke schaal en de conditie- en gedragsgegevens en de aanwezigheid van windturbines moeten worden vergeleken met metingen van broedsucces en sterfte, die worden gedaan in de broedkolonies. Deze zijn voor de zeekoet, de alk en de jan-van-gent langs de kust van het VK, wat de opzet van dit soort werk niet vergemakkelijkt. We hopen in de nabije toekomst meer inzicht te verkrijgen door middel van de inzet van gedragsmodellen van vogels. Maar het gedrag van deze zeer intelligente zeevogels in reactie op veranderende omstandigheden is moeilijk te voorspellen. Tot er meer van dit soort onderzoek beschikbaar is, zullen we het moeten doen met de inschattingen van deskundigen. In KEC 4 werden bijvoorbeeld 'Bradbury scores' gebruikt, die vergelijkbaar zijn met een sterfte van verdreven individuen van 1% voor de jan-van-gent en 5% voor de zeekoet. Als deze eerdere inschattingen worden gebruikt, wordt de ALI overschreden voor de zeekoet, maar niet voor de jan-van-gent.

De effecten van veranderingen in lokale dichtheden, door de tijd of door effecten van windturbines, worden niet meegenomen in de slachtofferberekeningen of de populatiemodellen. Door de verdrijving van vogels uit bepaalde gebieden met windturbines zullen hogere dichtheden ontstaan in andere gebieden. Door de toegenomen aantallen buiten de windmolen gebieden op zee kan de onderlinge concurrentie om ruimte en voedsel toenemen en zouden er lokale voedseltekorten kunnen ontstaan. In hoeverre zulke effecten ontstaan is afhankelijke van het lokale voedselaanbod, maar ook van het totale aantal windmolenparken dat wordt gebouwd. Dit is wederom lastig mee te nemen door een tekort aan kennis op dit gebied. Ook in de populatiemodellen is gekozen voor een dichtheidsonafhankelijke populatiegroei, omdat dit tot de meest voorzichtige inschatting zou leiden van de effecten. We missen echter voor veel soorten de kennis om dichtheidsafhankelijkheid in populatieontwikkeling goed mee te kunnen nemen.

De huidige assessment methode is gebaseerd op populatieprojecties over een periode van 40 jaar voorwaarts in de toekomst. De snelle veranderingen in zeevogelpopulaties door vogelgriep en klimaatverandering onderstrepen dat omgevingsfactoren populatie ontwikkeling sterk kunnen beïnvloeden. De uitkomsten van deze studie zijn de best mogelijke schattingen gebaseerd op de vogeltellingen en de assessment methode die op dit moment beschikbaar is. De uitkomsten zijn gebaseerd op de aanname dat de ruimtelijke zeevogel verspreiding niet strek zal veranderen in de komende 40 jaar, anders dan door de ontwikkeling van windparken op zee.

Summary

This report describes the impact assessment of planned offshore wind farm (OWF) development in the Dutch North Sea up to 2031 on marine bird populations (Chapter 1). The assessment considers the effects of displacement of marine birds from operational OWFs as a result of avoidance of these areas. The species in the assessment are the northern gannet, Sandwich tern, common guillemot, and razorbill. These species are known avoiders of OWFs, due to the presence of the structures, the noises related to turbine operation, or the presence of maintenance ships.

The assessment method is based on the methodology developed in KEC 2, 3 and 4. Compared to the previous assessment (Chapter 3), we:

1. used newly developed distribution maps for the national scenarios,
2. considered only four out of the original ten species because of the limited availability of the new maps,
3. updated many of the parameters in the model based on the most current knowledge,
4. implemented species specific effect distances for the displacement casualty calculations,
5. used a 'displacement matrix' approach instead of the 'Bradbury' approach that was used in KEC 4,
6. considered time-varying values for the population parameters rather different values per simulation for the parameter values that were fixed through time,
7. designed and implemented a new method for uncertainty analysis throughout the full model framework, and,
8. implemented the adjusted ALI methodology, which was updated based on external reviews, and used new, ALI values which were defined by LVVN.

The assessment was conducted for four scenarios.

- One scenario includes the OWFs that had already been built up to 2020 at the Dutch North Sea ('Basic scenario').
- One scenario includes the OWFs that had been permitted up to 2024 in the Dutch North Sea ('Basic plus scenario').
- One scenario includes the OWFs planned up to 2031 in the Dutch North Sea ('Total scenario').
- One scenario includes the OWFs planned up to 2031 in the southern North Sea at large, on the international level ('International scenario').

Based on seasonal distribution maps of the birds, we calculated the overlap between OWFs and marine bird distributions. The mortality due to displacement is highly uncertain, and we therefore used a range of displacement mortality values for the birds that were displaced from areas with OWFs. Using population models it was estimated how the additional mortality due to displacement affected population trends. The changes in population development due to the presence of OWFs were tested against acceptable levels of impact (ALIs), for which values were used in this report as determined by the ministry of LVVN ('Ministerie van Landbouw, Visserij, Voedselzekerheid en Natuur') (Chapter 3 and 4). For the international scenario, no ALI tests could be conducted due to the absence of uncertainty estimates for the international distribution maps.

Sandwich tern and razorbill population trends were predicted to be hardly affected by the presence of OWFs. None of the ALI thresholds were violated for these species. However, for the northern gannet and common guillemot, the models predicted a slower population growth with OWFs. The mortality due to displacement is highly uncertain, and we therefore used values of 1, 2, 5 and 10% for displacement mortality. For the scenario that included all planned OWFs up to 2031 in the Dutch North Sea and a displacement mortality of 10%, the mean northern gannet population size after 40 years was estimated to be 4.9% lower than without OWFs. For the common guillemot, this effect was even larger, the mean population size after 40 years was estimated to be 31% lower. The ALI thresholds were violated for the northern gannet for the national scenarios that included all planned OWFs up to 2031 and all approved OWFs in 2024 in the Dutch North Sea

for a displacement mortality of 10%, but not for 5, 2 and 1% displacement mortality. For the common guillemot, ALI violations were found for all national scenarios for 10% displacement mortality. For 5% displacement mortality, the ALI was violated for the scenario that included all OWFs planned up to 2031 and the scenario that included all OWFs approved up to 2024 in the Dutch North Sea. For 1 and 2% displacement mortality, no ALI violations were found for the common guillemot.

No ALI tests were conducted for the international scenario, but the mean population trends were found to be more affected by the international scenario than by the national scenarios for all four species (Chapter 4). For the international scenario, a higher mean overlap was found between the bird distributions and the OWFs than for the national scenarios. This is probably because the common guillemot, razorbill and northern gannet densities are, on average, higher in international waters close to the largest breeding colonies along the UK coast.

Many of the model components contain a lot of uncertainty and processes had to be based on strong assumptions. For example, the mortality level that was assumed strongly influences the outcomes of the assessment. Yet, it is difficult to determine which level of displacement mortality should be considered. While there are clear observations of changes in bird distributions due to the presence of OWFs, it is not clear in which way this will affect the individuals. To determine effects of OWF displacement on their vital rates, long term monitoring of bird behaviour around OWFs would have to be coupled to long term observations of energetic condition, breeding success and survival. So far, no studies exist that link these factors together for the same group of individuals. In KEC 4, we relied on mortality values based on expert opinions (‘Bradbury scores’). The scores that were used in KEC 4, were comparable to a displacement mortality of 1% for the northern gannet and 5% for the common guillemot. Based on these values, the ALI threshold would be violated for the common guillemot, but not for the northern gannet (Chapter 4).

One of the main uncertainties, the casualty calculation method of the effects of displacement, is hard to overcome, as so little information is available with regards to the effects of displacement on individual birds. In the current project, we attempted to develop a model with a more mechanistic consideration of displacement (Chapter 5). Unfortunately, we did not manage to develop the model to such a level that it could be included in the assessment.

Finally, displacement possibly leads to a density dependent effect on the food intake, as displaced birds are redistributed to areas outside OWFs. The locally higher densities may lead to increased competition for food. The casualty calculation methods should somehow include these effects, which is likely to become more important now that larger areas are being developed for wind energy. The population models are also not considering effects of increased density on the population parameters. This was a design choice based on the precautionary principle. However, density dependent processes probably do affect the populations. Again, the lack of detailed information on these processes makes them hard to implement.

The current assessment projects population developments forward over a period of 40 years. The rapid changes in marine bird populations due to avian influenza and climate change show that environmental change can strongly impact population trends. The estimates in this study give the best estimate possible based on the species distribution data and assessment framework available to us at the current moment and are based on the implicit assumption that seabird distributions will not drastically change in the next 40 years, for other reasons than offshore wind farm development.

1 Introduction

Offshore wind farms (OWFs) form an important part of the Dutch strategy to comply with the agreements on reducing CO_2 emissions, such as defined in the Paris Agreement. Around 2032, about 21 GW of energy should be generated by OWFs in the Dutch North Sea. This should provide the Netherlands with 16% of its energy requirements. In 2022, the Dutch government has indicated that the energy generated by offshore wind should be further extended to 50 GW by 2040 and eventually to 70 GW in 2050.

The 'Kader Ecologie en Cumulatie' (KEC) is an assessment that considers the ecological effects of current OWFs and future OWF developments. It aims to determine if OWFs lead to significant negative effects on the population trends and Marine Framework Strategy Directive objectives for marine birds, bats, migratory birds and marine mammals. These species are considered because they are expected to be most affected by OWFs and have a legal protected status in the Netherlands and are of high importance for the Dutch Environment and Planning Act. In the Offshore Wind Energy Act, the elaboration of the nature conservation articles from the Environment and Planning Act are incorporated. If necessary, regulations will be included in the site decisions of OWF areas to prevent or diminish the effects of OWF developments on these species. In the North Sea Agreement 2022-2027, the government has agreed to the formulation and application of the KEC. In the KEC, the cumulative ecological effects of OWF development on a selection of impact sensitive marine bird species with a policy relevant status are assessed.

Potential adverse effects of offshore wind farms on marine birds may result from mortality due to collisions with turbines and/or displacement from wind farm areas. The latter may lead to a loss of foraging habitat or barriers to both daily and seasonal movements of birds (Drewitt & Langston, 2006; Masden *et al.,* 2010). Species that show high avoidance of turbines at sea are thought to be more sensitive to displacement and habitat loss, while species that show low avoidance and typically fly at altitudes within rotor heights of the turbines are thought to be more sensitive to collision mortality. Here, the focus lies on potential adverse effects of wind farms on seabirds in relation to losses of foraging habitat. Another part of KEC 5 focuses on the effects of collision mortality (IJntema *et al.,* 2025). The potential adverse effects of wind farms as barriers to movement are currently not under consideration within KEC due to a lack of knowledge on this subject.

This report describes the KEC 5 impact assessment of habitat loss due to OWFs for the northern gannet, Sandwich tern, common guillemot and razorbill. The assessment considers the effects of existing OWFs and future OWFs in the search areas that will be permitted for OWF development up to 2031 in the Dutch North Sea (Jetten, 2024). The methodology largely follows the model framework such as previously described by Soudijn *et al.,* (2022b), which, in turn, was based on the methodologies that were applied for KEC 2 and 3 (Leopold *et al.,* 2014; van der Wal *et al.,* 2018). The main changes compared to the previous KEC version are described in Chapter 3. Chapter 4 describes the assessment itself and also all the methodological details of the current assessment.

In line with the previous assessments, the current KEC assessment will test the population-level effects of offshore wind farms on seabirds. For the last KEC, threshold detection methods were developed in the form of Acceptable Levels of Impacts (ALIs), which are based on population models (Potiek *et al.,* 2022). Since KEC 4, the ALI methodology has been updated (Hin *et al.,* 2024) and species-specific ALI threshold values were defined by LVVN (the Ministry of Agriculture, Fisheries, Food security and Nature).

2 Assignment

The aim of this project is to estimate population level effects of habitat loss due to existing OWFs and future OWF developments up to 2031 based on the most recent scientific knowledge. The species included in the KEC 5 habitat loss assessment are the northern gannet, Sandwich tern, common guillemot and razorbill. For these species, population level effects of habitat loss due to OWFs will be tested against acceptable level of impact thresholds, for which values have been defined by LVVN.

The KEC framework consists of four components: a. habitat suitability models of marine birds, b. casualty calculation models, c. population models, and, d. acceptable level of impact definitions. Following these steps, the assessment was carried out for each species. Component a. was conducted in a separate project (van Donk, 2024). Components b-c were conducted in the current project, while the method for the acceptable level of impacts was defined in a separate WOZEP project (Hin *et al.,* 2024).

The casualty calculation (component b) was adjusted to align better with the methods used in recent MER ('Milieu Effect Reportage': environmental impact assessment) procedures that are conducted for individual OWF areas. We used a method known as the Displacement Matrix (JNCC, 2022; Searle *et al.,* 2022a). In addition, we worked on the development of HALOMAR, an individual-based model for the estimation of effects of habitat loss that may eventually replace the current casualty calculation method. Due to the short timeline of this project we did not manage to finalise the HALOMAR model to be used in the current assessment. Finally, we developed an uncertainty analysis method for the full model framework, which includes uncertainty from all parameters in the different model components. It propagates the uncertainty across the steps in the estimation such that the uncertainty of all steps is included in the final estimation of the population trends. This development was necessary to align the method better with the newly defined ALI methodology (Hin *et al.,* 2024).

In consultation with the commissioner, we agreed on four scenarios for OWF development for the Dutch EEZ: Null, Basic, Basic plus and Total national scenarios. In addition, an International scenario was considered in the assessment.

As part of the assignment, we documented:

- A public-friendly summary (page 5),
- A knowledge base update, which describes the changes in methodology compared to KEC 4 (Chapter 3),
- The KEC assessment, which includes documentation of the methodology and the results of the effect estimations, as well as a discussion of shortcomings of the model framework (Chapter 4),
- The HALOMAR method, we document the current state of the model (Chapter 5),
- The data and code that was used for the estimations (stored publicly at: https://doi.org/10.5281/zenodo.14452359).

3 Knowledge base update

3.1 Outline

As part of the activities carried out within KEC 5, displacement of marine birds due to offshore wind farms (OWFs) in the operational phase, we provide written documentation of the update of the methodology compared to KEC 4. This written documentation describes:

1. How the species in the assessment were chosen,
2. Which scenarios were chosen and why,
3. Which effects of OWFs were considered,
4. Which recent knowledge has become available and why this knowledge was or was not taken on board in the current assessment,
5. How the effects on the chosen bird populations were estimated.

The methodology is designed to do a cumulative environmental impact assessment of the OWF plans up to 2031 (Jetten, 2024).

We report on updates in the following components of the KEC assessment:

1. Species selection
2. Model framework
3. Casualty calculation models
4. Population models
5. Uncertainty analysis
6. Input data and parameter values for the whole model framework
7. New insights regarding habituation based on empirical studies
8. Scenarios

The habitat suitability models, which were used in this project, are reported on in a separate document (van Donk, 2024), as these were part of a preceding KEC project. Detailed model descriptions are provided in dedicated chapters of the report.

3.2 Species selection

In KEC 4, the effect of displacement due to OWFs was considered for 10 species: diver sp., northern fulmar, northern gannet, great cormorant, common eider, common scoter, Sandwich tern, common guillemot, razorbill and Atlantic puffin. For KEC 5, we included the species for which new habitat suitability models were available (van Donk, 2024): northern gannet, Sandwich tern, razorbill and common guillemot (***Table 3-1***). In time, new distribution maps will be developed for additional species that need to be considered for effects of displacement. Due to time constraints, it was only feasible to make new maps for the four species that were considered the most relevant, all are common species within the Dutch part of the North Sea.

From the remaining six species, the great cormorant is, based on at sea observations, expected to be attracted rather than repelled by OWFs (Dierschke *et al.,* 2016; Lindeboom *et al.,* 2011). Surprisingly, this species did score high in the displacement sensitivity study conducted by Furness *et al.,* (2013). Nonetheless, we consider it unlikely for the great cormorant to suffer from displacement by OWFs based on the observed presence in OWFs at sea (Dierschke *et al.,* 2016; Lindeboom *et al.,* 2011). The diver sp., common eider and common scoter are species that occur most in nearshore waters (Bijlsma *et al.,* 2001) and are not expected to occur much in the zone where the new Dutch OWFs will occur (>10 km offshore). The northern fulmar and Atlantic puffin occur mostly in the most northern part of the Dutch waters (Bijlsma *et al.,* 2001) and thus, effects of Dutch OWFs are less likely to affect these species severely.

Table 3-1 *List of species included in KEC 5 for the assessment of effects of habitat loss due to OWFs on marine birds and references to the description of the models used for each of the steps in the model framework for effects of displacement.*

Species	**International distribution maps**	**National distribution maps**	**Casualty calculations**	**Population models**	**Uncertainty analysis**
northern Gannet	(Waggitt *et al.,* 2020)	(van Donk *et al.,* 2024)	Displacement matrix (JNCC, 2022; Soudijn *et al.*, 2022b)	Updated version of models described in Soudijn *et al,*. (2022b)	Newly developed
Sandwich tern	Updated version of maps described in Soudijn *et al.,* (2022b)				
razorbill	(Waggitt *et al.,* 2020)				
common guillemot	(Waggitt *et al.,* 2020)				

3.3 Model framework

The model framework to estimate population level effects of habitat loss due to offshore wind farms for marine birds consists of four components: a. habitat distribution models of marine birds, b. casualty calculation models, c. population models, and, d. acceptable level of impact definitions. For each of these components calculations are executed based on species-specific empirical data. The components a-c are all conducted within KEC and the method for the acceptable level of impacts was defined in a project that was not part of KEC (Hin *et al.,* 2024).

For the national scenarios, we used the new habitat suitability maps, which are focussed on the Dutch part of the North Sea (van Donk, 2024). For the international scenarios, we used the maps presented by Waggitt *et al.,* (2020), or when these were not available (Sandwich tern), we used maps based on the inverse distance weighting methodology described by Soudijn *et al.,* (2022b) (***Table 3-1***). The distribution maps used in the current KEC have a resolution of 10x10 km, rather than the 5x5 km resolution used in KEC 4 (van Donk *et al.,* 2024). For the casualty calculations, the displacement matrix was used (JNCC, 2022; Searle *et al.,* 2022a), which is a method similar to the one used in the previous KEC (Soudijn *et al.*, 2022b). The population models have remained almost the same in structure as for the last KEC. The acceptable level of impact (ALI) methodology has been updated (Hin *et al.,* 2024) and ALI norms have been defined by LVVN. For better alignment with the new ALI methodology, we developed an uncertainty analysis for the casualty calculations based on the uncertainty in the distribution maps and propagation of uncertainty from the distribution maps to the population models. We describe the details of this method in section 4.1.7.

3.3.1 Casualty calculation models

Effects of habitat loss are difficult to assess as these are typically slow, indirect effects on individuals that build up over time and space (Daunt *et al.,* 2020). As effects may interact with each other, it is not ecologically relevant to predict effects of habitat loss due to a single offshore wind farm in an impact assessment model. Both effects of the losses of foraging habitat and barrier effects, which together comprise habitat loss effects (Croll *et al.,* 2022; Exo *et al.,* 2003), affect individuals' energy budgets, either through reduced energy intake or increased energy costs. The severity of effects depends strongly on the condition of the individuals (which reflects food conditions, population status and level of competition). In case an indication of the severity of the consequences of habitat loss due to a single OWF needs to be determined, a statistical approach which indicates the importance of an area for a certain species may be more applicable than an impact assessment model (Mercker *et al.,* 2021).

Effect studies of habitat loss are commonly based on expert judgement (Bradbury *et al.,* 2014; Leopold *et al.,* 2014). Since the mortality of displaced individuals is unknown, a precautionary value is usually adapted (Leopold *et al.,* 2014; Searle *et al.,* 2022a). Modern statistical approaches have been used to estimate the number of individuals that could be displaced in the German part of the North Sea (Peschko *et al.,* 2020, 2024). These German studies are more representative and superior to the Dutch assessment methods, as studies of tagged birds of different species (common guillemot and northern gannet) of the colonies on Helgoland and long-term monitoring data covering the impacts of multiple wind farms are available. These methods do, however, not result in impact assessments in the form of casualty or mortality estimates. For the Scottish coast, individual-based models for common guillemot, razorbill, Atlantic puffin and black legged-kittiwake have been developed to calculate displacement impacts in the form of mortality during the breeding period (Searle *et al.*, 2014, 2018). We are currently developing an individual-based model to calculate displacement impacts for our study species for the Dutch part of the North Sea, but this model was not developed sufficiently yet to include it in the current assessment. This model is described in chapter 5 of this report.

3.3.1.1 Displacement casualties

For KEC 5, only the effect of displacement, in the form of the loss of foraging area, was considered. In the previous versions of KEC, casualty calculations were conducted with the 'Bradbury method'. This method was first proposed by Leopold *et al.,* (2014), following Bradbury *et al.,* (2014). This method calculates the number of displaced individuals by overlaying a density map with a map of the planned OWFs. The sum of the numbers of individuals on the map in each OWF area represents the number of displaced individuals. Subsequently, this number is multiplied by a displacement sensitivity score, which represents the tendency of the species to avoid OWFs and its habitat selectivity. The product of the two represents the number of casualties due to habitat loss. The displacement sensitivity score assumes a maximum of 10% mortality of displaced individuals. In recent MER procedures (van der Vliet *et al.,* 2022), the number of displaced individuals was calculated by multiplying the bird density in OWF areas with the displacement probability and an assumed displacement mortality probability of 10%.

For KEC 5, we will replace the "Bradbury method" with an approach based on a "displacement matrix" (JNCC, 2022; Searle *et al.,* 2022a). This approach was chosen because it allows for an explicit assessment of uncertainty. The displacement matrix method is similar to the method used in MER procedures, while adopting a range of values for the displacement mortality, barrier distance and displacement probability. The numbers of casualties were calculated at an annual scale, as opposed to the Bradbury method used in KEC 4, which was based on bimonthly casualty calculations (see section 4.1.4 for details).

3.3.2 Population models

Population viability analysis (PVA; Soule, 1986) is a computational method that is often used in conservation science and management to predict population trajectories and population extinction risk (Croll *et al.,* 2022). PVAs are considered best practice in assessments of effects of offshore renewable energy developments (OWFs) on marine birds (Horswill *et al.,* 2022; Searle *et al.,* 2023). Most PVAs that evaluate effects of OWFs on marine birds are stage- or age- classified matrix population models (Caswell, 2001) that translate estimates of demographic rates, particularly breeding success, age at first breeding and survival, to projections of population abundance. In addition, PVAs used in impact assessment often account for various sources of uncertainty and variability (Croll *et al.,* 2022) and distinguish between forms of variability that are reducible through collection of additional data or better parameter estimation (uncertainty) and variability that is irreducible, such as environmental variability in demographic rates, or demographic stochasticity (Searle *et al.,* 2023).

The PVAs that are used to quantify the consequences of habitat loss are age-structured matrix population models (Caswell, 2001). The structure of the population models and their assumptions were previously described by Van Kooten *et al.,* (2019), Hin and Soudijn (2021) and Soudijn *et al.,* (2022b). In these population models, uncertainty and environmental variability were accounted for by sampling parameters from probability distributions and executing many replicate simulations (the so-called Monte Carlo method). In the previous KEC (Soudijn *et al.*, 2022b), parameters were sampled only at the beginning of each simulation. For the current KEC, parameters will instead be sampled each time step (annually), based on the observation that seabird demographic rates tend to vary from year-to-year as a result of environmental variability (Jenouvrier *et al.,* 2005; Miller *et al.,* 2019). This will lead to less variation in the prediction of future population abundance, because years with good environmental conditions, in which population growth is high, will be alternated by years with poorer conditions and lower or even negative population growth (Hin *et al.,* 2023, 2024).

Sampling demographic parameters each year implies that the relative abundance (stage distribution) of the different stages fluctuates through time and is no longer stable. This has repercussions for the matrix models of the common guillemot, razorbill and Sandwich tern, in which some year classes were combined into a single stage, under the assumption of a stable age-distribution. For these matrix models, we have split the relevant stages into multiple year-classes, such that fluctuations in the age distributions are properly accounted for (see section 4.1.5 and *Table 4-2*). In addition, for the current KEC, we developed an improved and more structured treatment of the different forms of uncertainty (see section 3.3.4).

3.3.3 Acceptable level of impact

The Acceptable Level of Impact (ALI) methodology defines acceptable limits for the predicted population effects of mortality imposed by offshore wind farms (OWFs) on marine bird populations. The ALI methodology was first developed in 2021 as a replacement for the Potential Biological Removal (PBR) method (Potiek *et al.,* 2022) and was revised in 2024 (Hin *et al.,* 2024). The revised ALI has been subjected to extensive external review and was recently approved for use in KEC 5 (for details, see sections 4.1.5 and 4.1.6). Values for the thresholds were defined by Dutch policy-makers (LVVN) for use in KEC 5.

3.3.4 Uncertainty analysis

For the new ALI method, it is considered important that the effect of the uncertainty in all inputs of the assessment framework on the model outcomes are made explicit (Hin *et al.,* 2024). In other words, the possible ranges in model outcomes need to be taken into account. To achieve this, we considered the full model framework and defined the categories of uncertainty in all model inputs, distinguishing between natural variability and knowledge uncertainty. In addition, we developed a method for uncertainty propagation through the model framework based on Monte Carlo simulations. The interrelations in the model framework are illustrated in Figure 3-1. Parameter values were sampled from probability distributions for all parameters in the model, and the habitat distribution maps were sampled from a pool of 1000 possible maps per bimonthly period. These maps were generated based on the estimated probability distributions of the habitat models per grid cell.

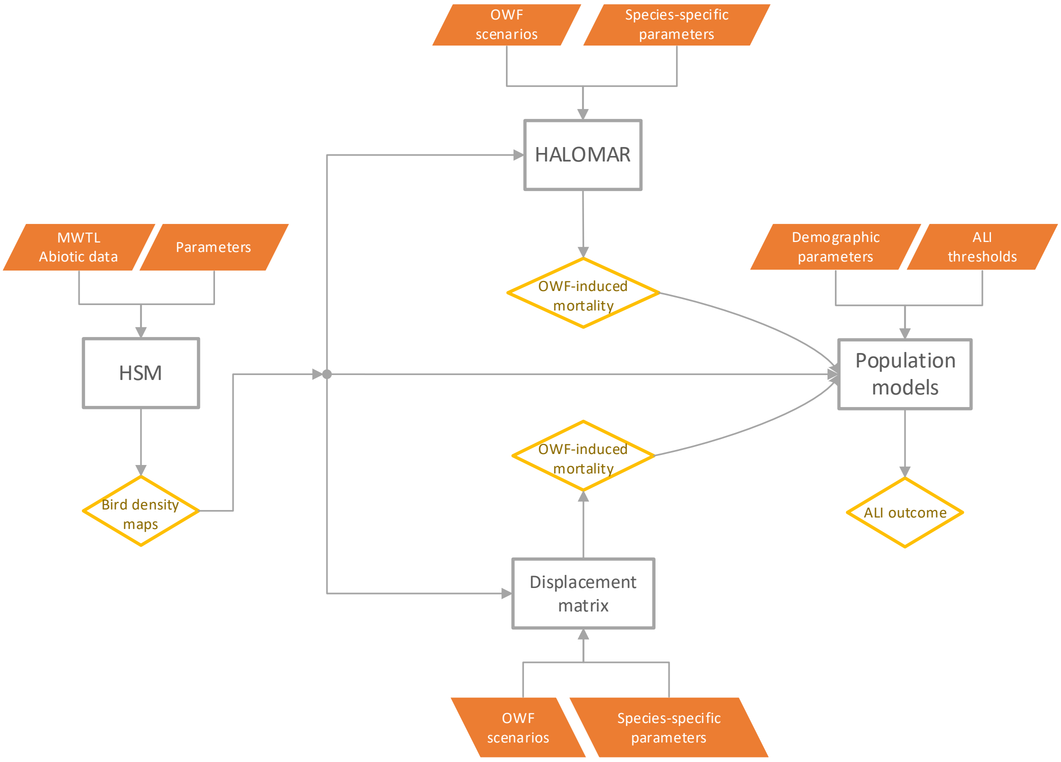

Figure 3-1 *Schematic overview of the model framework. Uncertainty in the input data and parameters in the orange boxes is considered throughout the framework. HSM is an abbreviation of habitat suitability model. KEC 5 is based on casualty calculations with the displacement matrix (Chapter 4), future KEC studies may use HALOMAR for the casualty calculations instead, the model is still in development (Chapter 5).*

3.4 Input data and model parameters

3.4.1 Data and parameter uncertainty

The displacement matrix calculations are based on habitat preference models, and calculations have been repeated multiple times based on a pool of 1000 possible bird distribution maps per bimonthly period that were provided by the KEC maps project (van Donk, 2024). Both casualty calculation methods provide a distribution of additional mortality probabilities per scenario, based on the uncertainties in the maps and parameters. The population models sample from these additional mortality distributions.

The probability distributions of the model parameter values follow beta, gamma or multinomial distributions. In most cases, the data underlying the parameter values are binomially distributed. The survival probability, breeding success, displacement probability and skipped breeding probability either take the value '0' or '1' on the individual level. As a consequence, the variability around the mean parameter values $\bar{\mu}$ and variance σ follows a beta distribution, with parameters $\alpha = \bar{\mu}^2 \left(\frac{1-\bar{\mu}}{\sigma^2} - \frac{1}{\bar{\mu}}\right)$ and $\beta = (\alpha - \bar{\mu}\,\alpha)/\bar{\mu}$. A beta distribution is appropriate to describe the uncertainty around the mean chance of a 'failure/success' type of process, such as survival and producing a single offspring. Categorical or discrete data, such as the start and end date of the breeding season or the number of foraging locations utilised per day follow a multinomial distribution and the probability of occurrence of each category needs to be defined.

3.4.2 Casualty calculations

The habitat distribution maps form the foundation of the displacement matrix, the displacement matrix calculations are conducted 1000 times, where a bimonthly map for each of the periods is randomly chosen for each calculation. The period with the maximum population density is determined per species as the period which has, on average, most individuals in total on the map. Since the mortality of displaced individuals is unknown, 10% mortality has been adopted as a precautionary value (Leopold *et al.,* 2014) for the displacement mortality. In the displacement matrix approach, we will use a range of possible values, 1%, 2%, 5% and 10%, to show the effect of this parameter on the predicted numbers of casualties.

For the boundary distance of displacement behaviour, 500 meters was used for all species in KEC 4. For KEC 5, studies with species specific information has become available, of which we selected worst case values or values of studies most representative for the Dutch North Sea (*Table 3-2*). In addition, we introduced a multiplication factor to correct for the expected decrease of the magnitude of bird displacement towards the edge of the affected area. We do not consider uncertainties for these parameters as no estimates were available. Due to the new effect distances, the OWF areas started to overlap and therefore, scaling factors (which translate the surface areas planned per OWF to the surface areas that are actually used), were now determined per scenario and per species rather than per OWF area, such as was done in KEC 4 (see section 4.1.3, *Table A-1* and *Table A-2*).

In KEC 4, all displacement probabilities were based on the review by Dierschke *et al.,* (2016b) of bird displacement by wind farms in European waters. In the review of Dierschke *et al.,* (2016b), the northern gannet was categorized as a 'strong avoider' of wind farms, while razorbill, guillemot and Sandwich tern were qualified as 'weak avoiders' of wind farms. In KEC 4, we used the information for individual OWF areas for which displacement was quantified and chose avoidance values on the high side, in line with the precautionary principle. This resulted in the values per species in KEC 4 as described in *Table 3-3*. Since KEC 4, new information has become available based on several case studies published in the scientific literature. In KEC 5, we used values and uncertainty ranges based on these new studies, the values and sources per species are described in *Table 3-3*. For the northern gannet, we used the value reported by Vanermen *et al.,* (2015), rather than the new value reported by Peschko *et al.,* (2021). We consider the study by Vanermen *et al.,* (2015) a more reliable estimate for displacement as they base their values on area counts including many individuals while Peschko *et al.,* (2021) based their estimate on tracking studies with few birds. No information on uncertainty of the displacement probability was available for the gannet, and we used the same standard deviation as for the Sandwich tern. For the Sandwich tern, we changed the value of the displacement probability based on a new study (van Bemmelen *et al.,* 2024), which was, however, in the same order of magnitude as the earlier reported values. For the razorbill, Grundlehner & Leopold (2024) report similar estimates as previously reported values. For the common guillemot, we chose the estimate based on Peschko *et al.,* (2024), as they were based on long term aerial and ship-based monitoring on a large spatial scale covering multiple offshore windfarms. Based on the analysis of aerial monitoring, Grundlehner & Leopold (2024) found a slightly higher value than earlier reported values.

Table 3-2 *Casualty calculation parameters, shapes of the distributions for the parameter values and sources from which the values were taken are indicated in the table.*

Description	**symbol**	**unit**	**northern gannet**	**Sandwich tern**	**razorbill**	**common guillemot**	**distribution**	**Sources**
Effect distance around OWFs	ϑ	m	1500	1500	2000	19500	No uncertainty considered	(Grundlehner & Leopold, 2024; Peschko *et al.,* 2024; Vanermen *et al.,* 2015; Welcker & Nehls, 2016)
Effect distance multiplication factor	ε	-	0.5	0.5	0.5	1	No uncertainty considered	(Grundlehner & Leopold, 2024; Peschko *et al.,* 2024; Vanermen *et al.,* 2015; Welcker & Nehls, 2016)
Displacement mortality probability	μ_d	-	1%, 2%, 5% & 10%	1%, 2%, 5% & 10%	1%, 2%, 5% & 10%	1%, 2%, 5% & 10%	Multiple values considered	-
Period with maximum density	$imax$	-	Aug-Sep	Aug-Sep	Feb-Mar	Aug-Sep	No uncertainty considered	Distribution maps

Table 3-3 *Displacement probabilities (*$\boldsymbol{\rho}$*), parameter values used in KEC 4 and KEC 5, both means and confidence intervals (CIs) are indicated, shapes of the distributions for the parameter values and sources from which the values were taken are indicated in the table.*

	KEC 4		**KEC 5**				
Species	Mean	Source	Mean	CI	sd	Distribution	Source
northern Gannet	0.8	(Dierschke *et al.,* 2016)	0.85	NA	0.69	beta	(Vanermen *et al.,* 2015; sd from Sandwich tern)
Sandwich tern	0.5	(Dierschke *et al.,* 2016)	0.54	0.35-0.70 (95% CI)	0.69	beta	(van Bemmelen *et al.,* 2024)
razorbill	0.5	(Dierschke *et al.,* 2016)	0.43	0.40 – 0.45 (99% credibility)	0.021	beta	(Grundlehner & Leopold, 2024)
common guillemot	0.5	(Dierschke *et al.,* 2016)	0.79	0.74 – 0.83 (95% CI)	0.097	beta	(Peschko *et al.,* 2024)

3.4.3 Population parameters

A search of the scientific and grey literature was performed to find new estimates of demographic rates (e.g. survival, breeding success) for the species that are being assessed for habitat loss, the northern gannet, Sandwich tern, common guillemot and razorbill. In addition, the method of parameter weighting was improved. Previous estimates of demographic rates were derived as the means of estimates from multiple colonies and weighted means were calculated based on the length of each time series. The sizes or locations of the colonies were previously not considered, which might create a bias if demographic rates vary considerably between colonies. Ringing recoveries indicate that northern gannet, common guillemot and razorbill occurring in the Dutch part of the North Sea originate from a wide area of colonies, including colonies on the west side of the UK and Ireland and further away (Leopold, 2017; Vogeltrekstation NIOO-KNAW & Ringersvereniging, 2024). Where possible, it was decided to derive population parameters from estimates of demographic rates for colonies closest to the Southern North Sea (these are specified per species below). In addition, if data from multiple colonies were used, these data were weighted with colony size based on the most recent published data. Where possible, weighted standard deviations were derived in a similar manner. In case no estimate of colony size was available for a certain year, an interpolated value was calculated based on linear interpolation. Lastly, estimates of demographic rates were as much as

possible confined to the period 2016-2021, as sea bird survey data from these years were used to create the distribution maps.
Compared to KEC 4, the stage-specific scaling factors that weigh the contribution of each life stage towards habitat loss induced mortality have also been adjusted (see section 4.1.5 and *Table 4-1*).

3.4.3.1 Northern gannet

For northern gannet, the colonies closest to the Southern North Sea are Bass rock (UK), Bempton Cliffs (UK) and Helgoland (Germany) (Jeglinski *et al.,* 2023). Yearly breeding productivity estimates from these colonies are given by the Flamborough and Filey Coast SPA Seabird monitoring programs (Aitken *et al.,* 2013, 2017; Babcock *et al.,* 2015, 2018; Cope *et al.,* 2021; Lloyd *et al.,* 2020; Lloyd & Aitken, 2019). Combining breeding success estimates according to the method described above yields a weighted mean value of 0.754, which is slightly higher than the previous UK-wide estimate of 0.7 (*Table 3-4*). The weighted standard deviation was derived as 0.031. An apparent adult survival rate of 0.94 (0.771-0.993 95% CI) was reported by Lane *et al.,* (2024) for the period 2011-2021 and adopted instead of the previous value of 0.918. The standard deviation of this survival estimate (0.0483) was derived from the reported confidence interval. No new values of juvenile and immature survival rate were found. With the newly derived parameter estimate the deterministic annual population growth rate as estimated by the northern gannet population model becomes 1.032 (3.2 per cent per year). The mean stochastic growth rate equals 1.031 (±0.042 SD).

Table 3-4 *Population parameters for the northern gannet, as used in KEC 5. In case values deviate from those used within KEC4 (Soudijn et al., 2022b), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.754 (0.7)	0.031 (0.082)	Beta	(Aitken *et al.,* 2013, 2017; Babcock *et al.,* 2015, 2018; Burnell *et al.,* 2023; Cope *et al.,* 2021; Dierschke *et al.,* 2016a, 2017, 2018, 2019, 2020, 2021; Lloyd *et al.,* 2020; Lloyd & Aitken, 2019)
Probability floater	P_F	0.05	0.125	Beta	KEC 4
Survival age 0	S_0	0.481	0.0853	Beta	KEC 4
Survival age 1	S_1	0.816	0.0393	Beta	KEC 4
Survival age 2	S_2	0.884	0.0293	Beta	KEC 4
Survival age 3	S_3	0.887	0.0301	Beta	KEC 4
Adult survival	S_A	0.940 (0.918)	0.0483 (0.0199)	Beta	(Lane *et al.,* 2024)

3.4.3.2 Sandwich tern

For the Sandwich tern, new estimates of demographic rates were taken from van Bemmelen *et al.,* (2022). Mean adult survival during years 2016-2020 was 0.94 for colonies in the South and North of the Netherlands. This value is identical to the previous mean adult survival based on 30 years of data (Schekkerman *et al.,* 2021). Annual immature survival for the period 2016-2020 was estimated at 0.6 and 0.69 for, respectively, Southern and Northern colonies in the Netherlands (van Bemmelen *et al.,* 2022). These estimates can, however, not be used directly in the Sandwich tern population model, because the model distinguishes survival of juvenile (age 0) birds from immature birds (age 1 and 2). Based on the estimates from van Bemmelen *et al.,* (2022), mean cumulative survival across the first 3 years of life equals $0.6^3 = 0.216$ (South Netherlands) and $0.69^3 = 0.329$ (North Netherlands). The previous parameters of the Sandwich tern population model (Soudijn *et al.*, 2022b) yielded a survival across ages 0-3 years of 0.307. To be conservative, we adjusted the juvenile survival of the population model to 0.357, such that survival across the first three years of life matched the cumulative survival of 0.216 as estimated by van Bemmelen *et al.,* (2022) for birds in the South of the Netherlands.
For the reproductive rate (breeding success) Schekkerman *et al.,* (2021) reported a weighted mean value of 0.49 for several colonies throughout the Netherlands in the period 2010-2019. Recent estimates from the Delta area suggest a higher breeding success for the years 2020 and 2021 (van Bemmelen *et al.,* 2022),

with a breeding success of 0.94 for some larger colonies. However, these estimates only represent the Delta area and no estimates of breeding success for the rest of the Netherlands could be obtained. We therefore adopted the values reported by Schekkerman *et al.,* (2021) as a representative estimate of breeding success for the whole of the Netherlands. The mean breeding success in the model is set to half this value, because the breeding success parameter in the Sandwich tern model only considers female offspring.
With the newly derived parameter estimate the deterministic annual population growth rate as estimated by the Sandwich tern population model becomes 0.986 (-1.4 per cent per year). The new mean stochastic growth rate for the Sandwich tern equals 0.986 (±0.0981 SD).

Table 3-5 *Population parameters for the Sandwich tern, as used in KEC 5. In case values deviate from those used in KEC4* (Soudijn *et al.*, 2022b)*, the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (female chicks only)	F_A	0.245* (0.325)	0.08 (0.160)	beta	Schekkerman *et al.,* (2021)
Probability floater	P_F	0.1	0.125	beta	KEC 4
Survival age 0	S_0	0.357 (0.508)	0.0917	beta	Derived from van Bemmelen *et al.,* (2022)
Survival age 1-2	S_{12}	0.777	0.0518	beta	KEC 4
Adult survival	S_A	0.94 (0.942)	0.108	beta	KEC 4; van Bemmelen *et al.,* (2022)

*this value represents the number of female chicks only. The original value reported by Schekkerman *et al.,* (2021) was 0.49 chicks per breeding pair.

3.4.3.3 Common guillemot and razorbill

For razorbill and common guillemot, the most important colony in the Southern North Sea is Bempton Cliffs with, respectively, approximately 44,000 and 106,000 individuals in 2022 (*Seabird Monitoring Programme Database*, 2024). For these two species we derived new mean breeding success parameters as the average and standard deviation of the breeding success estimates for years 2016-2021 (*Table 3-6* and *Table 3-7*). The literature search furthermore revealed updated survival estimates for razorbills breeding at Skomer (Newman *et al.,* 2021). These estimates were used to update the mean and standard deviation of the razorbill survival rate. With the newly derived parameter estimates the deterministic annual population growth rate as estimated by the razorbill population model becomes 1.003 (0.3 per cent per year). The new mean stochastic growth rate for the razorbill equals 1.003 (±0.0653 SD). For the common guillemot, the new deterministic growth rate equals 1.034, and the new stochastic growth rate equals 1.033, with a standard deviation of 0.0442.

Table 3-6 *Population parameters for the common guillemot as used in KEC 5. In case values deviate from those used within KEC4* (Soudijn *et al.*, 2022b)*, the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.622 (0.664)	0.045 (0.149)	beta	(Aitken *et al.,* 2017; Babcock *et al.,* 2015, 2018; Cope *et al.,* 2021; Lloyd *et al.,* 2020; Lloyd & Aitken, 2019)
Probability floater	P_F	0.07	0.03	beta	KEC 4
Survival age 0	S_0	0.608	0.132	beta	KEC 4
Survival age 1	S_1	0.774	0.112	beta	KEC 4
Survival age 2	S_2	0.858	0.0736	beta	KEC 4
Adult survival	S_A	0.949	0.0447	beta	KEC 4

Table 3-7 *Population parameters for the razorbill, as used in KEC 5. In case values deviate from those used within KEC4 (Soudijn et al., 2022b), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.615 (0.55)	0.085 (0.138)	beta	(Aitken *et al.,* 2017; Babcock *et al.,* 2015, 2018; Cope *et al.,* 2021; Lloyd *et al.,* 2020; Lloyd & Aitken, 2019)
Probability floater	P_F	0.03	0.125	beta	KEC 4
Immature survival	S_{01}	0.643	0.048	beta	KEC 4
Adult survival	S_A	0.911 (0.909)	0.0663 (0.0678)	beta	KEC 4; (Newman *et al.,* 2021)

3.5 Habituation

Habituation occurs when individual birds learn through experience that a disturbance source is harmless. Especially when the disturbance source is predictable in behaviour, birds can adjust their assessment of potential danger and therefore stop to respond. If habituation occurs after a repeated stimulus, individual birds show a reduced response; the individual 'gets used' to the disturbance (Goodship & Furness, 2024). So far, our model framework does not explicitly include habituation to specific OWFs, although habituation to OWFs in general may be included in recent observed displacement rates.

As a follow up of Dierschke *et al.,* (2016), in 2018, a review report was published which specifically considered habituation of seabirds (MMO, 2018). It contains few examples of habituation of seabirds to offshore wind farms. The examples presented all considered sea ducks or divers, including the famous incidental observation of sea ducks within Horns Rev Wind Farm (Petersen & Fox, 2007). However, also in these two species groups, mainly displacement effects were reported, and a strong tendency to habituation thus seems unlikely.
In case of offshore wind farms, habituation could be defined as the recovery in occurrence in relation to the presence of an offshore windfarm or a decreasing displacement effect of an offshore windfarm. According to this definition, no published evidence has so far been found that habituation takes place in the four species considered for habitat loss effects. Perhaps, the fraction of birds that is still making use of the wind farms could be regarded as birds which have been habituated to the presence of the wind farms, for example, for the northern gannet it was observed that the same group of individuals repeatedly attended a wind farm area (Peschko *et al.,* 2021). With the increase in offshore wind farms at the North Sea, the opportunity that seabirds learn that the movement of turning rotors bears no danger also increases. As soon as there is well documented evidence of such behaviour we could include it in the model framework, in the current setup we choose to consider no habituation as this is in line with a precautionary approach.

3.6 Scenarios

In KEC 5, we consider four versions of the national scenario, the Null, Basic, Basic plus and Total scenario and one international scenario (see *Table A-4* for a list of all considered OWFs per national scenario). The null scenario serves as a baseline situation without any offshore wind farms present. This is also the situation to which any OWF effects are compared. The Basic scenario considers all OWF areas that have already been built, the Basic plus scenario includes all OWFs that have been approved in 2024. The Total scenario includes all OWF plans up to 2031. We decided to split the scenarios into these four categories to allow for a comparison of any effect of the OWFs that have already been built or approved, to the full future OWF plans. The international scenario includes the Total scenario for the national area.

3.7 Summary

In summary, several changes have taken place in the KEC model framework for the assessment of effects of displacement by OWFs for marine birds:

1. The distribution maps that were provided are based on a new methodology for the national area. New maps are available for the common guillemot, the Sandwich tern, the northern gannet and the razorbill. Due to time constraints, these are only four out of the original ten species, but these are considered the most relevant species for the Dutch waters out of this list. The species list needs to be updated for the next KEC.
2. The original casualty calculation method has been adjusted to a displacement matrix method, to allow for a calculation of the uncertainty ranges in the mortality due to displacement.
3. The numbers of casualties were calculated at an annual scale, as opposed to the old bimonthly casualty calculations.
4. The structure of the population models is only slightly adjusted compared to the previous KEC. The main change is that stochasticity in the parameter values is now considered as an annual fluctuation in the parameter values rather than a fixed value per simulation.
5. The range of uncertainty in the outcomes of the model framework is now based on uncertainty of all the components of the model framework, from the distribution maps to the casualty calculation and the population parameters.
6. Population model parameters for breeding success and survival were updated based on the most recent published knowledge. Species specific parameter values were adopted for the effect distance around OWFs, displacement probabilities were updated and the age-specific weighting of displacement effects was adjusted, all based on recent publications.

4 KEC assessment

4.1 Materials and Methods

4.1.1 Bird distribution maps

Spatial bird distributions were used as the basis of the calculations. In a WOZEP project prior to this KEC, habitat suitability models were developed for the four seabird species considered in this assessment in the southern and central North Sea (van Donk *et al.,* 2024), based on Dutch (MWTL) bird count data. The habitat suitability models were designed to tackle the following issues:

- Handle high numbers of zero observations that are present in the data and over-dispersion of the density distribution;
- Provide uncertainty estimations of density distributions;
- Model the spatial-temporal autocorrelation of the density distributions;
- Predict densities at unobserved locations/times;
- Provide estimates at a small spatial scale relevant for OWFs (10x10 km);
- Provide seasonal maps.

Based on the habitat suitability models, bird distribution maps were predicted in the KEC maps project (van Donk, 2024), which form the basis of the effect calculations of KEC. Here, we broadly describe the maps that were used, the details of the maps are described by Van Donk (2024). The MWTL data used were collected in aerial surveys from 2016-2020 (van Donk, 2024). Bimonthly maps were predicted for the periods: Oct-Nov, Dec-Jan, Feb-Mar, Apr-May, Jun-Jul and Aug-Sep. For the international scenario, we used the monthly maps previously described by Waggitt *et al.,* (2020), except for the Sandwich tern, which was not considered by Waggitt *et al.,* (2020), and for which an inverse distance weighted map was used (*Table 3-1*). The spatial resolution of all the maps that were used was 10x10 km. This is a change compared to KEC 4, where a spatial resolution of 5x5 km was used.

4.1.2 OWF Scenarios

In KEC 5, we consider four versions of the national scenario: the Null, Basic, Basic plus and Total scenario, and an international scenario. The Basic scenario considers all OWF areas that have already been built up to 2020, the Basic plus scenario includes all OWFs that have been approved up to 2024. The Total scenario includes all OWF plans up to 2031. We decided to split the national scenarios into three categories to allow for a comparison between:

1. the period including any effect of the OWFs that were already present in the period from 2016-2020, which matches the period that was used for the bird distribution maps (Basic scenario),
2. the period in which the most recent OFWs have been approved up to and including 2024 (Basic plus scenario) and,
3. the period with the full realisation of future OWF plans up to 2031 (Total scenario).

From Basic to Total scenario, more OWF areas were considered, see Table A-4 for a list of all considered OWFs per national scenario. For the international scenario (International scenario), we only consider all future offshore windfarm development plans up to 2031, and it includes the Total scenario for the national area. All scenarios were compared with a situation with no OWFs present (Null scenario).

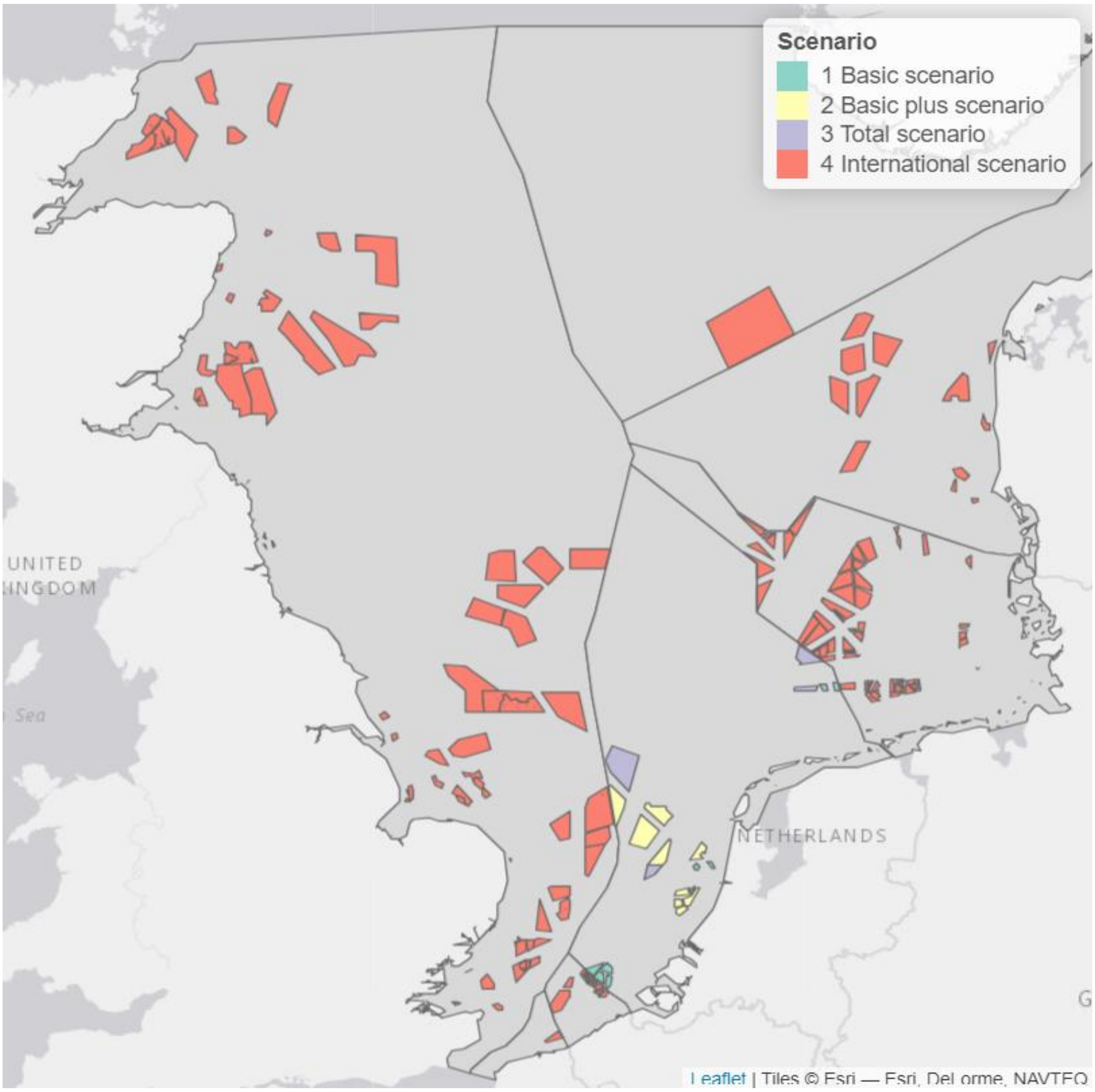

Figure 4-1 *Illustration of the national and international OWF scenarios. Note that the Basic plus scenario also includes the Basic scenario areas and that the Total scenario includes the Basic and Basic plus areas. The international scenario includes the areas indicated as the Basic, Basic plus and Total scenarios for the Dutch EEZ.*

4.1.3 OWF overlap

For each OWF, the impacted area was calculated as the surface area of the OWF plus a species-specific effect distance ϑ multiplied by the effect distance multiplier ε, resulting in an effect boundary around the OWF , (Chapter 3, *Table 3-2*). These effect zones around OWF search areas were found to overlap when OWFs were located relatively close to each other. Since the impact of an OWF decreases with distance from the OWF, we assumed that no cumulative displacement effect occurs in these overlapping areas, birds were thus assumed to be displaced only once by an OWF in the overlapping areas. As a consequence, OWF effect areas were often connected together.
The number of birds affected by an OWF B_{io}, per OWF area o and bimonthly period i, were determined by multiplying the impacted surface area with the local bird density. For vector grid-cells that had a partial overlap with the buffered OWF, the impacted number of birds was taken proportional to the part of the grid cell that was actually covered by the OWF effect area. Subsequently, the number of birds impacted by OWF search areas was corrected by a scale factor per scenario to correct for the difference in the area which is

under consideration vs. the area that is needed for the amount of power that is planned to be installed. Scale factors were calculated based on the contribution of each OWF to the total surface area within each scenario. The scale factors are close to 1 for fully commissioned OWFs and for OWFs that are in a late state of development (*Table A-4*). The OWFs in early development stages, labelled as concept/early planning resp. development zone, mostly have a scale factor well below 1.

RWS provided shapefiles of the OWF search areas, these delineate the outer edges where an offshore wind farm can be positioned (*Figure 4-1*). RWS provided also the surface area estimates for all OWF areas on which the scale factors were based (*Table A-4*). Areas of future OWFs were estimated by RWS based on planned turbine densities where available. In cases where no information was available regarding planned turbine density, they estimated the area based on probable future energy densities per area (i.e., 10 MW, 11 MW, or 12 MW/km²). The total surface areas covered by OWFs per scenario, both before and after taking the planned turbine densities into account, are presented in Annex 1, Table A-1. Species and scenario specific scaling factors applied in the calculations can be found in Table A-2.

4.1.4 Displacement matrix

In the current KEC assessment (KEC 5), the displacement matrix approach replaces the 'Bradbury' method that was used to estimate effects of displacement within previous KEC assessments (KEC 2: Leopold *et al.,* 2014; KEC 3: Van der Wal *et al.,* 2018; KEC 4: Soudijn *et al.,* 2022). The Bradbury method was based on a 'relative displacement risk score' (RDRS), which is a combined measure of the likelihood of a species to be displaced by OWFs and the species' sensitivity to habitat loss once it occurs (Bradbury *et al.,* 2014). A maximum mortality risk of 10% was assigned to species that scored highest on the ordinal scales of sensitivity and vulnerability to habitat loss (Leopold *et al.,* 2014).

As an alternative, the displacement matrix approach (JNCC, 2022; Searle *et al.,* 2022) calculates the number of casualties due to displacement by multiplying the number of birds in OWF areas by two parameters; *i*) the displacement probability (ρ) and *ii*) an annual displacement mortality risk (μ_d). This approach was adopted because it is more straightforward to parameterise and it allows a more explicit assessment of the associated uncertainty (Chapter 3). Empirical estimates of the displacement probability and their variability are represented in the displacement matrix approach by using a distribution for ρ, instead of a single value. Because empirical estimates of displacement mortality risk are unavailable, we used a range of values for μ_d: 1%, 2%, 5% and 10% per year for all four species (Chapter 3).

The number of displaced birds D_{io} per OWF area o and bimonthly period i was calculated as:

$$D_{io} = B_{io}\,\rho,$$

where B_{io} is the estimated number of birds in a certain OWF area from which birds are displaced and ρ is the displacement probability. On an annual basis, over six bimonthly periods, this leads to a mean number of casualties from displacement equal to:

$$\overline{C_o} = \frac{\sum_{i=1}^{6} D_{io}}{6}\,\mu_d.$$

The annual additional mortality due to displacement for the population, m_d, is calculated by adding the number of casualties over all OWF areas in a certain scenario, and dividing by the maximum population abundance N_{max}, i.e.:

$$m_d = \frac{\sum_{o=1}^{n} \overline{C_o}}{N_{max}},$$

Here, the value of N_{max} is determined from the distribution maps as the predicted total number of birds on the map of the bimonthly period that has the highest overall abundance per species. The bimonthly period with the highest overall abundance was selected for each species and shown in Table 3-2. The value of m_d is specific for each scenario because of the different OWF areas included in the different scenarios.

4.1.5 Population models

The same type of population models as used within KEC 4 (Soudijn *et al.,* 2021) were used for the current assessment. These are stochastic stage-based matrix population models without density dependence. This means that the projected population abundance either increases or decreases in an exponential fashion, depending on whether the discrete-time population growth rate (λ) is larger or smaller than 1. The baseline (without additional mortality from habitat loss) population growth rates are reported per species in section 3.4.3. Several aspects of the population models were changed for the current assessment. Some of these changes follow from the recommendations associated with the revised ALI methodology (Hin *et al.,* 2024). In contrast to KEC 4, the effect of environmental stochasticity was simulated as annual stochastic variation in population parameters, instead of on a per-simulation basis (Soudijn *et al.*, 2022b). Therefore, population growth rates varied between years within a single projection (simulation), as opposed to sampled population growth rates that varied between projections, but not between years of the same projection. This adjustment has repercussions for the structure of the matrix models of the Sandwich tern, the common guillemot and the razorbill. For these species, several age-classes were combined into a single stage, under the assumption of a stable age structure. Because demographic parameter values now change within a simulation, this assumption no longer holds. Therefore, the relevant stages were split into different age-classes. This resulted in a matrix of seven stages for the common guillemot and six stages for the other three species (*Table 4-2*)**.**
Also, the effect of additional mortality from habitat loss was evaluated while keeping constant all other processes that affected the baseline population projection. This was implemented by, for each species, simulating 10^5 replicate series of baseline annual projection matrices ($\mathbf{A}$), with 40 annual time steps (matrices) for each series. The species-specific baseline matrices represent different realizations of environmental stochasticity (annual variation in population parameters) and were used to calculate 10^5 baseline projections per species. Subsequently, the baseline projection matrices were modified based on the mortality values calculated for the relevant species and the combination of a scenario and mortality level (section 4.1.4 and below). The modified projection matrices ($\mathbf{A_m}$) were then used to calculate the scenario projections, leading to 10^5 impacted projections per combination of species, scenario and mortality-level. Final population abundances were derived from both the baseline and impacted population projections and used to assess violations of the ALI threshold values. Because the baseline and impacted projections shared the same realization of environment stochasticity, the pairwise differences in final population abundance between an impacted and the baseline scenario were now entirely driven by habitat-loss induced mortality (Hin *et al.,* 2024).
Upon modifying the baseline annual projection matrices, the sampled mortality values were first adjusted based on stage-specific scaling factors that weigh the contribution of each life stage towards habitat loss induced mortality. For example, for species that spend the first few years of their life outside the North Sea (e.g. Sandwich tern), the mortality effects of habitat loss were only applied on the adult life stages. Table 4-1 shows the stable stage distributions and the stage-specific weighing factors (m_i with $i \in \{1, \dots, n\}$ and n the number of life stages) that were used for each species. For the northern gannet, these factors were derived from the assumption that around 75% of gannet in OWFs are adults (Potiek *et al.,* 2019). For razorbills and guillemots, we made the precautionary assumption that habitat loss effects are experienced by the adult life stages only. This is in line with the general observation that adult seabirds occupy the core of the distribution area, while less experienced juvenile and immature individuals are displaced towards less competitive, lower-quality areas (e.g. Lane *et al.,* 2021; Leopold & Overmaat, 2023).

***Table 4-1** stable stage distribution based on pre-breeding census and the stage-specific weighing factor of mortality effects of habitat loss*

Species	**Number of stages (n)**	**Stable stage distribution (pre-breeding)**	**Weighing factor per life stage (m_i with $i \in \{1, \dots, n\}$)**
northern gannet	6	[0.0; 0.11; 0.088; 0.075; 0.064; 0.66]	[0.904;0.904;0.904;0.904;1.036;1.036]
Sandwich tern	6	[0.0; 0.066; 0.052; 0.041; 0.039; 0.80]	[0.0; 0.0; 0.0; 1.134; 1.134; 1.134]
common guillemot	7	[0.0; 0.11; 0.080; 0.067; 0.061; 0.056; 0.63]	[0.0; 0.0; 0.0; 1.231; 1.231; 1.231; 1.231]
razorbill	6	[0.0; 0.13; 0.08; 0.073; 0.066; 0.66]	[0.0; 0.0; 1.143; 1.143; 1.143; 1.143]

To include stage- or age-specific mortality effects, we constructed an impact matrix ($\mathbf{M}$; *Table 4-2*) for each species with the stage-specific weighing factors (*Table 4-1*) in places where stage-transitions (survival) could be affected by mortality from habitat loss. For a sampled annual habitat loss mortality value m_d, each baseline annual projection matrix $\mathbf{A}$ was modified according to:

$$\mathbf{A_M} = \mathbf{A} \circ (1 - m_d \cdot \mathbf{M})$$

Where $\circ$ is the Hadamard product (element-wise multiplication). The resulting annual projection matrix $\mathbf{A_M}$ includes the mortality effect of habitat-loss. A single value of m_d was used for each series of 40 matrices. Note that the annual projection matrix $\mathbf{A}$ is derived from the matrix product of the winter and summer matrix (*Table 4-2*). This order of matrix multiplication results in a pre-breeding census model with zero abundance of the first life stage (*Table 4-1*). The effects on first-year survival are therefore combined with the reproduction term, which appears in the second row of the resulting annual projection matrix. Correspondingly, the effect on first-year survival also appears on the second row of the impact matrix.

Table 4-2 *Summer, winter and impact matrices for the different species. Parameters of summer and winter matrices are specified in Table 3-4 to Table 3-6. The annual projection matrix A is the matrix product of the winter matrix and the summer matrix. The weighing factors of the impact matrices (m_i) are given in Table 4-1.*

Species	Summer matrix (A_s)	Winter matrix (A_w)	Impact matrix (M)
northern gannet	$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_1 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_2 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_3 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & m_6 \end{pmatrix}$
Sandwich tern	$\begin{pmatrix} 1 & 0 & 0 & \phi F_A(1-P_F) & \phi F_A(1-P_F) & F_A(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_{12} & 0 & 0 & 0 & 0 \\ 0 & 0 & S_{12} & 0 & 0 & 0 \\ 0 & 0 & 0 & S_A & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & m_1 & m_1 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & m_6 \end{pmatrix}$
common guillemot	$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_A & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & m_6 & m_7 \end{pmatrix}$
razorbill	$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_{01} & 0 & 0 & 0 & 0 & 0 \\ 0 & S_{01} & 0 & 0 & 0 & 0 \\ 0 & 0 & S_A & 0 & 0 & 0 \\ 0 & 0 & 0 & S_A & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & m_6 \end{pmatrix}$

4.1.5.1 Population parameters

Parameters of population models are shown in Table 3-4 to Table 3-7. When data availability allowed, these parameters were based on demographic data from the period 2016-2022 from colonies that were considered most relevant for the birds occupying the Southern North Sea (see section 3.4.3).

4.1.6 Acceptable Levels of Impact

For the current KEC 5 assessment, we used the revised 'Acceptable Level of Impact' (ALI) methodology (Hin *et al.,* 2024). Violation of the ALI was calculated based on the relative difference in population abundance between an impacted scenario and its baseline counterpart. For each of 10^5 pairwise projections the relative difference in final population abundance was calculated as:

$$\Delta = \frac{N_{40}^0 - N_{40}^I}{N_{40}^0}$$

With N_{40}^0 and N_{40}^I being, respectively, the baseline and impacted population abundance after 40 simulated years. Because of the stochastic nature of both the modelled impact and the baseline population dynamic, the result was a distribution of Δ. From the distribution of Δ, the fraction that exceeded the ALI X threshold was calculated as $P_{VI} = P(\Delta > X_{40})$. The ALI was violated if this fraction exceeded the ALI Y threshold, i.e. if

$P_{VI} > Y$. The ALI tests were conducted for all national scenarios but not for the international scenario, because the uncertainty could not be properly accounted for in this scenario.

Threshold values for the revised ALI methodology were supplied by LVVN. The threshold values for X denote the maximum acceptable percentage reduction in population abundance between the unimpacted (without OWF effects) and impacted (with OWF effects) population trajectories. The value of X varies between species and depends on an assessment performed by Sovon of the species' conservation status, which largely followed the Dutch conservation status (*Staat van Instandhouding*), but was also informed by the status according to OSPAR and the IUCN, and the impact of avian influenza. Species for which the assessment resulted in an unfavourable conservation status, an X threshold value of 5% was adopted. If conservation status was favourable, an X threshold value of 15% was chosen. These X threshold values were defined for a time period equal to the maximum of 10 years or three times the generation time of the species. All X threshold values were converted to a period of 40 years for the current assessment, using the formula in Hin *et al.,* (2024). Generation time was calculated as the "*mean age of the parents of the offspring produced by a cohort over its lifetime*" (see Caswell 2001), which depends on species-specific estimates of maximum age (*Table 4-3*). The ALI Y threshold value determined the maximum allowable probability of a population reduction larger than X and was set to 0.05 for all species (*Table 4-3*).

Table 4-3: *Threshold values for the Acceptable Level of Impact methodology as supplied by LVVN. Estimates of maximum age are taken from Birds of the World (2022) and used for calculating generation time.*

Species	**X threshold max(10, 3x T_G)**	**Generation time (T_G)**	**Maximum age (yrs)**	**X threshold (40 yrs)**	**Y threshold**
northern gannet	5%	14.2	30	4.71%	0.05
Sandwich tern	5%	13.7	30	4.88%	0.05
common guillemot	15%	18.8	44	10.9%	0.05
razorbill	15%	13.9	40	14.4%	0.05

4.1.7 Uncertainty propagation

We explicitly account for uncertainty in various steps of the casualty calculation. The uncertainty in the national distribution maps was captured by generating 1000 sets of six bimonthly density maps from the estimated distributions and for each set the value of N_{max} was determined as described above. In addition, we used beta distributions for the displacement probability ρ, with species-specific parameters as indicated in *Table 3-3* (see also Chapter 3, *Figure 3-1*). For each species, the thousand sets of six bimonthly maps were combined with the distributions of displacement probability and the four discrete levels of annual displacement mortality (μ_d; *Table 3-2*). This resulted in four distributions of m_d per species and scenario. Based on this uncertainty in the displacement mortalities, population models were run with their uncertainties in the population parameters, resulting in full uncertainty propagation for all model parameters, except for the effect distance parameters ϑ and ε. Due to the limited timeframe of the current project, it proved difficult to adjust the methodology to such an extent that the uncertainty in this parameter could be taken into account. Moreover, for the international scenarios we only had a single mean distribution map available, and we could therefore not account for the uncertainty in the international bird distributions. For the international scenario, only the uncertainty in the displacement matrix and population parameters has been accounted for.

4.2 Results

4.2.1 OWFs and bird distributions

For each of the four species, two sets of maps are presented. The first set shows the average distribution by two-month periods, data are visualized on a 10 x 10 km grid, including both an average and a maximum density across the six bimonthly periods (*Figure 4-2, Figure 4-4, Figure 4-6* and *Figure 4-8*). The second set shows maps of the international southern North Sea, with monthly average distribution for three species, and for Sandwich Tern, a bimonthly distribution (*Figure 4-3*, *Figure 4-5*, *Figure 4-7* and *Figure 4-9*). Each set of maps includes wind farm locations, with species-specific effect distances around the wind farms where displacement effects are anticipated (*Table 3-2*).

4.2.1.1 Northern gannet

In the Dutch North Sea, northern gannets exhibit the widest distribution during the post-breeding migration period (August–September and October–November). In winter, they are limited to the southern area and occur at very low densities (*Figure 4-2*). During the summer and migration seasons, the highest densities are seen in the western part of the Dutch North Sea, likely due to proximity to large colonies, while Helgoland’s colony with its limited size has little impact. Overlap with planned offshore wind farms (OWFs) mainly occurs far offshore in the western area. Internationally, distribution maps reflect the concentration of northern gannets near colonies on Scotland's east coast, particularly at Bass Rock, the largest colony (*Figure 4-3*). Consequently, high-density areas do not overlap with Dutch OWFs but are mainly found in UK waters.

4.2.1.2 Sandwich tern

Of the four species studied, the Sandwich tern has the most coastal distribution, both nationally (*Figure 4-4*) and internationally (*Figure 4-5*). Its presence closely follows breeding colony locations, which in the Netherlands are situated in the Delta region, along the North Holland coast, and on the Wadden Islands. Other North Sea nations also host colonies, mostly in major estuaries on the UK’s east coast and the German and Danish coastlines, similar to Dutch Wadden Island locations. Overlap with OWFs is thus primarily with those located closer to shore, Sandwich terns are no true seabirds as they return to land at night to rest.

4.2.1.3 Common guillemot

The common guillemot is the most abundant seabird in the Dutch North Sea. It breeds in large colonies along the UK and Ireland cliff coasts, with post-breeding migration bringing adults and young to Dutch waters, where the latter arrive swimming alongside parent birds across hundreds of kilometres. Numbers in Dutch waters rise in July (*Figure 4-6*), consisting of arriving juveniles accompanied with their parents, swimming from the colonies on the east coast of the UK. The non-breeding season distributions are generally widespread, resulting in substantial overlap with OWFs, though the highest densities in Dutch waters are found in northern areas where no wind farms are currently planned. Internationally, guillemot distributions during the summer months are closely tied to colonies on the UK’s east coast, and they stay relatively close to shore, exhibiting a much smaller foraging range from the colonies than northern gannets (*Figure 4-7* and *Figure 4-3*). By August, their distribution broadens, with peak densities in the northern international North Sea, an area that, like northern gannets, contains the largest colonies and can also be frequented by birds from the Norwegian coast in winter.

4.2.1.4 Razorbill

The razorbill is far less abundant than the guillemot and is almost absent from the Dutch North Sea between April and September (*Figure 4-8*). Unlike the guillemot, which can reach high numbers in this period, the razorbill's distribution shifts, with peak densities occurring relatively more south and west within the Dutch sector. This results in greater overlap with Dutch OWFs, though razorbill densities remain lower than those of guillemots. Internationally, highest razorbill densities are found along the UK’s east coast, particularly post-breeding (*Figure 4-9*). Their distribution then becomes more diffuse across the North Sea, lacking specific concentration zones.

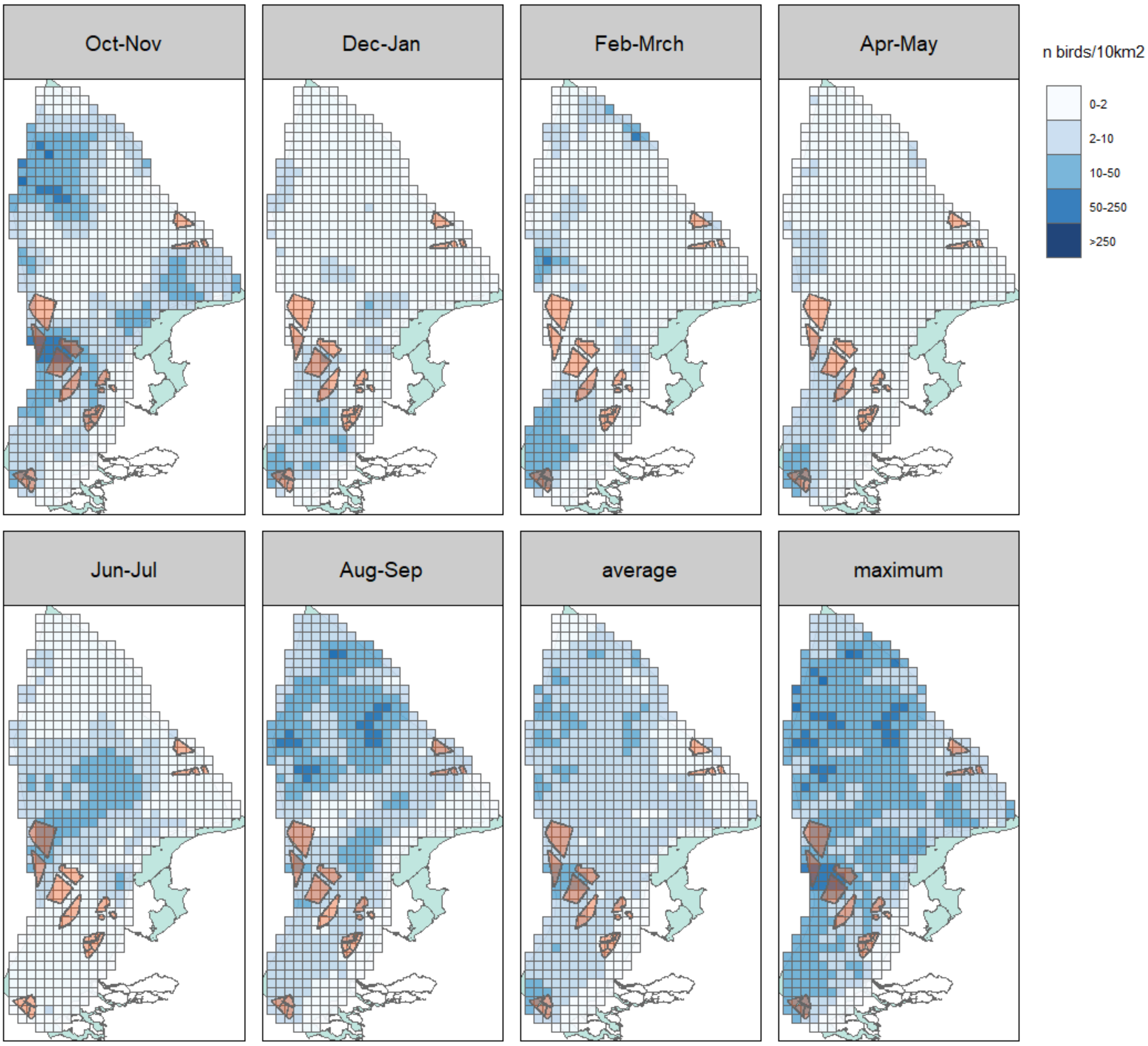

Figure 4-2 *Average distribution pattern of northern gannet per bimonthly period, on a 10x10 km^2 resolution, annual average and annual maximum over the period 2016-2020 (van Donk et al., 2024) for the Dutch part of the North Sea and the overlay of off shore wind farm search areas (orange) with a displacement impact buffer area (yellow, but hardly visible at this scale) based on effect distances published by Vanermen et al., (2015).*

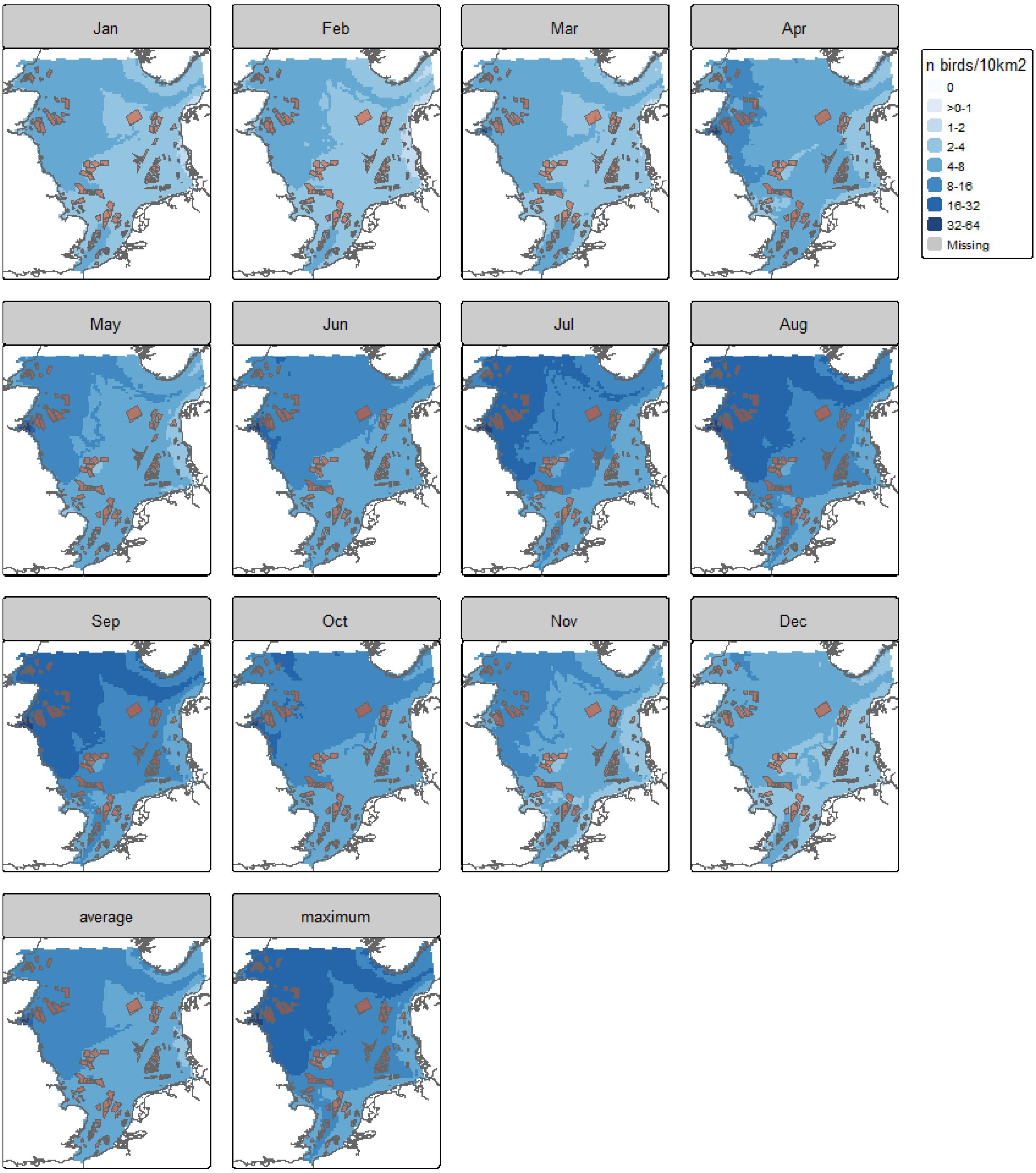

Figure 4-3 *Average distribution pattern of northern gannet per month, on a 10x10 km^2 resolution, annual average and annual maximum for the period 1980-2018 (Waggitt et al., 2020) of the international KEC area and the overlay of offshore wind farm search areas (orange) with a displacement impact buffer area (yellow, but hardly visible at this scale) based on effect distances published by Vanermen et al., (2015).*

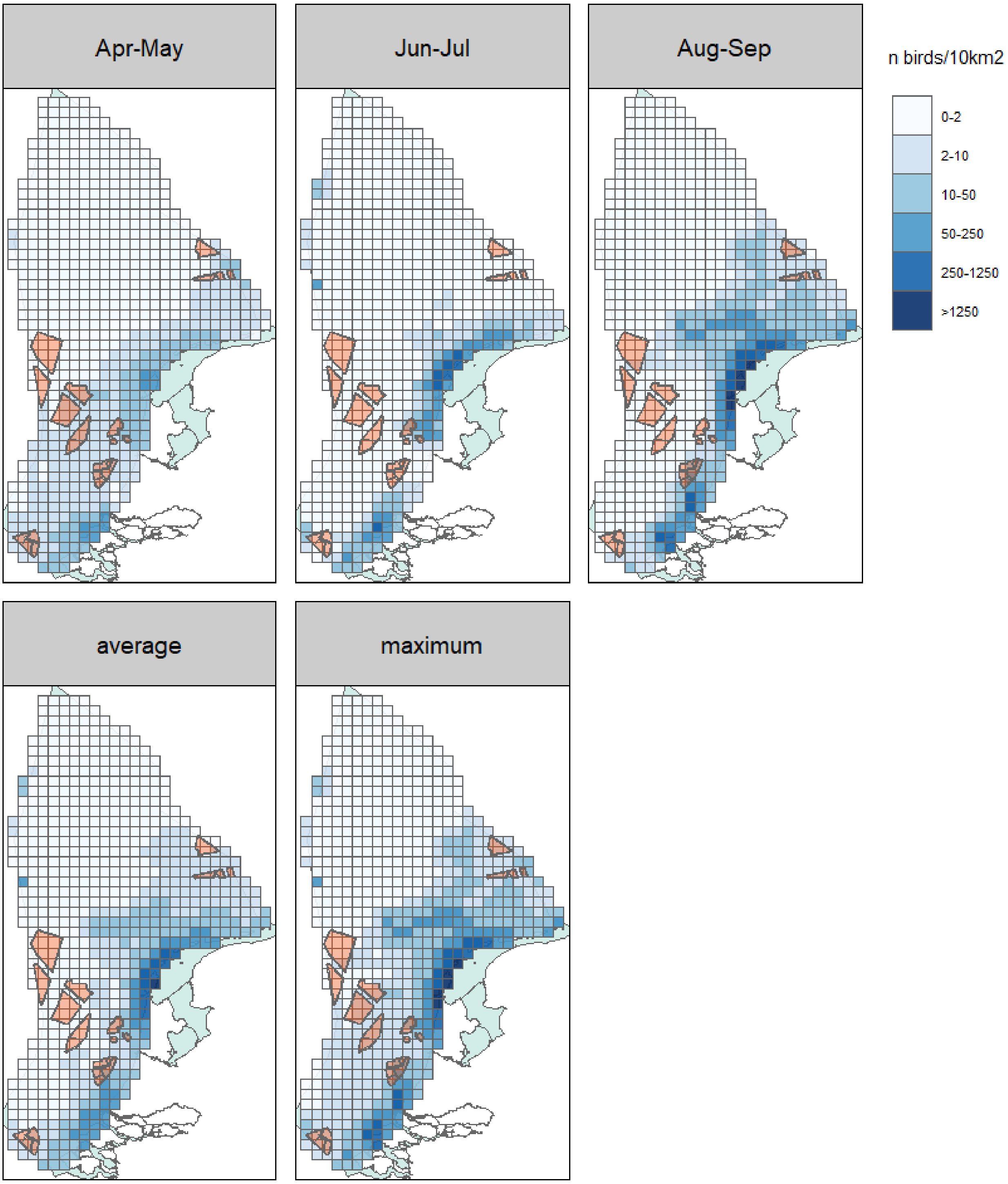

Figure 4-4 *Average distribution pattern of Sandwich tern per bimonthly period, on a 10x10 km² resolution, annual average and annual maximum over the period 2016-2020 (van Donk et al., 2024) for the Dutch part of the North Sea and the overlay of off shore wind farm search areas (orange) with a displacement impact buffer area (yellow, but hardly visible at this scale) based on effect distances published by Welcker & Nehls (2016).*

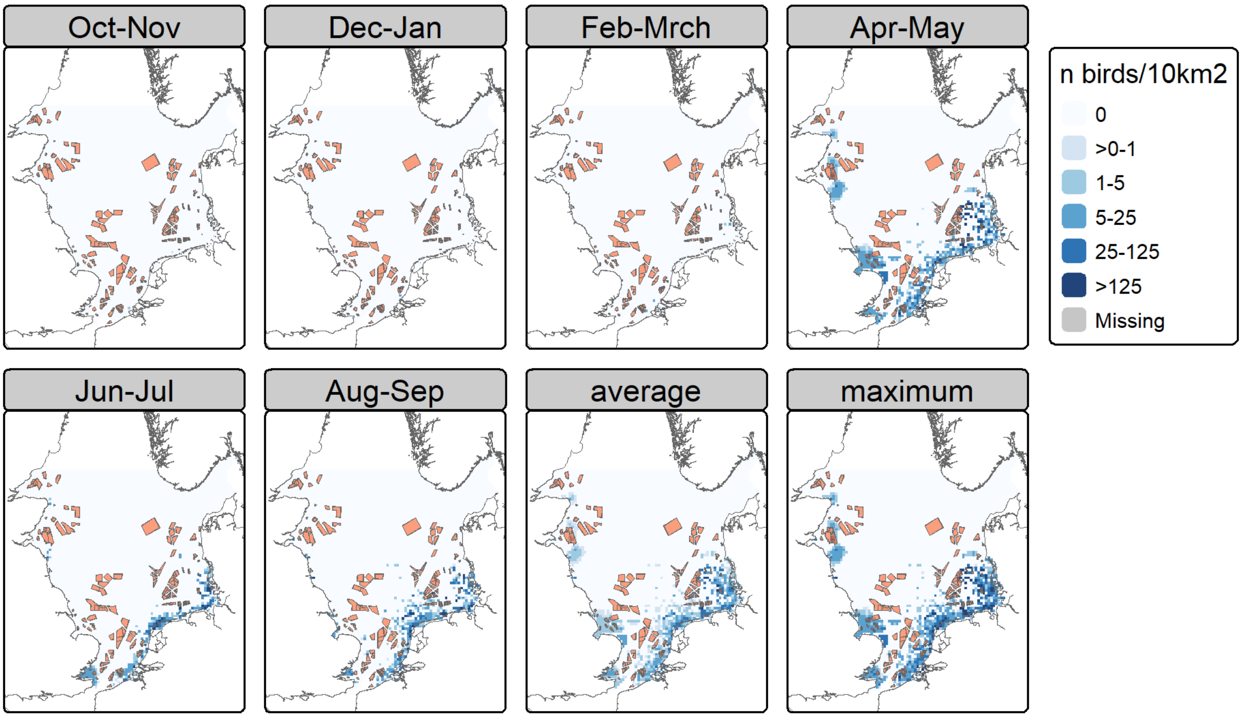

Figure 4-5 *Average distribution pattern of Sandwich tern per bimonthly period, on a 10x10 km*^{2} *resolution, annual average and annual maximum for the period 1980-2018 (Waggitt et al., 2020) of the international KEC area and the overlay of offshore wind farm search areas (orange) with a displacement impact buffer area (yellow, but hardly visible at this scale) based on effect distances published by Welcker & Nehls (2016) . Note that the techniques used to make this map are different from the other international maps (Inverse Distance Weighting).*

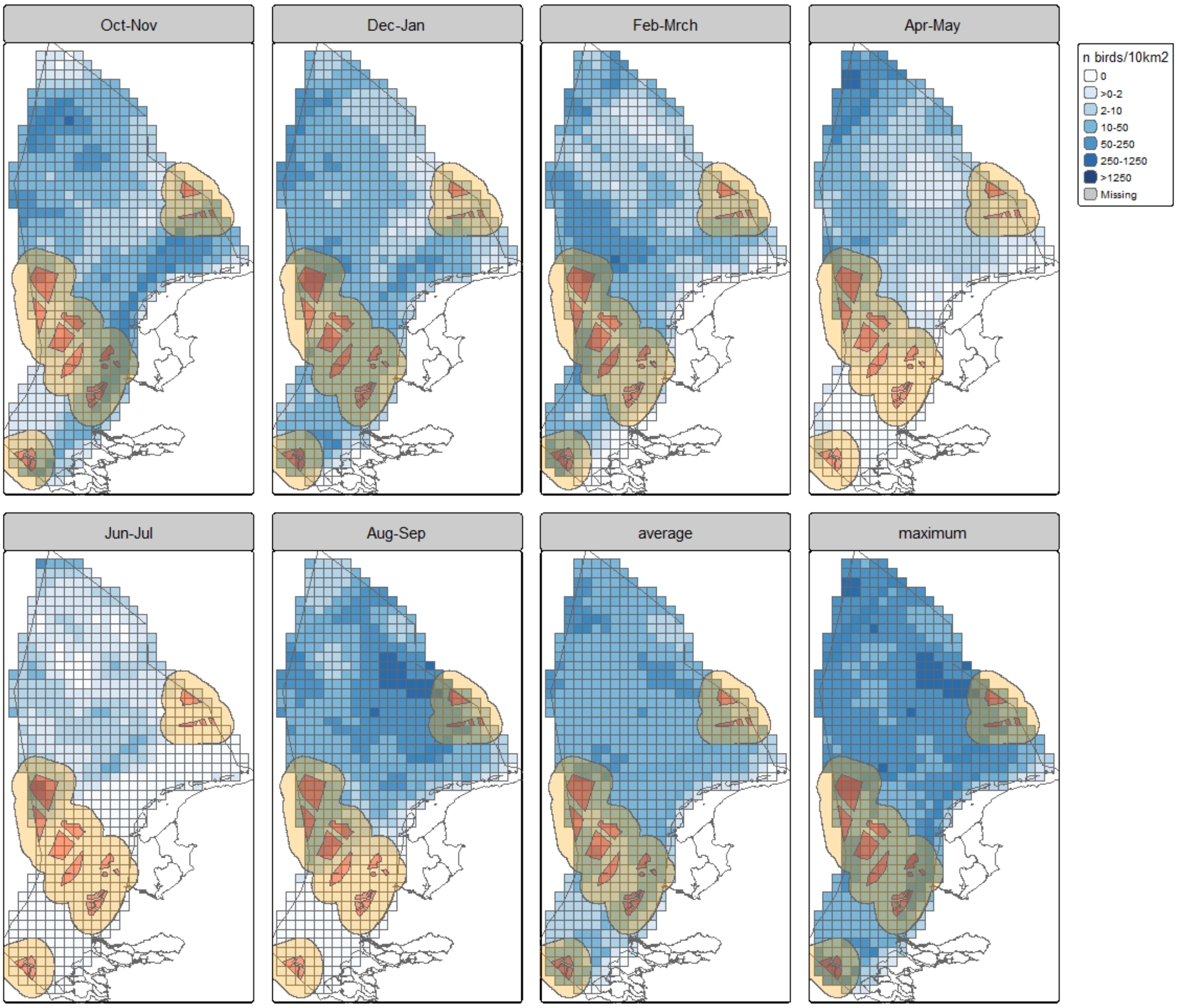

Figure 4-6 *Average distribution pattern of common guillemot per bimonthly period, on a 10x10 km^2 resolution, annual average and maximum for the period 2016-2020 for the Dutch part of the North Sea* (van Donk *et al.*, 2024) *and the overlay of offshore wind farm search areas (orange) for the Total scenario with the displacement impact buffer area (yellow) based on effect distances published by Peschko et al., (2024).*

Figure 4-7 *Average distribution pattern of common guillemot per month, on a 10x10 km^2 resolution, annual average and maximum for the period 1980-2018 (Waggitt et al., 2020) of the international KEC area and the overlay of off shore wind farm search areas (orange) with a displacement impact buffer area (yellow) based on effect distances published by Peschko et al., (2024).*

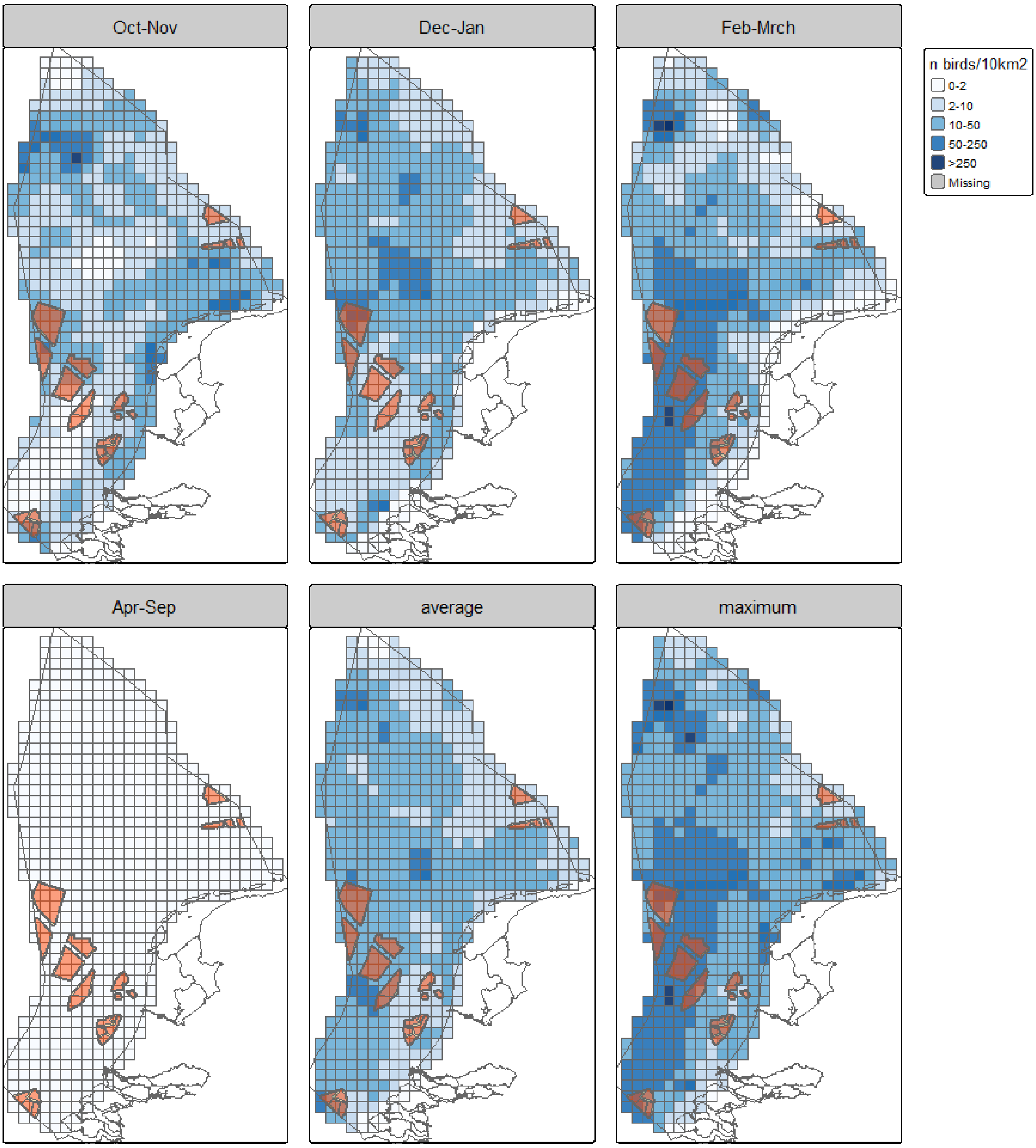

Figure 4-8 *Average distribution pattern of razorbill per bimonthly period, on a 10x10 km^2 resolution, annual average and annual maximum for the period 2016-2020 for the Dutch part of the North Sea (van Donk et al., 2024) and the overlay of offshore wind farm search areas for the Total scenario (orange) with a displacement impact buffer area (yellow, but hardly visible at this scale) based on effect distances published by Grundlehner & Leopold (2024).*

Figure 4-9 *Average distribution pattern of razorbill per month, on a 10x10 km^2 resolution, annual average and annual maximum for the period 1980-2018 (Waggitt et al., 2020) of the international KEC area and the overlay of offshore wind farm search areas (orange) with a displacement impact buffer area (yellow, but hardly visible at this scale) based on effect distances published by Grundlehner & Leopold (2024).*

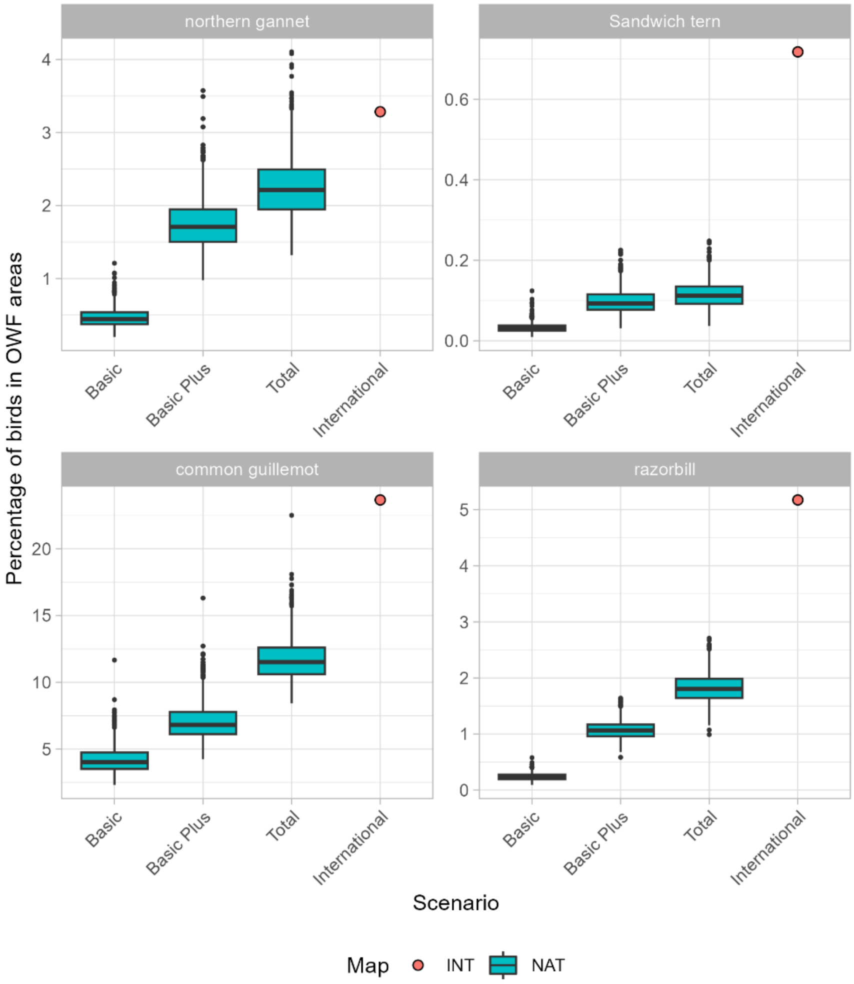

Figure 4-10 *Percentage of birds in offshore wind farm areas, including species-specific buffer zones for the different scenarios. For the national scenarios, the percentages are depicted as distributions (boxplots) derived from 1000 sampled density maps for each bimonthly period in 2016-2020 (van Donk et al., 2024) and are relative to the average maximum total number in the Dutch part of the North Sea of the bimonthly period with the maximum abundance. For the international scenario only a single map is used, resulting in an average point estimate over 12 months in 1980-2018 (Waggitt et al., 2020). Note the different scaling of the vertical axis per panel.*

4.2.2 Overlap with OWFs

The percentage of birds in offshore wind farm effect areas (including the effect distance around OWFs) varies considerably between species (*Figure 4-10*). The mean overlap between birds and OWFs was found to be the highest for the common guillemot, ranging from 4.2% - 24% depending on the scenario and the map that was used (national vs international), and lowest for the Sandwich tern, ranging from 0.033% - 0.72% (*Figure 4-10* and *Table A-3*). For the other two species, overlap estimates for the national scenarios are all under 4%. Note that the variation displayed stems from variation in the bird distribution maps only.

The international scenario is calculated based on a single map and therefore represented by a point estimate. For all four species, the estimates calculated for the international scenario are well above the mean percentages for the national scenarios. Based on the international map, 24% of the common guillemots are located within offshore wind farm-affected areas (including the effect distance around OWFs), while for the national 'Total' scenario, the mean percentage of guillemots in OWF-affected areas equals 12%, with 5% and 95% quantile values of respectively 10% and 15%.

4.2.3 Mortality estimations

4.2.3.1 National scenarios

The additional annual mortality resulting from displacement by offshore wind farms was calculated using the displacement matrix approach (section 4.1.4). The result is a distribution for the additional annual mortality from displacement for each mortality level. The distributions for the national scenarios are shown per species in Figure 4-11 to Figure 4-14 and values of the mean and standard deviations of the distributions are shown in Table A-5. The estimated mortality rate was found to be lowest for the Basic scenario and highest for the Total scenario. The estimated mortality rate is highest for the common guillemot, then for the northern gannet, then for the razorbill and lowest for the Sandwich tern. For the national Total scenario and a mortality level of 10%, the estimated mean annual additional mortality due to displacement was 0.0013 (~0.13%) for the northern gannet, 0.00006 (~0.006%) for the Sandwich tern, 0.0093 (~0.93%) for the common guillemot and 0.00078 (~0.078%) for the razorbill.

4.2.3.2 International scenario

The distribution for the international scenario is shown separately in Figure 4-15 (see also *Table A-5*). This distribution only reflects variation from displacement probability distributions, as only a single mean map per period was used for each species. The resulting mean mortality estimates are higher for the international scenario compared to any of the national scenarios.

The estimated mortality rate is highest for the common guillemot, then for the razorbill, then the northern gannet and lowest for the Sandwich tern. Again, the highest mean mortality of 0.019 (1.9%) was estimated for the common guillemot in the international scenario with a 10% mortality level. For the same mortality level, this was 0.0018 (0.18%) for the northern gannet, 0.00036 (0.036%) for the Sandwich tern and 0.0022 (0.22%) for the razorbill. Note that the mean background adult mortality probability of the guillemot in the population model is set to 0.051. A mean additional mortality of 0.019 is hence equal to 37% of the annual mortality probability of adult common guillemots with no OWFs.

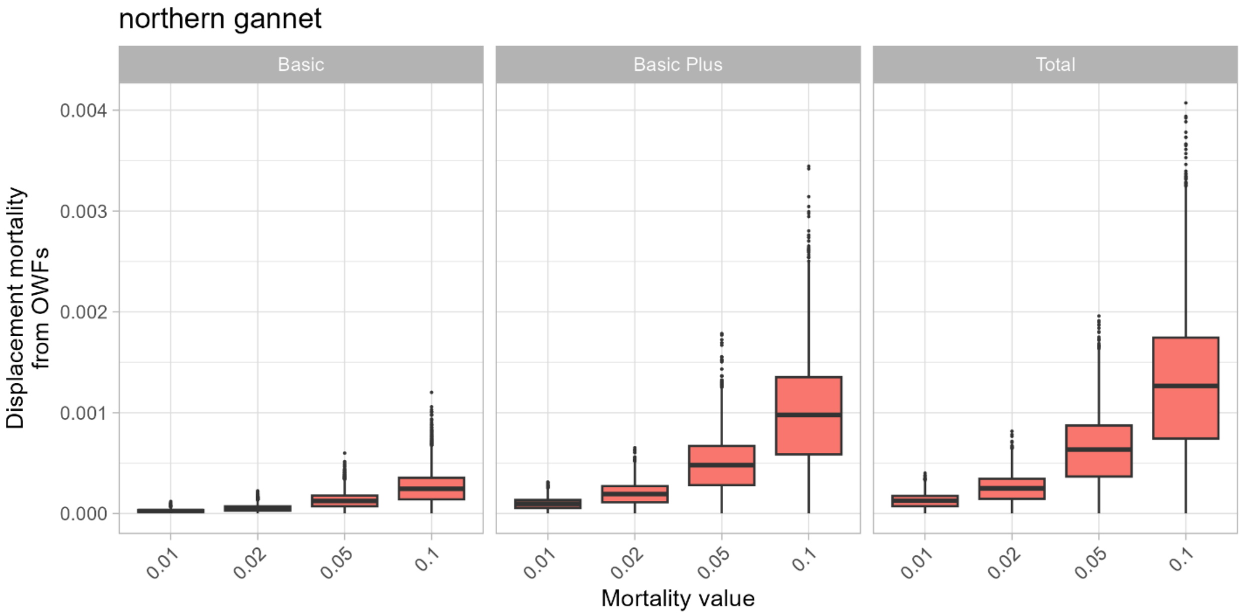

Figure 4-11 *Additional annual mortalities due to displacement from OWF areas for the northern gannet for the three national scenarios.*

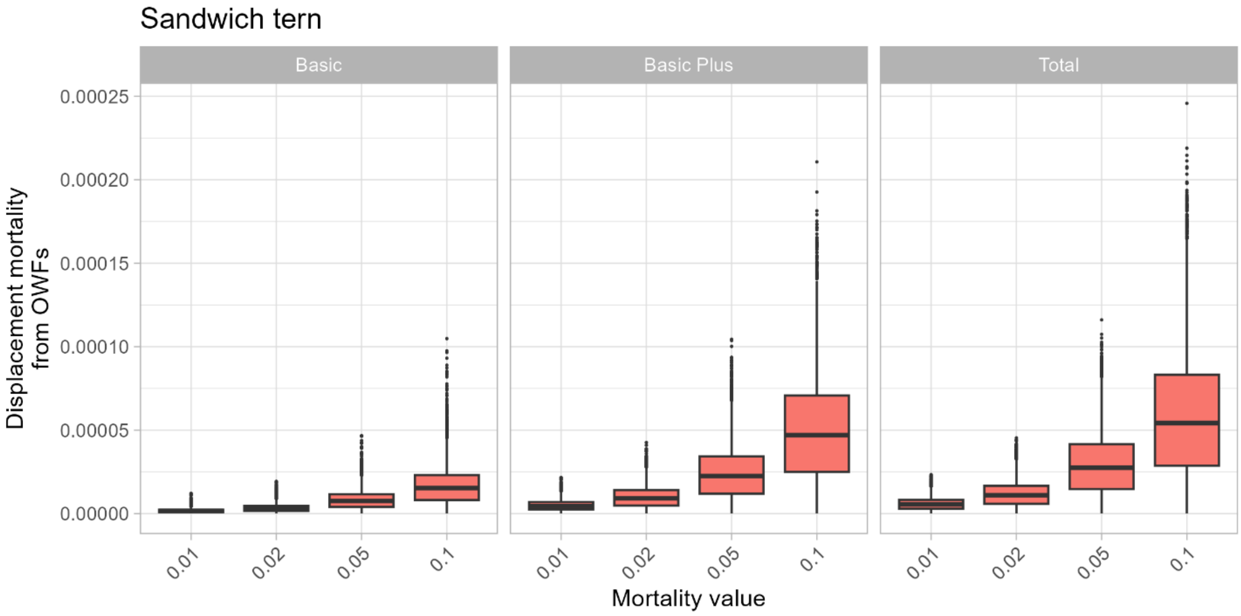

__Figure 4-12__ Additional annual mortalities due to displacement from OWF areas for the Sandwich tern for the three national scenarios.

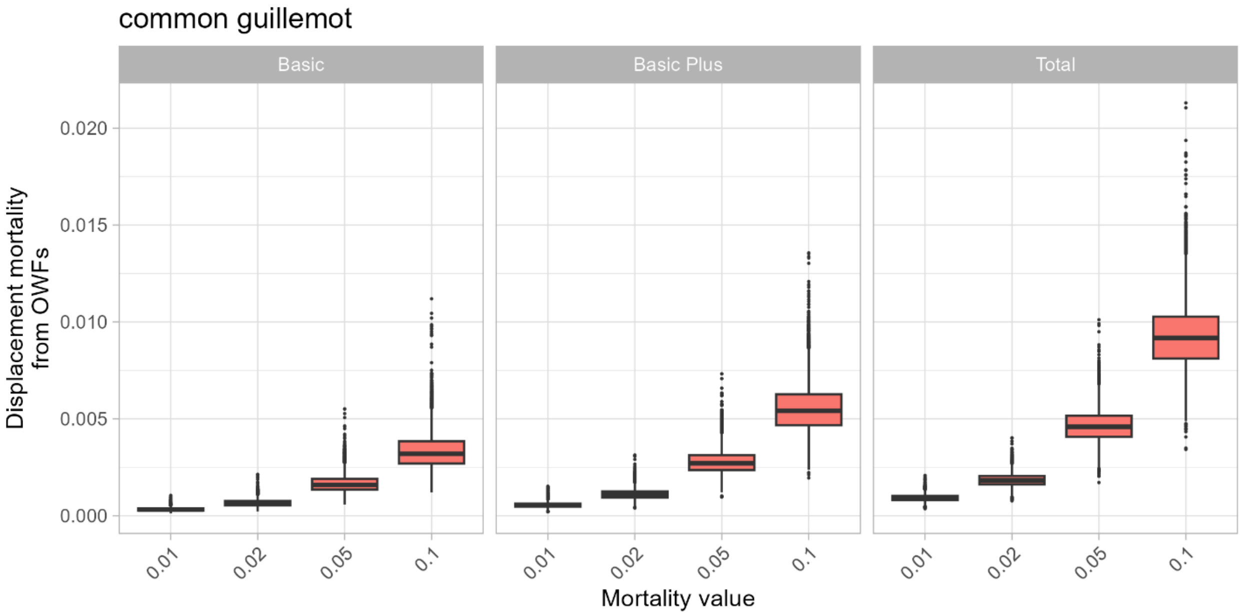

__Figure 4-13__ Additional annual mortalities due to displacement from OWF areas for the common guillemot for the three national scenarios.

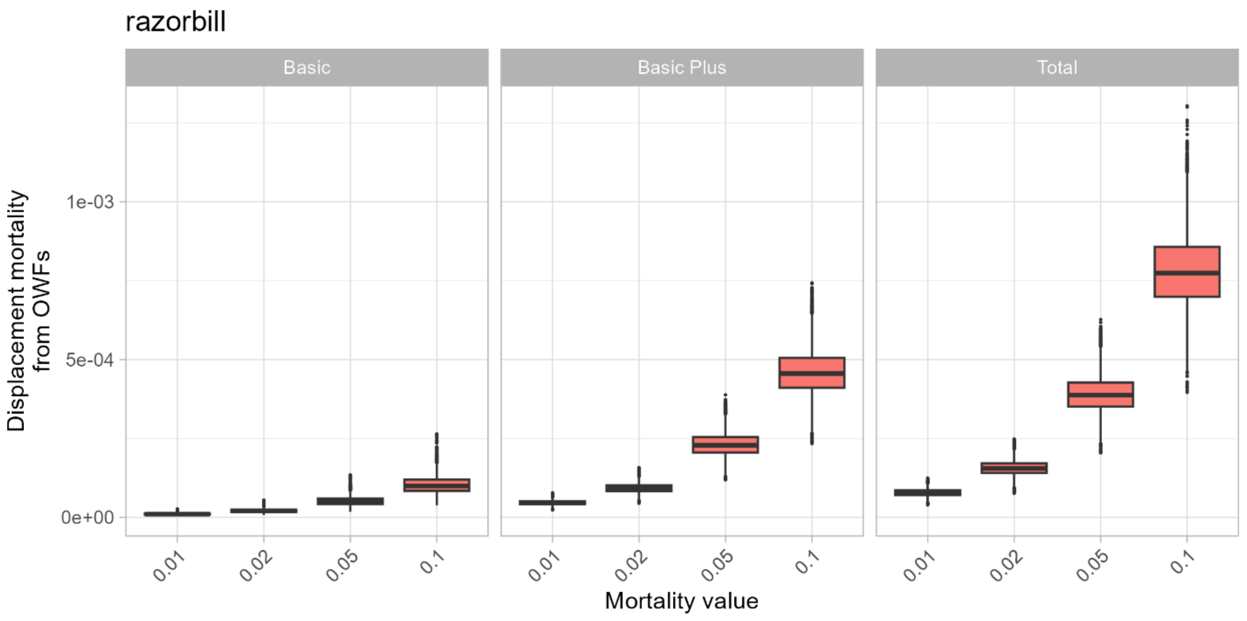

***Figure 4-14** Additional annual mortalities due to displacement from OWF areas for the razorbill for the three national scenarios.*

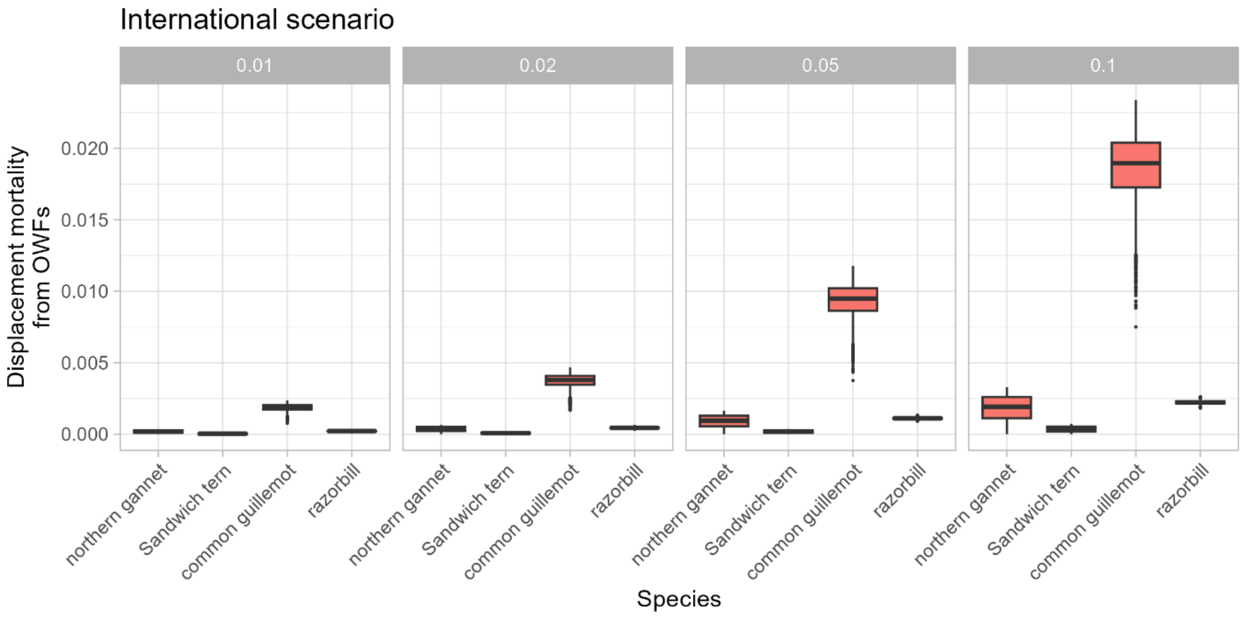

***Figure 4-15** Additional annual mortalities due to displacement from OWF areas for the international scenario per species and mortality level.*

4.2.4 Population models and ALI outcome

4.2.4.1 National scenarios

For the razorbill (*Figure 4-19*) and Sandwich tern (*Figure 4-17*), the population projections show little effect of the estimated additional mortality from displacement for the national scenarios. The projections of the Sandwich tern population abundance with no OWFs show a declining trend with a growth rate of 0.986 (~1.4% decline per year; *Table A-6*). Over 40 simulated years, this leads to a mean reduction in population abundance of 43% with no OWFs. For the northern gannet (*Figure 4-16*), population effects of displacement mortality are apparent for the Total scenario at higher mortality levels (5% and 10%). For the Null scenario, the mean gannet population growth rate was estimated at 1.032 (*Table A-6*). For the Total scenario at the 10% mortality level, the mean annual population growth rate reduces to 1.030, which results in a mean population abundance after 40 simulated years that is about 4.9% lower than for the Null scenario (*Table*

A-7). For the common guillemot (*Figure 4-18*), population effects are apparent, especially at higher mortality levels and scenarios with more OWFs. The projections of guillemot population abundance for the Null scenario show a mean population growth of 1.034 per year, which results in a population that is, on average, 3.75 times larger after 40 years (*Table A-6*). For the Total scenario and the 10% mortality level, the mean annual population growth rate reduces to 1.024 and the population increases by a factor of 2.58 over a time period of 40 years (*Table A-6*). This results in a mean population abundance after 40 simulated years that is approximately 31% lower for the Total scenario than the final population abundance under the Null scenario with no OWFs.

For the national scenarios, the population effects of displacement from OWFs were tested against ALI methodology, using the ALI threshold values as defined by LVVN (*Table 4-3*). No ALI violations were estimated for the Sandwich tern and the razorbill. The population effects of OWFs were predicted to exceed the ALI for the northern gannet and the common guillemot (*Figure 4-20*). For the northern gannet, ALI violations occur for the Total scenario and for the Basic plus scenario at a mortality level of 10%. For the common guillemot, ALI violations were found for each national scenario, but not at all mortality levels. In the Basic scenario, we found a 68% probability of exceeding the ALI X threshold for the highest mortality level (10%). The probabilities for exceeding the ALI X threshold were estimated below 5% for the other mortality levels (*Table A-7*). For the Basic plus scenario and the Total scenario, mortality levels of 5% and 10% were found to violate the ALI for the common guillemot (*Table A-7*). The probability of violating the X threshold equalled 100% under the Total scenario and the highest mortality level (10%). This implies that the distribution of the relative difference in final population abundance between the scenario and the Null scenario is completely above the ALI X threshold value.

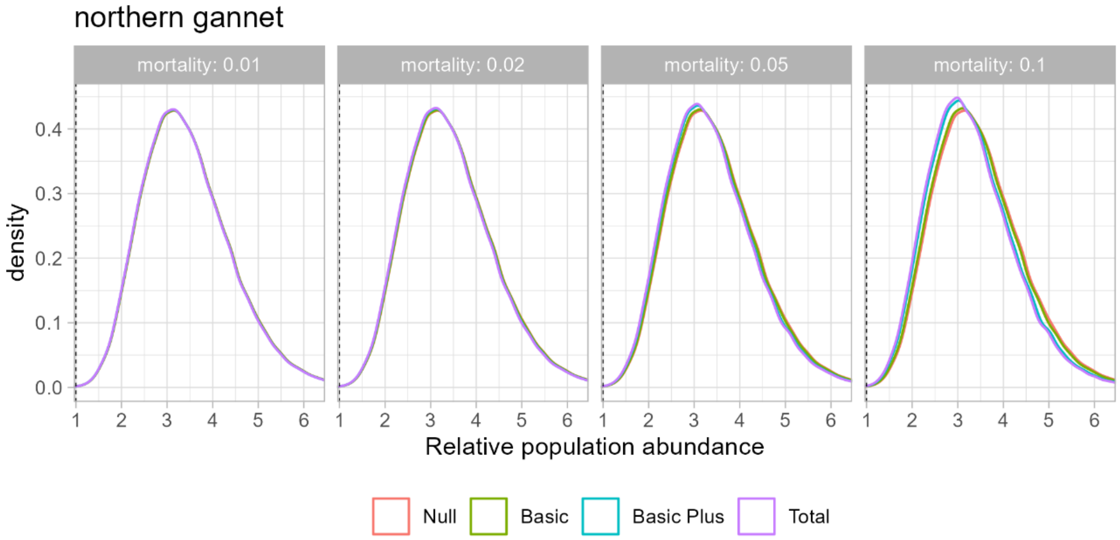

__Figure 4-16__ Probability distributions of population abundances after 40 simulated years, relative to the initial mean population abundance, for the northern gannet and the national OWF scenarios. Different panels show different mortality probabilities used in calculating the mortality effect of displacement. The distributions for the Null scenario are identical between panels. Each distribution is scaled such that mean initial population abundance equals 1 (vertical dashed line).

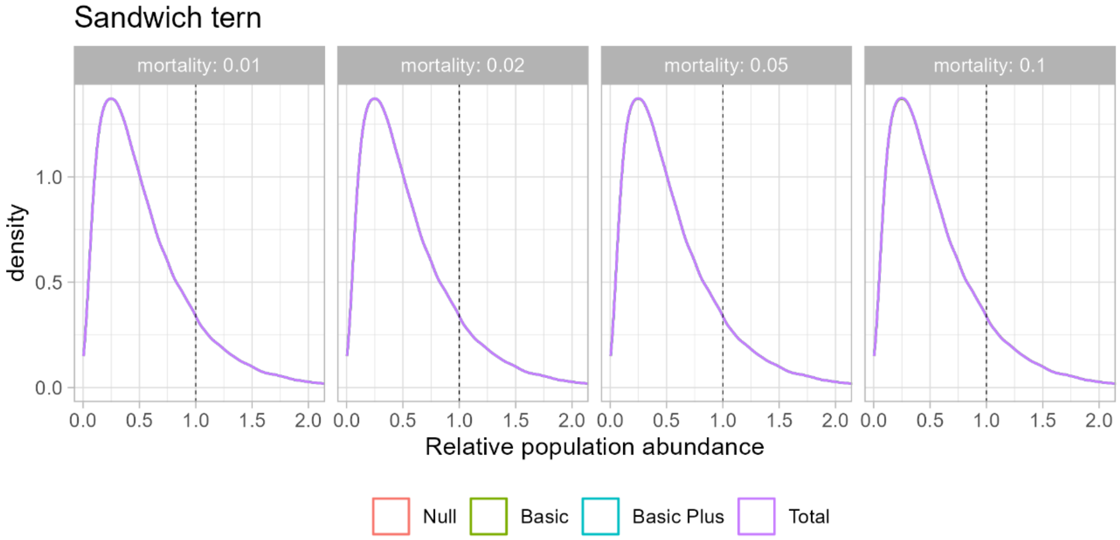

__Figure 4-17__ Probability distributions of population abundances after 40 simulated years, relative to the initial mean population abundance, for the Sandwich tern and the national OWF scenarios. Different panels show different mortality probabilities used in calculating the mortality effect of displacement. The distributions for the Null scenario are identical between panels. Each distribution is scaled such that mean initial population abundance equals 1 (vertical dashed line). Note that all lines are situated almost exactly on top of each other in these figures.

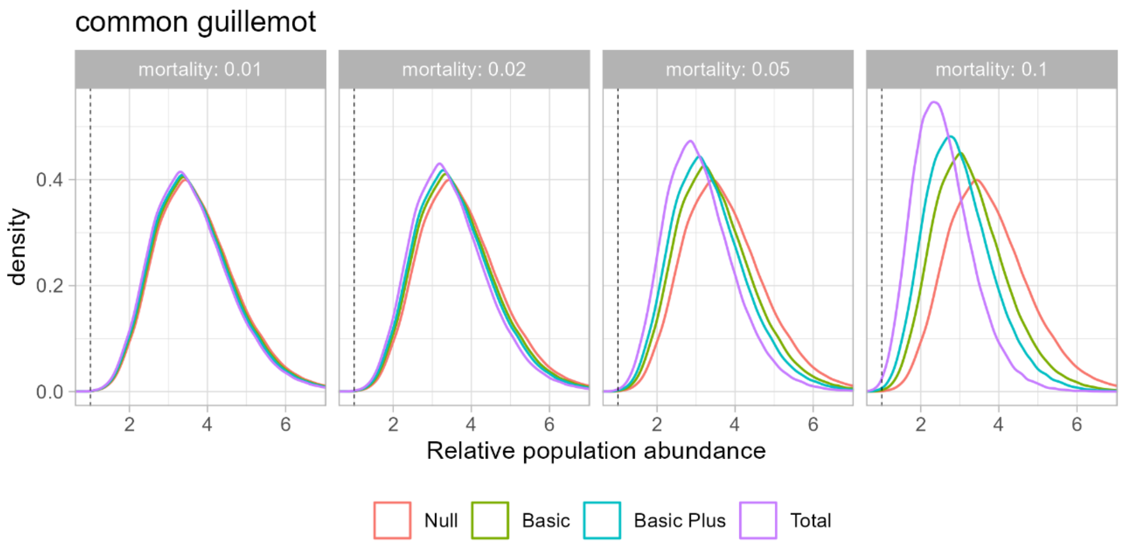

Figure 4-18 *Probability distributions of population abundances after 40 simulated years, relative to the initial mean population abundance, for the common guillemot and the national OWF scenarios. Different panels show different mortality probabilities used in calculating the mortality effect of displacement. The distributions for the Null scenario are identical between panels. Each distribution is scaled such that mean initial population abundance equals 1 (vertical dashed line).*

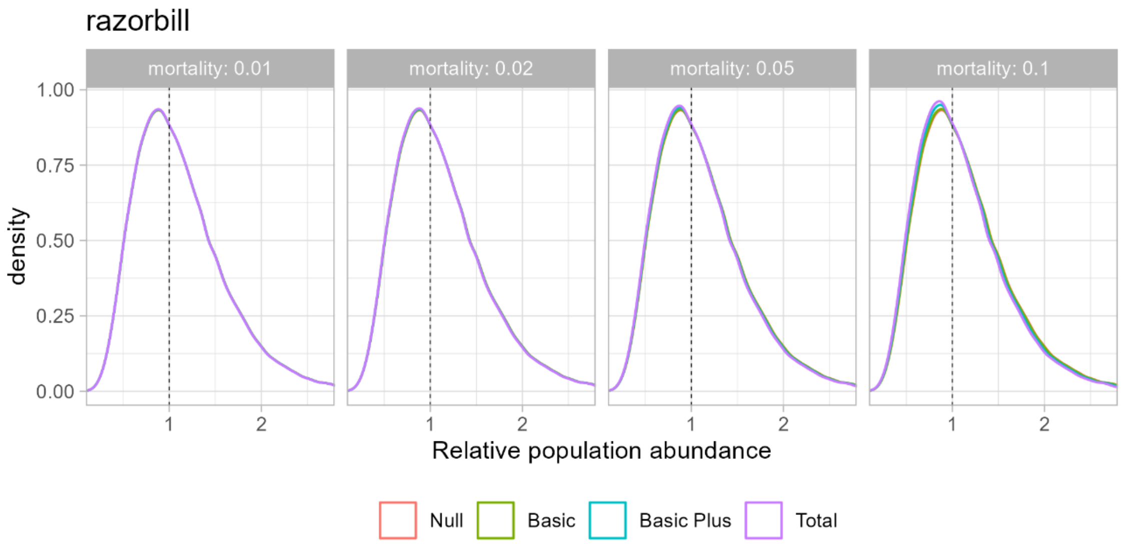

Figure 4-19 *Probability distributions of population abundances after 40 simulated years, relative to the initial mean population abundance, for the razorbill and the national OWF scenarios. Different panels show different mortality probabilities used in calculating the mortality effect of displacement. The distributions for the Null scenario are identical between panels. Each distribution is scaled such that mean initial population abundance equals 1 (vertical dashed line).*

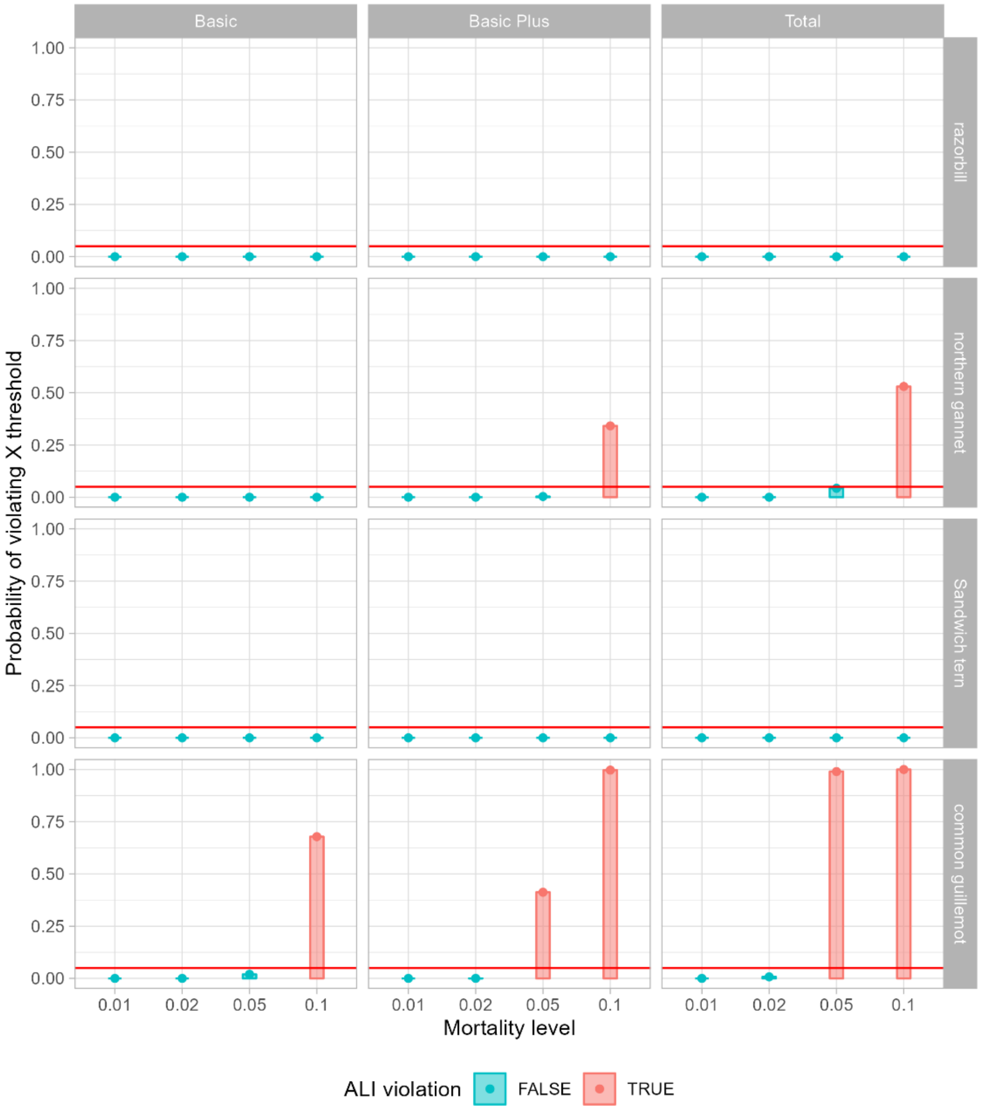

Figure 4-20 *Probabilities of violating the ALI X thresholds per species (row panels), scenario (column panel) and mortality level (horizontal axis) for the national scenarios. An ALI violation occurs if the probability of violating the ALI X threshold, exceeds the ALI Y threshold, 0.05 (horizontal red lines). Violations of the ALI threshold are indicated with red coloured bars, while no ALI violation occurs for blue coloured bars.*

4.2.4.2 International scenario

For the international scenario, population effects of OWF mortality from displacement are apparent for the razorbill and the northern gannet at high mortality levels. For the Sandwich tern, the population effects are smaller, with at maximum a 1.4% reduction in final population abundance relative to the Null scenario (*Table A-6*). The estimated population effects for common guillemot are substantial, especially for higher mortality levels (*Figure 4-21*). For the 10% mortality level, the mean annual population growth rate is reduced from 1.034 (Null scenario) to 1.014 (International scenario)(*Table A-6*). Although the mean estimated population trend of the guillemot is still increasing even for the highest mortality level, the mean projected population abundance after 40 years is around 53% lower for the International scenario at the 10% mortality level compared to the projected population abundance for the Null scenario.

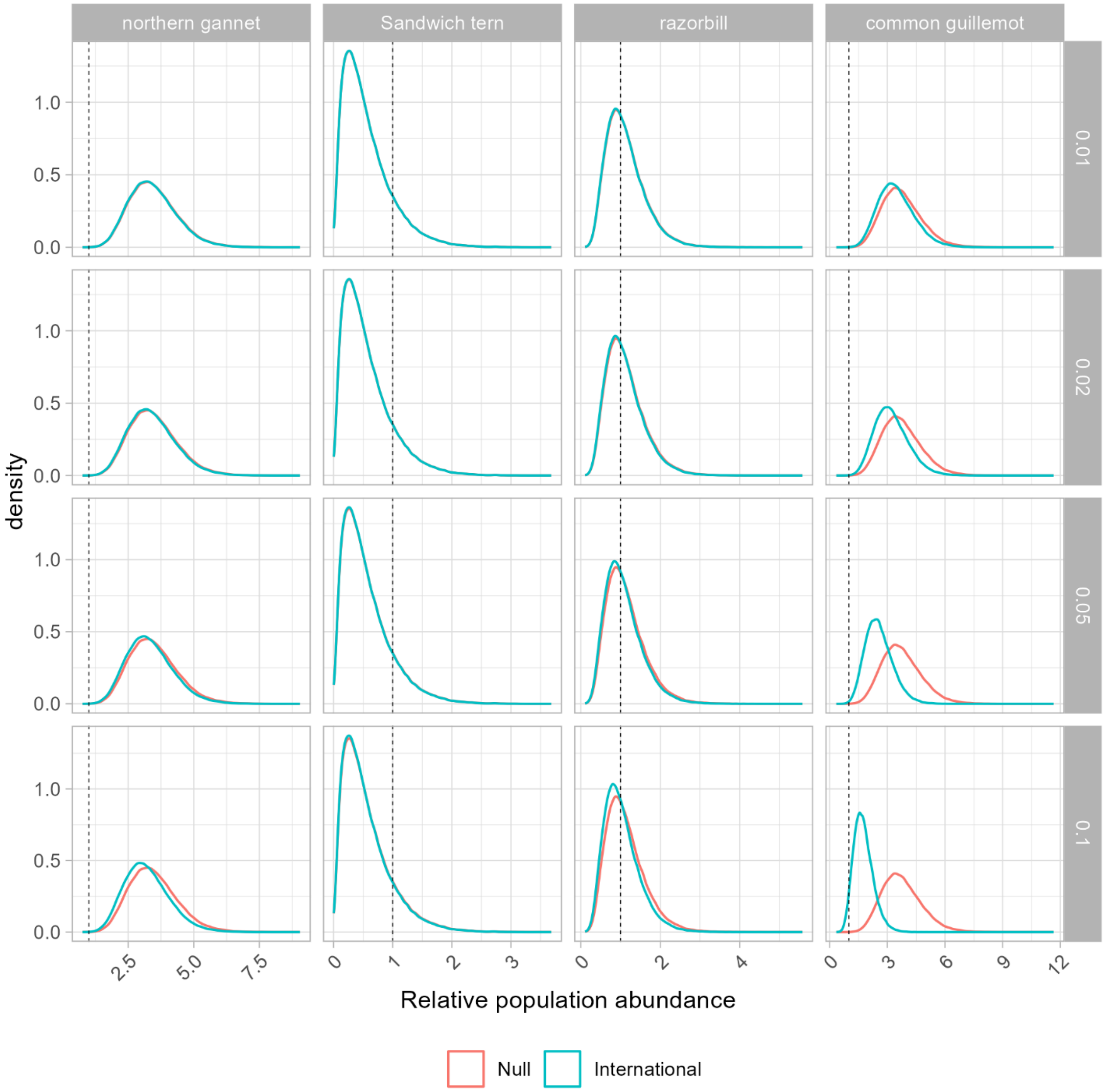

Figure 4-21 *Distributions of population abundances after 40 simulated years, relative to the initial population abundance, for a situation with no OWFs (red) and the international OWF scenario (cyan) per species. Abundances were scaled such that the mean initial abundance equals 1 (vertical dashed line). Note that the x-axis are scaled differently per species.*

4.3 Discussion

Compared to the previous KEC, we have updated several of the population parameters, the displacement probabilities and effect distances around OWFs (see Chapter 3). In addition, we have changed the casualty calculation method from the Bradbury method (Soudijn *et al.*, 2022b) to a displacement matrix method (JNCC, 2022; Searle *et al.,* 2022). This approach was chosen because it allows for an explicit assessment of uncertainty and because it allows for a transparent inclusion of factors such as displacement mortality and displacement probabilities. The Bradbury multiplication factors (RDRS), are based on a range of ecological factors (Bradbury *et al.,* 2014), but it is not straightforward to update their values based on new empirical knowledge. The parameter values in the current casualty calculation method can be updated based on new knowledge with regards to displacement probabilities and displacement mortalities. The explicit consideration of uncertainty in the casualty calculations was important in relation to the revised ALI methodology (Hin *et al.,* 2024), and allows for the propagation of uncertainty throughout all model components. Finally, we have implemented an uncertainty analysis method that propagates the uncertainty from the bird distribution maps all the way to the population modelling (section 3.3.4). This uncertainty propagation is also important in relation to the revised ALI methodology (Hin *et al.,* 2024), which was used in the current assessment.

4.3.1 Summary results

We considered three national scenarios, the Basic (all OWFs built up to 2020), Basic plus (all OWFs permitted up to 2024) and Total scenario (all OWFs plans up to 2031) and an International scenario (including the Total national scenario). The OWF scenarios were compared with a Null scenario without OWFs. The effect calculations estimated no ALI violations for the national scenarios for the Sandwich tern and the razorbill. For the northern gannet, ALI violations were estimated for the Total and Basic Plus scenarios for the 10% displacement mortality level. For the common guillemot, ALI violations were predicted for the 10% displacement mortality level for all scenarios. For the 5% mortality level, ALI violations were estimated for the Basic plus and the Total scenarios. For the 1% and 2% displacement mortality levels, no ALI violations were predicted for any of the species (*Figure 4-20*).

With no OWFs, the populations of the northern gannet, common guillemot and razorbill were predicted to increase over time, while the Sandwich tern population was predicted to decrease. The OWF scenarios led to slower increases in the projected population trends for the northern gannet and common guillemot, while the effects on the population trends of the Sandwich tern and razorbill were very small. Thus, despite the decreasing trend in the baseline population trajectory of the Sandwich tern, the effect of displacement from OWF developments does not lead to a violation of the ALI. In addition, for the scenarios where ALI violations occurred, the populations of the northern gannet and common guillemot were still projected to increase with OWFs, but at a slower pace.

For the international scenario, distribution maps were used as described by Waggitt *et al.,* (2020) rather than the maps described by van Donk *et al.,* (2024), which only cover the Dutch part of the North Sea and could thus only be used for the national scenarios. Waggitt *et al.,* (2020) did not provide uncertainty ranges in the maps and we have therefore conducted the international scenario on the mean maps only. For the Sandwich tern, an international map based on reversed distance weighing was used (van Donk, 2024), but also here only a single mean map was available. As a consequence, the international scenario could only be calculated using a lower quality representation of uncertainty for all four species. The international maps are likely to contain more uncertainty than the national maps as the sampling effort for the international areas of the North Sea is poor (van Donk *et al.,* 2024). We decided that no ALI tests should be conducted for the international scenario, because the uncertainty in the outcomes was not properly represented, and the revised ALI methodology depends strongly on a proper representation of the uncertainty (Hin *et al.,* 2024). While a formal ALI test was not performed, it should be noted that the effect of the international scenario on the predicted population trends through time were estimated to be higher than for the national scenarios for all species (*Figure 4-21*, *Table A-6*). In other words, the effect of OWFs on the mean bird population trends was found to be strongest for the international scenario. Seabird densities of the northern gannet, common guillemot and razorbill are highest in UK waters, because of their close proximity to the breeding colonies. Logically, any OWFs planned around there will affect more individuals of these species than in the Dutch North Sea, as described earlier in 4.2.1.

Displacement mortality is the most uncertain parameter in the model framework and the value chosen strongly affects the outcome of this modelling exercise. Empirical knowledge on the effect of displacement on mortality is unfortunately lacking. To our knowledge, it has so far only been determined based on expert opinions to allow quantification of the effects of displacement (Bradbury *et al.*, 2014; Leopold *et al.*, 2014; Soudijn *et al.*, 2022b). Effects of displacement are likely to be slow, and may cumulate over time (Daunt *et al.,* 2020). To take this uncertainty into account, we have implemented four levels of displacement mortality in our results (1%, 2%, 5% and 10%). Historically, 10% mortality has been designated, based on expert opinions, as a precautionary value, which should not underestimate the effects (Bradbury *et al.,* 2014; Leopold *et al.,* 2014). It is open for debate what a realistic value for displacement mortality is, and, whether this value in reality depends on the total area that is occupied by OWFs.

In the previous KEC, RDRS values were considered instead of explicit parameters for the displacement probabilities and displacement mortalities (Soudijn *et al.*, 2022b). The RDRS were originally not quantitative measures of the effects of habitat loss, but rather qualitative indications of species sensitivities to habitat loss (Bradbury *et al.,* 2014). In a way, these RDRS were, in KEC 4, implemented as quantities that included both the displacement mortality and displacement probability parameters that we have now implemented as separate factors in our analysis. Table 4-4 shows the RDRS values as used in previous KEC versions for the four species in the current assessment. In addition, we calculated species-specific benchmark values of the displacement mortality μ_d, based on the RDRS values used in previous KEC assessments by dividing the RDRS value by the mean displacement probability as derived in Chapter 3 (*Table 4-4*). These values allow for a comparison of displacement mortality with the RDRS such as applied in previous KEC assessments. If mortality values such as used in the previous KEC would be adopted, the closest calculated displacement mortalities would be 0.01 for the northern gannet, 0.05 for the Sandwich tern and common guillemot, and 0.1 for the Razorbill (corresponding to 1%, for the gannet, 5% for the tern and guillemot and 10% for the razorbill when expressed in percentages). For these mortality levels, no ALI violations were predicted for the northern gannet, Sandwich tern and razorbill (*Figure 4-20*). For the common guillemot, for a mortality level of 5%, ALI violations were predicted for the national Basic plus and Total scenarios.

Table 4-4*: Displacement mortality values per species as used in KEC 4, calculated as the ratio between the RDRS value used in KEC 4 and the mean displacement probability. The asterisk* indicates the displacement mortality value that matches the value that was used in KEC 4.*

Species	**Displacement mean (ρ)**	**RDRS (KEC 4)**	**RDRS μ_d**	**Displacement mortality probabilities (μ_d)**
northern gannet	0.85	0.008	0.0094	0.01*, 0.02, 0.05, 0.1
Sandwich tern	0.54	0.024	0.044	0.01, 0.02, 0.05*, 0.1
common guillemot	0.79	0.036	0.045	0.01, 0.02, 0.05*, 0.1
razorbill	0.43	0.036	0.084	0.01, 0.02, 0.05, 0.1*

4.3.2 Species selection

In this assessment, we considered effects of habitat loss due to OWFs for the northern gannet, Sandwich tern, common guillemot, and the razorbill. In KEC 4, the effect of displacement due to OWFs was considered for 6 more species: diver sp., northern fulmar, great cormorant, common eider, common scoter and Atlantic puffin. The smaller selection of species was due to time constraints in making the new distribution maps. Yet, it is probably wise to reconsider the species that should be part of the assessment when it is repeated in the future. The great cormorant is likely attracted rather than repelled by OWFs (Dierschke *et al.,* 2016; Lindeboom *et al.,* 2011). Although it did score high in the displacement sensitivity study conducted by Furness *et al.,* (2013), we consider it unlikely for the great cormorant to suffer from displacement by OWFs. The northern fulmar and Atlantic puffin are species that are likely sensitive to displacement and are common in offshore Dutch waters, it would make sense to include them in future impact assessment studies. The diver sp., common eider and common scoter are common in coastal Dutch waters, where there are so far no plans to develop wind energy. The diver sp. do occur more offshore in the international German waters (Garthe *et al.,* 2015, 2023) and could thus be an important species for the international scenario. Another species that may warrant further assessment for displacement impacts in future evaluations is the black-legged kittiwake. This species exhibits a sharp decline across extensive areas along the UK's east coast and Norway's cliffs (Anker-Nilssen *et al.,* 2022; Burnell *et al.,* 2023), with colonies experiencing consecutive years of poor breeding success, at several locations ultimately leading to complete site abandonment. These shifts are linked to changes in the availability of small pelagic fish (Frederiksen *et al.,* 2004), influenced both by intensive fishing and climate-driven ecosystem alterations (Daunt & Mitchell, 2013; Frederiksen *et al.,* 2007). Additionally, the species has an at-sea ecological association with the most abundant species in the Dutch North Sea, the common guillemot, benefiting from its foraging behaviour (Camphuysen & Webb, 1999). As the common guillemot is sensitive to displacement impacts, the black-legged kittiwake might as well suffer indirectly. It may therefore be wise to include this species in future assessments of displacement effects.

4.3.3 Spatial bird distribution maps

In the previous KEC studies, international and national seabird distribution maps were generated by interpolating ship- and airplane based counts of seabirds in the North Sea (Soudijn *et al.*, 2022b). For this, 'Inverse Distance Weighted' interpolation was used, which is a deterministic method that results directly from raw (averaged) counts (Leopold *et al.,* 2014). However, this method does not provide any information on robustness or statistical uncertainty of the interpolated count data. Another shortcoming of the deterministic method was that rare observations with high numbers of birds get a relatively large influence on the density at a certain location. Furthermore, ecological covariates that might explain the distribution of birds were not taken into account. Therefore, a statistical GLMM for estimating bird distributions was developed to address these issues (van Donk *et al.,* 2024). The resulting maps are based on statistical models in which the bird distributions in space and time per species are predicted based on statistical correlations between relevant covariates (abiotic and biotic conditions and human activities) and a random spatial temporal factor. In addition to the predicted distributions, the maps also provide information about uncertainty regarding the predictions. By including covariates, this new method provides a deeper knowledge on the ecological processes underlying the observed and expected seabird distributions. The spatial-temporal factor in the model explained, relative to the fixed covariates, the most variation for almost all species and time-periods (van Donk *et al.,* 2024). Yet, covariates like water depth and sea surface temperature were related to the distribution of many seabird species, while for some species, distance to shipping lanes (disturbance) or fishing intensity (source of food) played a role.

The survey data used to model habitat suitability were collected only at specific days of the year (van Donk *et al.,* 2024). Additionally, the model estimated an averaged bimonthly density using pooled data across the years (in order to obtain larger sample sizes). Therefore, it is inappropriate to interpret the estimated distributions as absolute population abundances. Instead, the spatial-seasonal patterns or the detected relative/average abundances are more informative to be used in the risk assessment. So far, not much has been published on habitat suitability modelling on the large spatial temporal scale of the entire North Sea over the last 30 years. Waggitt *et al.,* (2020) published estimated density maps using a GLM for several seabird species using international observational data. Soudijn *et al.,* (2022a) applied a GLM model under a Bayesian framework using the R-package 'INLA' for the herring gull. However, the uncertainty of the estimation and the ability of the model to predict densities were not extensively taken into consideration, and the model runs were highly time-consuming. Although a number of covariates were included in the suitability model to explain causal relationships, overall the abiotic covariates (water depth, Chlorophyll, sea surface temperature, etc) showed a weak link with the sea birds' distributions. This is likely due to the high variability in the data collection process (small sample size), as well as the high natural variability in birds behaviour. Furthermore, essential covariates like prey distribution are missing. Better data collection procedures on bird behaviour (e.g. tracking data) as well as information of prey distributions (fish distribution) are desired for future studies.

4.3.4 Effect calculations

In this report, we have chosen to model a range of increased mortality scenarios, presented here as the displacement matrix. It is a slight improvement compared to the Bradbury method that was applied in the previous KEC 4 (Soudijn *et al.*, 2022b), mostly because it is a more transparent method that is more easily updated with new empirical knowledge. The advantage of the approach is that it illustrates the relative impact of different levels of displacement mortality on the assessment outcomes. While this method is still considered simplistic, a better approach is currently not available due to the lack of knowledge of increased mortality from displacement. This is not surprising, as it is extremely difficult to study this in the field. It has been shown that spatial redistribution occurs in response to OWFs (Garthe *et al.,* 2023; Peschko *et al.,* 2024). How fitness of such displaced individuals is affected is not known, and it is also unclear how the redistribution of birds affects all individuals, as they may suddenly experience more conspecifics in certain areas. Compared to the method we used, many more birds actually travel through the OWF areas than we assume based on the static picture of the bird distribution maps. In reality, the maps indicate hot spots of birds that are continuously moving, and a larger part of the birds are likely to be affected by displacement than we estimated, but only for a short amount of time.

Key questions are whether:

1. birds can find sufficient alternative foraging locations and density-dependent mechanisms are at play (this section),
2. redistributions are dependent on the condition of the individuals and whether they are permanent (section 4.3.8), and,
3. redistributions depend on OWF characteristics such as the distances between turbines (section 4.3.10).

Knowledge regarding competition at sea is gradually developing for seabirds during the breeding season, with indications that inter-colony competition or the avoidance of such competition appears to be a driving factor behind the location and size of colonies and the utilization of marine areas within a specific foraging range. This has been particularly highlighted for the northern gannet (Wakefield *et al.,* 2013, 2017). In the case of displacement caused by OWFs, it is theoretically expected that displaced birds lead to increased densities elsewhere and, consequently, increased competition for food. The question then arises whether local food supplies can support such an increase in bird density. Effects of local food shortages may be partly compensated by bird behaviour and could build up over time, which makes these effects difficult to quantify in the field. Interestingly, the largest OWF displacement effect distances for the common guillemot have been reported in periods of highest guillemot density (Peschko *et al.,* 2024). This indicates that displacement is somehow a density dependent process, and it also implies that for periods with high bird densities, competition is expected to be more intense after displacement occurs.

A mechanistic underpinning of the habitat displacement casualties estimation would be advisable. Simple IBMs or a quantitative estimate of the loss in habitat quality and food quantity based on bird density maps or habitat suitability maps could be a way forward to make estimates of displacement casualties based more on ecological processes (van Kooten *et al.,* 2019). In the current project we worked on developing HALOMAR, where we use an energetic model with virtual birds following behavioural rules to perform population simulations (Chapter 5). Additional work is needed before the model can be included in the assessment process (see also Soudijn *et al.*, 2022b).

4.3.4.1 Effect distance values

The effect distance parameter in our model framework determines which area of the sea is affected by OWFs. In the previous KEC studies, an effect distance of 500 m was used for all the species that were considered in the assessment (Leopold *et al.*, 2014; Soudijn *et al.*, 2022b; van der Wal *et al.*, 2018). Recently, several studies were published that explicitly studied effect distances of OWFs for the species considered here (Grundlehner & Leopold, 2024; Peschko *et al.,* 2024; Vanermen *et al.,* 2015; Welcker & Nehls, 2016). It was therefore decided to update this parameter in the model framework. The effect distance varies between species, and we used a value of 1500 m for the northern gannet and Sandwich tern, 2000 m for the razorbill, while for the common guillemot, an effect distance of 19.5 km was adopted (*Table 3-2*). The effect distance determined which area of the sea is affected by OWFs. The parameter likely strongly affects our study's outcomes. Unfortunately, we did not manage to perform a sensitivity analysis of the parameter's effect in the current project's time limit. It is advisable to do so before the next assessment. In addition, it was complex to include a range of values for this parameter in the model framework and we have therefore not taken uncertainty in this parameter into consideration in the outcomes of the study.

For the common guillemot, we followed the precautionary principle by adopting the largest observed effect distance of 19.5 km (Peschko *et al.,* 2024). In addition, we consider the study conducted by Peschko *et al.,* (2024) the most extensive study available, covering a large spatial area, with data from multiple seasons, years and more than ten offshore wind farms in German waters included. For most sites, multi-year pre- and post-construction comparisons were made. The study distinguishes autumn (mid-July to September) and winter (October to February) effects (Peschko *et al.,* 2024). During the autumn period, guillemot numbers peak in the German and Dutch EEZ. The 19.5 km effect distance was found for this time of the year. A substantial effect distance, of larger than 10 km, was also reported for the common guillemot at the Gemini wind farm in the Dutch part of the North Sea (Grundlehner & Leopold, 2024). The Gemini study, however, covered one wind farm in a single winter season and only considered guillemot densities up to 10 km from the OWF. A smaller effect distance of 16.5 km was found in winter by Peschko *et al.,* (2024), and, it remains unclear which effect distance may be found around Gemini in August-September. Although the study by Peschko *et al.,* (2024) was conducted in German waters, we consider the study area to be representative for the Dutch context, as it is likely that guillemots in German waters are ecologically connected to those in Dutch waters. A large part of the populations that breed along the UK coast are involved, with birds moving both through German and Dutch waters. In addition, the studied wind farms are similar in design and located in similar ecological conditions (e.g., depth, sediment) as the Dutch wind farms, including the Gemini park. Two caveats should be noted regarding the study by Peschko *et al.,* (2024). First, we think it is uncertain whether the observed effect can be fully attributed to offshore wind farms alone. Especially the impacts of altered patterns of shipping not related to the wind farms remain unknown. Second, the study examines a generation of wind farms similar to Gemini, yet, future wind farms will feature much larger turbines spaced further apart. So far, it is unknown how these new characteristics will affect marine bird behaviour (see section 4.3.10).

4.3.5 Population models

The matrix population models used in the current assessment are a well-established framework for modelling population dynamics and are used frequently in conservation science (Caswell, 2001; Croll *et al.,* 2022; Heppell *et al.,* 2000). The population models were adjusted compared to the previous KEC to consider stochastic fluctuations in the parameter values through time (section 3.3.2). Hereby, they account for natural variation in demographic parameters (survival, breeding success, age at first reproduction). This variation leads, together with the uncertainties in the other model parameters, to distributions of population growth rates. These were used to calculate probabilities of OWF related violations of the population abundance thresholds. For some species, there are few data on immature/juvenile survival, age at first breeding or incidence of breeding (proportion of floater individuals). Parameter values of these variables are therefore poorly substantiated. It should be stressed, that continued monitoring and efforts to collect the life history parameters used in the models, are imperative for making accurate population assessments and, in addition, provide an indispensable way for monitoring the health of natural populations, especially with the advent of rapid environmental changes induced by climate change. In case OWFs have a strong effect on the marine bird populations, these effects should lead to changes in life history parameters. Declining trends in demographic parameters could lead to an underestimation of the population effects of OWFs (Horswill *et al.,* 2022) and it is therefore important to consider potential trends in timeseries of these parameters. Within the revised ALI methodology it was shown that an ALI violation will occur at a lower level of OWF-induced mortality when the baseline population growth rate is also lower (Hin *et al.,* 2024). A precautionary take on the population parameters should therefore always be considered.

Tagging studies and ringing recoveries indicate that northern gannet (Furness *et al.,* 2018; Pollock *et al.,* 2021), common guillemot (Dunn *et al.,* 2020a; Harris *et al.,* 2015) and razorbill (Mead, 1974; St. John Glew *et al.,* 2019) occurring in the Dutch part of the North Sea originate from a wide area of colonies, including colonies on the west side of the UK and Ireland and further away (Vogeltrekstation NIOO-KNAW & Ringersvereniging, 2024). Where possible, it was decided to derive population parameters from estimates of demographic rates for colonies closest to the Southern North Sea (these were specified per species, chapter 3), choosing those as being most representative for the Dutch national scenarios. Ideally, data from more colonies should be considered for the international scenario, but on that scale much information is probably also lacking.

In some assessments, population models (PVAs or more classical population models) are tuned such that the predicted population abundance matches observed abundance trends from for example colony counts (Croll *et al.,* 2022; Poot *et al.,* 2011). This can be achieved by adjusting, for example, the parameter with largest uncertainty. However, the observed population trend may be affected by processes that are not captured by most population models used in OWF impact assessments, such as dispersal. For example, large-scale redistribution of birds might lead to increasing bird numbers in colonies with low intrinsic growth capacity, which then act as a sink population. Rates of immigration are notoriously hard to quantify and not properly accounting for such processes in population models might lead to an overestimation of the true population growth rate when population models are tuned to observed trends. This would lead to an overestimation of the number of causalities that such a population can withstand before acceptable levels of population impact are exceeded. Adjusting demographic rates, such as breeding success or survival, should therefore only be conducted if all the processes that affect population growth can be quantified with sufficient accuracy. This requires a proper understanding of seabird metapopulation dynamics on a larger spatial scale, including the role of density-dependent regulation and mechanisms for immigration. Recently, Jeglinski *et al.,* (2023) used a Bayesian hierarchical state-space model to unravel metapopulation dynamics for the northern gannet. This revealed that metapopulation regulation for gannets is more complex than previously assumed and more thorough understanding of such processes will require further analysis.

The current population models are not developed to provide realistic predictions of absolute population trends and any biases in these predictions are likely to be on the precautionary side (lower baseline growth rates). Because not all important population processes could be quantified with sufficient accuracy, the baseline population trends predicted by the population models might not necessarily reflect observed trends. For example, without OWF impact, the population of the sandwich tern was predicted to decline with an annual rate of 1.4% per year, which is likely to be pessimistic. However, the current ALI methodology is a relative approach and only weakly sensitive to the direction and value of the absolute population growth rate (Hin *et al.,* 2024). To maintain on the precautionary side, it is crucial that modelled population trends are more pessimistic (lower population growth rate) than observed trends. This is the case for all four species addressed in the current assessment.

4.3.5.1 Density dependence

PVAs or more simplified population models used in ecological impact assessments for consenting activities most often lack density-dependent processes, despite the available evidence of their existence and importance (Croll *et al.,* 2022; Horswill *et al.,* 2017; Horswill & Robinson, 2015). The exclusion of density dependence in population models is motivated from the uncertainty about the strength and direction of density dependent population regulation and the argument that density-independent models provide a precautionary approach to the assessment (Green *et al.,* 2016; Merrall *et al.,* 2024). However, the latter argument only holds when density-dependent processes affect population development in a compensatory manner, i.e. a reduction in population abundance leads to an increase in demographic performance (survival, productivity). This might for example occur if a loss of breeding pairs increases the recruitment rate of individuals from the non-breeding part of the population. Compensatory density dependence leads is also called negative population regulation. When density-dependence acts in a depensatory manner (or "positive population regulation"), a reduction in population abundance will further endanger population persistence. For example, loss of individuals from a breeding colony might increase vulnerability to nest predation for the remaining individuals and thus decrease breeding success. Not accounting for density-dependence in impact assessments would therefore overestimate effects of the activity in case of compensatory density dependence, and underestimate the effects when density dependence acts in a depensatory way (Green *et al.,* 2016; Horswill *et al.,* 2017; Merrall *et al.,* 2024; Stephens & Sutherland, 1999).

A recent study by Merrall *et al.,* (2024) explored the strength and direction of density-dependence in seabird populations in the UK and Ireland, and the consequences of including (or excluding) density-dependence on the outcome of PVAs similar to those used in ecological impact assessment for offshore wind energy developments. A number of interesting observations were made by Merrall and coworkers. First of all, the availability of data on concurrent breeding success and population counts over longer times is low. Of the 599 population (colonies) identified, only 31 populations had concurrent data on breeding success and population counts for a minimum of ten years, which was used as a threshold for detecting a density dependent relationship. Of these populations, only three (10%) showed strong evidence of density dependence, all with positive regulation (Merrall *et al.,* 2024). An additional five populations (16%) showed weak density dependence, of which three negative and two positive. Based on these results, Merrall *et al.,* (2024) recommend to use density-dependent PVA-based assessments for seabirds populations as this will prevent unconstrained population growth in case of negative density-dependence, and maintain a precautionary approach for population with positive regulation. It should furthermore be noted that Merrall *et al.,* (2024) only looked at the relationship between breeding success and colony counts, and it is likely density dependence also operates on other demographic parameters, such the age at first breeding or the probability of breeding in populations with a large proportion of floaters.

4.3.6 Uncertainty analysis

In the current assessment, a first attempt was made to propagate uncertainty across the different model components (i.e. habitat suitability models, casualty calculation, population models). Quantifying uncertainty about the magnitude of the impact of habitat loss was also a requirement put forward by the revised ALI methodology (Hin *et al.,* 2024) as the outcome of the ALI is primarily determined by the magnitude and variability of the impacts from OWFs, as opposed to the absolute population trends. A substantial improvement in the current assessment compered to KEC4 was the use of 1000 possible distribution maps, which quantified variability in the distribution and abundance of birds, as well as uncertainty about its estimation. Another source of variability was the displacement parameter, which was quantified as a probability distribution, as opposed to a single number. Similar to KEC 4, variation due to environment stochasticity was accounted in the population models, although in a different manner (annual vs. per-simulation, see section 3.3.2). These three source of variability ultimately determined the distribution of the relative difference in population abundance between impacted and baseline scenarios, which was used to evaluate the ALI outcome. The implemented uncertainty analysis should be regarded as a first attempt that may need to be improved upon in the following KEC assessments. An obvious goal for the next KEC assessment is to include uncertainty of the one model parameter that is currently represented as a single number: the effect distance. In addition to quantifying uncertainty of all model parameters, more structural types of uncertainty could potentially be quantified, but this will require changing (parts of) the model assessment framework. For example, including uncertainty about the role of density dependence requires density-dependent population models. In addition, the mortality effect of OWFs might also be a density-dependent process, in which a direct coupling between the casualty calculation and population modelling is required.

4.3.7 ALI methodology

In KEC 4, the cumulative population-level effects were tested against Acceptable Levels of Impacts (ALIs; Potiek *et al.,* 2022) for the first time. In KEC 2 and 3, species-specific Potential Biological Removal (PBR) reference points were used (Leopold *et al.,* 2014; van der Wal *et al.,* 2018). The PBR approach was criticized because it implies a fixed level of density dependence in the populations that leads to compensation of mortality which may not actually occur (O'Brien *et al.,* 2017). Thresholds based on actual population models that explicitly consider the population dynamics of species were considered more reliable and precautionary. Unlike the PBR approach, the ALI approach explicitly links impact assessment to population dynamics (Hin *et al.,* 2024; Potiek *et al.,* 2022). For the current KEC, the ALI methodology was improved (Hin *et al.,* 2023, 2024). Threshold values for the revised ALI methodology were supplied by LVVN (*Table 4-3*).
The ALI method is based on population models that assess whether the impact of wind farms leads to an unacceptable population trajectory according to pre-defined policy criteria. These criteria involve two threshold values, i) the X threshold denotes the maximum acceptable percentage decline in population size relative to a population trajectory without OWF effects, and ii) the Y threshold denotes the level of statistical certainty that the predicted population effect will not exceed the X threshold. It should be stressed that the ALI methodology considers the population effect of the mortality from OWFs relative to the population size that could potentially be reached over a period of 40 years in absence of this impact (Hin *et al.,* 2024; Potiek *et al.,* 2022). As a consequence, the ALI threshold can be exceeded when total population size may be well above a nature conservation population target level (such as defined under the Dutch 'staat van instandhouding'). Also, the ALI may be violated even when the impacted population trajectory is still increasing, which for example occurs for the northern gannet and common guillemot in the current study. In contrast, the ALI threshold was not exceeded for the Sandwich tern, which has a negative population trend but where the displacement impacts by OWFs turn out to be negligible.

4.3.8 Behaviour in response to OWFs

Recent studies are quite convincingly showing redistributions of marine birds when comparing bird distributions before and after OWF constructions (Garthe *et al.,* 2023; Grundlehner & Leopold, 2024; Peschko *et al.,* 2024). The extent of displacement is quantified as the percentage which the observed abundance represents compared to the expected abundance in the undisturbed situation at different distances from the OWFs. Of course, these densities still only tell part of the story. The densities are the result of a change in behaviour of the birds, of which we do not know the exact mechanism or motivation. It is, for example, unclear whether the redistributions occurs because part of the birds always avoids a certain area, or if all the birds avoid the area part of the time. The difference between the two is not futile, as birds may still use the OWF areas for foraging if the latter is true. For northern gannets, it has been shown that individuals tend to be consistent in their avoidance behaviour (Peschko *et al.,* 2021), but for the Sandwich tern, common guillemot and razorbill such information is so far not available.

Little is known about what specifically deters birds from wind farms at sea. Peschko *et al.,* (2020), found strong OWF avoidance in common guillemots irrespective of blade rotation, suggesting that the rotation speed of the blades was not the main cause of the avoidance reaction. It could be, that the presence of very tall, moving vertical structures by themselves, which are unusual for open marine areas, has a deterring effect on the birds. The increase of displacement effects with turning rotors was limited, 75% compared to 63% when the rotors were not moving. Displacement could be due to visual stimuli (e.g. rotating blades, vertical structures) or (underwater) noise generated by the same rotors of the turbines. In addition, in circumstances of little wind, increased vessel activity in relation to maintenance works occurs in and around the OWFs may contribute to disturbance of marine birds (Dierschke *et al.,* 2016b; Mendel *et al.,* 2019; Vanermen *et al.,* 2015). Finally, ecosystem effects could play a role, for example, if underwater noise affects the presence or behaviour of prey fish. The effects of different factors cannot be disentangled using observational bird density surveys alone. Since we lack this basic knowledge, it is important to conduct further research to determine precisely what causes birds to avoid wind farms. A better understanding of the mechanism behind displacement of marine birds from OWFs could perhaps also indicate ways to mitigate the effects of OWFs on marine birds.

It is an interesting question why seabird species differ in their reaction towards offshore wind turbines. Why have cormorants and several gull species quickly learned to make us of the wind turbines and do not show the displacement effects as e.g. in common guillemots? Because the presence of offshore wind farms offers perching places for the more coastal species and in this way enhances the offshore occurrence of cormorants and gulls the behaviour of these species is qualified as attraction. Recently, two remarkable examples have been found of habituation behaviour in offshore windfarms. Although both are anecdotical, they illustrate the potential consequences in case birds learn that offshore windfarms offer perching or other opportunities and adapt to live in offshore wind farms (as cormorants and gulls do). The first one, is the observation of a juvenile White-tailed Eagle visiting a Danish offshore windfarm and resting on the platform basis of a wind turbine. This species is known as especially vulnerable for collisions and specifically for this species the black rotor tip has been developed to mitigate the relatively high collision risks (May *et al.,* 2020). In case White-tailed Eagles learn to make use of the wind turbine as resting places, and hunt for fish and birds in or near offshore windfarms, potentially increased risk of collisions will occur. The second example, is for the Sandwich Tern, one of our study species, which has been observed to use the wind turbine platform basis for perching. Also common/arctic terns performed the same behaviour during two spring migration seasons of 2021 and 2022 at offshore wind farm Borselle in Belgium waters (Stienen *et al.,* 2024). The terns were observed using a buoy within the wind farm as a resting site, while some also perched on the railing and platform of the wind turbines. In these instances, foraging activities were performed close to the turbine base. On a few occasions, male terns were observed transporting captured pelagic prey fish (sand eels) to females waiting on the turbine (Stienen *et al.,* 2024). Although the potential effects of these behaviours on collision risk remain unclear, the potential habituation in this species could, as for the White-tailed Eagle, increase the risk for collisions, but may decrease the effects of displacement.

Figure 4-22 *Juvenile white-tailed eagle visiting a Danish offshore windfarm. Picture by Uwe Wowk/Vattenvall.*

4.3.9 Ecological changes

Our results indicate that the greatest impact of displacement due to OWFs is expected for the common guillemot. The common guillemot is the most abundant seabird species on the Dutch Continental Shelf (DCS). Directly across the sea, on the southeast coast of England, lies Bempton Cliffs (Flamborough and Filey Coast), which now hosts almost the largest guillemot colony in the UK (111,925 breeding pairs in 2022, Clarkson *et al.,* 2022). This guillemot colony is experiencing significant growth, a trend that mirrors the increase seen of at sea presence of guillemots in the relatively nearby Dutch sector of the North Sea (Van Bemmelen *et al.,* 2023). However, the breeding population parameters of Bempton Cliffs colony do not explain this increase in abundance, suggesting that immigration from other areas is likely. The most logical explanation is a southward shift of birds from the north, where the last couple of years breeding success was low for several species and dominant decreasing trends occur (Burnell *et al.,* 2023). Such a shift, with e.g. increasing numbers in colonies in the south of the east coast of the UK, could also explain the increase in guillemot numbers in our part of the North Sea. In the northern part of the greater North Sea, numerous factors are hypothesized to contribute to this redistribution, with climate change and overfishing playing central roles (Daunt & Mitchell, 2013).

Rising sea temperatures and changes in ocean currents have been hypothesized to affect the food chain of the marine ecosystem ultimately changing the availability of prey species to seabirds (Frederiksen *et al.,* 2004). Studies suggest that warming waters may be driving the displacement of key prey species, such as sandeels, which are a primary food source for many seabird species including the black-legged kittiwake (Frederiksen *et al.,* 2007) and the common guillemot. Such shifts in prey abundance and distribution are especially important as food for the growing chicks on the breeding cliffs are likely to influence the breeding success and foraging strategies of seabirds (Searle *et al.,* 2022c, 2022b), causing them to seek more favourable conditions further south.

What is remarkable about this situation is that the carrying capacity of our section of the North Sea appears to accommodate the increasing numbers present outside the breeding season.

In addition to climate-related changes, intensive commercial fishing for sand eels in the North Sea has been implicated in the decline of seabird populations. Sand eels are a critical prey species, particularly for surface-feeding seabirds. The overexploitation of this resource, particularly in areas near important breeding colonies, is thought to have had direct negative impacts on seabird reproductive success. Black-legged kittiwakes, for instance, have shown significant declines in productivity in areas where sand eel stocks have been depleted (Carroll *et al.,* 2017). Moreover, in 2022, avian influenza outbreaks among northern gannet and Sandwich terns affected their European populations quite drastically. At Bass Rock, northern gannet nest occupation decreased by over 71%, breeding success fell by approximately 66% relative to the long-term UK average, and adult survival appeared significantly lower than in the previous decade (Lane *et al.,* 2024). In affected Sandwich tern colonies in the Netherlands, nearly all chicks perished (Rijks *et al.,* 2022). In 2023, avian influenza remained active in both species, though with a less devastating effect than in 2022. Many Sandwich tern colonies exhibited improved breeding success in early 2023, but showed severe mortality among nearly fledging chicks. Through an influx of floater birds, the total number of breeding pairs remained relatively high in this species (Knief *et al.,* 2024; Rijks *et al.,* 2022). Future impacts of avian influenza on both species' population dynamics remain to be evaluated. The population projections in this project do not include the effects of avian influenza, because the life history parameters are mostly based on values collected before the epidemic started. Especially the effect of avian influenza on survival rates will only become clear after several years of monitoring.

In summary, seabird distributions, but also life history parameters, may change drastically under the influence of various environmental factors. It is unpredictable how seabird population dynamics and distributions at the North Sea will develop in the near future. The current estimates of the effects of offshore wind farms on seabirds are based on the assumption that the seabird distributions and life history parameters will remain as they are. The estimates in this study are the best possible, but based on the species distribution data and assessment framework available to us at the current moment, and on the implicit assumption that gross seabird distributions will not drastically change in the next 40 years.

4.3.10 Technological developments OWFs

The technological advancements in offshore wind turbines, particularly in terms of size, energy generation, and spatial configuration, might have implications for displacement impacts on seabirds. As turbines increase in size, with rotor diameters exceeding 200 meters and turbine capacities up to 21 MW, seabirds are more likely to notice the structures from larger distances. However, the increased spacing between these large turbines, often exceeding 1.5 to 2 kilometres, provide seabirds with larger areas between the individual turbines, which possibly lets birds perceive wind farm less as continuous obstacles and more as isolated structures, allowing them to navigate through or nearer to the wind farms. This could lessen the overall displacement impact. However, the reaction of individuals to changes in the configurations are highly uncertain, as the mechanisms behind seabird displacement are still unclear. As indicated earlier (see section 4.3.8), it is uncertain whether displacement is primarily driven by visual deterrence from the turbines themselves or if other factors, such as underwater noise (Duarte *et al.,* 2021). Larger turbines generate more underwater noise during operation (Tougaard *et al.,* 2020), which could be an additional factor influencing seabird displacement, particularly for species that rely on auditory observational cues for prey detection. In this case, noise may disrupt the seabirds' ability to forage. Underwater noise from turbines may disturb foraging behaviour of seabirds, especially for diving species like the common guillemot and razorbill (Anderson Hansen *et al.,* 2020; Mooney *et al.,* 2020). The specific role of underwater noise in driving displacement is not yet known. In general, the reaction of marine birds to turbines with different characteristics is unknown, and research is needed to investigate its impact.

5 HALOMAR

5.1 Motivation

Displacement mortality is generally unknown, and the overlap between a bird distribution map and OWF shapefile is not an actual estimate of the number of individuals that come across OWF areas (birds are mobile and not static). It was therefore considered desirable to develop another method to estimate the effect of displacement on marine birds, based on individual energetics. Individual-based bioenergetic models have previously been developed to estimate the effects of habitat loss in the breeding season of the northern gannet in the English Channel (Warwick-Evans *et al.,* 2018), and of the common guillemot, razorbill, Atlantic puffin, and black-legged kittiwake along the Scottish coast (Searle *et al.*, 2018).These types of models have furthermore been developed for the non-breeding season in the North Sea, for the northern gannet, red throated diver, Sandwich tern, razorbill (van Kooten *et al.,* 2019) and common guillemot (Layton-Matthews *et al.,* 2023; van Kooten *et al.,* 2019). These types of models are considered suitable tools to assess the consequences of behavioral changes on individuals because they can simulate individual behaviour through time and space, while keeping track of the energetics of the individuals. For the effect of displacement by OWFs, food intake can be simulated through time, while also considering the energetic storage or energy supplementation of the young. Of course, some kind of assumption has to be made regarding the spatial distribution of food availability and the food available to individuals in OWFs.
The models developed for the breeding season for the northern gannet, common guillemot and razorbill (Searle *et al.*, 2018; Warwick-Evans *et al.*, 2018) are not suitable for our purpose because these species do not use Dutch waters during the breeding season. The model developed for the non-breeding season (van Kooten *et al.,* 2019) is not considered suitable yet for inclusion in an assessment due to unexplainable patterns in the model outcomes (Soudijn *et al.*, 2022b). An adjusted model for is thus desirable for an assessment of habitat loss effects in Dutch waters. The HALOMAR (HAbitat LOss MARine birds) model described here, is a simple, individual-based, bioenergetics model, similar to the model that was developed for the non-breeding season previously described by van Kooten *et al.,* (2019). In KEC 5, the new model could not be implemented yet, more time is needed for its development before it can be included in an assessment. In this project, a small step was made in the development of a basic model framework to assess the effects of habitat loss due to OWFs on common guillemots, northern gannets and Sandwich terns.

5.2 Model description

The model builds upon the individual-based model that was previously described by Van Kooten *et al.,* (2019). The model tests how a change in the quality of foraging habitats experienced by individuals, due to displacement by OWFs, affects individual energetic condition and the probability of starvation mortality or failed breeding. The model predicts effects on the vital rates due to the OWF scenarios for all the OWF areas together, which can subsequently be used in the KEC population models to determine the effect of the additional mortality on the population trends and test for an ALI violation. No density dependence is considered in the model; food sources are not depleted due to foraging and no competition between birds occurs. As a proxy for food availability, the model relies on bird distribution maps. Different from van Kooten *et al.,* (2019), foraging behaviour of the birds is mimicked by choosing a weighted random location for foraging at each time step from the total map, rather than limiting their foraging movements to a region around their location. In simulations using priorly fitted maintenance costs, the effects of OWFs are tested by either not selecting locations with OWFs for foraging, or not foraging for a full-time step when an OWF is encountered. The model considers the non-breeding season for common guillemots and northern gannets and the breeding season for Sandwich terns. In line with the precautionary principle, HALOMAR assumes, for the common guillemot and northern gannet, that all natural mortality occurs in the non-breeding season and that it is solely caused by starvation. On the other hand, for the Sandwich tern, it assumes that all failed breeding is caused by starvation of the young. The model is fitted on published values of annual survival probabilities of adults (common guillemot, northern gannet) or the hatching and fledging success of a nest (Sandwich tern) to predict a value for the maintenance costs for a situation without OWFs present.

5.2.1 Birds

HALOMAR is a discrete time model, which tracks the energy storage, food intake and energy expenditure of individuals through time. Since we consider the breeding season for the Sandwich tern, we consider nest energy rather than adult energy storage in the body, based on the idea that adults will abandon their nest before they would starve. The energetic storage (E_t) of an individual bird or nest at time t follows:

$$E_t = E_{t-1} + (I_t - m)/c, \tag{1}$$

where E_t depends on the energy in the previous timestep E_{t-1}. The energy increases with food intake I_t and decreases with maintenance costs m, and this change in energy storage is rescaled with the number of foraging locations c. The initial energy per bird or nest at $t = 0$ is equal to E_0, and the bird or nest dies when its energy reaches $E_t \leq 0$.

Per day, c foraging locations are visited, and equation (1) is executed c times per day. Which foraging locations are chosen depends on weighted random selection from the map. Birds avoid proportion σ, the habitat selectivity parameter, of the locations with the lowest food values. The birds are thus assumed to have a preference towards locations with higher food availability based on knowledge about the location of these areas. Within the proportion $1 - \sigma$ of the highest food locations, there is an equal probability of selecting each location. Food intake for common guillemot and northern gannet depends on the food encountered by an individual. For Sandwich terns, the food intake of a nest depends on the food encountered by both parents.

The maintenance costs are constant through time. Its value is determined by fitting the model, with no OWFs present, to species-specific annual survival probabilities for common guillemot and northern gannet, and to annual hatching-fledging probability for Sandwich terns (for the fitting method see 5.2.5).

5.2.2 OWF effects

OWF effects were considered on the mean survival and the mean energy storage over a thousand simulations of a thousand birds. For any given combination of parameters, OWF effects were calculated using maintenance costs that were estimated through model fitting (see section 5.2.5) for a situation with no OWFs. This allows for a comparison of the stored energy, adult survival or hatching-fledging probability without and with OWFs. The effect of OWFs was considered in two ways: i) the probability of selecting a location for foraging decreased with displacement probability ρ when an OWF is present, or ii) when a location in an OWF area was selected for foraging, there was a probability equal to the displacement probability ρ that no foraging occurred for that particular time step. For the scenarios with OWFs, the national ‘Total scenario’ was used as defined for the KEC (section 0).

5.2.3 Environment

We used the mean bimonthly bird distribution maps of the Dutch North Sea for the environment, that were also used for the habitat loss calculations in KEC (section 4.1.1). The maps were rescaled to a resolution of 5x5 km by projecting the original 10x10 km values over multiple grid cells to allow for the fine spatial effects that might be caused on the scale of a single OWF area. The periods for which maps were used depends on the species (see section 5.2.4). The bird density values derived from the maps were converted to proxies for food availability. Density values above the 80th percentile were assigned the density value of the 80th percentile in order to exclude outliers. For the sake of simplicity, we do not consider seasonal fluctuations in the energy requirements of the birds, the maximum food intake, or total food availability of the birds. Density values were thus rescaled across bimonthly maps per species to equalise the sum of all values across the bimonthly periods, such that the total food densities were in the same order of magnitude throughout the study period. Subsequently, the values were rescaled to feeding level values between 0 and 1. Here, a value of 1 represents a food availability that is sufficient to cover the maximum food intake rate. The rescaling also ensured that the maximum food intake of the birds was equal between the bimonthly periods. We restricted the map of Sandwich terns to 40 km out of the Dutch coast as adult birds tending to a nest usually have a foraging range of no more than 40 km (Fijn *et al.,* 2017; van Bemmelen *et al.,* 2022). Moreover, adult Sandwich terns generally do not go on prospecting flights outside of the 40 km range, neither during breeding nor if their nest fails (Fijn *et al.,* 2014).

OWF effect areas were calculated as being the search area plus an effect distance of ϑ. For the simulations with OWFs, each grid cell of the maps that has an overlap with the OWF effect area of at least 55% was considered affected by OWFs. A threshold value of 55% overlap was selected as this was found to result in a similar total area coverage of the study area with OWFs as the total OWF effect area based on the original search areas.

5.2.4 Parameter derivation

HALOMAR contains parameters for the number of cells used for foraging per day, the habitat selectivity, the initial energy a bird is given at the beginning of the period of interest, the population level survival probability or hatching and fledging probability, the start and end dates of the non-breeding season or the breeding season, the effect distance around OWF areas where displacement occurs, and the displacement probability. All parameters were assumed constant among individuals except for the initial energy storage, which we varied following a beta distribution with shape parameters a and b, that we set respectively to 5 and 0.5 to create a left-skewed distribution (*Table 5-1*). We thus chose for a distribution in which most individuals have a condition close to the mean, while some individuals have quite a low condition compared to the mean and no individuals with a condition far higher than the mean. The distribution of the mean body condition, a proxy for the energy storage, of northern gannets in September, just before the start of the non-breeding period, shows a left-skewed distribution (Fitzgerald *et al.,* 2022). Other than this, not much information was available on the distribution of energy storage in the bird populations. The value of the habitat selectivity σ was difficult to determine, as we did not know which information to base it on. We chose a value of 0.15 for all species, indicating that individuals would exclude 15% of the worst foraging locations a priori from foraging based on their knowledge of the environment. We investigated the importance of this parameter value in the sensitivity analysis (see section 5.2.6).

Table 5-1 *Default parameter values HALOMAR for all species.*

Parameter	Symbol	Value	Description
Habitat selectivity parameter	σ	0.15	Proportion of low food cells not visited
Shape parameter 1	a	5	Beta distribution shape parameter for the distribution of the initial energy storage
Shape parameter 2	b	0.5	Beta distribution shape parameter for the distribution of the initial energy storage

5.2.4.1 Northern gannet

For the start and end dates of the non-breeding season for the northern gannet, we considered observations on behaviour in and around the breeding colonies in the UK and Germany (Camphuysen, 2001; Fort *et al.,* 2012; Garthe *et al.,* 2012; Kubetzki *et al.,* 2009). We assumed the non-breeding period for northern gannets to last from November through January (Fort *et al.,* 2012; Garthe *et al.,* 2012; Kubetzki *et al.,* 2009). An annual background survival probability of 0.94 was used (*Table 5-2*), which is the same value that was used in the population models for adult northern gannets (*Table 3-4*). No information was found regarding the mean energy condition at the start of the non-breeding season. Therefore, we adopted the same value as for the common guillemot (*Table 5-2*). Since the northern gannet is a highly mobile species, we assumed that the number of foraging locations it may visit per day is limited by the amount of time it spends at each location rather than by its travel time or speed. We further assumed that, in the North Sea during the non-breeding season, a northern gannet has approximately nine hours of daylight per day in which it can forage (Garthe *et al.,* 2012). Yet, there is no information of the daily time budget for foraging during the non-breeding season available for northern gannets. As daily travelling activity values in the non-breeding season lie closely to the values registered in tracking studies during the breeding season, we based the foraging time on values from the breeding season, when about 25% of the day is spend foraging (Amélineau *et al.,* 2014; Garthe *et al.,* 2012; Grecian *et al.,* 2018; Lane *et al.,* 2019). This amounts to about 2.3 h/d for the nine hours daylight in the non-breeding season, and with a foraging speed of 15 km/h (loosely based on breeding season values from (Hamer *et al.,* 2000) and a grid cell length of 5 km, this results in seven different foraging locations per day. We used an effect distance ϑ of 1500 m around the OWFs and a displacement probability ρ of 0.85 (*Table 5-2*) the same as was used for the displacement matrix calculations in Chapter 4 (*Table 3-2* and *Table 3-3*).

Table 5-2 *Default parameter values HALOMAR for the northern gannet.*

Parameter	Symbol	Value	Unit	Source
Non-breeding period		Nov – Jan	-	(Camphuysen, 2001; Fort *et al.,* 2012; Garthe *et al.,* 2012; Kubetzki *et al.,* 2009)
Adult survival probability	S_A	0.94	-	*Table 3-4*
Initial mean energy	E_0	6	-	Based on the same value as for guillemots (Clairbaux *et al.,* 2021)
Foraging locations	*c*	7	cells/day	(Amélineau *et al.,* 2014; Garthe *et al.,* 2012; Grecian *et al.,* 2018; Hamer *et al.,* 2000; Lane *et al.,* 2019)
Effect distance of impact around OWFs	ϑ	1500	m	*Table 3-2*
Displacement probability	ρ	0.85	-	*Table 3-3*

5.2.4.2 Sandwich tern

For the start and end dates of the breeding season, we considered observations on behaviour in and around the Dutch colonies. We identified the breeding season of Sandwich terns to last from April 20th to July 10th (*Table 5-3*) after which the young are fledged and leave the nest to gather in crèches where they are still fed by their parents for several weeks (Brenninkmeijer & Stienen, 1992; Fijn & Kemper, 2023; Stienen, 2006; Vanermen *et al.,* 2013). Instead of using the annual adult survival probability to calibrate the maintenance costs, we derived a combined hatching-fledging probability of 0.301, based on a mean annual breeding success of 0.49 chicks/year (Schekkerman *et al.,* 2021) and a mean clutch size of 1.63 eggs/nest based on observations in the Dutch Voordelta colonies (Poot *et al.,* 2014). For the mean initial nest energy condition, we considered that failure and death of the chicks may occur within one to two days without food intake (*Table 5-3*), as chicks are not expected to have large energy reserves yet. Given maintenance costs of about 0.5, we set $E_0 = 1$. As Sandwich terns are central-place foragers during the breeding season (Fijn *et al.,* 2022), we assumed that a single foraging location (grid cell of the map) can be used per foraging trip. If adults take turns going out for food, there are 17 hours per day of possible foraging activity per breeding pair (Stienen *et al.,* 2000). Based on a mean trip duration of 2.5 h (Fijn *et al.,* 2017; van Bemmelen *et al.,* 2022), this means six foraging locations can be visited per day per nest. For the boundary distance of impact ϑ, we chose 1500 m and for the displacement probability ρ we chose 0.54 (*Table 5-3*), the same values as were used in the displacement matrix (*Table 3-2* and *Table 3-3*).

Table 5-3 *Default parameter values HALOMAR for the Sandwich tern.*

Parameter	Symbol	Value	Unit	Source
Breeding period		20.4. – 10.7.	-	(Brenninkmeijer & Stienen, 1992; Fijn & Kemper, 2023; Stienen, 2006; Vanermen *et al.,* 2013)
Annual hatching-fledging probability		0.301	-	(Poot *et al.,* 2014; Schekkerman *et al.,* 2021)
Initial mean energy	E_0	1	-	-
Foraging locations	*c*	6	cells/day	(Fijn *et al.,* 2017; Stienen *et al.,* 2000; van Bemmelen *et al.,* 2022)
Effect distance of impact around OWFs	ϑ	1500	m	*Table 3-2*
Displacement probability	ρ	0.54	-	*Table 3-3*

5.2.4.3 Common guillemot

For the start and end dates of the non-breeding season, we considered observations on behaviour of common guillemots in and around the colonies, as well as activity patterns characterising the seasons (Dunn *et al.,* 2020a, 2020b). Considering the timing of colony-absence at one of their UK breeding colonies (Isle of

May) and their moulting period in August-September, we identified the period from October through December as non-breeding season for common guillemots (Dunn *et al.,* 2020b). An annual background survival probability of 0.949 was used (*Table 5-4*), which is the same value as used in the population model for adult common guillemots (*Table 3-6*). The mean energy condition at the start of the non-breeding season has not explicitly been reported on. However, Clairbaux *et al.,* (2021) reported that under cyclonic conditions in the North Atlantic in the winter period (e.g. Newfoundland and Labrador Sea), a common guillemot can endure 8.1 days of fasting. As the conditions in the southern North Sea are normally not cyclonic and probably less harsh than in the North Atlantic, the energy requirements of common guillemots in the Dutch North Sea are likely a bit lower, and we assumed they can sustain a fasting period of about 11 days. Based on this maximum fasting period and the maintenance costs that were usually estimated around 0.5-0.6, we assumed a mean initial energy of $E_0 = 6$ (*Table 5-4*). On average, common guillemots were described to spend 4.85 h daily on diving (Dunn, Wanless, *et al.,* 2020b). Assuming on average ten daylight hours per day during their non-breeding period (https://www.timeanddate.com/sun/@54.092,3.252 for 54°05'31"N, 3°15'07"E), this means they can maximally spend about 5.15 h per day travelling between foraging locations. Common guillemots travel exclusively by swimming during the non-breeding period (Dunn *et al.,* 2020b), and we assumed that they use their entire daily travel budget for travelling between foraging locations. This assumption will result in the maximum possible number of foraging locations per day as it is not known which proportion of the day common guillemots are spending not travelling or foraging, for example resting on the water. Combined with a swimming speed of approximately 6 km/h during the breeding season (Amélineau *et al.,* 2021), common guillemots can travel through about six grid cells with a length of 5 km per day. For the boundary distance around the OWF, we chose a value of 19.5 km, and we chose a displacement probability value of 0.79 (*Table 5-4*), which were both also used for the displacement matrix calculations (*Table 3-2* and *Table 3-3*).

Table 5-4 *Default parameter values HALOMAR for the common guillemot.*

Parameter	Symbol	Value	Unit	Source
Non-breeding period		Oct – Dec	-	(Dunn *et al.,* 2020a, 2020b)
Annual background survival probability adults	S_A	0.949	-	*Table 3-6*
Initial mean energy	E_0	6	-	(Clairbaux *et al.,* 2021)
Foraging locations	c	6	cells/day	(Amélineau *et al.,* 2021; Dunn *et al.,* 2020b)
Effect distance of impact around OWFs	ϑ	19500	m	*Table 3-2*
Displacement probability	ρ	0.79	-	*Table 3-3*

5.2.5 Model simulations

For any given combination of parameters, the maintenance costs needed to be estimated by fitting the model to the adult survival probability (common guillemot, northern gannet) or the hatching-fledging probability (Sandwich tern) for a model with no OWFs. After the maintenance costs were estimated, simulations could be conducted to estimate the effect for both OWF effect approaches (see 5.2.2). To estimate the maintenance costs, we conducted a bisection method routine, in which different values of the maintenance costs m were used in the model to find a value for the predicted survival probability $S_N(m)$ as close as possible to the given species-specific survival S. The estimation of m was done in a situation without OWF effects. For each of the calculation steps, we did 1000 simulations with 1000 birds; this process was repeated as many times as deemed necessary by the algorithm. The predicted survival probability $S_N(m)$ is defined as:

$$S_N(m) = \sum_{1}^{I} \frac{h_N(m)}{I}$$

where $m \in [0,1]$, $h_N(m)$ is the number of surviving birds given by a single run of the HALOMAR model, I is the number of iterations, and N the number of simulated birds. The parameter m is estimated by solving the following equation for $m \in [0..1]$

$$S - S_N(m) = 0.$$

5.2.6 Sensitivity analysis

To test the robustness of the model, we conducted a sensitivity analysis in the form of parameter sweeps for the displacement probability, the avoidance of low food cells, the number of cells visited for foraging per day, and the initial energy. We also varied the length of the simulated period. Parameter values were varied one by one, while the other parameters were kept at their default values. For each parameter combination that was tested, the full model sequence was conducted: first fitting the maintenance costs on a map without OWFs, and subsequently running effect simulations with OWFs.

As we were especially uncertain about values for the habitat selectivity parameter, and for the initial energy and foraging locations parameters, we considered a wide range of values for these parameters. The duration and timing of the breeding and non-breeding seasons are relatively well documented. We used a range of values that can be considered realistic for those parameters. For the survival and hatching-fledging probabilities and the displacement probabilities we did not conduct a sensitivity analysis. The survival and hatching-fledging probabilities are only linked to the estimated maintenance costs and are unlikely to affect how the OWF effect plays out. The displacement probabilities likely have a linear relationship with the OWF effect and were therefore considered less important to simulate extensively.

Table 5-5 *Parameter ranges and periods used for the sensitivity analysis, with step sizes in parentheses. For default values of the parameter values see Table 5-4 - Table 5-3.*

	Habitat selectivity	**Foraging locations**	**Mean initial energy**	**Period**
Symbol	σ	c	E_0	
Units	-	Cells/day	-	Months
northern gannet	0-0.72 (0.01)	5-20 (1)	1-20 (1)	1: Nov-Dec; 2: Oct-Jan; 3: Oct-Dec
Sandwich tern	0-0.75 (0.01)	2-11 (1)	1-20 (1)	1: May-June; 2: April-July
common guillemot	0-0.68 (0.01)	1-19 (1)	1-20 (1)	1: Sep-Jan; 2: Oct-Dec; 3: Sept-Feb; 4: Sept-Dec; 5: Oct-Feb; 6:Oct-Jan

5.3 Results

5.3.1 Simulations

Over the period of the non-breeding season, the model predicts a gradual decline in the mean energy storage for the northern gannet. The decline is more rapid in the last two months of the non-breeding season, in line with a lower mean food intake over this period (*Figure 5-1* a-b). For the common guillemot, the model predicts an initial decline and then an increase in energy storage in the final month of the non-breeding season. This is caused by a higher food intake in this last period (*Figure 5-1* c-d). The Sandwich tern nest energy condition initially increases rapidly but then starts to decline somewhere halfway during the breeding season. This is caused by a drop in the food availability in the same period (*Figure 5-1* e-f). The patterns in food intake are probably due to changes in the food availability maps per bimonthly period, and it is likely that the mean food densities vary a bit between the maps.

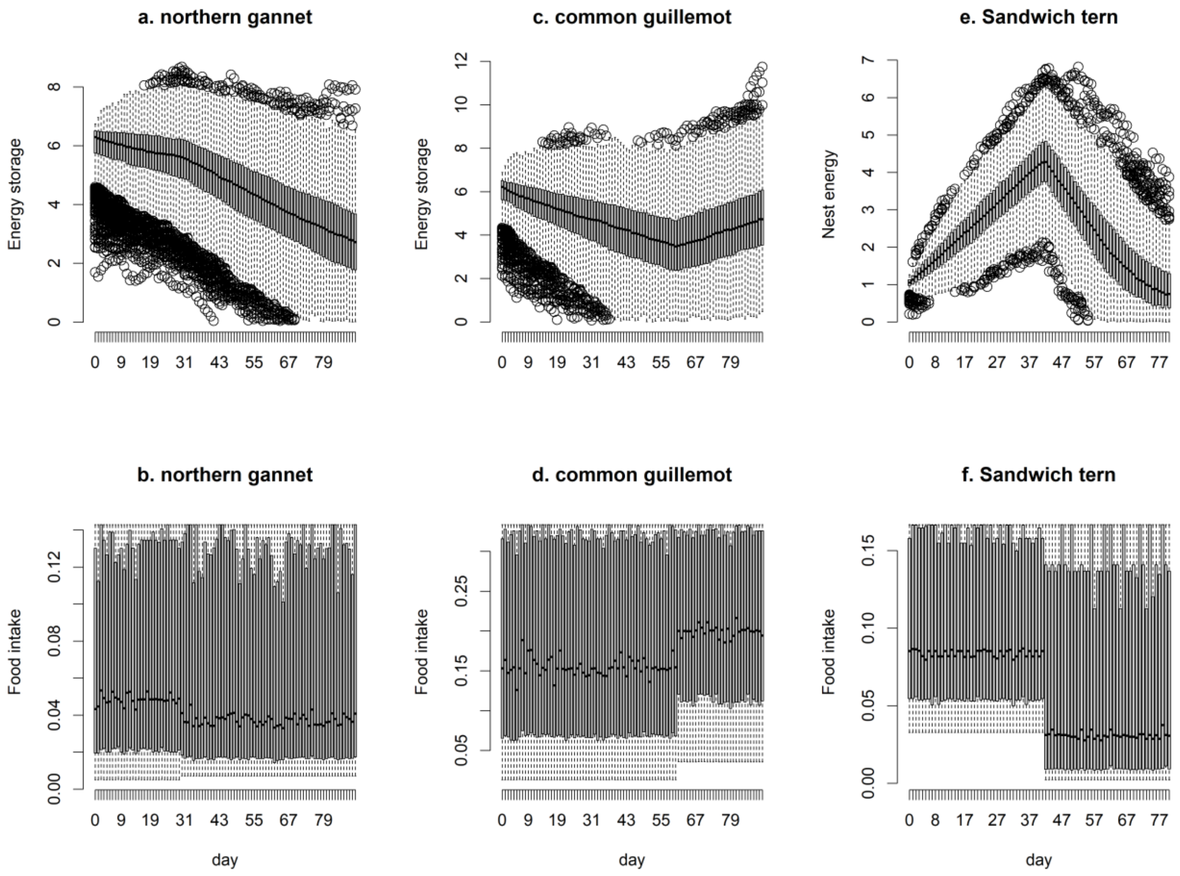

Figure 5-1 *Change in energy storage, nest condition and food intake of surviving birds over time for a simulation with 1000 birds with no OWFs, for the a,b) northern gannet, c,d) common guillemot and the e,f) Sandwich tern. Food intake is considered at the first of the foraging locations per day, the condition is given at the start of the day. Note that while for the northern gannet and the common guillemot the body condition is plotted, for the Sandwich tern the nest condition is indicated in the plots. Default parameter values were used for all simulations as given in Table 5-4, Table 5-2 and Table 5-3. For the northern gannet and common guillemot, the simulation was run on the bimonthly maps for Oct/Nov and Dec/Jan. For the Sandwich tern, these were April/May and Jun/Jul.*

5.3.2 OWF effects

The condition of surviving birds at the end of the non-breeding season (northern gannet, common guillemot) and at the end of the breeding season (Sandwich tern) is predicted to be negatively affected by OWFs for the northern gannet (*Figure 5-2* a). This effect is visible for both the scenario where birds avoid OWFs without costs and when they skip foraging for a time step when they encounter an OWF. For the common guillemot, on the other hand, the body condition is only predicted to be negatively affected when they cannot forage when they encounter an OWF. In case OWFs are simply avoided without costs, the body condition is actually higher than without OWFs (*Figure 5-2* b). The Sandwich tern shows a similar pattern, the nest condition at the end of the breeding season is higher with OWFs than with no OWFs when they are avoided without penalty but lower than with no OWFs when foraging is skipped when an OWF is encountered (*Figure 5-2*). A similar pattern is visible in the additional mortality (*Figure 5-3*), where for the northern gannet, mortality is higher in the model for both scenarios with OWFs compared to a situation with no OWFs, for the common guillemot and Sandwich tern there is only additional mortality of adults and nests when foraging is skipped for a timestep when birds encounter an OWF. For the northern gannet, the additional mortality when avoiding areas with OWFs without costs is estimated to be 6.7% on average, where for the scenario without foraging on OWF encounter, this is 64%. For the common guillemot this additional mortality due to OWFs with no foraging on OWF encounter is quite extreme, 94.6%, and for the Sandwich tern, additional nest failure for the same scenario is 13.9% (*Figure 5-3*).

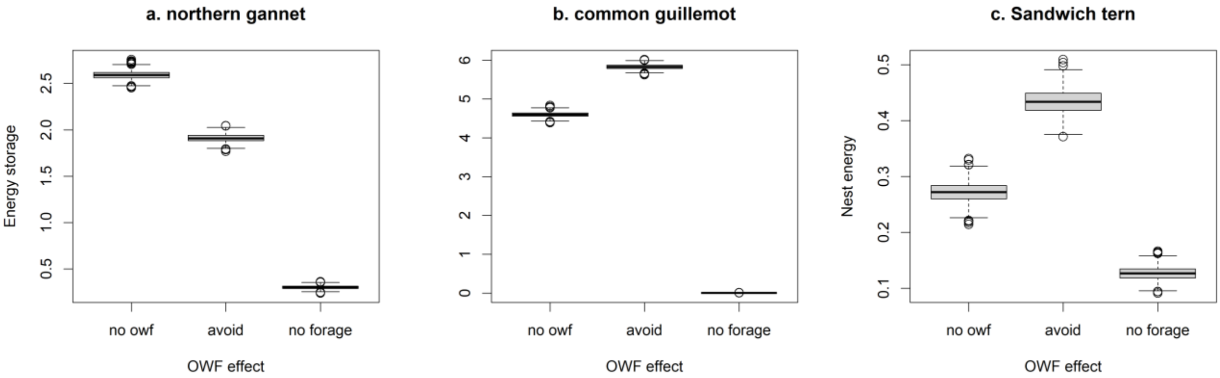

Figure 5-2 *Energy storage of surviving birds and nests at the end of a 1000 simulation with 1000 individuals per simulation with and without OWF effects, for the a) northern gannet, b) common guillemot and c) Sandwich tern. Note that while for the northern gannet and the common guillemot the mean body condition is plotted, for the Sandwich tern the mean nest condition is indicated. Default parameter values were used for all simulations as given in Table 5-4, Table 5-2 and Table 5-3.*

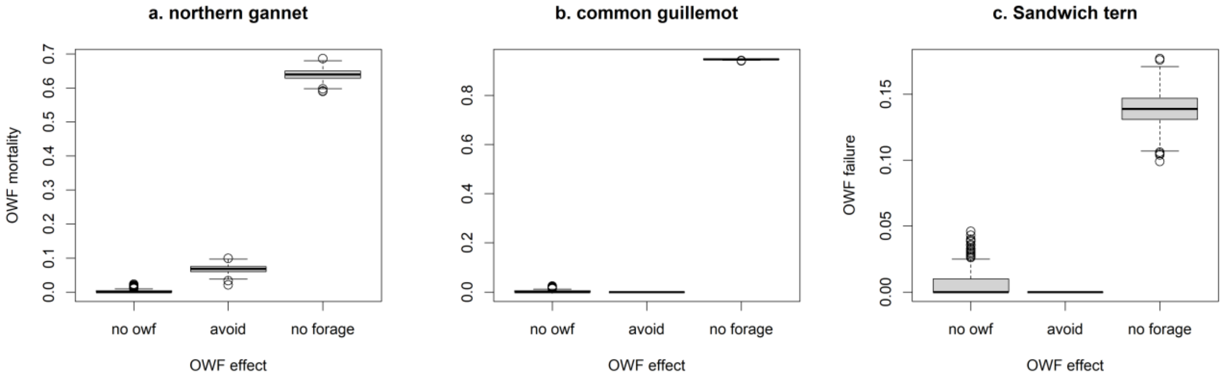

Figure 5-3 *Mean additional mortality and nest failure, compared to the standard default background mortality and nest failure, based on 1000 simulations with 1000 birds at the end of simulations with no OWFs, avoidance of OWFs and no foraging in OWFs, for the a) northern gannet, b) common guillemot and c) Sandwich tern. Note that while for the northern gannet and the common guillemot the mean mortality of adults at the end of the simulations is plotted, for the Sandwich tern the nest failure is indicated in the plots. Default parameter values were used for all simulations as given in Table 5-4, Table 5-2 and Table 5-3.*

The food intake over the full simulation period shows quite a wide range of variation between simulations (*Figure 5-4*). A relatively subtle change in the mean food intake in the model can have a strong effect on the estimated mortality probability. For example, for the northern gannet, the mean food intake is only slightly lower for the scenario with avoidance of OWFs than for the scenario with no OWFs, but this still results in an additional 6-7% mortality (*Figure 5-3*). For the northern gannet and the common guillemot, there is a clear decrease in food intake for the scenario with no foraging in OWFs, and this indeed also results in a strong increase in mortality (*Figure 5-3*). For the Sandwich tern and common guillemot, there is a slight increase visible in food intake for the scenario where OWFs are avoided (*Figure 5-4*), which can explain why no negative effects of this scenario are predicted by the model. The effects of the no foraging in OWFs scenario on the Sandwich tern food intake seems very subtle, but does have a strong impact on nest survival, about 15% additional nest failure occurs (*Figure 5-3*). Perhaps this is due to the low energy condition of the Sandwich tern nests that the model predicts in the last period of the breeding season (*Figure 5-1*).
For the Sandwich tern and common guillemot, relatively more areas on the distribution maps are covered by OWFs than for the northern gannet *(Figure 5-5).* However, for the northern gannet, relatively more areas with high bird density (and thus high food availability) are covered by OWFs than for the common guillemot and Sandwich tern. As a result, the food intake and energy storage of the northern gannet are negatively affected by OWF avoidance, while it is the other way around for the common guillemot and Sandwich tern.

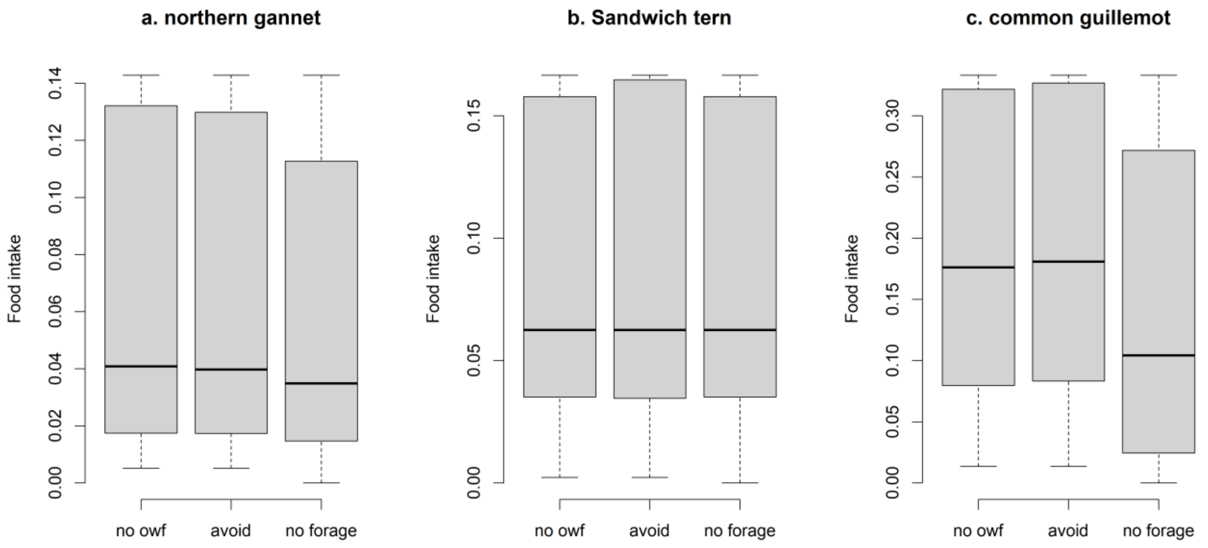

Figure 5-4 *Boxplots of the food intake over a full period for a single simulation with 1000 birds, for a situation with no OWFs, avoidance of OWFs and with no foraging in OWFs, for the a) northern gannet, b) Sandwich tern and c) common guillemot. Default parameter values were used for all simulations as given in Table 5-4, Table 5-2 and Table 5-3.*

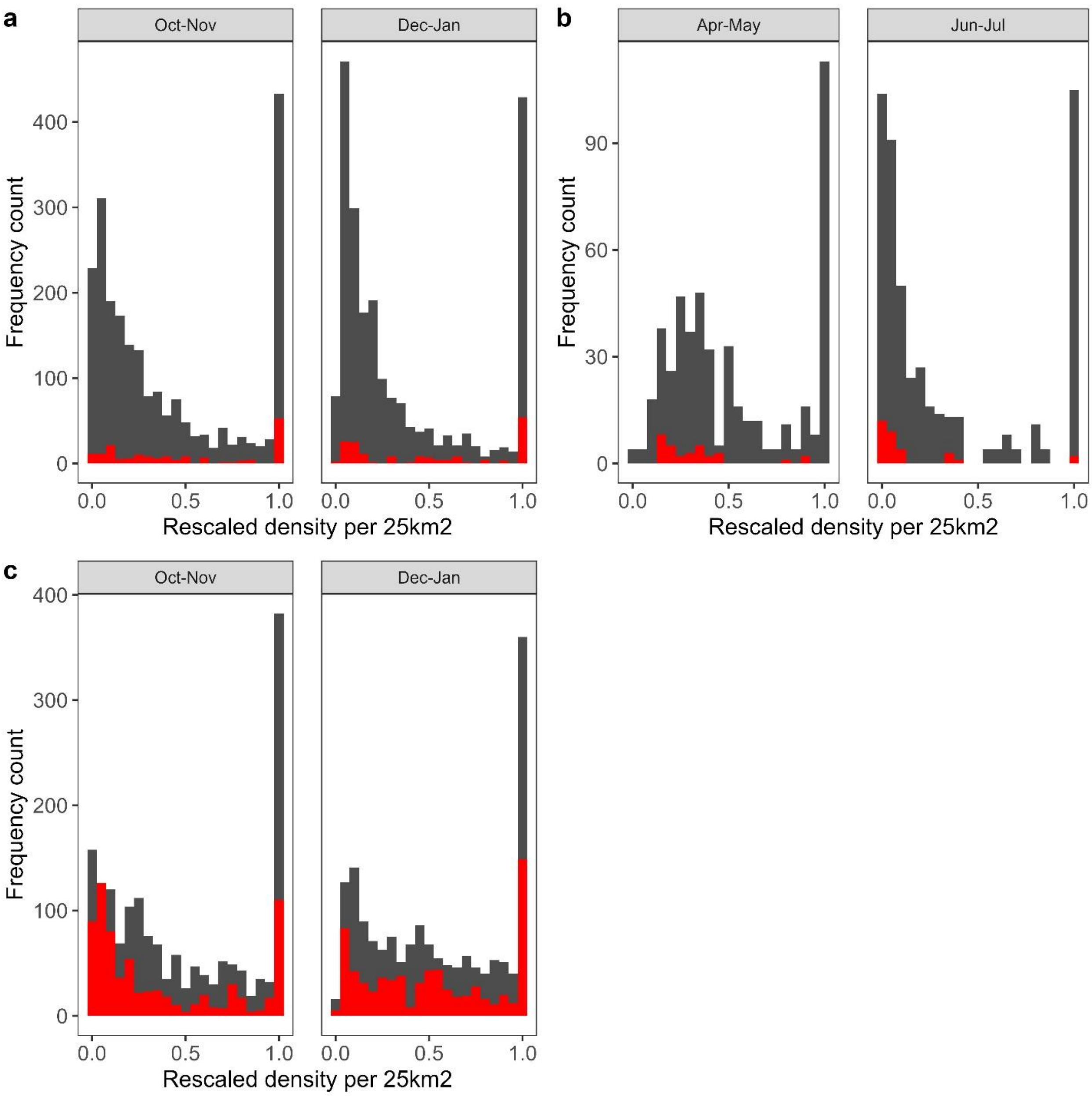

Figure 5-5 *Frequency of map grid cells with a certain value for the food availability for the a) northern gannet, b) Sandwich tern, and c) common guillemot densities per grid cell (5x5 km). In dark grey, cells with no OWFs, and in red cells with OWFs.*

5.3.3 Sensitivity analysis

In the sensitivity analysis, we tested the effect of changes in the parameter values for the number of foraging locations per day, the habitat selectivity, and the length of the simulation period. In addition, we considered the mean value of the initial energy. Both the effect on the fitted maintenance costs as well as the additional mortality or nest failure were considered. Here, we show examples of the sensitivity analysis results for the Sandwich tern. Plots for the other species can be found in Annex 4, as they show remarkably comparable results.

Overall, the longest simulation period leads to the lowest estimate of the maintenance costs (period 2, *Figure 5-6* d). This is probably due to the pattern of decreasing energy storage for the birds over time, with a lower maintenance costs, individuals can survive for longer. This pattern was also found for the northern gannet and common guillemot (Annex 4, *Figure A-5* and *Figure A-6*). Increasing habitat selectivity and the mean initial energy generally lead to higher maintenance costs; a higher food intake or energy storage at the start of the simulation allows the birds to sustain higher maintenance costs (*Figure 5-6*, *Figure A-5* and *Figure A-6*). These effects and the shape of the relationships are the same for all species. For the number of foraging locations per day, the effect 'levels off', but for the Sandwich tern, maintenance costs decrease as the number of locations visited per day increases, while maintenance costs increase with the number of foraging locations per day for the northern gannet and common guillemot (*Figure 5-6*, *Figure A-5* and *Figure A-6*).

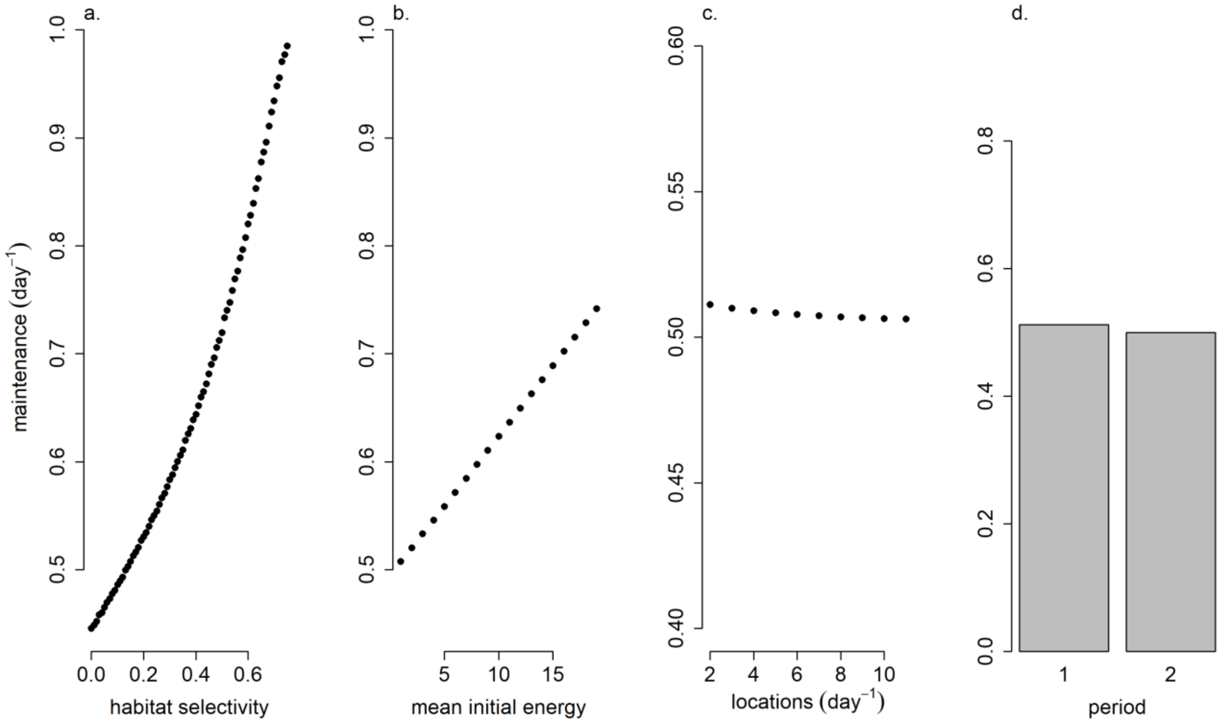

Figure 5-6 *Sensitivity analysis of the fitted maintenance costs per parameter for the Sandwich tern for simulations with no OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-3, the default value of the mean maintenance cost is 0.51. The periods considered here are, 1: May-June; 2: April-July. For the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulations with 1000 birds were conducted to estimate the maintenance costs.*

When avoiding OWFs without additional costs, the Sandwich terns are estimated to have a better hatching-fledging success than with no OWFs (Figure 5-7). When the Sandwich terns are more selective in their habitat, OWFs have a less positive effect (Figure 5-7 a). This is likely because most OWFs are located in areas with low densities of Sandwich tern, which are considered 'poor food areas' (Figure 5-5). As a result, when the birds are more selective, the positive effect of the OWFs is less pronounced. The number of areas birds can use for foraging per day increases the positive effect on the survival. Perhaps because there are now more instances for which the poorer food areas inside OWF areas are avoided. The initial energy condition of the birds reduces the positive effect of OWFs (Figure 5-7 b). While the exact reason is unclear, it may be because the increased food intake from avoiding OWF areas is partly offset by the higher maintenance costs associated with this parameter setting. A longer simulation period results in a more positive effect of avoiding areas with OWFs, likely because the positive effect accumulates over a longer time period. The patterns of the OWF avoidance effect are similar for the adult survival of the common guillemot (Annex 4, Figure A-9). However, for the northern gannet, avoiding OWF areas negatively affects adult survival (Figure A-7). This effect is lessened by a higher habitat selectivity, probably because OWFs are located in food-rich areas, and avoiding these areas is less detrimental when the birds are more selective in their foraging habitat choice to begin with (mean food intake is increased). The effect of OWFs is also lessened by higher mean initial energy but strengthened by the number of foraging locations per day. A longer simulation period results in a more negative effect of avoiding areas with OWFs for the northern gannet (Figure A-7).

When birds do not forage in OWF areas, OWFs always have a negative effect on the vital rates. Regardless of the parameters used, the hatching-fledging success for the Sandwich tern is always below the default value of 0.301 (*Figure 5-8*). The same occurs for adult survival of the northern gannet (*Figure A-8*) and the common guillemot (*Figure A-10*), where OWFs always lead to lower adult survival. For all species, habitat selectivity results in a more negative effect of OWFs. Similarly, with increasing number of foraging locations per day, the presence of OWFs decreases the hatching-fledging success for terns and the survival for northern gannet and common guillemot. This probably occurs because areas with OWFs are encountered more often with a higher number of foraging locations per day, and the timesteps without foraging are, as a result, increasing. With a higher initial energy, the negative effect of OWFs is reduced for all species, possibly because with the increased initial energy birds can sustain longer periods without food. Finally, the length of the simulation period now has a negative effect, as the negative impact of the OWFs builds up over a longer period (*Figure 5-8, Figure A-8, Figure A-10*).

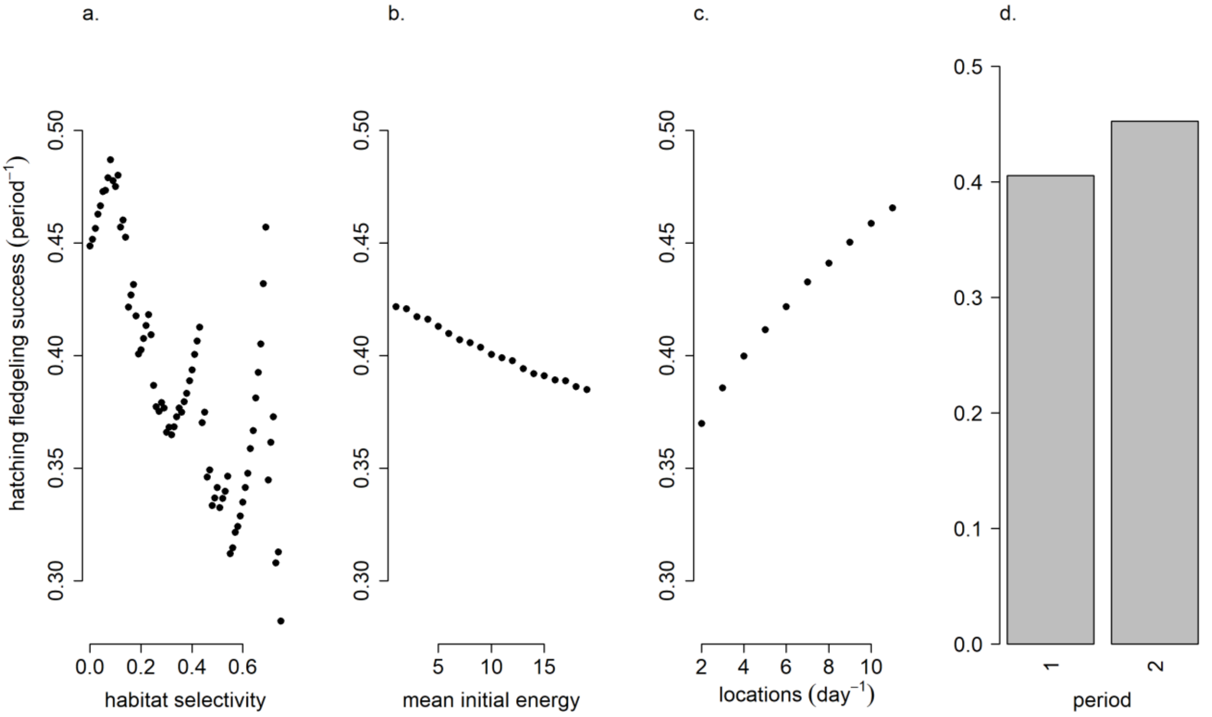

Figure 5-7 *The effect of changes in the parameters on the hatching-fledging probability for the scenario in which the Sandwich terns avoid OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-3, the default value of the hatching-fledging success is 0.301. The periods considered here are, 1: May-June; 2: April-July. For the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulation with 1000 birds were conducted.*

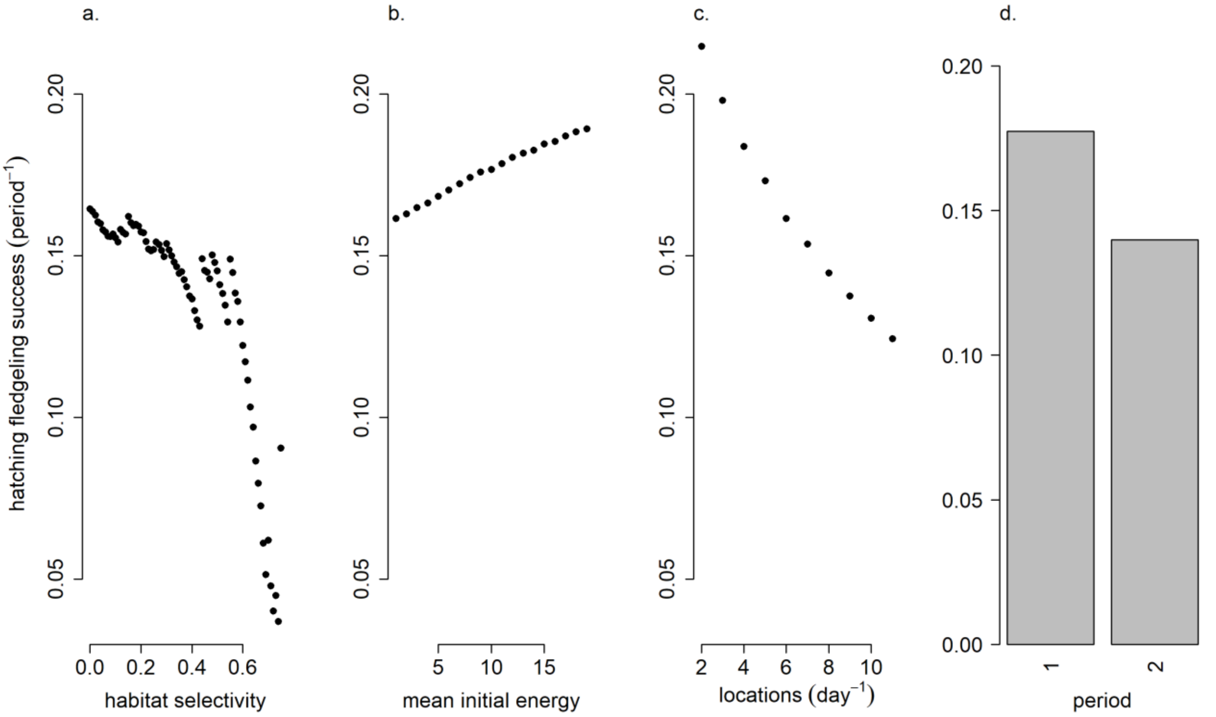

Figure 5-8 *The effect of changes in the parameters on the hatching-fledging probability in the scenario where the Sandwich tern does not forage in OWFs when they encounter them. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-3, the default value of the hatching-fledging success is 0.301. The periods considered here are, 1: May-June; 2: April-July, for the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulation with 1000 birds were conducted.*

5.4 Discussion

Originally, we aimed to include HALOMAR in the KEC assessment framework. However, we decided that the model is not yet sufficiently developed for this step. For some scenarios, the model predicts unrealistically high mortality values from displacement, which we cannot fully explain. Additionally, several key parameters are both highly influential and difficult to assign appropriate values to. This chapter describes the current status of the model and the developments achieved during the current project. Here, we briefly interpret the model results, and discuss the main model uncertainties and potential future steps for improvement.

Brief summary of the results

HALOMAR is a model based on individual-level assumptions regarding food intake, energy use and storage to estimate the effect of OWFs on the vital rates of marine birds. The model predicts strong negative effects on bird survival and nest success when we assume that birds skip foraging for a single time step whenever they encounter an OWF. This is understandable, as these interruptions strongly affect their mean food intake. However, in reality, birds adapt their behaviour quickly, therefore making continuous skipped foraging unlikely. We tested this extreme assumption in the model because, to our knowledge, there is no information available of the effect of displacement by OWFs on marine bird foraging behaviour or body condition. Therefore, we wanted to visualise the potential effect of an extreme. At the other end of the spectrum, we considered a scenario where displacement comes with no costs, and birds simply forage at alternative locations. In this case, we can see that the effect of OWFs largely depends on their placement in locations with high or low food availability. When OWFs decrease the mean food intake, even slightly, impacts on the survival or nesting success still occurs. For this second mechanism, the effect depends on how well bird densities correlate and translate to food availability and the extent to which birds can compensate by finding the same amount of food in foraging areas that are considered less optimal, as suggested by the bird distribution maps.

The sensitivity analysis of HALOMAR highlights that when birds avoid OWFs without any costs, the effects of OWFs strongly depends on habitat selectivity. With high habitat selectivity, the negative effect of the OWFs on northern gannet adult survival suddenly disappears. This finding illustrates the model's sensitivity to the exact overlap between OWFs and areas with high food availability. Moreover, in the model, the total area occupied by OWFs no longer really matters, but instead the mean food quality of the areas not covered by OWFs determines the effect. In case an additional OWF blocks a location with low food, leading to a higher mean food intake, it can actually lessen the effect of the other OWFs.

In the model, birds cannot compensate for periods with a lower food intake, which is likely not reflecting real-life behaviour. This illustrates, once again, the importance of a better understanding of bird foraging behaviour to refine the model. Which locations are most important for their food intake in which season? How do birds adjust their foraging behaviour in response to the presence of OWFs? Can they easily find their food elsewhere and thereby compensate for habitat loss? And, to what extent does food availability limit their survival?

If food intake is indeed proportional to bird densities in reality, and birds are not capable of finding additional food outside the OWF areas, our outcomes show that OWF displacement can affect food availability. For the northern gannet, we found that the overlap between OWF areas and the northern gannet distributions in the model led to a decrease in mean food intake with OWF displacement. In other words, OWFs seem to be planned in areas where high northern gannet densities are currently observed.

Bird density maps

The translation of bird densities to food availability is not straightforward. Unfortunately, bird observations are currently available at higher spatial resolution than data of their prey (Trifonova & Scott, 2023), making it challenging approximate prey availability for marine birds (see also section 4.3.3). While the bird distribution maps vary by bimonthly period but remain static across years, food landscapes are likely a dynamic environment and continuously changing in reality. The translation of bird densities to food availability and intake probably needs further work and could benefit from the inclusion of synoptic surveys of prey fish, birds, oceanography, and bird behaviour. For the current study, we decided on an approach in which the 20% of the map with the highest bird densities represents areas where birds can obtain their maximum food intake in a single time step. However, it is possible that more locations could also provide maximum food intake. We have not yet investigated the sensitivity of our outcomes to this assumption.

Density dependence

Right now, we do not consider any density dependent effects of displacement in the model. This is probably not a major issue as long as the number of displaced birds is limited. In such cases, birds may just redistribute themselves without a negative impact on their individual food intake. However, as larger areas become disturbed, it becomes more and more likely that bird densities will increase outside of the disturbed areas. This has already been shown for common guillemots in the German Bight (Peschko *et al.,* 2024). In areas where very high bird densities arise due to redistributions, it is likely that individuals will begin to experience direct competition for food. HALOMAR does not incorporate such density-dependent effects yet. For a further discussion of density-dependent effects of displacement, see sections 4.3.4 and 4.3.5.1.

Breeding and non-breeding

For northern gannets and common guillemots, we determined the non-breeding periods on observations of their presence at their breeding colonies and their (migratory) behaviour (Camphuysen, 2001; Dunn *et al.,* 2020a, 2020b; Fort *et al.,* 2012; Kubetzki *et al.,* 2009). In contrast, Sandwich terns breed in multiple colonies along the Dutch coast (Stienen, 2006; van Bemmelen *et al.,* 2022) and winter in warmer regions between France and South Africa (Fijn & Kemper, 2023). Therefore, unlike the northern gannet and common guillemot, we used the breeding period of Sandwich terns within HALOMAR. Since the breeding activity is well-monitored in the Netherlands, we were able to base the start and end dates of the breeding period on Sandwich tern presence at the Dutch colonies and the observed courtship period (Fijn *et al.,* 2014, 2017; Fijn & Kemper, 2023; Stienen, 2006; van Bemmelen *et al.,* 2022). While the quality of the observations for the Sandwich tern are somewhat better than for the other species, the observations for northern gannet, Sandwich tern, and common guillemot provide strong confidence that the chosen periods are representative. Additionally, the sensitivity analysis suggests that the model outcomes are not overly sensitive to the exact length of these periods. Generally, extending the season just results in a more pronounced effect in the same direction already observed.

The consideration of the breeding season for the Sandwich tern is a new implementation of the model. We have not explicitly account for central-place foraging behaviour, but instead restricted foraging to map areas close to the coast. We feel that this allowed best for comparability between species for the current step in model development. However, since Sandwich terns exhibit central-place foraging during the breeding season explicitly modelling this behaviour in future development steps would allow for a more realistic representation of their foraging areas use and the impacts of displacement.

While common guillemots are present in the North Sea year-round and Sandwich terns are seasonal inhabitants, the seasonal migration patterns of northern gannet are less well-defined. Based on the distribution maps, they are frequently observed in Dutch waters from August to November (*Figure 4-2* and *Figure 4-3*). Northern gannets can migrate widely during the non-breeding season, although the primary wintering sites are located in the North Sea and migrating long distances seems less common for older individuals and (Garthe *et al.,* 2024; Veron & Lawlor, 2009). Following the precautionary principle, we assume that northern gannets are exposed to the OWF areas in the Dutch North Sea throughout the entire non-breeding period. Yet this assumption may overestimate the effects, as gannets may spend part of the non-breeding season outside the area.

Foraging behaviour

The values for the number of foraging locations used by the species were based on indirect measures of behaviour, such as activity data and movement speed (Amélineau *et al.,* 2014, 2021; Dunn *et al.,* 2020b; Fijn *et al.,* 2017; Grecian *et al.,* 2018; Hamer *et al.,* 2000; Lane *et al.,* 2019; van Bemmelen *et al.,* 2022). These data likely provide a rough indication of the areas available for foraging. They informed our choice of values for the number of foraging locations parameter, and the sensitivity analysis shows that the model outcomes is minimally affected by this parameter unless unrealistically low values are used.

In addition to the number of foraging locations, foraging behaviour was influenced by the habitat selectivity parameter. Selecting a realistic value for this parameter was challenging. The bird density maps provide an indication of foraging locations, assuming that the birds spend more time in a location when foraging and are thus more likely to be observed there. Yet, the density maps do not give an idea about the number of locations with unsuccessful visits or about the underlying behavioural patterns. Habitat selectivity was implemented to explore the model's sensitivity to the assumption of random location selection beside the OWF areas. The sensitivity analysis showed that this parameter indeed strongly affects the model estimates. In the end, it would perhaps be better if we can implement more realistic patterns of behaviour, such that we can track individuals experience over a trajectory on the maps. Random weighted movement over limited regions, such as was previously implemented, resulted in unexplainable outcomes (Soudijn *et al.*, 2022b; van Kooten *et al.*, 2019). Perhaps more refined movement patterns based on empirical observations would overcome these difficulties also. For the Sandwich tern, the availability of several tracking studies for the breeding season (Fijn *et al.,* 2017; van Bemmelen *et al.,* 2022) should allow for the implementation of such adjustments. However, for northern gannets and common guillemots during the non-breeding season, no tracking data are currently available for the Dutch North Sea.

Energy budget

The modelled energy conditions of northern gannet and common guillemot show a decrease throughout the non-breeding season. It is, however, unclear how realistic this outcome is, as there is no published data available on energy condition through time for these species. This trend is a direct consequence of the decision to calibrate the maintenance costs to match the food intake from the bird density maps. To further improve the model, it may be beneficial to make the energy budget of the model more realistic, allowing the same type of simulations without forcing the energy conditions to decrease.

The mean initial energy values used in the model were educated guesses, and the sensitivity analysis shows that this parameter is highly influential in the model. Predictions may vary a lot depending on the chosen initial energy values. Unfortunately, information about the energetic condition of these birds is scarce. The only relevant study we found used a mechanistic model, estimating that common guillemots can sustain on average 8.1 (±0.1) days of fasting under cyclonic conditions off North Newfoundland (Clairbaux *et al.,* 2021). No comparable information was available for northern gannets or Sandwich terns.

We used a left-skewed distribution for the initial energy of all three species in HALOMAR. This is based on the notion that some individuals are likely to be 'weak' or 'unlucky' and die regardless of external conditions, consistent with observations of numerous dead common guillemots that are washed ashore each winter (Camphuysen, 2022). Supporting this, a study from Great Saltee Island, Wexford, Ireland found a skewed distribution of body condition in breeding northern gannets during the late chick-rearing period in September

(Fitzgerald et al 2022) – just two months before the northern gannet simulation period in HALOMAR. In contrast, the body mass of common tern chicks (*Sterna hirundo*) on the German Wadden Sea island Neufelderkoog appeared to follow a normal distribution (Thyen *et al.,* 1998). While we did not conduct a sensitivity analysis on the assumed skewness during this project, the effects of the initial conditions on the model predictions highlight the need to investigate this parameter further.

Future steps

At this stage, HALOMAR is not yet suitable for use in an assessment framework. We need to consider the importance of several components of the model to improve its applicability:

1. The consideration of an actual energy budget, where maintenance costs and food intake are explicitly accounted for, represented by energetic parameters that can be based on published studies on bird energetics.
2. The choices in the convergence of the bird distributions to food availability maps. How do we rescale bird densities to food availabilities? And which bird density values are potentially representative for a maximum food intake?
3. The consideration of movement restricted to a part of the map rather than the full area. Can we find information that informs us about the movements of the birds outside the breeding season? Can we start with the Sandwich tern central-place foraging behaviour for the breeding season to investigate the effect of spatial restrictions in the foraging locations?
4. A sensitivity analysis of some additional components in the model, such as the initial energy distribution. Data of weight distributions of Sandwich terns during ringing in the breeding season and weights and condition scores of stranded birds (common guillemot) in the non-breeding season could be used as a basis for more realistic choices in the shapes of these distributions.

6 Quality Assurance

Wageningen Marine Research utilises an ISO 9001:2015 certified quality management system. The organisation has been certified since 27 February 2001. The certification was issued by DNV.

References

Aitken, D., Babcock, M., Barratt, A., Clarkson, C., & Prettyman, S. (2017). *Flamborough and Filey Coast pSPA Seabird Monitoring Programme 2017*. RSPB Bempton Cliffs.

Aitken, D., Babcock, M., Clarkson, K., & Hibbert, T. (2013). *Flamborough Head and Bempton Cliffs SPA Seabird Monitoring Programme 2013 Report*. RSPB Bempton Cliffs.

Amélineau, F., Merkel, B., Tarroux, A., Descamps, S., Anker-Nilssen, T., Bjørnstad, O., Bråthen, V., Chastel, O., Christensen-Dalsgaard, S., Danielsen, J., Daunt, F., Dehnhard, N., Ekker, M., Erikstad, K., Ezhov, A., Fauchald, P., Gavrilo, M., Hallgrimsson, G., Hansen, E., … Strøm, H. (2021). Six pelagic seabird species of the North Atlantic engage in a fly-and-forage strategy during their migratory movements. *Marine Ecology Progress Series*, *676*, 127–144. https://doi.org/10.3354/meps13872

Amélineau, F., Péron, C., Lescroël, A., Authier, M., Provost, P., & Grémillet, D. (2014). Windscape and tortuosity shape the flight costs of northern gannets. *Journal of Experimental Biology*, *217*(6), 876–885. https://doi.org/10.1242/jeb.097915

Anderson Hansen, K., Hernandez, A., Mooney, T. A., Rasmussen, M. H., Sørensen, K., & Wahlberg, M. (2020). The common murre (Uria aalge), an auk seabird, reacts to underwater sound. *The Journal of the Acoustical Society of America*, *147*(6), 4069–4074. https://doi.org/10.1121/10.0001400

Anker-Nilssen, T., Hanssen, S. A., Moe, B., Barrett, R., Bustnes, J. O., Christensen-Dalsgaard, S., Dehnhard, N., Descamps, S., Erikstad, K.-E., Langset, M., Layton-Matthews, K., Lorentzen, E., Reiertsen, T. K., & Strøm, H. (2022). Key-site monitoring in Norway 2021, including Svalbard and Jan Mayen. *Seapop Short Report*, *2022*(1).

Babcock, M., Aitken, D., Jackson, S., & Clarkson, K. (2015). *Flamborough and Filey Coast pSPA Seabird Monitoring Programme 2015* (2015 Report). RSPB Bempton Cliffs.

Babcock, M., Aitken, D., Lloyd, I., Wischnewski, S., Baker, R., Duffield, H., & Barratt, A. (2018). *Flamborough and Filey Coast SPA Seabird Monitoring Programme 2018* (2018 Report). RSPB Bempton Cliffs.

Bijlsma, R., Hustings, F., & Camphuysen, C. (2001). *Algemene en schaarse vogels van Nederland* (Avifauna van Nederland 2). KNNV Uitgeverij, GMB Uitgeverij.

Billerman, S. M., Keeney, B. K., Rodewald, P. G., & Schulenberg, T. S. (Eds.). (2022). *Birds of the World*. Cornell Laboratory of Ornithology. https://birdsoftheworld.org/bow/home

Bradbury, G., Trinder, M., Furness, B., Banks, A. N., Caldow, R. W. G., & Hume, D. (2014). Mapping seabird sensitivity to offshore wind farms. *PLoS ONE*, *9*(9), e106366. https://doi.org/10.1371/journal.pone.0106366

Brenninkmeijer, A., & Stienen, E. W. M. (1992). *Ecologischprofielvandegrote stern (Sterna sandvicensis)* (R.I.N-Rapport 92/17). DLO-Insituut voor Bos- en Natuuronderzoek.

Burnell, D., Perkins, A., Newton, S., Bolton, M., Tierney, T. D., & Dunn, T. E. (2023). *Seabirds Count A census of breeding seabirds in Britain and Ireland (2015–2021)*. Lynx Nature Books.

Camphuysen, C. (2001). Northern Gannets Morus bassanus found dead in The Netherlands, 1970-2000. *Atlantic Seabirds*, *3*(1), 15–30.

Camphuysen, C. (2022). Mission accomplished: Chronic North Sea oil pollution now at acceptable levels, with Common Guillemots as sentinels. *Seabird Journal*, *34*. https://doi.org/10.61350/sbj.34.1

Camphuysen, C., & Webb, A. (1999). Multi-species feeding associations in North Sea Seabirds: Jointly exploiting a patchy environment. *Ardea*, *87*.

Carroll, M. J., Bolton, M., Owen, E., Anderson, G. Q. A., Mackley, E. K., Dunn, E. K., & Furness, R. W. (2017). Kittiwake breeding success in the southern North Sea correlates with prior sandeel fishing mortality. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*(6), 1164–1175. https://doi.org/10.1002/aqc.2780

Caswell, H. (2001). *Matrix population models: Construction analysis and interpretation* (2nd ed.). Sinauer Associates.

Clairbaux, M., Mathewson, P., Porter, W., Fort, J., Strøm, H., Moe, B., Fauchald, P., Descamps, S., Helgason, H. H., Bråthen, V. S., Merkel, B., Anker-Nilssen, T., Bringsvor, I. S., Chastel, O., Christensen-Dalsgaard, S., Danielsen, J., Daunt, F., Dehnhard, N., Erikstad, K. E., … Grémillet, D. (2021). North

Atlantic winter cyclones starve seabirds. *Current Biology*, *31*(17), Article 17. https://doi.org/10.1016/j.cub.2021.06.059

Clarkson, K., Aitken, D., Cope, R., & O'Hara, D. (2022). *Flamborough & Filey Coast SPA: 2022 seabird colony count and population trends*. RSPB.

Cope, R., Aitken, D., & O'Hara, D. (2021). *Flamborough and Filey Coast SPA Seabird Monitoring Programme 2021* (2021 Report). RSPB Bempton Cliffs.

Croll, D. A., Ellis, A. A., Adams, J., Cook, A. S. C. P., Garthe, S., Goodale, M. W., Hall, C. S., Hazen, E., Keitt, B. S., Kelsey, E. C., Leirness, J. B., Lyons, D. E., McKown, M. W., Potiek, A., Searle, K. R., Soudijn, F. H., Rockwood, R. C., Tershy, B. R., Tinker, M., … Zilliacus, K. (2022). Framework for assessing and mitigating the impacts of offshore wind energy development on marine birds. *Biological Conservation*, *276*, 109795. https://doi.org/10.1016/j.biocon.2022.109795

Daunt, F., Fang, Z., Howells, R., Harris, M. P., Wanless, S., Searle, K., & Elston, D. (2020). Improving estimates of seabird body mass survival relationships. *Scottish Marine and Freshwater Science*, *11*(13), CR/2018/12.

Daunt, F., & Mitchell, I. (2013). Impacts of climate change on seabirds [Pdf]. *MCCIP Science Review 2013*, 9 pages. https://doi.org/10.14465/2013.ARC14.125-133

Dierschke, J., Dierschke, V., Schmaljohann, H., & Stühmer, F. (2016a). Ornithologischer Jahresbericht 2015 für Helgoland. *Ornithologischer Jahresbericht*, *26*, 3–83.

Dierschke, J., Dierschke, V., Schmaljohann, H., & Stühmer, F. (2017). Ornithologischer Jahresbericht 2016 für Helgoland. *Ornithologischer Jahresbericht*, *27*, 1–97.

Dierschke, J., Dierschke, V., Schmaljohann, H., & Stühmer, F. (2018). Ornithologischer Jahresbericht 2017 für Helgoland. *Ornithologischer Jahresbericht*, *28*, 1–96.

Dierschke, J., Dierschke, V., Schmaljohann, H., & Stühmer, F. (2019). Ornithologischer Jahresbericht 2018 für Helgoland. *Ornithologischer Jahresbericht*, *29*, 1–100.

Dierschke, J., Dierschke, V., & Stühmer, F. (2020). Ornithologischer Jahresbericht 2019 für Helgoland. *Ornithologischer Jahresbericht*, *30*, 1–97.

Dierschke, J., Dierschke, V., & Stühmer, F. (2021). Ornithologischer Jahresbericht 2020 für Helgoland. *Ornithologischer Jahresbericht*, *31*, 1–89.

Dierschke, V., Furness, R., & Garthe, S. (2016b). Seabirds and offshore wind farms in European waters: Avoidance and attraction. *Biological Conservation*, *202*, 59–68. https://doi.org/10.1016/j.biocon.2016.08.016

Drewitt, A. L., & Langston, R. H. W. (2006). Assessing the impacts of wind farms on birds: Impacts of wind farms on birds. *Ibis*, *148*, 29–42. https://doi.org/10.1111/j.1474-919X.2006.00516.x

Duarte, C. M., Chapuis, L., Collin, S. P., Costa, D. P., Devassy, R. P., Eguiluz, V. M., Erbe, C., Gordon, T. A. C., Halpern, B. S., Harding, H. R., Havlik, M. N., Meekan, M., Merchant, N. D., Miksis-Olds, J. L., Parsons, M., Predragovic, M., Radford, A. N., Radford, C. A., Simpson, S. D., … Juanes, F. (2021). The soundscape of the Anthropocene ocean. *Science*, *371*(6529), eaba4658. https://doi.org/10.1126/science.aba4658

Dunn, R. E., Green, J. A., Wanless, S., Harris, M. P., Newell, M. A., Bogdanova, M. I., Horswill, C., Daunt, F., & Matthiopoulos, J. (2020a). Modelling and mapping how common guillemots balance their energy budgets over a full annual cycle. *Functional Ecology*, *36*(7), 1612–1626. https://doi.org/10.1111/1365-2435.14059

Dunn, R. E., Wanless, S., Daunt, F., Harris, M. P., & Green, J. A. (2020b). A year in the life of a North Atlantic seabird: Behavioural and energetic adjustments during the annual cycle. *Scientific Reports*, *10*(1), 5993. https://doi.org/10.1038/s41598-020-62842-x

Exo, K.-M., Huppop, O., & Garthe, S. (2003). Birds and offshore wind farms: A hot topic in marine ecology. *Wader Study Group Bulletin*, *100*(April), 50–53.

Fijn, R. C., De Jong, J., Courtens, W., Verstraete, H., Stienen, E. W. M., & Poot, M. J. M. (2017). GPS-tracking and colony observations reveal variation in offshore habitat use and foraging ecology of breeding Sandwich Terns. *Journal of Sea Research*, *127*, 203–211. https://doi.org/10.1016/j.seares.2016.11.005

Fijn, R. C., & Kemper, J. (2023). Sandwich Tern feeds juvenile on wintering grounds in southern Namibia. *Ardea*, *111*, 549–553. https://doi.org/10.5253/arde.2023.a2

Fijn, R. C., Thaxter, C. B., Aarts, G., Adema, J., Middelveld, R. P., & Van Bemmelen, R. S. A. (2022). Relative effects of static and dynamic abiotic conditions on foraging behaviour in breeding Sandwich terns. *Marine Ecology Progress Series*, *692*, 137–150. https://doi.org/10.3354/meps14076

Fijn, R. C., Wolf, P., Courtens, W., Verstraete, H., Stienen, E. W. M., Iliszko, L., & Poot, M. J. M. (2014). Post-breeding prospecting trips of adult Sandwich Terns *Thalasseus sandvicensis*. *Bird Study*, *61*(4), 566–571. https://doi.org/10.1080/00063657.2014.950942

Fitzgerald, M., Lynch, S. A., & Jessopp, M. (2022). Breeding stage impacts on chronic stress and physiological condition in northern gannets (Morus bassanus). *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology*, *274*, 111305. https://doi.org/10.1016/j.cbpa.2022.111305

Fort, J., Pettex, E., Tremblay, Y., Lorentsen, S.-H., Garthe, S., Votier, S., Pons, J. B., Siorat, F., Furness, R. W., Grecian, W. J., Bearhop, S., Montevecchi, W. A., & Grémillet, D. (2012). Meta-population evidence of oriented chain migration in northern gannets (Morus bassanus). *Frontiers in Ecology and the Environment*, *10*(5), Article 5. https://doi.org/10.1890/110194

Frederiksen, M., Edwards, M., Mavor, R., & Wanless, S. (2007). Regional and annual variation in Black-legged Kittiwake breeding productivity is related to sea surface temperature. *Marine Ecology Progress Series*, *350*, 137–143. https://doi.org/10.3354/meps07126

Frederiksen, M., Wanless, S., Harris, M. P., Rothery, P., & Wilson, L. J. (2004). The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. *Journal of Applied Ecology*, *41*(6), 1129–1139. https://doi.org/10.1111/j.0021-8901.2004.00966.x

Furness, R. W., Hallgrimsson, G. T., Montevecchi, W. A., Fifield, D., Kubetzki, U., Mendel, B., & Garthe, S. (2018). Adult Gannet migrations frequently loop clockwise around Britain and Ireland. *Ringing & Migration*, *33*(1), 45–53. https://doi.org/10.1080/03078698.2018.1472971

Furness, R. W., Wade, H. M., & Masden, E. A. (2013). Assessing vulnerability of marine bird populations to offshore wind farms. *Journal of Environmental Management*, *119*, 56–66. https://doi.org/10.1016/j.jenvman.2013.01.025

Garthe, S., Ludynia, K., Hüppop, O., Kubetzki, U., Meraz, J. F., & Furness, R. W. (2012). Energy budgets reveal equal benefits of varied migration strategies in northern gannets. *Marine Biology*, *159*(9), 1907–1915. https://doi.org/10.1007/s00227-012-1978-6

Garthe, S., Peschko, V., Fifield, D. A., Borkenhagen, K., Nyegaard, T., & Dierschke, J. (2024). Migratory pathways and winter destinations of Northern Gannets breeding at Helgoland (North Sea): Known patterns and increasing importance of the Baltic Sea. *Journal of Ornithology*, *165*(4), 869–880. https://doi.org/10.1007/s10336-024-02192-x

Garthe, S., Schwemmer, H., Markones, N., Müller, S., & Schwemmer, P. (2015). Verbreitung, Jahresdynamik und Bestandsentwicklung der Seetaucher Gavia spec. In der Deutschen Bucht (Nordsee). *Vogelwarte*, *53*.

Garthe, S., Schwemmer, H., Peschko, V., Markones, N., Müller, S., Schwemmer, P., & Mercker, M. (2023). Large-scale effects of offshore wind farms on seabirds of high conservation concern. *Scientific Reports*, *13*(1), Article 1. https://doi.org/10.1038/s41598-023-31601-z

Goodship, N., & Furness, R. (2024). *NatureScot Research Report 1283 - Disturbance Distances Review: An updated literature review of disturbance distances of selected bird species* (NatureScot Research Report 1283; Issue NatureScot Research Report 1283.). Mac Arthur Green.

Grecian, W. J., Lane, J. V., Michelot, T., Wade, H. M., & Hamer, K. C. (2018). Understanding the ontogeny of foraging behaviour: Insights from combining marine predator bio-logging with satellite-derived oceanography in hidden Markov models. *Journal of the Royal Society Interface*, *15*(143). https://dx.doi.org/10.6084/m9.figshare.c.4105319

Green, R. E., Langston, R. H. W., McCluskie, A., Sutherland, R., & Wilson, J. D. (2016). Lack of sound science in assessing wind farm impacts on seabirds. *Journal of Applied Ecology*, *53*(6), 1635–1641. https://doi.org/10.1111/1365-2664.12731

Grundlehner, A., & Leopold, M. (2024). *Habitatverlies van alken en zeekoeten in en rondom offshore windpark Gemini: Onderzoek naar de ruimtelijke verspreiding van zeevogels middels digitale vliegtuig surveys in een groot referentiegebied en een robuuste statistische analyse in R-INLA* (Wageningen Marine Research report C083/23a). Wageningen Marine Research. https://doi.org/10.18174/643585

Hamer, K., Phillips, R., Wanless, S., Harris, M., & Wood, A. (2000). Foraging ranges, diets and feeding locations of gannets Morus bassanus in the North Sea:evidence from satellite telemetry. *Marine Ecology Progress Series*, *200*, 257–264. https://doi.org/10.3354/meps200257

Harris, M. P., Wanless, S., Ballesteros, M., Moe, B., Daunt, F., & Erikstad, K. E. (2015). Geolocators reveal an unsuspected moulting area for Isle of May Common Guillemots Uria aalge. *Bird Study*, *62*(2), 267–270. https://doi.org/10.1080/00063657.2015.1006164

Heppell, S. S., Crouse, D. T., & Crowder, L. B. (2000). Using matrix models to focus research and management efforts in conservation. In *Quantitative Methods for Conservation Biology* (pp. 148–168). Springer-Verlag. https://doi.org/10.1007/0-387-22648-6_10

Hin, V., IJntema, G., van Kooten, T., & Potiek, A. (2024). *A revised methodology for quantifying 'Acceptable Level of Impact' from offshore wind farms on seabird populations* (Wageningen Marine Research report C034/24). Wageningen Marine Research. https://doi.org/10.18174/659871

Hin, V., Kooten, T. van, Potiek, A., & Kraal, J. (2023). *Further development of the 'Acceptable Level of Impact' framework for effects of offshore wind farms on seabirds: Extended method development and sensitivity analysis* (Wageningen Marine Research report C088/23). Wageningen Marine Research. https://doi.org/10.18174/644082

Hin, V., & Soudijn, F. H. (2021). *Feasibility study of the development of population models for the northern fulmar (Fulmarus glacialis) and the Atlantic puffin (Fratercula arctica)* (Wageningen Marine Research report C046/21). Wageningen Marine Research. https://doi.org/10.18174/546670

Horswill, C., Miller, J. A. O., & Wood, M. J. (2022). Impact assessments of wind farms on seabird populations that overlook existing drivers of demographic change should be treated with caution. *Conservation Science and Practice*, *4*(4), Article 4. https://doi.org/10.1111/csp2.12644

Horswill, C., O'Brien, S. H., & Robinson, R. A. (2017). Density dependence and marine bird populations: Are wind farm assessments precautionary? *Journal of Applied Ecology*, *54*(5), 1406–1414. https://doi.org/10.1111/1365-2664.12841

Horswill, C., & Robinson, R. A. (2015). *Review of seabird demographic rates and density dependence* (JNCC Report No. 552; Issue JNCC Report No. 552).

IJntema, G. J., Heida, N., Leemans, J. J., Gyimesi, A., & Potiek, A. (2025). *Collision effects of North Sea wind turbines on bird species within the "Kader Ecologie & Cumulatie (KEC) 5.0"* (Waardenburg Ecology Report 24-163). Waardenburg Ecology.

Jeglinski, J. W. E., Wanless, S., Murray, S., Barrett, R. T., Gardarsson, A., Harris, M. P., Dierschke, J., Strøm, H., Lorentsen, S.-H., & Matthiopoulos, J. (2023). Metapopulation regulation acts at multiple spatial scales: Insights from a century of seabird colony census data. *Ecological Monographs*, *93*(2), e1569. https://doi.org/10.1002/ecm.1569

Jenouvrier, S., Barbraud, C., Cazelles, B., & Weimerskirch, H. (2005). Modelling population dynamics of seabirds: Importance of the effects of climate fluctuations on breeding proportions. *Oikos*, *108*(3), 511–522. https://doi.org/10.1111/j.0030-1299.2005.13351.x

Jetten, R. (2024, April 25). *Kamerbrief over update aanvullende routekaart wind op zee*. https://www.rijksoverheid.nl/documenten/kamerstukken/2024/04/25/update-aanvullende-routekaart-wind-op-zee

JNCC. (2022). *Joint SNCB Interim Displacement Advice Note 2022*. Https://Hub.Jncc.Gov.Uk/Assets/9aecb87c-80c5-4cfb-9102-39f0228dcc9a.

Knief, U., Bregnballe, T., Alfarwi, I., Ballmann, M. Z., Brenninkmeijer, A., Bzoma, S., Chabrolle, A., Dimmlich, J., Engel, E., Fijn, R., Fischer, K., Hälterlein, B., Haupt, M., Hennig, V., Herrmann, C., Veld, R. in 't, Kirchhoff, E., Kristersson, M., Kühn, S., … Courtens, W. (2024). Highly pathogenic avian influenza causes mass mortality in Sandwich Tern Thalasseus sandvicensis breeding colonies across north-western Europe. *Bird Conservation International*, *34*, e6. https://doi.org/10.1017/S0959270923000400

Kubetzki, U., Garthe, S., Fifield, D., Mendel, B., & Furness, R. (2009). Individual migratory schedules and wintering areas of northern gannets. *Marine Ecology Progress Series*, *391*, 257–265. https://doi.org/10.3354/meps08254

Lane, J. V., Jeglinski, J. W. E., Avery-Gomm, S., Ballstaedt, E., Banyard, A. C., Barychka, T., Brown, I. H., Brugger, B., Burt, T. V., Careen, N., Castenschiold, J. H. F., Christensen-Dalsgaard, S., Clifford, S., Collins, S. M., Cunningham, E., Danielsen, J., Daunt, F., D'entremont, K. J. N., Doiron, P., … Votier, S. C. (2024). High pathogenicity avian influenza (H5N1) in Northern Gannets (Morus bassanus):

Global spread, clinical signs and demographic consequences. *Ibis*, *166*(2). https://doi.org/10.1111/ibi.13275

Lane, J. V., Pollock, C. J., Jeavons, R., Sheddan, M., Furness, R. W., & Hamer, K. C. (2021). Post-fledging movements, mortality and migration of juvenile northern gannets. *Marine Ecology Progress Series*, *671*, 207–218. https://doi.org/10.3354/meps13804

Lane, J. V., Spracklen, D., & Hamer, K. (2019). Effects of windscape on three-dimensional foraging behaviour in a wide-ranging marine predator, the northern gannet. *Marine Ecology Progress Series*, *628*, 183–193. https://doi.org/10.3354/meps13089

Layton-Matthews, K., Buckingham, L., Critchey, E. J., Nilsson, A. L. K., Ollus, V. M. S., Ballesteros, M., Christensen-Dalsgaard, S., Denhard, N., Fauchauld, P., Hanssen, F., Helberg, M., Masden, E., May, R. F., Sandvik, H., Tarroux, A., & Reiertsen, T. K. (2023). *Development of a Cumulative Impact Assessment tool for birds in Norwegian Off-shore Waters: Trollvind OWF as a case study* (NINA Report 2295). Norwegian Institute for Nature Research.

Leopold, M. F. (2017). *Seabirds? What seabirds? An exploratory study into the origin of seabirds visiting the SE North Sea and their survival bottlenecks.* (Wageningen Marine Research report C046/17). Wageningen Marine Research. https://doi.org/10.18174/416194

Leopold, M. F., Boonman, M., Collier, M. P., Davaasuren, N., Fijn, R. C., Gyimesi, A., de Jong, J., Jongbloed, R. H., Jonge Poerink, B., Kleyheeg-Hartman, J. C., Krijgsveld, K. L., Lagerveld, S., Lensink, R., Poot, M. J. M., J.T, van der W., & Scholl, M. (2014). *A first approach to deal with cumulative effects on birds and bats of offshore wind farms and other human activities in the Southern North Sea* (IMARES Report C166/14). IMARES.

Leopold, M. F., & Overmaat, W. (2023). Zeekoeten verdrinken in staand want: Nieuwe kansen voor dieetonderzoek? *Limosa*, *96*(4), 174–187.

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*(3), 035101. https://doi.org/10.1088/1748-9326/6/3/035101

Lloyd, I., & Aitken, D. (2019). *Flamborough and Filey Coast SPA Seabird Monitoring Programme 2019* (2019 Report). RSPB Bempton Cliffs.

Lloyd, I., Aitken, D., & O’Hara, D. (2020). *Flamborough and Filey Coast SPA Seabird Monitoring Programme* (2020 Report). RSPB Bempton Cliffs.

Masden, E. A., Haydon, D. T., Fox, A. D., & Furness, R. W. (2010). Barriers to movement: Modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. *Marine Pollution Bulletin*, *60*(7), 1085–1091. https://doi.org/10.1016/j.marpolbul.2010.01.016

May, R., Nygård, T., Falkdalen, U., Åström, J., Hamre, Ø., & Stokke, B. G. (2020). Paint it black: Efficacy of increased wind turbine rotor blade visibility to reduce avian fatalities. *Ecology and Evolution*, *10*(16), 8927–8935. https://doi.org/10.1002/ECE3.6592

Mead, C. J. (1974). The Results of Ringing Auks in Britain and Ireland. *Bird Study*, *21*(1), 45–86. https://doi.org/10.1080/00063657409476401

Mendel, B., Schwemmer, P., Peschko, V., Müller, S., Schwemmer, H., Mercker, M., & Garthe, S. (2019). Operational offshore wind farms and associated ship traffic cause profound changes in distribution patterns of Loons (Gavia spp.). *Journal of Environmental Management*, *231*(October 2018), 429–438. https://doi.org/10.1016/j.jenvman.2018.10.053

Mercker, M., Dierschke, V., Camphuysen, K., Kreutle, A., Markones, N., Vanermen, N., & Garthe, S. (2021). An indicator for assessing the status of marine-bird habitats affected by multiple human activities: A novel statistical approach. *Ecological Indicators*, *130*, 108036. https://doi.org/10.1016/J.ECOLIND.2021.108036

Merrall, E., Green, J. A., Robinson, L. A., Butler, A., Wood, M. J., Newell, M. A., Black, J., Daunt, F., & Horswill, C. (2024). Incorporating density-dependent regulation into impact assessments for seabirds. *Journal of Applied Ecology*, *61*(10), 2510–2524. https://doi.org/10.1111/1365-2664.14750

Miller, J. A. O., Furness, R. W., Trinder, M., & Matthiopoulos, J. (2019). The sensitivity of seabird populations to density-dependence, environmental stochasticity and anthropogenic mortality. *Journal of Applied Ecology*, *56*(9), Article 9. https://doi.org/10.1111/1365-2664.13448

MMO. (2018). *Displacement and habituation of seabirds in response to marine activities. A report produced for the Marine Management Organisation.* (MMO 1139). Natural Power.

Mooney, T. A., Smith, A., Larsen, O. N., Hansen, K. A., & Rasmussen, M. (2020). A field study of auditory sensitivity of the Atlantic puffin, Fratercula arctica. *Journal of Experimental Biology*, *223*(15), jeb228270. https://doi.org/10.1242/jeb.228270

Newman, L., Blockley, F., Hewitt, J., & Wood, M. (2021). *Seabird monitoring on Skomer Island in 2021* (Skomer Seabird Report 2021).

O'Brien, S. H., Cook, A. S. C. P., & Robinson, R. A. (2017). Implicit assumptions underlying simple harvest models of marine bird populations can mislead environmental management decisions. *Journal of Environmental Management*, *201*, 163–171. https://doi.org/10.1016/j.jenvman.2017.06.037

Peschko, V., Mendel, B., Mercker, M., Dierschke, J., & Garthe, S. (2021). Northern gannets (*Morus bassanus*) are strongly affected by operating offshore wind farms during the breeding season. *Journal of Environmental Management*, *279*, 111509. https://doi.org/10.1016/j.jenvman.2020.111509

Peschko, V., Mendel, B., Müller, S., Markones, N., Mercker, M., & Garthe, S. (2020). Effects of offshore windfarms on seabird abundance: Strong effects in spring and in the breeding season. *Marine Environmental Research*, *162*, 105157. https://doi.org/10.1016/j.marenvres.2020.105157

Peschko, V., Schwemmer, H., Mercker, M., Markones, N., Borkenhagen, K., & Garthe, S. (2024). Cumulative effects of offshore wind farms on common guillemots (Uria aalge) in the southern North Sea—Climate versus biodiversity? *Biodiversity and Conservation*, *33*, 9479–9970. https://doi.org/10.1007/s10531-023-02759-9

Petersen, I. K., & Fox, A. D. (2007). *Changes in bird habitat utilisation around the Horns Rev 1 offshore wind farm, with particular emphasis on Common Scoter*. University of Aarhus.

Pollock, C. J., Lane, J. V., Buckingham, L., Garthe, S., Jeavons, R., Furness, R. W., & Hamer, K. C. (2021). Risks to different populations and age classes of gannets from impacts of offshore wind farms in the southern North Sea. *Marine Environmental Research*, *171*, 105457. https://doi.org/10.1016/j.marenvres.2021.105457

Poot, M. J. M., Fijn, R. C., Boudewijn, T. J., De Jong, J., van Horssen, P. W., Japink, M., van de Boogaard, B., Bergsma, J., Stienen, E. W. M., Courtens, W., Vanermen, N., Verstraete, H., Wolf, P. A., Hoekstein, M. S. J., & Lilipaly, S. J. (2014). *PMR monitoring natuurcompensatie Voordelta—Eindrapport 1e fase 2009-2013 Deel B: Perceel Vogels – Grote stern en visdief* (Deltares report 1200672-000-ZKS-0043). Deltares.

Poot, M. J. M., Horssen, P. van W. V., Collier, M. P., Lensink, R., & Dirksen, S. (2011). *Effect studies offshore wind Egmond aan Zee: Cumulative effects on seabirds* (Bureau Waardenburg report 11-026; p. 220). Bureau Waardenburg.

Potiek, A., IJntema, G. J., Kooten, T. van, Leopold, M. F., & Collier, M. P. (2022). *Acceptable Levels of Impact from offshore wind farms on the Dutch Continental Shelf for 21 bird: A novel approach for defining acceptable levels of additional mortality from turbine collisions and avoidance-induced habitat loss* (Bureau Waardenburg Report 21-0120). Bureau Waardenburg.

Potiek, A., Vanermen, N., Middelveld, R. P., de Jong, J., Stienen, E., & Fijn, R. C. (2019). *Spatial and temporal distribution of different age classes of seabirds in the North Sea. Analysis of ESAS database.* (Bureau Waardenburg report 19-129). Bureau Waardenburg.

Rijks, J. M., Leopold, M. F., Kühn, S., Veld, R. in 't, Schenk, F., Brenninkmeijer, A., Lilipaly, S. J., Ballmann, M. Z., Kelder, L., Jong, J. W. de, Courtens, W., Slaterus, R., Kleyheeg, E., Vreman, S., Kik, M. J., Gröne, A., Fouchier, R. A., Engelsma, M., Jong, M. C. de, … Beerens, N. (2022). Mass Mortality Caused by Highly Pathogenic Influenza A(H5N1) Virus in Sandwich Terns, the Netherlands, 2022. *Emerging Infectious Diseases*, *28*(12), 2538. https://doi.org/10.3201/eid2812.221292

Schekkerman, H., Arts, F., Buijs, R.-J., Courtens, W., van Daele, T., Fijn, R., van Kleunen, A., van der Jeugd, H., Roodenberg, M., Stienen, E., de Vries, L., & Ens, B. (2021). *Geïntegreerde populatie-analyse van vijf soorten kustbroedvogels in het Zuidwestelijk Deltagebied* (Sovon-rapport 2021/03 CAPS-rapport 2021/01). Sovon Vogelonderzoek Nederland.

Seabird Monitoring Programme database. (2024, April 4). SMP, BTO. https://app.bto.org/seabirds/public/data.jsp

Searle, K., Butler, A., Mobbs, D., Trinder, M., McGregor, R., Cook, A., McCluskie, A., Caneco, B., & Daunt, F. (2022a). *Study to Examine how Seabird Collision Risk, Displacement and Barrier Effects Could be Integrated for Assessment of Offshore Wind Developments* (Marine Scotland Science ITQ-0246). UK Centre for Ecology & Hydrology.

Searle, K., Butler, A., Waggitt, J., Evans, P., Quinn, L., Bogdanova, M., Evans, T., Braithwaite, J., & Daunt, F. (2022b). Potential climate-driven changes to seabird demography: Implications for assessments of marine renewable energy development. *Marine Ecology Progress Series*, *690*, 185–200. https://doi.org/10.3354/meps14045

Searle, K., Butler, A., Waggitt, J. J., Evans, P. G. H., Bogdanova, M. I., Hobbs, N. T., Daunt, F., & Wanless, S. (2022c). Opposing effects of spatiotemporal variation in resources and temporal variation in climate on density dependent population growth in seabirds. *Journal of Animal Ecology*, *91*(12), 2384–2399. https://doi.org/10.1111/1365-2656.13819

Searle, K., Mobbs, D., Butler, A., Bogdanova, M., Freeman, S., Wanless, S., & Daunt, F. (2014). *Population Consequences of Displacement from Proposed Offshore Wind Energy Developments for Seabirds Breeding at Scottish SPAs* (Marine Scotland Science CR/2012/03).

Searle, K., Mobbs, D. C., Butler, A., Furness, R. W., Mcgregor, H., & Evans, T. (2018). *Finding out the Fate of Displaced Birds (FCR/2015/19): Scottish Marine and Freshwater Science Vol 9 No 8*. https://doi.org/10.7489/12118-1

Searle, K., O’Brien, S. H., Jones, E. L., Cook, A. S. C. P., Trinder, M. N., McGregor, R. M., Donovan, C., McCluskie, A., Daunt, F., & Butler, A. (2023). A framework for improving treatment of uncertainty in offshore wind assessments for protected marine birds. *ICES Journal of Marine Science*, fsad025. https://doi.org/10.1093/icesjms/fsad025

Soudijn, F. H., Chen, C., Potiek, A., & Donk, S. van. (2022a). *Density maps of the herring gull for the Dutch continental shelf: Memo to supplement the seabird assessment reports within KEC ("Kader Ecologie en Cumulatie")*. https://doi.org/10.18174/566013

Soudijn, F. H., van Donk, S., Leopold, M. F., van der Wal, J. T., & Hin, V. (2022b). *Cumulative population-level effects of habitat loss on seabirds 'Kader Ecologie en Cumulatie 4.0'* (Wageningen Marine research report C007/22). Wageningen Marine Research. https://doi.org/10.18174/553775

Soule, M. E. (1986). *Conservation Biology: The Science of Scarcity and Diversity*. Sinauer Associates Inc.

St. John Glew, K., Wanless, S., Harris, M. P., Daunt, F., Erikstad, K. E., Strøm, H., Speakman, J. R., Kürten, B., & Trueman, C. N. (2019). Sympatric Atlantic puffins and razorbills show contrasting responses to adverse marine conditions during winter foraging within the North Sea. *Movement Ecology*, *7*(1), 33. https://doi.org/10.1186/s40462-019-0174-4

Stephens, P. A., & Sutherland, W. J. (1999). Consequences of the Allee effect for behaviour, ecology and conservation. *Trends in Ecology & Evolution*, *14*(10), 401–405. https://doi.org/10.1016/S0169-5347(99)01684-5

Stienen, E. W. M. (2006). *Living with gulls. Trading off food and predation in the Sandwich Tern* Sterna sandvicensis. [PhD-thesis, Rijksuniversiteit Groningen]. http://www.vliz.be/imis/imis.php?module=ref&show=html&refid=78620

Stienen, E. W. M., Courtens, W., Van de walle, M., Verstraete, H., & Vanermen, N. (2024). Frequent use of offshore wind farms in the southern North Sea by migrating terns. *Ardea*, *112*(2). https://doi.org/10.5253/arde.2024.a7

Stienen, E. W. M., van Beers, P. W. M., Brenninkmeijer, A., Habraken, J. M. P. M., Raaijmakers, M. H. J. E., & van Tienen, P. G. M. (2000). Reflections of a specialist: Patterns in food provisioning and foraging conditions in sandwich terns Sterna sandvicensis. *Ardea*, *88*(1), 33–49.

Tougaard, J., Hermannsen, L., & Madsen, P. T. (2020). How loud is the underwater noise from operating offshore wind turbines? *The Journal of the Acoustical Society of America*, *148*(5), 2885–2893. https://doi.org/10.1121/10.0002453

Trifonova, N., & Scott, B. (2023). Ecosystem indicators: Predicting population responses to combined climate and anthropogenic changes in shallow seas. *Ecography*, *2024*. https://doi.org/10.1111/ecog.06925

Van Bemmelen, R., de Jong, J., Arts, F., Beuker, D., Hoekstein, M., Horst, Y., Kuiper, K., Leemans, J., Sluijter, M., Straalen, D., & Fijn, R. (2023). *Verspreiding, abundantie en trends van zeevogels en zeezoogdieren op het Nederlands Continentaal Plat in 2021-2022.*

van Bemmelen, R. S. A., Courtens, W., Collier, M., & Fijn, R. C. (2022). *Sandwich Terns in the Netherlands in 2019-2021. Distribution, behaviour, survival and diet in light of (future) offshore wind farms.* (Bureau Waardenburg Report 21-310 Bureau Waardenburg Report 21-310; Bureau Waardenburg Report 21-310).

van Bemmelen, R. S. A., Leemans, J. J., Collier, M. P., Green, R. M. W., Middelveld, R. P., Thaxter, C. B., & Fijn, R. C. (2024). Avoidance of offshore wind farms by Sandwich Terns increases with turbine density. *Ornithological Applications*, *126*(1), Article 1. https://doi.org/10.1093/ornithapp/duad055 van der Vliet, R., Bakker, E., Potiek, A., Kraal, J., Leemans, J. J., Boonman, M., & Gyimesi, A. (2022). *Ecologisch achtergronddocument windenergiegebied IJmuiden Ver. Kavel I en kavel II: vogels, vleermuizen, vissen en benthos.* (Rapport 22-104). Bureau Waardenburg, Culemborg.

van der Wal, J. T., van Puijenbroek, M. E. B., & Leopold., M. F. (2018). *Cumulatieve effecten van offshore wind parken: Habitatverlies zeevogels: Update voor vijf zeevogelsoorten tot 2030.* (Wageningen Marine Research rapport C059/18). Wageningen Marine Research.

van Donk, S. (2024). *Knowledge-update KEC5 density maps seabirds* (Wageningen University and Research Report C044/24). Wageningen Marine Research.

van Donk, S., van Bemmelen, R., Chen, C., Tulp, I., & Melis, E. (2024). *Seabird maps of the North Sea: A short description of methodology* (Wageningen Marine Research Report C024/24a). Wageningen Marine Research. https://doi.org/10.18174/657263

van Kooten, T., Soudijn, F., Tulp, I., Chen, C., Benden, D., & Leopold, M. (2019). *The consequences of seabird habitat loss from offshore wind turbines, version 2: Displacement and population level effects in 5 selected species* (Wageningen Marine Research Report C063/19). Wageningen Marine Research. https://doi.org/10.18174/496173

Vanermen, N., Onkelinx, T., Courtens, W., Van de walle, M., Verstraete, H., & Stienen, E. W. M. (2015). Seabird avoidance and attraction at an offshore wind farm in the Belgian part of the North Sea. *Hydrobiologia*, *756*(1), Article 1. https://doi.org/10.1007/s10750-014-2088-x

Vanermen, N., Stienen, E. W. M., Courtens, W., & Van de Onkelinx, T. (2013). *Bird monitoring at offshore wind farms in the Belgian part of the North Sea* (Report INBO).

Veron, P. K., & Lawlor, M. P. (2009). The dispersal and migration of the Northern Gannet Morus bassanus from Channel Islands breeding colonies. *Seabird*, *22*(June 2009), 37–47.

Vogeltrekstation NIOO-KNAW, & Ringersvereniging. (2024). *Vogeltrekatlas*. Accesed 2024-06-21. www.vogeltrekatlas.nl

Waggitt, J. J., Evans, P. G. H., Andrade, J., Banks, A. N., Boisseau, O., Bolton, M., Bradbury, G., Brereton, T., Camphuysen, C. J., Durinck, J., Felce, T., Fijn, R. C., Garcia-Baron, I., Garthe, S., Geelhoed, S. C. V., Gilles, A., Goodall, M., Haelters, J., Hamilton, S., … Hiddink, J. G. (2020). 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

Wakefield, E. D., Bodey, T. W., Bearhop, S., Blackburn, J., Colhoun, K., Davies, R., Dwyer, R. G., Green, J. A., Grémillet, D., Jackson, A. L., Jessopp, M. J., Kane, A., Langston, R. H. W., Lescroël, A., Murray, S., Nuz, M. L., & Patrick, S. C. (2013). Space Partitioning Without Territoriality in Gannets. *Science*, *68*(2013), 16–19. https://doi.org/10.1126/science.1236077

Wakefield, E. D., Owen, E., Baer, J., Carroll, M. J., Daunt, F., Dodd, S. G., Green, J. A., Guilford, T., Mavor, R. A., Miller, P. I., Newell, M. A., Newton, S. F., Robertson, G. S., Shoji, A., Soanes, L. M., Votier, S. C., Wanless, S., & Bolton, M. (2017). Breeding density, fine-scale tracking and large-scale modeling reveal the regional distribution of four seabird species. *Ecological Applications*, *0*(0), 1–18. https://doi.org/10.1002/eap.1591

Warwick-Evans, V., Atkinson, P. W., Walkington, I., & Green, J. A. (2018). Predicting the impacts of wind farms on seabirds: An individual-based model. *Journal of Applied Ecology*, *55*(2), 503–515. https://doi.org/10.1111/1365-2664.12996

Welcker, J., & Nehls, G. (2016). Displacement of seabirds by an offshore wind farm in the North Sea. *Marine Ecology Progress Series*, *554*(July), 173–182. https://doi.org/10.3354/meps11812

Justification

Report C094/24 A
Project Number: 4316100357

The scientific quality of this report has been peer reviewed by a colleague scientist and a member of the Management Team of Wageningen Marine Research

Approved: Drs. M.F. Leopold
Researcher

Signature: Signed by:
Mardik Leopold
24CF1855A39B432...

Date: February 27th, 2025

Approved: Dr. Ir. K.E. v.d. Wolfshaar
Researcher

Signature: Signed by:
7E4E358EE96248C...

Date: February 27th, 2025

Approved: C.J. Wiebinga, PhD
Business Manager Projects

Signature: Signed by:
D41E9304A710493...

Date: February 27th, 2025

Annex 1 OWF overlap with bird distributions

Table A-1 *Surfaces areas in km^2 for offshore wind farm development search areas as presented in the maps (search area (km^2), Figure 4-1) and the realistic expected surface area based on the type of turbines and the related densities of future wind farms (wind farm area (km^2)). Based on information provided by RWS.*

Scenarios	**Search area (km^2)**	**Wind farm area (km^2)**	**Difference (km^2)**	**% difference**
Basic	260	246	-13	-5.1%
Basic plus	791	673	-118	-15.0%
Total	3,389	2,336	-1,053	-31.1%
International	40,384	15,600	-24,783	-61.4%

Table A-2 *Effect distance and scenario specific scaling factors to correct for the differences in OWF effect areas between the OWF search areas and smaller areas actually occupied by wind turbines after OWF construction. The effect distances used in the calculations differ per species (Table 3-2).*

Scenarios	**19,500 m**	**2,000 m**	**1,500 m**
Basic	0.9723	0.8753	0.8679
Basic plus	0.8735	0.6835	0.6755
Total	0.8631	0.6996	0.6929
International	0.8910	0.8366	0.8373

Table A-3 *Mean percentage of birds in offshore wind energy area per scenario and species. Column 'Map' indicates whether national (NAT) or international maps (INT) were used. Last four columns show mean values, standard deviations (Sd), and 5% (q5) and 95% (q95) quantile values. The latter three estimates are only available for national maps.*

Species	Map	Scenario	Mean	Sd	q5	q95
northern gannet	NAT	Basic	0.467	0.135	0.288	0.721
northern gannet	NAT	Basic Plus	1.744	0.341	1.271	2.362
northern gannet	NAT	Total	2.255	0.420	1.673	3.010
northern gannet	INT	International	3.284	NA	NA	NA
Sandwich tern	NAT	Basic	0.033	0.012	0.019	0.055
Sandwich tern	NAT	Basic Plus	0.098	0.029	0.059	0.151
Sandwich tern	NAT	Total	0.116	0.033	0.070	0.177
Sandwich tern	INT	International	0.718	NA	NA	NA
common guillemot	NAT	Basic	4.224	1.036	2.902	6.127
common guillemot	NAT	Basic Plus	7.060	1.347	5.292	9.561
common guillemot	NAT	Total	11.729	1.604	9.507	14.559
common guillemot	INT	International	23.652	NA	NA	NA
razorbill	NAT	Basic	0.239	0.063	0.151	0.354
razorbill	NAT	Basic Plus	1.075	0.162	0.822	1.370
razorbill	NAT	Total	1.818	0.256	1.423	2.251
razorbill	INT	International	5.175	NA	NA	NA

Table A-4 *Mean number of birds per Dutch OWF area per species, with standard deviation of the mean (Sd), based on the full range of maps used for the calculations. The scenario in which the OWF is included is indicated in the table (Total includes the OWFs of the Basic and Basic plus scenarios, Basic plus includes the OWFs of the Basic scenario). The OWF areas are composed of the search area with the species specific effect distance (Table* 3-2*).*

OWF	Scenario	Species	OWF area (km2)	Scaling factor	Mean	Sd
Borssele Kavel I	Basic	northern gannet	92.4	0.7843	6.7	7.5
Borssele Kavel II	Basic	northern gannet	90.6	0.8117	3.7	3.9
Borssele Kavel III	Basic	northern gannet	102.4	0.7699	6.4	6.6
Borssele Kavel IV	Basic	northern gannet	108.7	0.7171	9.8	10.4
Borssele Kavel V	Basic	northern gannet	4.9	1	0.5	0.5
Buitengaats Gemini I	Basic	northern gannet	54.8	1	1.1	1.6
Luchterduinen	Basic	northern gannet	31	1	0.6	1
Offshore windpark Egmond aan Zee	Basic	northern gannet	44.9	1	0.3	0.4
Prinses Amalia Windparken	Basic	northern gannet	36.3	0.9985	0.5	0.8
ZeeEnergie Gemini II	Basic	northern gannet	53.5	1	1.1	1.8
HKN Kavel V	Basic plus	northern gannet	168.1	0.5811	2.1	3.1
HKZ Kavel I	Basic plus	northern gannet	101.3	0.6611	1.2	1.8
HKZ Kavel II	Basic plus	northern gannet	70.3	1	1	1.4
HKZ Kavel III	Basic plus	northern gannet	78.4	1	1.2	1.7
HKZ Kavel IV	Basic plus	northern gannet	130.3	0.4383	0.8	1.4
Hollandse Kust west noordelijk deel	Basic plus	northern gannet	261	0.7538	4.3	4.4
IJmuiden Ver Noord	Basic plus	northern gannet	298.8	0.6667	18.9	37.6
IJmuiden Ver versie 2021	Basic plus	northern gannet	529.3	0.674	33.9	53.9
Nederwiek zuid	Basic plus	northern gannet	336.7	0.3846	19.7	34
Doordewind	Total	northern gannet	250.8	1	3.3	4.5
Hollandse Kust west zuidelijk deel	Total	northern gannet	171.9	0.3366	1.7	1.7
Nederwiek noord	Total	northern gannet	755.7	0.4115	15.5	27.8
Ten noorden van de Wadden west	Total	northern gannet	107.3	0.9863	2.1	3.5
Borssele Kavel I	Basic	Sandwich tern	92.4	0.7843	0.7	1.9

Borssele Kavel II	Basic	Sandwich tern	90.6	0.8117	1.2	2.7
Borssele Kavel III	Basic	Sandwich tern	102.4	0.7699	0.9	2.2
Borssele Kavel IV	Basic	Sandwich tern	108.7	0.7171	1	2.5
Borssele Kavel V	Basic	Sandwich tern	4.9	1	0	0.1
Buitengaats Gemini I	Basic	Sandwich tern	54.8	1	2.4	6.4
Luchterduinen	Basic	Sandwich tern	31	1	1.1	2.1
Offshore windpark Egmond aan Zee	Basic	Sandwich tern	44.9	1	3.9	4.6
Prinses Amalia Windparken	Basic	Sandwich tern	36.3	0.9985	0.5	0.7
ZeeEnergie Gemini II	Basic	Sandwich tern	53.5	1	1.9	3.8
HKN Kavel V	Basic plus	Sandwich tern	168.1	0.5811	5.1	9.6
HKZ Kavel I	Basic plus	Sandwich tern	101.3	0.6611	1.5	3
HKZ Kavel II	Basic plus	Sandwich tern	70.3	1	3.3	7.3
HKZ Kavel III	Basic plus	Sandwich tern	78.4	1	8.9	20.2
HKZ Kavel IV	Basic plus	Sandwich tern	130.3	0.4383	4.6	10
Hollandse Kust west noordelijk deel	Basic plus	Sandwich tern	261	0.7538	1.5	4.6
IJmuiden Ver Noord	Basic plus	Sandwich tern	298.8	0.6667	1.5	4.3
IJmuiden Ver versie 2021	Basic plus	Sandwich tern	529.3	0.674	1.9	5.2
Nederwiek zuid	Basic plus	Sandwich tern	336.7	0.3846	0.4	1.5
Doordewind	Total	Sandwich tern	250.8	1	1.9	4.5
Hollandse Kust west zuidelijk deel	Total	Sandwich tern	171.9	0.3366	0.5	1.4
Nederwiek noord	Total	Sandwich tern	755.7	0.4115	1.2	3.5
Ten noorden van de Wadden west	Total	Sandwich tern	107.3	0.9863	2.5	4.8
Borssele Kavel I	Basic	common guillemot	1930.3	0.9442	1451.3	1648.6
Borssele Kavel II	Basic	common guillemot	1917.1	0.9521	1152.3	1348.8
Borssele Kavel III	Basic	common guillemot	1989.8	0.937	1183.3	1418.2
Borssele Kavel IV	Basic	common guillemot	2087.3	0.9163	1128	1471.8
Borssele Kavel V	Basic	common guillemot	1261	1	1022	1246.7
Buitengaats Gemini I	Basic	common guillemot	1715	1	913.8	1126.1
Luchterduinen	Basic	common guillemot	1556.5	1	1435.5	2594.6
Offshore windpark Egmond aan Zee	Basic	common guillemot	1643	1	2424.9	4850.2
Prinses Amalia Windparken	Basic	common guillemot	1549.1	0.9998	1829	3356.6
ZeeEnergie Gemini II	Basic	common guillemot	1705.4	1	1387.1	1653.1
HKN Kavel V	Basic plus	common guillemot	2283.6	0.8583	2682.8	4773.5
HKZ Kavel I	Basic plus	common guillemot	2055.9	0.8983	1300.2	2115.4
HKZ Kavel II	Basic plus	common guillemot	1783	1	1165.5	1796.3
HKZ Kavel III	Basic plus	common guillemot	1841.2	1	1230.3	1932.7
HKZ Kavel IV	Basic plus	common guillemot	2262.2	0.7994	1493.7	2634.8
Hollandse Kust west noordelijk deel	Basic plus	common guillemot	2764.9	0.9067	1133.4	1286.1
IJmuiden Ver Noord	Basic plus	common guillemot	2824.7	0.8676	1307.2	959.8
IJmuiden Ver versie 2021	Basic plus	common guillemot	3399.2	0.8546	1686.8	1703
Nederwiek zuid	Basic plus	common guillemot	3154.7	0.7168	1368.4	876.2
Doordewind	Total	common guillemot	2658.6	1	2108.2	4179.7
Hollandse Kust west zuidelijk deel	Total	common guillemot	2317.1	0.7547	710.8	892.8
Nederwiek noord	Total	common guillemot	3954.6	0.695	4665.7	3568
Ten noorden van de Wadden west	Total	common guillemot	2174.2	0.996	2477	3209.6
Borssele Kavel I	Basic	razorbill	102.4	0.7939	5.5	10.5
Borssele Kavel II	Basic	razorbill	100.5	0.8202	7.1	12.2
Borssele Kavel III	Basic	razorbill	113.5	0.7799	10.3	18.5
Borssele Kavel IV	Basic	razorbill	120.7	0.7289	17.3	39.3
Borssele Kavel V	Basic	razorbill	7.1	1	0.5	0.9
Buitengaats Gemini I	Basic	razorbill	62.4	1	6.6	13.4
Luchterduinen	Basic	razorbill	36.9	1	3	3.7
Offshore windpark Egmond aan Zee	Basic	razorbill	51.9	1	3.4	4.2
Prinses Amalia Windparken	Basic	razorbill	42	0.9986	1.9	2.6
ZeeEnergie Gemini II	Basic	razorbill	61	1	6	10.2
HKN Kavel V	Basic plus	razorbill	182.5	0.5938	5.8	7.8
HKZ Kavel I	Basic plus	razorbill	113.4	0.6756	4.1	5.7
HKZ Kavel II	Basic plus	razorbill	78.6	1	4.3	5.9
HKZ Kavel III	Basic plus	razorbill	87.4	1	4.9	6.1
HKZ Kavel IV	Basic plus	razorbill	144.5	0.4578	5.7	6.9
Hollandse Kust west noordelijk deel	Basic plus	razorbill	279.8	0.7612	27.7	63.7

IJmuiden Ver Noord	Basic plus	razorbill	318.4	0.6755	43.1	79.5
IJmuiden Ver versie 2021	Basic plus	razorbill	553	0.6802	90.1	188
Nederwiek zuid	Basic plus	razorbill	359.8	0.3986	27.7	44.9
Doordewind	Total	razorbill	268.6	1	5.2	8.7
Hollandse Kust west zuidelijk deel	Total	razorbill	186	0.3542	13.3	29.8
Nederwiek noord	Total	razorbill	783.8	0.4197	90.8	121.2
Ten noorden van de Wadden west	Total	razorbill	120.4	0.987	10.9	13.9

Annex 2 Mortality estimates

Table A-5 *Mean and standard deviation (Sd) of the annual mortalities due to displacement from OWF areas (and the boundaries around them) per species and scenario. The column 'map' indicates whether national (NAT) or international maps (INT) were used.*

Species	**Map**	**Scenario**	**Mortality level**	**Mean**	**Sd**
northern gannet	NAT	Basic	0.01	0.0000261	0.0000154
northern gannet	NAT	Basic	0.02	0.0000523	0.0000306
northern gannet	NAT	Basic	0.05	0.0001303	0.0000764
northern gannet	NAT	Basic	0.1	0.0002573	0.0001539
northern gannet	NAT	Basic Plus	0.01	0.0000966	0.0000528
northern gannet	NAT	Basic Plus	0.02	0.0001946	0.0001060
northern gannet	NAT	Basic Plus	0.05	0.0004845	0.0002634
northern gannet	NAT	Basic Plus	0.1	0.0009834	0.0005242
northern gannet	NAT	Total	0.01	0.0001261	0.0000678
northern gannet	NAT	Total	0.02	0.0002489	0.0001341
northern gannet	NAT	Total	0.05	0.0006314	0.0003373
northern gannet	NAT	Total	0.1	0.0012635	0.0006761
northern gannet	INT	International	0.01	0.0001819	0.0000903
northern gannet	INT	International	0.02	0.0003676	0.0001802
northern gannet	INT	International	0.05	0.0009127	0.0004519
northern gannet	INT	International	0.1	0.0018388	0.0009020
Sandwich tern	NAT	Basic	0.01	0.0000017	0.0000011
Sandwich tern	NAT	Basic	0.02	0.0000033	0.0000023
Sandwich tern	NAT	Basic	0.05	0.0000083	0.0000056
Sandwich tern	NAT	Basic	0.1	0.0000167	0.0000114
Sandwich tern	NAT	Basic Plus	0.01	0.0000049	0.0000032
Sandwich tern	NAT	Basic Plus	0.02	0.0000100	0.0000065
Sandwich tern	NAT	Basic Plus	0.05	0.0000244	0.0000159
Sandwich tern	NAT	Basic Plus	0.1	0.0000502	0.0000320
Sandwich tern	NAT	Total	0.01	0.0000059	0.0000038
Sandwich tern	NAT	Total	0.02	0.0000118	0.0000075
Sandwich tern	NAT	Total	0.05	0.0000295	0.0000187
Sandwich tern	NAT	Total	0.1	0.0000586	0.0000376
Sandwich tern	INT	International	0.01	0.0000362	0.0000200
Sandwich tern	INT	International	0.02	0.0000726	0.0000400
Sandwich tern	INT	International	0.05	0.0001824	0.0000996
Sandwich tern	INT	International	0.1	0.0003619	0.0002012
common guillemot	NAT	Basic	0.01	0.0003334	0.0000916

common guillemot	NAT	Basic	0.02	0.0006680	0.0001850
common guillemot	NAT	Basic	0.05	0.0016676	0.0004641
common guillemot	NAT	Basic	0.1	0.0033454	0.0009456
common guillemot	NAT	Basic Plus	0.01	0.0005590	0.0001287
common guillemot	NAT	Basic Plus	0.02	0.0011165	0.0002556
common guillemot	NAT	Basic Plus	0.05	0.0027863	0.0006336
common guillemot	NAT	Basic Plus	0.1	0.0055727	0.0012843
common guillemot	NAT	Total	0.01	0.0009264	0.0001722
common guillemot	NAT	Total	0.02	0.0018506	0.0003386
common guillemot	NAT	Total	0.05	0.0046469	0.0008518
common guillemot	NAT	Total	0.1	0.0092752	0.0017176
common guillemot	INT	International	0.01	0.0018637	0.0002316
common guillemot	INT	International	0.02	0.0037377	0.0004581
common guillemot	INT	International	0.05	0.0093485	0.0011528
common guillemot	INT	International	0.1	0.0186893	0.0022766
razorbill	NAT	Basic	0.01	0.0000103	0.0000027
razorbill	NAT	Basic	0.02	0.0000206	0.0000055
razorbill	NAT	Basic	0.05	0.0000515	0.0000138
razorbill	NAT	Basic	0.1	0.0001030	0.0000274
razorbill	NAT	Basic Plus	0.01	0.0000461	0.0000073
razorbill	NAT	Basic Plus	0.02	0.0000926	0.0000146
razorbill	NAT	Basic Plus	0.05	0.0002317	0.0000372
razorbill	NAT	Basic Plus	0.1	0.0004606	0.0000722
razorbill	NAT	Total	0.01	0.0000782	0.0000115
razorbill	NAT	Total	0.02	0.0001561	0.0000232
razorbill	NAT	Total	0.05	0.0003905	0.0000579
razorbill	NAT	Total	0.1	0.0007818	0.0001168
razorbill	INT	International	0.01	0.0002224	0.0000109
razorbill	INT	International	0.02	0.0004452	0.0000217
razorbill	INT	International	0.05	0.0011120	0.0000542
razorbill	INT	International	0.1	0.0022262	0.0001082

Annex 3 Population effects

Table A-6 *Estimated annual population growth and population growth factor over 40 years per species, scenario and mortality level. The column 'Map' indicates whether national (NAT) or international maps (INT) were used.*

Species	Map	Scenario	Mortality level	Annual pop. growth	Growth factor (40 yrs)
northern gannet	INT	Null	0.0	3.458	1.032
northern gannet	INT	International	0.01	3.432	1.031
northern gannet	INT	International	0.02	3.407	1.031
northern gannet	INT	International	0.05	3.334	1.031
northern gannet	INT	International	0.1	3.215	1.030
northern gannet	NAT	Null	0.0	3.458	1.032
northern gannet	NAT	Basic	0.01	3.454	1.031
northern gannet	NAT	Basic	0.02	3.451	1.031
northern gannet	NAT	Basic	0.05	3.440	1.031
northern gannet	NAT	Basic	0.1	3.422	1.031
northern gannet	NAT	Basic Plus	0.01	3.444	1.031
northern gannet	NAT	Basic Plus	0.02	3.431	1.031
northern gannet	NAT	Basic Plus	0.05	3.392	1.031
northern gannet	NAT	Basic Plus	0.1	3.328	1.031
northern gannet	NAT	Total	0.01	3.441	1.031
northern gannet	NAT	Total	0.02	3.423	1.031
northern gannet	NAT	Total	0.05	3.373	1.031
northern gannet	NAT	Total	0.1	3.290	1.030
Sandwich tern	INT	Null	0.0	0.560	0.986
Sandwich tern	INT	International	0.01	0.559	0.986
Sandwich tern	INT	International	0.02	0.558	0.986
Sandwich tern	INT	International	0.05	0.556	0.985
Sandwich tern	INT	International	0.1	0.552	0.985
Sandwich tern	NAT	Null	0.0	0.560	0.986
Sandwich tern	NAT	Basic	0.01	0.560	0.986
Sandwich tern	NAT	Basic	0.02	0.560	0.986
Sandwich tern	NAT	Basic	0.05	0.560	0.986
Sandwich tern	NAT	Basic	0.1	0.560	0.986
Sandwich tern	NAT	Basic Plus	0.01	0.560	0.986
Sandwich tern	NAT	Basic Plus	0.02	0.560	0.986
Sandwich tern	NAT	Basic Plus	0.05	0.560	0.986
Sandwich tern	NAT	Basic Plus	0.1	0.559	0.986
Sandwich tern	NAT	Total	0.01	0.560	0.986
Sandwich tern	NAT	Total	0.02	0.560	0.986
Sandwich tern	NAT	Total	0.05	0.560	0.986

Sandwich tern	NAT	Total	0.1	0.559	0.986
common guillemot	INT	Null	0.0	3.746	1.034
common guillemot	INT	International	0.01	3.473	1.032
common guillemot	INT	International	0.02	3.219	1.030
common guillemot	INT	International	0.05	2.563	1.024
common guillemot	INT	International	0.1	1.753	1.014
common guillemot	NAT	Null	0.0	3.746	1.034
common guillemot	NAT	Basic	0.01	3.696	1.033
common guillemot	NAT	Basic	0.02	3.647	1.033
common guillemot	NAT	Basic	0.05	3.503	1.032
common guillemot	NAT	Basic	0.1	3.276	1.030
common guillemot	NAT	Basic Plus	0.01	3.663	1.033
common guillemot	NAT	Basic Plus	0.02	3.582	1.032
common guillemot	NAT	Basic Plus	0.05	3.349	1.031
common guillemot	NAT	Basic Plus	0.1	2.994	1.028
common guillemot	NAT	Total	0.01	3.609	1.033
common guillemot	NAT	Total	0.02	3.477	1.032
common guillemot	NAT	Total	0.05	3.108	1.029
common guillemot	NAT	Total	0.1	2.580	1.024
razorbill	INT	Null	0.0	1.126	1.003
razorbill	INT	International	0.01	1.117	1.003
razorbill	INT	International	0.02	1.107	1.003
razorbill	INT	International	0.05	1.078	1.002
razorbill	INT	International	0.1	1.032	1.001
razorbill	NAT	Null	0.0	1.126	1.003
razorbill	NAT	Basic	0.01	1.126	1.003
razorbill	NAT	Basic	0.02	1.126	1.003
razorbill	NAT	Basic	0.05	1.124	1.003
razorbill	NAT	Basic	0.1	1.122	1.003
razorbill	NAT	Basic Plus	0.01	1.124	1.003
razorbill	NAT	Basic Plus	0.02	1.122	1.003
razorbill	NAT	Basic Plus	0.05	1.116	1.003
razorbill	NAT	Basic Plus	0.1	1.106	1.003
razorbill	NAT	Total	0.01	1.123	1.003
razorbill	NAT	Total	0.02	1.120	1.003
razorbill	NAT	Total	0.05	1.109	1.003
razorbill	NAT	Total	0.1	1.092	1.002

Table A-7 *ALI threshold values per species, scenario and mortality level for the national scenarios. X_q50, X_q95 and X_q5 denote that median, 5% and 95% quantile values of the relative difference in final population abundance between each scenario and the relevant null scenario. PVI_ALI is the probability of an exceedance of the ALI X threshold. The ALI is violated if PVI_ALI > ALI Y (0.05) and if X_q95 > ALI X (40yrs).*

Species	**ALI X (40 yrs)**	**ALI Y**	**Scenario**	**Mortality level**	**X_q50**	**X_q95**	**X_q5**	**PVI_ALI**	**ALI violation**
northern gannet	4.715	0.05	Basic	0.01	0.100	0.212	0.014	0.000	FALSE
northern gannet	4.715	0.05	Basic	0.02	0.199	0.423	0.029	0.000	FALSE
northern gannet	4.715	0.05	Basic	0.05	0.497	1.055	0.072	0.000	FALSE
northern gannet	4.715	0.05	Basic	0.10	0.993	2.099	0.143	0.000	FALSE
northern gannet	4.715	0.05	Basic Plus	0.01	0.389	0.734	0.057	0.000	FALSE
northern gannet	4.715	0.05	Basic Plus	0.02	0.777	1.463	0.114	0.000	FALSE
northern gannet	4.715	0.05	Basic Plus	0.05	1.932	3.618	0.284	0.003	FALSE
northern gannet	4.715	0.05	Basic Plus	0.10	3.828	7.108	0.567	0.342	TRUE
northern gannet	4.715	0.05	Total	0.01	0.504	0.941	0.074	0.000	FALSE
northern gannet	4.715	0.05	Total	0.02	1.006	1.874	0.148	0.000	FALSE
northern gannet	4.715	0.05	Total	0.05	2.498	4.621	0.369	0.043	FALSE
northern gannet	4.715	0.05	Total	0.10	4.934	9.033	0.737	0.530	TRUE
Sandwich tern	4.884	0.05	Basic	0.01	0.006	0.015	0.001	0.000	FALSE
Sandwich tern	4.884	0.05	Basic	0.02	0.012	0.030	0.001	0.000	FALSE
Sandwich tern	4.884	0.05	Basic	0.05	0.030	0.074	0.004	0.000	FALSE
Sandwich tern	4.884	0.05	Basic	0.10	0.060	0.148	0.007	0.000	FALSE
Sandwich tern	4.884	0.05	Basic Plus	0.01	0.018	0.043	0.002	0.000	FALSE
Sandwich tern	4.884	0.05	Basic Plus	0.02	0.037	0.086	0.004	0.000	FALSE
Sandwich tern	4.884	0.05	Basic Plus	0.05	0.091	0.214	0.011	0.000	FALSE
Sandwich tern	4.884	0.05	Basic Plus	0.10	0.183	0.427	0.022	0.000	FALSE
Sandwich tern	4.884	0.05	Total	0.01	0.022	0.050	0.003	0.000	FALSE
Sandwich tern	4.884	0.05	Total	0.02	0.044	0.101	0.005	0.000	FALSE
Sandwich tern	4.884	0.05	Total	0.05	0.109	0.252	0.013	0.000	FALSE
Sandwich tern	4.884	0.05	Total	0.10	0.217	0.502	0.026	0.000	FALSE
common guillemot	10.893	0.05	Basic	0.01	1.286	2.019	0.855	0.000	FALSE
common guillemot	10.893	0.05	Basic	0.02	2.557	3.999	1.704	0.000	FALSE
common guillemot	10.893	0.05	Basic	0.05	6.272	9.705	4.206	0.020	FALSE
common guillemot	10.893	0.05	Basic	0.10	12.158	18.487	8.239	0.678	TRUE
common guillemot	10.893	0.05	Basic Plus	0.01	2.174	3.136	1.530	0.000	FALSE
common guillemot	10.893	0.05	Basic Plus	0.02	4.303	6.175	3.037	0.000	FALSE
common guillemot	10.893	0.05	Basic Plus	0.05	10.419	14.743	7.423	0.412	TRUE
common guillemot	10.893	0.05	Basic Plus	0.10	19.773	27.353	14.306	0.997	TRUE
common guillemot	10.893	0.05	Total	0.01	3.650	4.824	2.665	0.000	FALSE
common guillemot	10.893	0.05	Total	0.02	7.169	9.421	5.261	0.007	FALSE
common guillemot	10.893	0.05	Total	0.05	16.989	21.945	12.649	0.990	TRUE
common guillemot	10.893	0.05	Total	0.10	31.142	39.154	23.727	1.000	TRUE
razorbill	14.402	0.05	Basic	0.01	0.039	0.060	0.025	0.000	FALSE
razorbill	14.402	0.05	Basic	0.02	0.078	0.121	0.051	0.000	FALSE

razorbill	14.402	0.05	Basic	0.05	0.195	0.301	0.126	0.000	FALSE
razorbill	14.402	0.05	Basic	0.10	0.389	0.601	0.253	0.000	FALSE
razorbill	14.402	0.05	Basic Plus	0.01	0.179	0.233	0.138	0.000	FALSE
razorbill	14.402	0.05	Basic Plus	0.02	0.359	0.466	0.275	0.000	FALSE
razorbill	14.402	0.05	Basic Plus	0.05	0.894	1.161	0.687	0.000	FALSE
razorbill	14.402	0.05	Basic Plus	0.10	1.781	2.309	1.369	0.000	FALSE
razorbill	14.402	0.05	Total	0.01	0.304	0.386	0.237	0.000	FALSE
razorbill	14.402	0.05	Total	0.02	0.608	0.771	0.474	0.000	FALSE
razorbill	14.402	0.05	Total	0.05	1.513	1.917	1.181	0.000	FALSE
razorbill	14.402	0.05	Total	0.10	3.004	3.798	2.348	0.000	FALSE

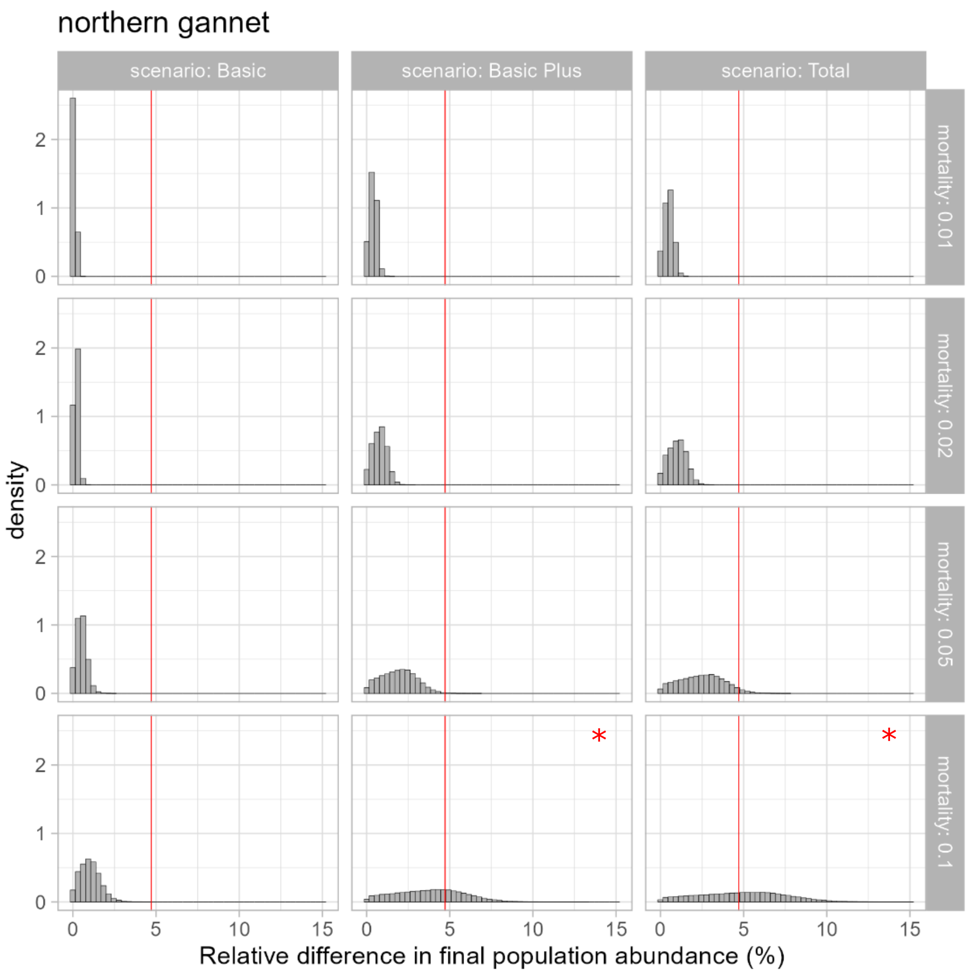

Figure A-1 *Relative difference in final population abundance in percentage between each scenario and the baseline (null) scenario for the northern gannet (Morus bassanus). The red vertical lines demark the ALI X threshold for 40 years. Histograms are based on 1e5 replicate simulations. ALI violations are indicated with a red asterisk (*).*

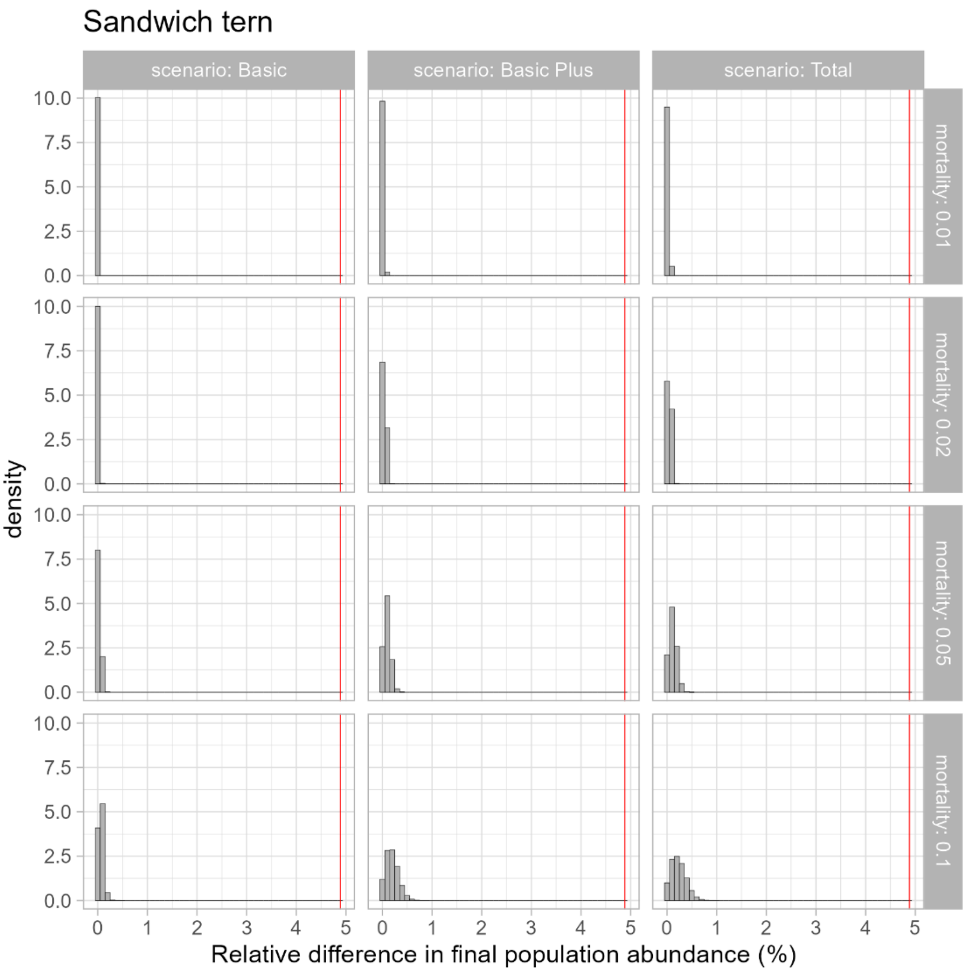

Figure A-2 *Relative difference in final population abundance in percentage between each scenario and the baseline (null) scenario for the Sandwich tern (Thalasseus sandvicensis). The red vertical lines demark the ALI X threshold for 40 years. Histograms are based on 1e5 replicate simulations. ALI violations are indicated with a red asterisk (*).*

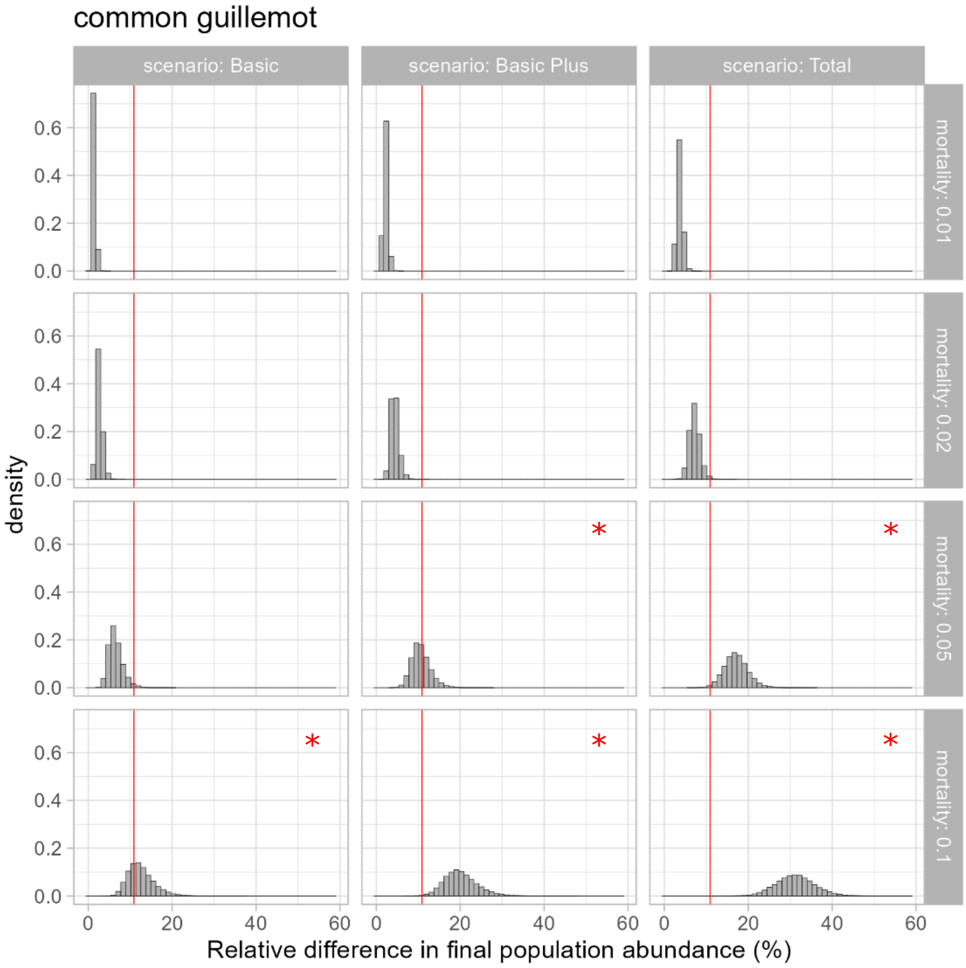

Figure A-3 *Relative difference in final population abundance in percentage between each scenario and the baseline (null) scenario for the common guillemot. The red vertical lines demark the ALI X threshold for 40 years. Histograms are based on 1e5 replicate simulations. ALI violations are indicated with a red asterisk (*).*

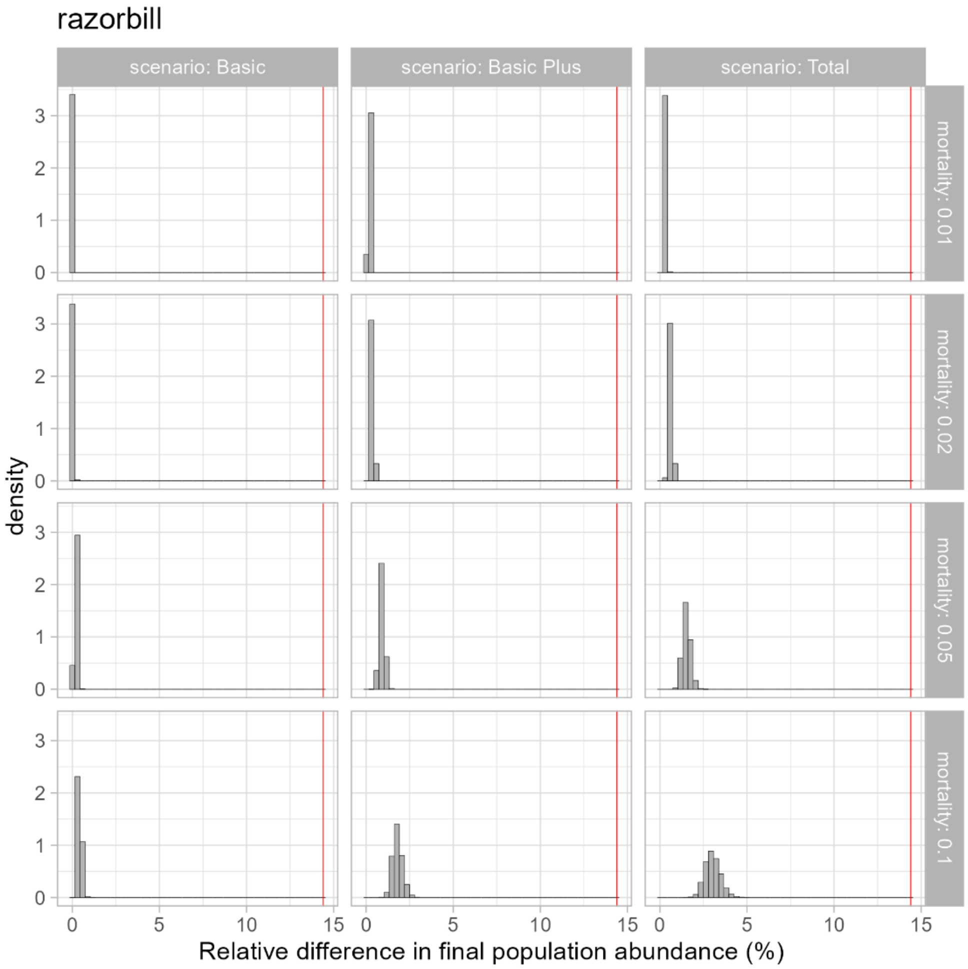

Figure A-4 *Relative difference in final population abundance in percentage between each scenario and the baseline (null) scenario for the razorbill. The red vertical lines demark the ALI X threshold for 40 years. Histograms are based on 1e5 replicate simulations. ALI violations are indicated with a red asterisk (*).*

Annex 4 Halomar sensitivity analysis

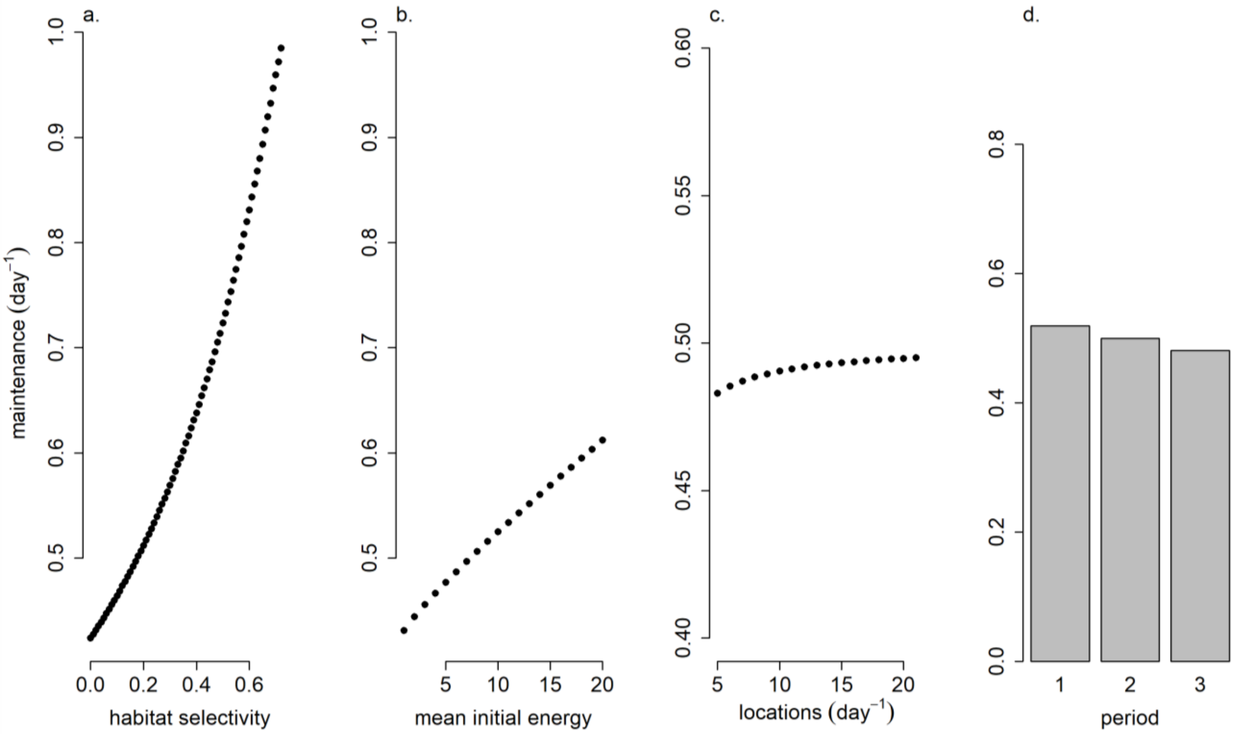

Figure A-5 *Sensitivity analysis of the fitted maintenance costs per parameter for the northern gannet, for simulations with no OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-2****,*** *the default value of the maintenance costs is 0.49. The periods considered here are: 1: Nov-Dec; 2: Oct-Jan; 3: Oct-Dec. For the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulations with 1000 birds were conducted to estimate the maintenance costs.*

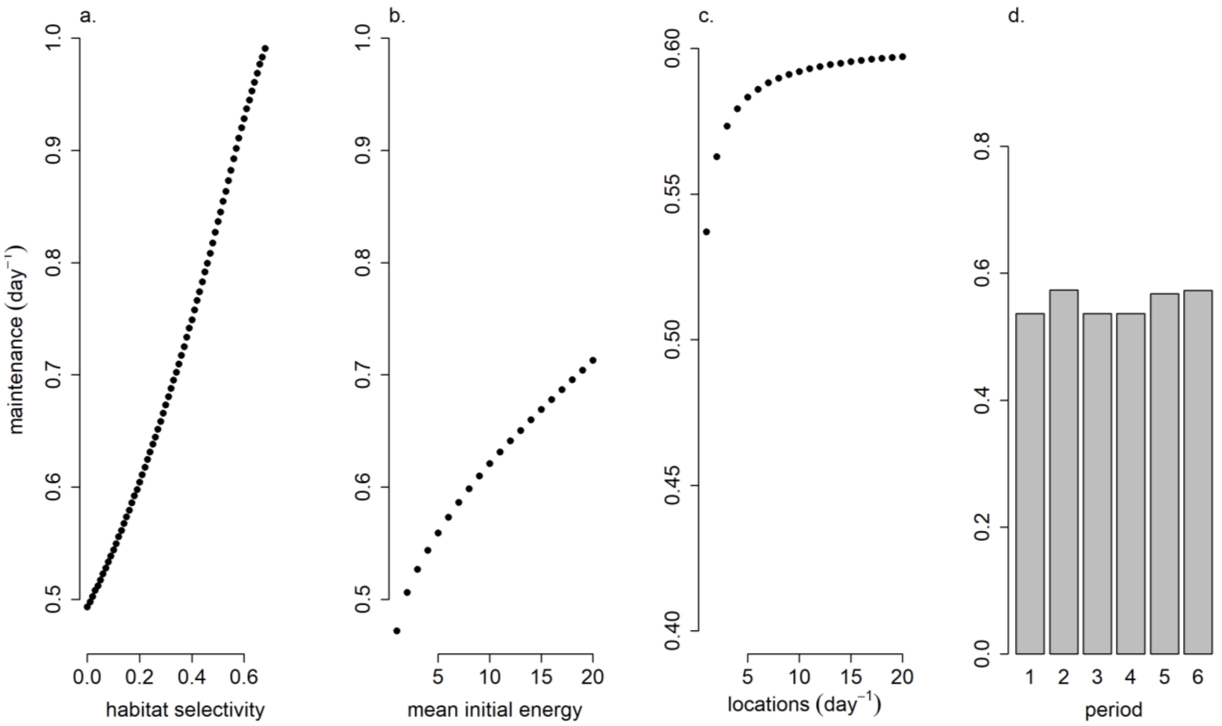

Figure A-6 *Sensitivity analysis of the fitted maintenance costs per parameter for the common guillemot, for simulations with no OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-4**,** the default value of the maintenance costs is 0.57. The periods considered here are: 1: Sep-Jan; 2: Oct-Dec; 3: Sept-Feb; 4: Sept-Dec; 5: Oct-Feb; 6:Oct-Jan. For the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulations with 1000 birds were conducted to estimate the maintenance costs.*

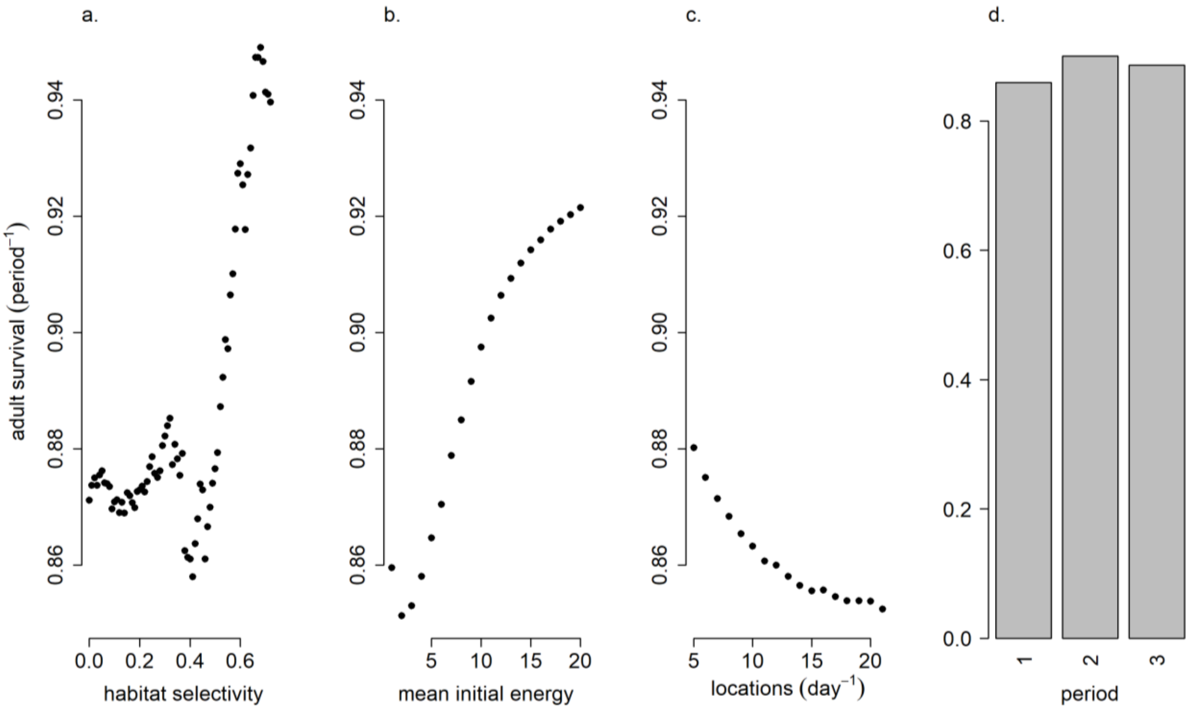

Figure A-7 *The effect of changes in the parameters on the survival probability in the scenario where the northern gannet avoids OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-2, the default value of the survival with no OWFs is 0.94. The periods considered here are, 1: Nov-Dec; 2: Oct-Jan; 3: Oct-Dec, for the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulation with 1000 birds were conducted.*

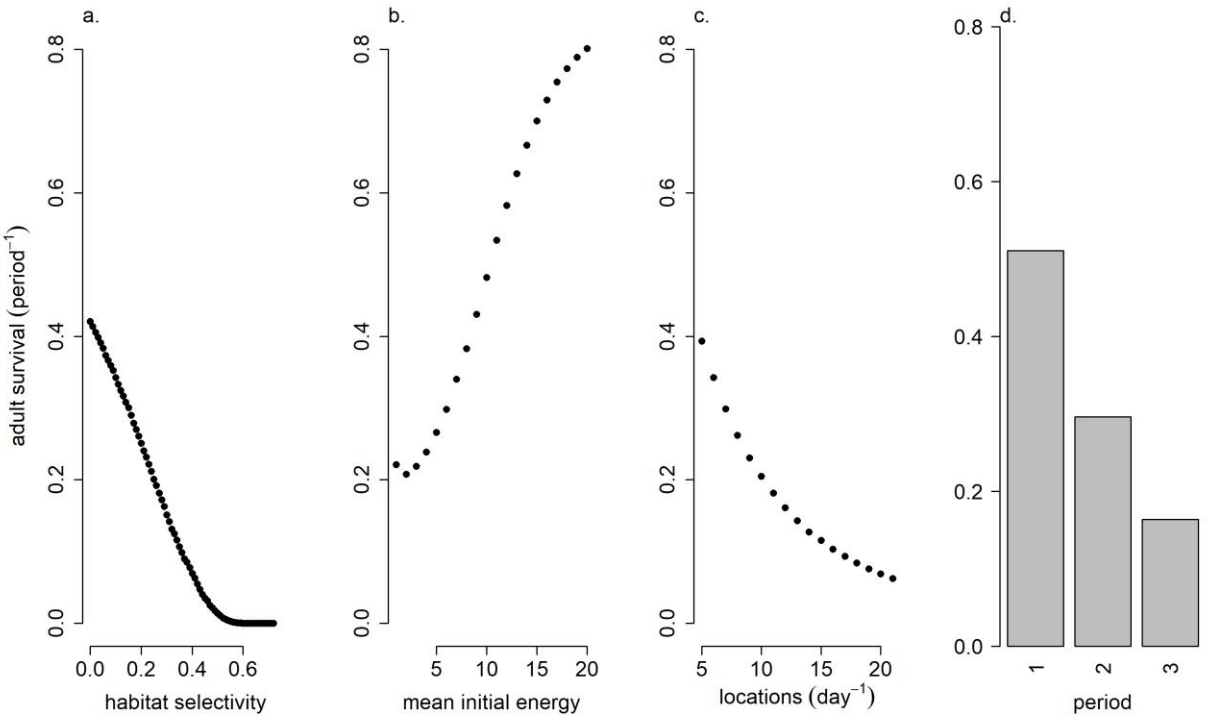

Figure A-8 *The effect of changes in the parameters on the survival probability in the scenario where the northern gannet does not forage in OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-2, the default value of the survival with no OWFs is 0.94. The periods considered here are, 1: Nov-Dec; 2: Oct-Jan; 3: Oct-Dec, for the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulation with 1000 birds were conducted.*

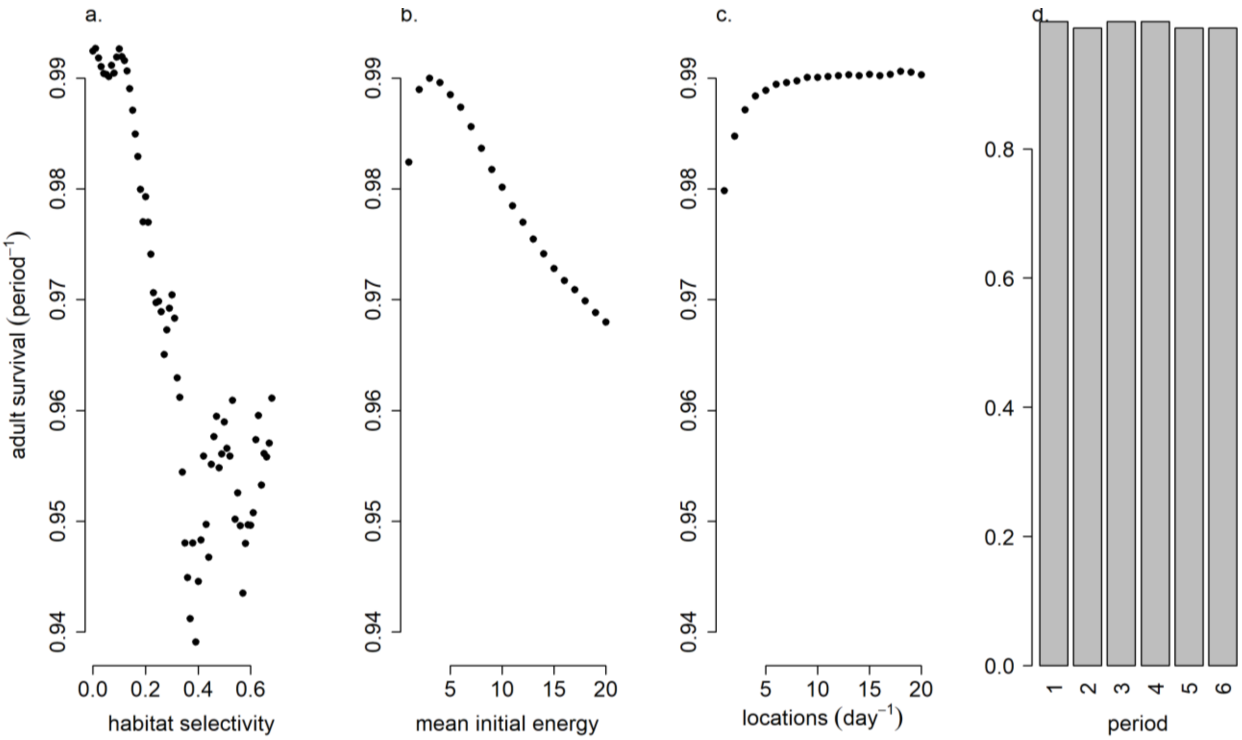

Figure A-9 *The effect of changes in the parameters on the survival probability in the scenario where the common guillemot avoids OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-4, the default value of the survival with no OWFs is 0.95. The periods considered here are: 1: Sep-Jan; 2: Oct-Dec; 3: Sept-Feb; 4: Sept-Dec; 5: Oct-Feb; 6:Oct-Jan. For the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulation with 1000 birds were conducted.*

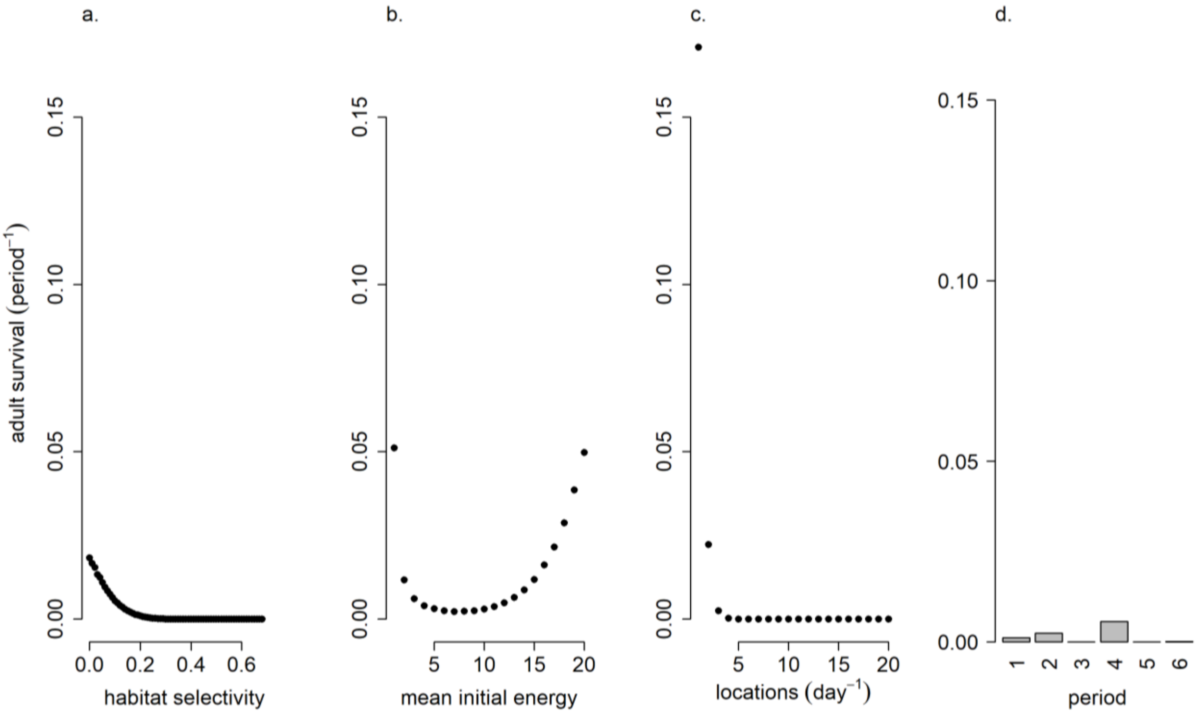

***Figure A-10** The effect of changes in the parameters on the survival probability in the scenario where the common guillemot does not forage in OWFs. One parameter was changed at a time, using the values as indicated on the x-axis of the plots. For the other parameters, default parameter values were used as given in Table 5-4, the default value of the survival is 0.95. The periods considered here are: 1: Sep-Jan; 2: Oct-Dec; 3: Sept-Feb; 4: Sept-Dec; 5: Oct-Feb; 6:Oct-Jan. For the ranges and step sizes of the varied parameters, see Table 5-5. For each combination of parameter values, 1000 simulation with 1000 birds were conducted.*

Wageningen Marine Research
T +31 (0)317 48 09 00
E imares@wur.nl
www.wur.nl/marine-research

Visitors’adress
- Ankerpark 27 1781 AG Den Helder
- Korringaweg 7, 4401 NT Yerseke
- Haringkade 1, 1976 CP IJmuiden