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Knowledge update and description of methodology

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Summary

The Dutch KEC assessment (Kader Ecologie en Cumulatie; Rijkswaterstaat 2025) aims to quantify the ecological effects of offshore wind farms (OWFs). The KEC assessment for habitat loss quantifies the population consequences of the displacement of seabirds from OWF areas by means of a spatial overlap analysis between (future) OWF sites and bird density maps, in combination with the displacement matrix approach and population modelling. This knowledge update report brings the KEC assessment for habitat loss up to date with the latest scientific knowledge. Compared to the last KEC, the uncertainty analysis is extended to include uncertainty regarding displacement effects. Also, the assessment framework is updated to deal with OWF scenarios that change in time, depending on the construction and decommissioning dates of individual wind farms. Furthermore, demographic parameters of the population models were updated and new information regarding displacement effects was evaluated and incorporated. In addition, an exhaustive description of the methodology to quantify effects of habitat loss is provided. The publication of this knowledge update will be accompanied by software code that can be used to carry out the assessment as described in this report.

1 Introduction

The Dutch KEC assessment (Kader Ecologie en Cumulatie; Rijkswaterstaat 2025) aims to quantify the ecological effects of offshore wind farms (OWFs). The latest KEC assessment (KEC 5) was commissioned in 2024 and published in 2025, and included assessments of the effects of habitat loss and collisions for several seabird species (Soudijn et al. 2025; IJntema et al. 2025). The KEC assessment for habitat loss quantifies the population consequences of the displacement of seabirds from OWF areas using spatial overlap analysis between (future) OWF sites and bird density maps, in combination with the displacement matrix approach (JNCC 2015; Searle et al. 2025) and population modelling. The KEC assessments use the latest scientific knowledge, and the methodology as used within the KEC forms the basis for the ecological impact assessments performed for permitting individual wind farm sites (Rijkswaterstaat 2025). Therefore, the methodology as used within KEC requires frequent updates to accommodate new scientific knowledge. This knowledge update report brings the KEC assessment for habitat loss up to date with the latest scientific knowledge. This will be done for six seabird species (northern gannet, Sandwich tern, common guillemot, razorbill, Atlantic puffin, and northern fulmar). Compared to KEC 5, the uncertainty analysis is extended to include uncertainty regarding displacement effects. Also, the assessment framework is updated to deal with OWF scenarios that change in time, depending on the construction and decommissioning dates of each OWF. Furthermore, demographic parameters of the population models were updated and new information regarding displacement effects was evaluated and incorporated. In addition, an exhaustive description of the methodology to quantify effects of habitat loss is provided, but assessment itself is not performed and no assessment results are presented. The publication of this knowledge update will be accompanied by software code that can be used to carry out the assessment as described in this report. This software code will be available from the online WOZEP gitlab repository[1].

[1] https://wozep.nl/git/

2 Assignment

The assignment is twofold:

1. Bring the methodology of the KEC assessment for habitat loss up to date with the latest scientific knowledge regarding the effects of displacement of seabirds by OWFs. This will be done for six seabird species (northern gannet, Sandwich tern, common guillemot, razorbill, Atlantic puffin, and northern fulmar) see 3.2.2 - Species included in current knowledge update.
2. Present a full description of the methodology.

Points 1 and 2 are combined into a single chapter. This description hence presents the full methodological details, updated with the latest scientific knowledge. This knowledge update only describes how new knowledge is incorporated into the KEC assessment methodology. The assessment itself is not performed and no assessment results are presented.

3 Knowledge base update and description of methodology

3.1 Model framework overview

In this knowledge update we present a full, updated, description of the KEC framework to estimate population effects of habitat loss for seabirds. This framework consists of four distinct steps and the output of each step is used as input for the next step (Figure 1). An integrated treatment of uncertainty across the entire framework was developed as part the last KEC (Soudijn et al. 2025). In the following sections, we consider the four steps of the framework in separate sections, in addition to the topic of 'species selection'.

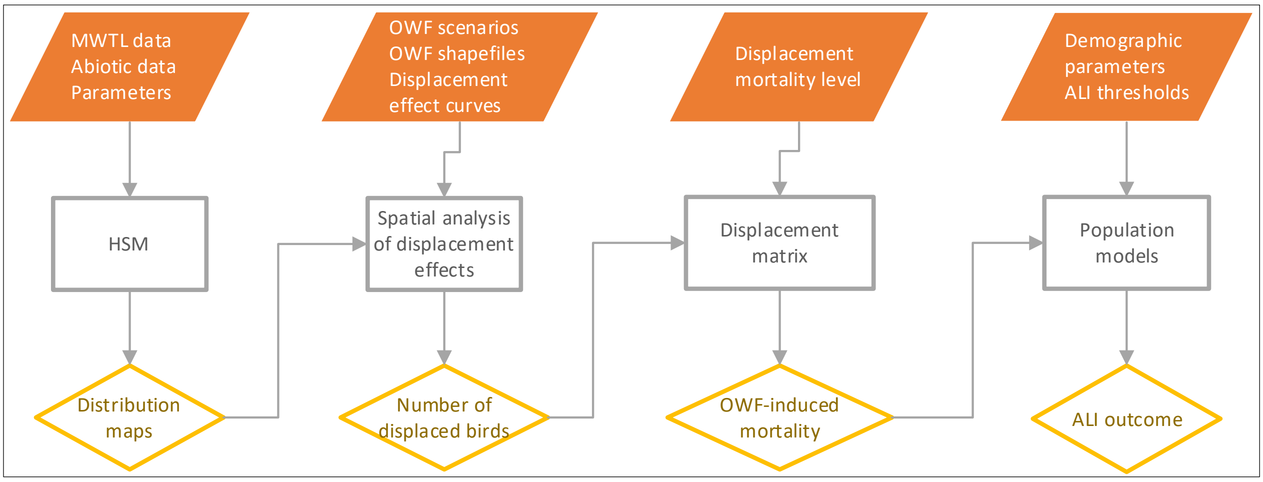

Figure 1 *Outline of the model framework for estimating population effects of displacement for seabirds, modified after Soudijn et al. (2025). Orange filled diamonds indicate inputs required for each step. Grey boxes denote the analysis performed, and the yellow diamonds show the output that are transferred to the next step. HSM = habitat suitability model, ALI = Acceptable Levels of Impact, MWTL (Dutch: Monitoring Waterstaatkundige Toestand des Lands) refers to national surveys in the Netherlands that provides data on bird distribution.*

3.2 Species selection

3.2.1 Sensitivity scoring

The list of bird species considered in the previous KEC assessments has changed over the years, depending on species occurrence in relation to the wind farms in the future windfarm scenarios. In order to make species selection for the KEC more explicit, a new methodology has been developed. This methodology was based on a a sensitivity index scoring method, which prioritises species for inclusion in future KEC assessments. This scoring system is based on species' vulnerability towards collisions, displacement by OWFs, and overall species sensitivity (Heße and Melis 2025). Each individual species was scored using thirteen factors that ranged from zero to one, representing low to high vulnerability. The scoring of each factor was preferably based on the scientific literature, and expert judgement was used if no published information was available.

Heße and Melis (2025) identified four marine and coastal bird species that were not considered within KEC 4 to which they applied their sensitivity scoring approach (Table 3-1). These were yellow-billed loon (*Gavia adamsii*), little auk (*Alle alle*), Manx shearwater (*Puffinus puffinus*), and Arctic tern (*Sterna paradisaea*). The final sensitivity scores of all four species ranked in the 'medium' to 'high' categories, with final scores ranging from 0.46 (little auk) to 0.84 (yellow-billed loon). Heße and Melis (2025) therefore suggested the inclusion of these four species for displacement effects in future KEC assessments.

However, Heße and Melis (2025) also identified uncertainties and issues with species-coverage within the bird count data (MWTL data) for each of these species which serves as the base of the bird distribution maps that are the first step of the KEC framework (see Figure 1 and section 3.3). There are no species-specific observations for the yellow-billed loon in the MWTL data, making it impossible to generate distribution maps for this species and therefore to include it in the KEC assessment as it currently exists. For Manx shearwater and little auk, the number of observations available per species is too limited to generate meaningful species-specific distribution maps. For the Arctic tern, however, Heße and Melis (2025) suggested that there might be sufficient data available to generate a distribution map using the inverse distance weighing approach (see section 3.3), but it is unclear what level of detail these maps would provide and if they could be used in the KEC assessments. These issues have to be addressed before it is possible to make a final decision on including these marine bird species into the next KEC assessment.

3.2.2 Species included in current knowledge update

The number of species included in the KEC habitat loss assessments has varied over the last KEC assessments. While population level effects of seven marine and coastal bird species were included in the KEC 4 assessment for habitat loss (Soudijn *et al*. 2022; Table 3-1), only four species were assessed for OWF effects at population level in KEC 5, due to the limited availability of updated distribution maps.

The four new species that were identified by Heße and Melis (2025) have not been included in any prior KEC assessments and were suggested for inclusion in any future KEC assessment. However, due to issues regarding species coverage in the MWTL data and the potential consequent issues in generating species-specific distribution maps (Heße and Melis 2025), the current knowledge update does not consider these species. A knowledge update for these species should be performed when necessary and if sufficient data is available to make distribution maps.

A knowledge update has been performed for the following six species, which were all included in KEC 4: northern gannet (*Morus bassanus*), Sandwich tern (*Thalasseus sandvicensis*), common guillemot (*Uria aalge*), razorbill (*Alca torda*), Atlantic puffin (*Fratercula arctica*), and northern fulmar (*Fulmarus glacialis*) (Table 3-1). For an additional five species that were considered in KEC 4 (common eider, common scoter, red-throated diver and great cormorant), the OWF sensitivity scoring approach as developed by Heße and Melis (2025) will be applied in a future project (Table 3-1). Based on that outcome, and the data available to construct meaningful distribution maps, a knowledge update for these species will be performed.

Table 3-1 *Species overview. Species included in an assessment are indicated by X. A '!' indicates that a species was included but no population modelling was done. Species for which a decision on inclusion has not been made yet are indicated by a question mark. This table refers to national maps only.*

Species	Dutch name	Most recent national map	KEC 4 2022	KEC 5 2024	National map update 2026	KEC 6 2026
Atlantic puffin – *Fratercula arctica*	Papegaaiduiker	KEC 4 IDW map (2021)	X		New map (HSM or IDW)	X
Common eider – *Somateria mollissima*	Eider	KEC 4 IDW map (2021)	!		?	?
Common guillemot – *Uria aalge*	Zeekoet	HSM map 2024 (2016-2020)	X	X	HSM map (2019-2024)	X
Common scoter – *Melanitta nigra*	Zwarte zee-eend	KEC 4 IDW map (2021)	!		?	?
Red-throated diver - *Gavia stellata*	Roodkeelduiker	KEC 4 IDW map (2021)	X		?	?
Great cormorant – *Phalacrocorax carbo*	Aalscholver	KEC 4 IDW map (2021)	!		?	?
Northern fulmar – *Fulmarus glacialis*	Noordse stormvogel	KEC 4 IDW map (2021)	X		New map (HSM or IDW)	X

Razorbill – *Alca torda*	Alk	HSM map 2024 (2016-2020)	X	X	HSM map (2019-2024)	X
Sandwich tern – *Thalasseus sandvicensis*	Grote stern	HSM map 2024 (2016-2020)	X	X	HSM map (2019-2024)	X
Northern gannet – *Morus bassanus*	Jan-van-gent	HSM map 2024 (2016-2020)	X	X	HSM map (2019-2024)	X
Manx shearwater – *Puffinus puffinus*	Noordse pijlstormvogel	-			?	?
Arctic tern – *Sterna paradisaea*	Noordse stern	-			?	?
Yellow-billed loon – *Gavia adamsii*	Geelsnavelduiker	-			?	?
Little auk – *Alle alle*	Kleine alk	-			?	?

3.3 Bird distribution maps

The first step of the framework is the creation of bird distribution maps (Figure 1). The details of this step are described elsewhere (Van Donk 2024; Van Donk et al. 2024). In summary, there are different types of bird distribution maps, and different types of maps were used for the Dutch part of the North Sea (national maps, based on MWTL data) and the international part of the North Sea (Southern and Central North Sea, based on ESAS data, see Van Donk 2024). Depending mostly on data availability, different methodologies were used to create national bird density maps. For species for which sufficient data were available, habitat suitability models (HSM) were constructed. These models take into account ecological and abiotic covariates to predict bird distribution at locations that were not surveyed and provide a measure of uncertainty of the estimated bird distribution. Maps derived from HSMs are referred to as 'HSM-maps' (Van Donk 2024). If data availability is insufficient to make HSM-maps, the method of inverse distance weighing (IDW) is often applied. This deterministic technique produces density estimates directly frow raw (or averaged) bird counts (Van Donk 2024; Leopold et al. 2014). So-called IDW-maps lack a measure of uncertainty.

For the KEC 5 assessment on habitat loss, national 'HSM-maps' were used for all four species (common guillemot, razorbill, gannet and Sandwich tern). These maps were based on data from the most recent five year period (2016 – 2020; Table 3-1) and separate maps were created for each bimonthly period within the year ('December-January', 'February-March', 'April-May', 'June-July', 'August-September', 'October-November'), except for razorbill for which one map was made for the period April to September due to low number of observations during this time of year. To quantify uncertainty throughout the assessment framework, 1000 species-specific distribution maps were generated for each bimonthly period, as opposed to only using a map describing the mean or median bird distribution as was done in KEC 4 (Soudijn et al. 2022). For the upcoming assessment (KEC 6), updated HSM-maps will be made for at least the four species of KEC 5. These maps will be based on survey data from 2019 to 2024. Depending on data and time considerations, 'Inverse Distance Weighted' (IDW) interpolation will be used to create maps for several other species (Table 3-1).

International maps were either created using IDW techniques, or directly derived from Waggitt et al (2020). Within KEC 5, only the mean abundance estimates provided by Waggitt et al. (2020) were used. Therefore, there was no measure of uncertainty or variation in the distribution of birds for either the international IDW maps, or the Waggitt maps. This was the main reason that no ALI tests were performed for the international scenario. In additional, there are also no internationally agreed ALI threshold values.

3.4 Spatial analysis of displacement effects

3.4.1 Input

In this step of the analysis framework, the number of displaced birds is estimated. This calculation is done for a particular OWF scenario and with particular bird distribution maps. The input for this step consists of the bird distribution maps from the previous assessment step, which for the national analysis consists of 1,000 replicate bird distribution maps per bimonthly period and species. Because there are six bimonthly periods per year, the assessment for a national OWF scenario is based on 6,000 maps per species. For the international analysis the uncertainty in bird distribution could not be quantified and only a single map was used per period and species. This step also requires OWF scenarios, which should include a list of OWFs included in each scenario and information on their spatial location, number of turbines, total area, date of construction, and other relevant information. Finally, information of displacement effects per species is required. This is subject is treated in more detail below.

3.4.2 Modelling displacement effects

The spatial analysis performed in this step relies on assumptions about displacement effects per species. Within KEC 5, displacement effects were calculated using a species-specific effect distance ϑ, which is defined as the minimum distance from the boundary of an OWF at which there is no longer any significant reduction of bird density (Szostek et al. 2024). Birds within a range of ϑ m of the boundary of an OWF could potentially be displaced and, as a consequence, might experience elevated mortality. The probability that a bird is displaced is given by a species-specific displacement probability distribution. This distribution both reflects uncertainty about the true displacement effect, and biological variation that reflects different rates of displacement due to, for example, varying ecological conditions. For KEC 5, this distribution was invariant with respect to the distance from the OWF. Hence, birds close to the OWF had the same probability of being displaced as bird further away from the OWF, but still within ϑ.

In reality, the displacement effect is a gradient, with high displacement close to the OWF that decreases with distance until no significant displacement effect is measurable (Szostek et al. 2024). Instead of a single effect distance combined with a displacement probability, the displacement effect of OWFs on bird density can therefore be better represented by a *displacement effect curve*, which describes the continuous change in probability of displacement (relative effect on bird density) with increasing distance from the boundary of an OWF (Hin and Ransijn 2025). A sensitivity analysis by Hin and Ransijn (2025) analysed the impact of using different displacement effect curves for the common guillemot on the outcome of the KEC 5 assessment. This showed that – at least for the common guillemot - the type of displacement effect curve considered has a large impact of the number of displaced birds and the estimated population effect, with repercussions for whether this effect was deemed as acceptable or not.

Furthermore, studies that report displacement effect curves for the common guillemot show an influence of season, with lower displacement effects in winter compared to autumn when guillemot densities are higher in the southern part of the North Sea (Szostek et al. 2024; Peschko et al. 2024). Within KEC 5 (Soudijn et al. 2025), a single effect distance value was used per species, although this parameter will likely change depending on season, location, or other factors. Hin and Ransijn (2025) argued that the main reason for the attenuation of population impacts for the common guillemot was the use of season-specific displacement effects, as opposed to using a fixed effect distance. Their recommendation is to "use precautionary estimates of displacement that are based on data with a high spatio-temporal representation (multiple years, seasons, and large area), and as much as possible include uncertainty and known sources of variability such as seasonality and distance from OWFs" (Hin and Ransijn 2025). Where possible, we therefore incorporate season-specific displacement effect curves in the KEC assessment methodology for habitat loss, as opposed to using single fixed effect distances per species. However, we also note that displacement effect curves might not be available for all species or seasons, and that in several cases, single effect distances need to be used instead. The displacement effects per species are discussed in section 3.4.3 – Displacement effect per species.

Hin and Ransijn (2025) also developed a method to include uncertainty with respect to the displacement effect curve in the KEC 5 assessment. Although a methodology for dealing with uncertainty was developed by Soudijn et al. (2025), uncertainty in the effect distance was not incorporated in the framework. In the current methodological update we also include uncertainty in displacement effects based on the method as described by Hin and Ransijn (2025).

The spatial analysis of displacement effects is done separately for each bird distribution map, which represents a particular period (either bimonthly, or monthly). This bird distribution map is overlayed by a map that contains the polygons of all wind farms that belong to a certain scenario. These polygons are first adjusted based on the ratio between the realized OWF size (derived from the turbine layout) and the OWF search area, which is often larger for future OWF sites. In KEC 5, the collection of wind farms within a scenario was fixed. Recently, as method has been developed to allow for scenarios in which the collection of wind farms can change through time, dependent on the construction and decommissioning dates of each OWF. This so-called calendar approach is discussed in more detail in IJntema et al. (2025) and requires that the spatial analysis of the number of displaced birds is performed per each unique combination of OWFs that are operational within a certain scenario. Spatial interpolation techniques are then used to derive the number of displaced birds per OWF (Hin and Ransijn 2025). The resulting output that is transferred to the next step consists of a data file with the number of displaced birds per species, combination of OWFs, density map ID, and period within the year associated with the density map.

3.4.3 Displacement effect per species

3.4.3.1 Effect distances and displacement effect curves

There has been little new evidence published regarding effect distances surrounding OWFs since KEC 5. Here, we present the newly published values for each marine bird species separately. In addition, we found a presentation by Garthe et al. (2022) in which effect distances of gannet, guillemot and northern fulmar (among others) were presented. In the species-specific discussion, we used some of these values, although they have not been subject to peer-review yet. New values that could inform the quantification of displacement effects may become available for some of the marine bird species soon (V. Peschko pers. comm.). If so, we recommend that these should be explored and considered for a potential additional update of the displacement probabilities and/or effect distances prior to any future assessment. An overview of all values is provided in Annex Table A1.

Northern gannet

For the northern gannet, new effect distances have been published. For instance, Searle et al. (2025) reported assumed displacement areas of 2000 m for seven out of nine UK OWF sites, averaging up to 1560 m. Although it is unclear what these values are based on, and the authors state that these values are project-specific and "shouldn't be considered general guidance applicable to other wind farm applications" (Searle et al. 2025), they are in line with the effect distance used in KEC 5 based on Vanermen et al. (2015). The values from Searle et al. (2025) further fall in the range of distance values reported by Vanermen et al. (2015) (500–3000 m). For both winter and summer (no month range specified for either of the seasons), additional effect distance ranges from 0–3000 m were identified (Garthe et al. 2022). The mean effect distance value of 1500 m used in KEC 5 is supported by these newly identified ranges and is recommended to be used in the KEC methodology for all bimonthly seasons, along with the effect distance ranges of 0–3000 m as reported by Garthe et al. (2022) (**Table 3-2**).

Sandwich tern

For Sandwich tern, no new information was found on effect distances. Therefore, the effect distance value for Sandwich tern of 1500 m during the breeding season and the following beginning of prospecting and migration season (Apr–May to Aug–Sep) remains unchanged (**Table 3-2**). No additional effect distance ranges have been found.

Common guillemot

For the common guillemot, assumed displacement values were published for two UK OWF sites with 1 and 2 km, respectively (Searle et al. 2025), although it is unclear where these values come from. As follow-up of their 2024 report, Grundlehner et al. (2025) published their effect distances for common guillemot (>10km) reported for the Gemini OWF in the Netherlands from October to March. This, however, did not provide new information compared to the effect distance values considered in KEC 5 for common guillemot. Since the last KEC, effect distances for common guillemots of 6–12 km from July to September and shorter effects distances from October to February (0.4–2 km) were published by Szostek et al. (2024). Both these values are lower than the values reported by Peschko et al. (2024) which reported effect distances between 18–21 km from mid-July to September (mean = 19.5 km) and between 15–18 km (mean 16.5 km) for the months October to February. In KEC 5, the mean effect distance value of 19,500 m from Peschko et al. (2024) was used which is based on long-term data in the German EEZ. Due to its high spatio-temporal resolution and Before-After-Impact-Approach (BACI), the data and results of Peschko et al. (2024) are preferred over the values reported by Searle et al. (2025). Following the precautionary principle and the recommendation of Hin and Ransijn (2025) to aim for using the data with the highest spatio-temporal resolution, the mean values and ranges from Peschko et al. (2024) are recommended to be used in over the values from Szostek et al. (2024) for the months Aug–Mar. From April to July, the values from Searle et al. (2025) are recommended as the majority of guillemots breeds along the UK coast for which these values were reported (**Table 3-2**).

Razorbill
Grundlehner et al. (2025) also published razorbill effect distance values of 2000 m for the Gemini OWF in the Netherlands (Oct–Mar) since the last KEC. This does not provide new information compared to the value used in KEC 5 based on Grundlehner & Leopold (2024). Searle et al. (2025) reported an assumed displacement area of 1 km for one UK OWF, as presented in the respective application for this OWF. This value is lower than the identified 2km by Grundlehner et al. (2025) and could be an underestimation of razorbill effect distances in the autumn and winter months. This is supported by the findings of Szostek et al. (2024). They found effect distances from 6–11 km from Jul–Sep, while they did not detect any significant effect of OWFs in winter (Oct–Feb) for razorbills and consequently no effect distances. The report by Szostek et al. (2024) used data from 2014–2021 across the German EEZ and is therefore the study using the most detailed information that was available for the current knowledge update. It is therefore recommended to use the range of 6,000–11,000 m reported by Szostek et al. (2024) as effect distance in Aug–Sep. While Szostek et al. (2024) did not find any significant effects of OWFs during winter, we recommend following the precautionary principle and applying the reported 2000 m effect distance reported by Grundlehner et al. (2025) to the bimonthly periods Oct–Nov, Dec–Jan and Feb–Mar. During breeding season, most razorbills breed along the UK coast. Therefore, we recommend using the 1000 m effect distance reported for the UK coast by Searle et al. (2025) for the period from April to July (**Table 3-2**).

Atlantic puffin
Atlantic puffin was not included in KEC 5. Therefore, no effect distance values have been reported or updated in the last knowledge update. For Atlantic puffin, little information on OWF effects and OWF effect distances is available. As they don’t visit Dutch or German waters very often, this information on them is not available for these parts of the North Sea. The only identified values for this species are assumed displacement areas, as reported by Searle et al. (2025). These values are based on the effect distance values assumed in the respective applications for two UK OWFs and are also categorised as project-specific by the authors. These assumed displacement area values average at 1500 m, with an assumed range of 1000–2000 m (Searle et al. 2025). Due to the lack of alternative information, these values are included in the methodology for displacement calculations throughout the year (**Table 3-2**). No season-specific information is currently available for Atlantic puffin.

Northern fulmar
Northern fulmar was not included in KEC 5 either. The evidence availability for this species is almost as limited as for Atlantic puffin. Searle et al. (2025) reported values for assumed displacement areas for three UK OWF site applications, ranging from 0–2000 m (mean 1333 m). Slightly larger effect distances have been reported for summer (0–3000 m; no month ranges available for this publication) by Garthe et al. (2022). For spring, this range is increased to 3000–6000 m (Garthe et al. 2022). As the underlying data covers a larger part of the German EEZ and likely multiple study years, we recommend using the ranges by Garthe et al. (2022), following the precautionary principle. We recommend using the range 0–3000 m for the period Jun–Jul, and 3000–6000 m for all other bimonthly periods (**Table 3-2**).

3.4.3.2 Displacement probabilities

New values for displacement probabilities have been found for northern gannet, Sandwich tern, and razorbill since KEC 5.

Northern gannet

For northern gannet, a mean displacement probability of 0.85±0.69 was used in KEC 5 (Vanermen et al. 2015; van Bemmelen et al. 2024). For summer, new values have been published since. Trinder et al. (2024) reported a reduction in mean abundance at the Beatrice wind farm (Moray Firth, Scotland) of 33% from May to August, which gives a mean displacement rate of 67%. Searle et al. (2025) reported an assumed displacement probability rate at UK OWFs of 70%. Garthe et al. (2022) present a displacement effect curve for gannets in summer (no months specified), with displacement probabilities of 75% at 1 km and 32% at 5 km distance from the OWFs. The values from Vanermen et al. (2015) and Trinder et al. (2024) are based on one OWF in the Belgian and one OWF in the UK North Sea, respectively, while values from Garthe et al. (2022) are based on multi-year sampling across multiple OWFs in the German EEZ. Therefore, we recommend using the displacement effect curve from Garthe et al. (2022) for northern gannets for the summer period Jun–Jul. For autumn and winter months (Aug–Mar), no displacement effect curves could be obtained, and we therefore adopt the season-unspecific value of 0.85 as used in KEC 5. As the majority of gannets using Dutch waters gather at their colonies along the UK coast in spring, we recommend using the assumed displacement rate of 70% reported by Searle et al. (2025) for bimonthly period Apr-May, and using the more precautionary summer value of 75% from Garthe et al. (2022) for the period Jun–Jul (**Table 3-2**).

Sandwich tern

For Sandwich tern, a mean displacement probability of 0.54±0.69 (0.39–0.70 95% CI; van Bemmelen et al. 2024) was used in KEC 5. Since KEC 5, a more temporally detailed version of this macro-avoidance data was published, providing season-specific values (van Bemmelen and Fijn 2024). The van Bemmelen and Fijn (2024) study was based on data from tagged Sandwich terns and the displacement effect was not studied as a function of the distance from OWFs. They reported season-specific macro-avoidance rates at 0.67 (0.51–0.78 95% CI) for Apr–May, 0.52 (0.41-0.62 95% CI) for Jun–Jul, and at 0.49 (0–0.74 95% CI) for Aug–Oct. These season-specific values are recommended for use (**Table 3-2**).

Common guillemot

Searle et al. (2025) reported assumed displacement probabilities of 30% for different UK OWFs. Peschko et al. (2024) provide autumn (mid-Jul–Sep) and winter (Oct–Feb) displacement effect curves for the common guillemot. The effect distances and displacement probabilities associated with the autumn displacement curve were used within KEC 5 (**Table 3-2**). The mean winter displacement effect as reported by Peschko et al. (2024) was lower than the displacement in autumn. No new displacement probability values were found for common guillemot. For the bimonthly periods Dec–Jan and Feb–Mar, we recommend using Peschko et al.’s (2024) autumn values of 0.51±0.16 (0.42–0.58 95% CI), and their winter values (0.79±0.097 (0.74–0.83 95% CI)) for the periods Aug–Sep and Oct-Nov. For the periods Apr–May and Jun–Jul, when most guillemots are breeding along the UK coast, we recommend using the UK values provided by Searle et al. (2025). Hin and Ransijn (2025) implemented the season-specific displacement effect curves from Peschko et al. (2024), as opposed to using single effect distances and probability probabilities. This approach will be include in the assessment methodology.

Razorbill

Displacement probabilities for razorbill were previously derived from Grundlehner & Leopold (2024) with a mean value of 0.43±0.021 (0.40–0.45 99% credibility intervals). Inside OWFs, Grundlehner et al. (2025) reported a 40.2% reduction of razorbills, comparable to the value used in KEC 5 and the 40% assumed displacement rate Searle et al. (2025) reported for one UK OWF. Garthe et al. (2022) reported a winter displacement effect curve for razorbills in the German EEZ, with 55% and 47% reduction in bird densities at 1 km and 5 km from German OWFs, respectively. The information in Garthe et al. (2022) is, however, insufficient for the construction of a displacement effect curve and we therefore use the precautionary displacement probability of 0.55 for razorbill from Aug–Mar. During breeding season, Apr–Jul, values from Searle et al. (2025) can be used (**Table 3-2**).

Atlantic puffin
As Atlantic puffin was not included in KEC 5, no displacement probabilities were included in the KBU prior KEC 5 for these two species. For Atlantic puffin, two values were found within this knowledge-update. Searle et al. (2025) reported the use of an assumed displacement rate of 0.40 for two UK OWF sites, while NatureScot (2023) reported a displacement rate value of 0.60 to be used in the UK OWF assessments. This latter value is included in the methodology for all bimonthly periods (**Table 3-2**), following the precautionary principle and common practice as done in the UK. There currently exists no information on how the displacement probability varies with distances from OWFs for the Atlantic puffin.

Northern fulmar
For northern fulmar, which was also not included in KEC 5, multiple values were found. Searle et al. (2025) reported an assumed displacement rate value of 0.30 to be used in two UK OWF applications, while for one site they reported values between 0.01–0.1. For spring and summer, Garthe et al. (2022) reported values at 1 km and 5 km distance from German OWFs which can be used for season-specific displacement probabilities. In spring, they reported reduced densities of 0.91 and 0.84 at 1 km and 5 km from the OWFs, while in summer those values were lower (0.64 and 0.43, respectively). Following the precautionary principle, a value of 0.91 is adopted as displacement probability for the months Aug–May, and 0.64 is used as value for the period Jun–Jul (**Table 3-2**). There currently exists no information on how the displacement probability varies with distances from OWFs for the northern fulmar.

3.4.3.3 Season-specific displacement effects

Displacement effect distances or ranges and displacement probabilities are not available for every season. We see two possible ways to address these missing values. Firstly, missing displacement probabilities could be interpolated from seasonal abundance data and the available probability values. This is based on the assumption that the displacement probability increases with higher bird abundance. However, the form of this relationship remains uncertain, and it may not follow a linear pattern. Investigating this relationship, however, is outside the scope of this assignment. Here, we propose a second approach on dealing with missing seasonal values for both effect distances/ranges and displacement probabilities that takes species' spatio-temporal distributions across the North Sea into account.

For instance, Sandwich terns only frequent the (Dutch) North Sea from around April onwards until they leave their colonies for prospecting flights and then migrate to their wintering areas around October (Tree 2011; Fijn et al. 2014). Therefore, distribution maps for Sandwich terns only consider the months from April to late summer, and no displacement values are required for the remaining months of the year.

The largest numbers of northern gannets, razorbills, Atlantic puffins, and common guillemots in Dutch waters during autumn and winter originate from breeding colonies along the east coast of Great Britain. In summer, these species are found predominantly at their breeding colonies and their foraging ranges hardly exceed outside UK waters. Therefore, where available, spring and summer values for these species can be used from Searle et al. (2025) who focussed on UK OWFs. This means that, for instance, missing displacement distance value(s) and displacement probability rates for northern gannet in spring, and for razorbill in spring and summer can be used from Searle et al. (2025) (**Table 3-2**). The species are therefore expected to show the greatest overlap in distribution with UK OFWs, making the UK displacement figures the most applicable. For all other species-season combinations with missing parameters values, the most precautionary value for each species is recommended. For instance, the effect distance range for northern fulmar in autumn is recommended to be set to 3000–6000 m as for spring (**Table 3-2**). In cases for which there are two values originating from the UK are available for the same season, i.e. Atlantic puffin displacement probabilities from NatureScot (2023) and Searle et al. (2025), the more precautionary value is recommended (**Table 3-2**).

Table 3-2 *Updated parameters recommended to model displacement effects per marine bird species and shapes of the distributions for the parameter values if available. Effect distances and displacement probabilities are presented by bimonthly period. Mean values, value ranges, standard deviation (SD), 95% confidence intervals (CI) or 99% credibility intervals (CrI) are provided where possible. Values without an indication are general values without further specification provided in the respective sources. Values also used in KEC 5 are indicated with an asterisk. Sources for each value are indicated by the letters a–n behind each value.*

Description	Symbol	Unit	Distribution	Bimonthly **period** map	Northern gannet	Sandwich tern	Common guillemot	Razorbill	Atlantic puffin	Northern fulmar
Effect distance around OWFs	θ	m								
				December–January	mean = 1,500 (n), range = 0–3,000 (a)	-	mean = 16,500 (f), range = 15,000–18,000 (f)	2,000 (d)*	mean = 1,500 (g), range = 1,000–2,000 (g)	range = 3,000–6,000 (a)
				February–March	mean = 1,500 (n), range = 0–3,000 (a)	-	mean = 16,500 (f), range = 15,000–18,000 (f)	2,000 (d)*	mean = 1,500 (g), range = 1,000–2,000 (g)	range = 3,000–6,000 (a)
				April–May	mean = 1,500 (n), range = 0–3,000 (a)	1,500 (k)*	range = 1,000–2,000 (g)	1,000 (g)	mean = 1,500 (g), range = 1,000–2,000 (g)	range = 3,000–6,000 (a)
				June–July	mean = 1,500 (n), range = 0–3,000 (a)	1,500 (k)*	range = 1,000–2,000 (g)	1,000 (g)	mean = 1,500 (g), range = 1,000–2,000 (g)	range = 0–3,000 (a)
				August–September	mean = 1,500 (n), range = 0–3,000 (a)	1,500 (k)*	mean = 19,500 (f)*, range = 18,000–21,000 (f)	range = 6,000–11,000 (h)	mean = 1,500 (g), range = 1,000–2,000 (g)	range = 3,000–6,000 (a)
				October–November	mean = 1,500 (n), range = 0–3,000 (a)	-	mean = 16,500 (f), range = 15,000–18,000 (f)	2,000 (d)*	mean = 1,500 (g), range = 1,000–2,000 (g)	range = 3,000–6,000 (a)
Displacement probability	ρ	-	beta							
				December–January	mean = 0.85 (n)*, SD = 0.69 (from Sandwich tern (k)*)	-	mean = 0.51 (f), CI = 0.42–0.58 (f), SD = 0.16 (inferred from CI (f))	0.55 (a)	0.60 (e)	0.91 (a)
				February–March	mean = 0.85 (n)*, SD = 0.69 (from Sandwich tern (k)*)	-	mean = 0.51 (f), CI = 0.42–0.58 (f), SD = 0.16 (inferred from CI (f))	0.55 (a)	0.60 (e)	0.91 (a)
				April–May	0.70 (g)	mean = 0.67 (m), CI = 0.51–0.78 (m)	0.30 (g)	0.40 (g)	0.60 (e)	0.91 (a) (at 1km from OWF)
				June–July	0.75 (a) (at 1km from OWF)	Jun–Jul: mean = 0.52 (m), CI = 0.41–0.62 (m)	0.30 (g)	0.40 (g)	0.60 (e)	0.64 (a) (at 1km from OWF)
				August–September	mean = 0.85 (n)*, SD = 0.69 (from Sandwich tern (k)*)	Aug–Oct: mean = 0.49 (m), CI = 0.00–0.74 (m)	mean = 0.79 (f)*, CI = 0.74–0.83 (f) * SD = 0.097 (inferred from CI (f)*)	0.55 (a)	0.60 (e)	0.91 (a)
				October–November	mean = 0.85 (n)*, SD = 0.69 (from Sandwich tern (k)*)	-	mean = 0.79 (f)*, CI = 0.74–0.83 (f) * SD = 0.097 (inferred from CI (f)*)	0.55 (a)	0.60 (e)	0.91 (a)
Sources	(a) Garthe et al. (2022), (b) Garthe et al. (2023), (c) Grundlehner & Leopold (2024), (d) Grundlehner et al. (2025), (e) NatureScot (2023), (f) Peschko et al. (2024), (g) Searle et al. (2025), (h) Szostek et al. (2024), (i) Trinder et al. (2024), (k) van Bemmelen et al. (2024), (m) van Bemmelen & Fijn 2024), (n) Vanermen et al. (2015)									

3.5 Methodology of casualty calculation

3.5.1 Input

To calculate the number of casualties, the inputs needed are the number of displaced birds per species, unique OWF combination, distribution map, and period within the year associated with the bird distribution map. The unique OWF combination refers to the set of OWFs that are operational at a certain moment in time. Note that this step requires the number of birds displaced per OWF scenario, instead of the number of birds in (potential) OWF areas (+ buffer zone). As opposed to KEC 5, the probability of displacement for a bird located within an OWF area (+ buffer) is applied during the spatial analysis in the previous step (see 3.4 Spatial analysis of displacement effects). The only other required inputs for this step are displacement mortality rates, which quantify the annual mortality probability that birds experience as a consequence of displacement.

3.5.2 Displacement matrix approach

Within KEC 5, mortality from displacement by OWFs was calculated using the 'displacement matrix approach' (JNCC 2015; Searle et al. 2022; SNCB 2022). Basically, this approach multiplies the fraction of the population within proposed wind farm sites by the percentage that is expected to be displaced and the percentage of displaced birds that are expected to die from displacement (Searle et al. 2022). The displacement matrix approach replaced the 'Bradbury method' used within earlier KEC assessment (KEC 4: Soudijn et al. 2022), because it separates the probability of displacement from its mortality consequences. This separation allows for a better treatment of the uncertainty of these two distinct processes. Within KEC 5, the probability of displacement was represented by a probability distribution, while four discrete mortality levels were used (1%, 2%, 5% and 10%) to cover a broad range of mortality outcomes.

We implement displacement effect curves in the KEC assessment methodology, if such information is available. Otherwise, fixed effect distances are used. The existing displacement matrix approach and the displacement probability now varies with season (potentially) and distance to the OWF. The number of displaced birds is calculated as part of the spatial analysis in the previous step (see 3.4 Spatial analysis of displacement effects). The analysis in the current step focuses on aggregating the number of displaced birds across the different periods within a year and multiplying this with a displacement mortality rate. Specifically, if the number of displaced birds for a particular species, OWF scenario and density map is denoted by D_i, with i indicating the period of the year (monthly or bimonthly), then the annual additional mortality due to displacement for the population is given by:

$$m_d = \frac{\sum_{i=1}^{n} D_i / n}{N_{max}} \mu_d$$

Here, μ_d represents the adopted displacement mortality probability, n denotes the number of distinct periods per year (either 6 or 12 depending of the bird distribution maps), and N_{max} represents the total population abundance, quantified as sum of bird abundance of the distribution map from the period within the year where population abundance is highest. Per species, OWF scenario, and mortality level (μ_d) we obtain 1000 replicate estimates of m_d. One for each replicate map. This is used as input for the analysis of population effects (see 3.6: Quantifying population effects).

3.5.3 Displacement mortality values

The assumed mortality level associated with displacement (μ_d) is probably the most influential, unknown parameter of the current assessment framework and has great influence on the assessment outcome. In KEC 5 (Soudijn et al. 2025), four different levels were adopted (1%, 2%, 5% and 10%) and ALI violations were found for the northern gannet and common guillemot at a mortality levels of 5% and 10%, but not for 1% and 2%. For each species, a benchmark mortality level was calculated as the ratio between the Relative Displacement Risk Score (RDRS) as used in KEC 4, and the mean displacement value (Soudijn et al. 2025).

Table 3-3 shows the updated benchmark mortality levels for each species, including the Atlantic puffin and the northern fulmar that were not assessed in KEC 5.

Table 3-3 *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. Per species, the maximum displacement value of the different seasons was taken from* ***Table 3-2****.*

Species	**Maximum displacement value (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	0.67	0.024	0.036	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	0.55	0.036	0.065	0.01, 0.02, 0.05, 0.1*
Atlantic puffin	0.60	0.024	0.04	0.01, 0.02, 0.05*, 0.1
Northern fulmar	0.91	0.004	0.0044	0.01*, 0.02, 0.05, 0.1

A recent project funded by the ORJIP Offshore Wind programme reviewed existing information on displacement mortality rates for six species, which were considered to be at risk of displacement mortality (black-legged kittiwakes, common guillemot, razorbill, Atlantic puffin, red-throated diver and northern gannet) and was unable to retrieve any empirical evidence for the mortality of seabirds that have been displaced by OWFs (Searle et al. 2025). The authors concluded that displacement mortality likely hasn't been studied to date, rather than not been reported. Displacement mortality rates commonly used within the displacement matrix approach were 1%-10% for most species, except 1-5% for the northern gannet and 1-3% for the black-legged kittiwake (Searle et al. 2025). An expert elicitation (EE) workshop was conducted within the same project, in which the 'bulk of belief' for the displacement mortality rate for individuals (breeding or non-breeding) was on values less than 10% (Searle et al. 2025). However, there was considerable variation between experts regarding the upper limit of the displacement mortality rate. The mortality rate of dependent chicks at the nest was also considered in the EE process, and was estimated as being substantially greater than for mature individuals, although there was less agreement and certainty among the experts' responses (Searle et al. 2025). Lastly, model-based displacement mortality rates were estimated using SeabORD (Searle et al. 2014; 2018), an individual-based model of seabird behaviour, energetics, demography and windfarm interactions in the breeding season. Across different scenarios of Special Protection Areas (SPAs) and OWF setups, the SeabORD model estimated that the maximum levels of mean impact (per scenario) on population-level mortality rates are highest for black-legged kittiwake (an 0.079 and 0.005 increase in mortality rates for, respectively chicks and adults), followed by razorbill (0.007 chicks; 0.003 adults) and common guillemot (0.006 chicks; 0.001 adults). However, mortality estimates from both the EE procedure and the SeabORD IBM cannot be used within the displacement matrix calculation, because these mortality rates apply to systematically different populations. This systematic difference arises because the EE and the SeabORD IBM account for turnover in space use of birds at sea, whereas the current displacement matrix approach does not (Searle et al. 2025). The mortality estimates derived using the SeabORD IBM and the EE procedure were therefore not used to revise the mortality probabilities as used in the KEC assessment.

3.5.4 Alternative approaches to quantify displacement effects

As an alternative to the displacement matrix approach, the SeabORD individual-based model simulates individual seabird movement and energetics to estimate the costs of displacement and barrier effects to individual seabirds in terms of survival and productivity (Searle et al. 2018; 2025). SeabORD was developed for the Forth and Tay region of Scotland to estimate additional mortality from OWFs during the chick rearing period during the breeding season and is currently parameterized for four species (black-legged kittiwake, common guillemot, razorbill, Atlantic puffin; Searle et al. 2025). There is currently no well-tested simulation tool or IBM to estimated additional mortality resulting from OWF displacement for the full annual cycle. An individual-based energetic model for an adult common guillemot and an adult gannet that covers the non-breeding season is currently being developed at Wageningen Marine Research. The goal of this model development is to replace the current displacement matrix approach within the KEC assessment for habitat loss. Until this model is available the displacement matrix method will be used with the KEC assessment for habitat loss.

3.6 Quantifying population effects

3.6.1 Population models

Population effects of habitat loss are quantified using stochastic, stage-based, matrix population models without density-dependence (Caswell 2001). The specific structure of the population models as used within the KEC for seabird habitat loss and collisions effects has been described earlier by Van Kooten et al. (2019), Hin and Soudijn (2021), Soudijn et al. (2022) and Soudijn et al. (2024). No changes to the structure of the population models have been made since the previous KEC assessment (KEC 5; Soudijn et al. 2024). Atlantic puffin and northern fulmar were not considered in the last KEC, and the modification made to the matrix models used in the last KEC were also applied to the population models for these two species. This involved the splitting of life stages into distinct age-classes to appropriately accommodate fluctuations in the stage-distribution that can arise from the use of yearly variations in demographic rates due to environmental stochasticity. The breeding and non-breeding matrices of the six species considered in this knowledge update are summarized in Table 3-4 and Table 3-5. An update of the parameters is given in the next section (3.6.2: Demographic parameters). A detailed description of the analysis of population effects is given below.

3.6.1.1 Matrix structure

The processes described by the matrix model are captured by two separate matrices. The summer matrix $\mathbf{A_S}$ describes the breeding season and deals with reproduction. The winter matrix $\mathbf{A_W}$ describes the non-breeding period and deals with survival. Together the summer and winter matrices describe the demographic processes within a single year. Each matrix element p_{ij} within row i and column j describes the probability that an individual within stage i gives rise to an individual in stage j. This can be a transition between stages due to survival, or the creation of new individuals through reproduction. The matrices use a consistent parameter notation between species, where F_A denotes breeding success, P_F the probability that an adult skips breeding and S_i the survival probability of an individual in stage i. Reproduction is oftentimes represented as the product of breeding success and the probability of breeding: $\frac{F_A}{2} \cdot (1 - P_F)$, where the division by 2 corrects for the proportion of female chicks. Because of the stochastic nature of the population models, each parameter is specified by a distribution (see section 3.6.2 - Demographic parameters), as opposed to a single, fixed number.

3.6.1.2 Baseline population projections

The first step in the analysis of population effects is to run baseline population projections. This is done a large number of times (in KEC 5 n = 100,000 replicate projections were simulated) to represent the annual variation of demographic rates as a result of environmental stochasticity. Parameters of the winter and summer matrices required to run the replicate simulations were generated once. Each annual time step requires a different set of parameter values. As a result, n population projections across t years require n * t samples of each parameter. The sampled parameters are used to create n sequences of baseline winter and summer matrices, each of length t. These sequences are then vector-multiplied into sequences of annual projection matrices $\mathbf{A}$, where

$$\mathbf{A} = \mathbf{A_W} \cdot \mathbf{A_S}$$

The sequences of annual projection matrices are stored and used to run n population projections across t years.

Besides a sequence of matrices, a population projection also requires an initial population state. For this, the stable stage distribution is calculated from the deterministic annual projection matrix, which is an annual projection matrix with all parameters set at their mean value (Table 3-6). To calculate the number of individuals per stage in the initial time step, the stable stage distribution is multiplied by the measure of total population abundance as derived from the bird distribution maps (see 3.5 Methodology of casualty calculation). Within KEC 5 there were 1,000 replicate bird distribution maps (for the national scenarios) resulting in 1,000 estimates of total population abundance. With 100,000 replicate population simulations each estimate of population abundance was used a hundred times on average (using random selection).

3.6.1.3 Impacted population projections

The impacted population projections are generated from the same matrices that were used for the baseline population projections, but modified based on the estimated OWF-induced mortality. Each sequence of baseline matrices therefore has a paired sequence of impacted matrices. This ensures that all processes unrelated to OWF effects remain identical for a particular pair of unimpacted and impacted projections, as recommended by Hin et al. (2024). Each matrix within a sequence is modified by using the impact matrix $\mathbf{M}$ (Table 3-4 and Table 3-5) with entries m_i being stage-specific weighing factors (Table 3-6). These factors determine how OWF-induced mortality modify the survival probability of each life stage. These weighing factors divide the estimated OWF-mortality across the different life stages of a species. If, for a particular species, the immature stages do not inhabit the North Sea, the weighing factor for those stages will be zero, and the estimated mortality will only be applied to the other life stages. For a particular mortality value m_d, each baseline annual projection matrix $\mathbf{A}$ is modified according to:

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

Here, $\circ$ is the Hadamard product (element-wise multiplication). The resulting annual projection matrix $\mathbf{A_M}$ includes the mortality effects of OWFs on each stage of the matrix $\mathbf{A}$ weighted by the weighing factors m_i.

Within KEC 5, a particular OWF scenario consisted of a fixed set of wind farms, irrespective of the dates of wind farm construction or decommissioning. Consequently, a single value of m_d was used for each sequence of $t = 40$ matrices. In the current methodological update, scenarios are modified to include a calendar approach that accounts for the date of construction and expected lifetime of each OWF. As a result, the set of OWFs that impose mortality upon a particular seabird population will change throughout each projection. This is incorporated into the analysis by making m_d time dependent, i.e. $m_{d,i}$ for $i \in (0,1,\ldots,t)$.

Table 3-4 *Summer, winter and impact matrices for the different species. Parameters of summer and winter matrices are specified in Table 3-7 to Table 3-11. 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 3-6.*

Species	Summer matrix ($\mathbf{A_S}$)	Winter matrix ($\mathbf{A_W}$)	Impact matrix ($\boldsymbol{M}$)
Northern gannet - *Morus bassanus*	$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_1 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_2 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_3 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & m_6 \end{pmatrix}$
Sandwich tern – *Thalasseus sandvicensis*	$\begin{pmatrix} 1 & 0 & 0 & \phi F_A(1-P_F) & \phi F_A(1-P_F) & F_A(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_{12} & 0 & 0 & 0 & 0 \\ 0 & 0 & S_{12} & 0 & 0 & 0 \\ 0 & 0 & 0 & S_A & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & m_1 & m_1 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & m_6 \end{pmatrix}$
Common guillemot - *Uria aalge*	$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_A & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & m_6 & m_7 \end{pmatrix}$
Razorbill - *Alca torda*	$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_{01} & 0 & 0 & 0 & 0 & 0 \\ 0 & S_{01} & 0 & 0 & 0 & 0 \\ 0 & 0 & S_A & 0 & 0 & 0 \\ 0 & 0 & 0 & S_A & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & m_6 \end{pmatrix}$
Atlantic puffin – *Fratercula arctica*	$\begin{pmatrix} 1 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_{F4}) & \frac{F_A}{2}(1-P_{F5}) & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ S_{03} & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_{03} & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_{03} & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_{34} & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_{45} & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_5 & S_A \end{pmatrix}$	$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & m_6 & m_7 \end{pmatrix}$

Table 3-5 *Summer, winter and impact matrices for the northern fulmar. Parameters of summer and winter matrices are specified in Table 3-12. 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 3-6.*

Summer matrix

$$\begin{pmatrix} 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & \frac{F_A}{2}(1-P_F) \\ 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 1 \end{pmatrix}$$

Winter matrix

$$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ S_J & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & S_J & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_J & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_J & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_J & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_A(1-P_5) & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & S_A(1-P_6) & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & S_A(1-P_7) & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & S_A(1-P_8) & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & S_A(1-P_9) & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & S_A(1-P_{10}) & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & S_A(1-P_{11}) & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & S_AP_5 & S_AP_6 & S_AP_7 & S_AP_8 & S_AP_9 & S_AP_{10} & S_AP_{11} & S_A & S_A \end{pmatrix}$$

impact matrix

$$\begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ m_1 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & m_1 \\ 0 & m_2 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & m_3 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & m_4 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & m_5 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & m_6 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & m_7 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & m_8 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & m_9 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & m_{10} & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 1 & 0 & 0 & 0 & 0 & 0 & m_{11} & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & m_{12} & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & m_6 & m_7 & m_8 & m_9 & m_{10} & m_{11} & m_{12} & m_{13} & m_{14} \end{pmatrix}$$

Table 3-6 *Stable stage distribution and the stage-specific weighing factor of mortality effects of habitat loss. The stage distribution is based on a pre breeding census (population counts right before reproduction), which leads to zero abundance in the first age class.*

Species	**Number of stages (n)**	**Stable stage distribution (pre-breeding)**	**Weighing factor per life stage (m_i with $i \in \{1, \ldots, n\}$)**
Northern gannet	6	[0.0; 0.107 0.085; 0.073; 0.063; 0.67]	[0.93; 0.93; 0.93; 0.93; 1.026; 1.026]
Sandwich tern	6	[0.0; 0.096; 0.074; 0.057; 0.053; 0.72]	[0; 0; 0; 1.205; 1.205; 1.205]
Common guillemot	7	[0.0; 0.106; 0.081; 0.068; 0.062; 0.057; 0.625]	[0; 0; 0; 1.23; 1.23; 1.23; 1.23]
Razorbill	6	[0.0; 0.119; 0.078; 0.071; 0.065; 0.668]	[0; 0; 1.135; 1.135; 1.135; 1.135]

Species	Number of stages (n)	Stable stage distribution (pre-breeding)	Weighing factor per life stage (m_i with $i \in \{1, \dots, n\}$)
Atlantic puffin	7	[0.0; 0.148; 0.106; 0.075; 0.054; 0.042; 0.575]	[0; 0; 0; 1.339; 1.339; 1.339; 1.339]
Northern fulmar	14	[0.0; 0.077; 0.069; 0.061; 0.055; 0.049; 0.04; 0.028; 0.016; 0.008; 0.003; 0.001; 0; 0.594]	[0; 0; 0; 0; 0; 1.356; 1.356; 1.356; 1.356; 1.356; 1.356; 1.356; 1.356; 1.356]

3.6.2 Demographic parameters

3.6.2.1 General approach

For KEC 5, it was decided that model parameters should be based on demographic rates from colonies closest to the Southern North Sea. In case there were multiple relevant colonies for a particular species, data from these colonies were weighted by colony size (number of breeding pairs or occupied nests). Standard deviations were weighted in a similar manner, if data availability allowed. Linear interpolation was used in case no estimate of colony size was available for a certain year. In the KEC 5 knowledge base update (Chapter 3 in Soudijn et al. 2025), model parameters were based on demographic rates from the period 2016-2021, in line with the seabird survey data that were used to create the bird density maps.

The current update of demographic parameters was conducted in an identical manner, but based on demographic data from more recent years (2019 – 2025). The Seabird Monitoring Program[2] database was consulted for the most recent data on productivity (breeding success) and colony sizes. To calculate the breeding success parameter, productivity estimates that reported number of fledged chicks per monitored site were divided by counts of the number of breeding pairs or apparently occupied nests (AON) for that site. Survival estimates can only be reliably measured over longer time periods and were therefore not constrained to the period 2019 – 2025. Specific data sources and resulting model parameters are discussed per species below.

3.6.2.2 Northern gannet

Productivity values were collated from the colonies at Bempton Cliffs, Bass rock and Helgoland from the period 2019 - 2024. This includes two years (2022 & 2023) when breeding success was significantly depressed due to high pathogenic avian influenza virus (HPAIV). For now, we consider the recent outbreak of HPAIV as a single aberrant event. Incorporating demographic estimates from two years of HPAIV outbreak within a five year period, would lead to parameter estimates that are not representative for the long-term growth rate of the gannet population. We therefore exclude the years 2022 and 2023 for the estimate of the breeding success. The breeding success parameter was therefore based on the years 2019, 2020, 2021 and 2024, which resulted in a mean value of 0.72, with a standard deviation of 0.094 (Table 3-7). Future knowledge updates should pay attention to the possibility that HPAIV outbreaks, or similar epidemics, could have longer term or more frequent effects on demographic rates.

For adult survival we retain the pre-HPAIV estimate by Lane et al. (2024) for the period 2011-2021. No new original data could be find for the other parameters.

Table 3-7 *Parameters for the northern gannet population model. In case values deviate from those used within KEC 5 (Soudijn et al. 2025), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.72 (0.754)	0.094 (0.031)	Beta	SMP 2025; Dierschke pers. comm.
Probability floater	P_F	(0.05)	(0.125)	Beta	KEC 5
Survival age 0	S_0	(0.481)	(0.0853)	Beta	KEC 5
Survival age 1	S_1	(0.816)	(0.0393)	Beta	KEC 5

[2] https://app.bto.org/seabirds/public/index.jsp

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Survival age 2	S_2	(0.884)	(0.0293)	Beta	KEC 5
Survival age 3	S_3	(0.887)	(0.0301)	Beta	KEC 5
Adult survival	S_A	0.940	(0.0483)	Beta	(Lane et al. 2024)

3.6.2.3 Sandwich tern

A new value for the Sandwich tern breeding success parameter was derived from data in Lilipaly et al. (2024), who reported number of fledged chicks per pair in the Dutch Delta area for the years 2020-2024. In 2022 and 2023, breeding success was significantly depressed due to HPAIV (Lilipaly et al. 2025). These years were excluded for the same reason as described above for the northern gannet. The reported productivity for years 2020, 2021 and 2024 resulted in a mean breeding success of 0.79 (SD = 0.125). Accounting for female chicks only leads to a mean breeding success parameter of 0.40 (SD = 0.063; Table 3-8)

No new values for the other parameters could be obtained.

Table 3-8 *Parameters for the Sandwich tern population model. In case values deviate from those used within KEC 5 (Soudijn et al. 2025), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (female chicks only)	F_A	0.40 (0.245)	0.063 (0.08)	beta	(S J Lilipaly et al. 2024)
Probability floater	P_F	(0.1)	(0.125)	beta	KEC 4
Breeding success scalar	ϕ	(0.3)	0.0	Fixed	KEC 5
Survival age 0	S_0	(0.357)	(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)	(0.108)	beta	KEC 4; van Bemmelen *et al.,* (2022)

3.6.2.4 Common guillemot

The new parameter for common guillemot breeding success was derived from the mean breeding success at the Bempton Cliffs colony over the years 2019 – 2024 (*Table 3-9*). The current estimate for adult survival (0.949) was an average of data from five colonies that are all within the range of 0.93 – 0.96. The most recent values for four out of these five colonies are from 2006, and for a single colony (Skomer Island, Wales) data up to 2011 was used. More recent adult survival values are reported by Laurenson et al. (2025) for Skomer Island (up to 2020) and Wanless et al. (2023) for Isle of May (up to 2019). This resulted in a mean survival estimate of 0.931 with a standard deviation of 0.037 (*Table 3-9*). No new original values for the other parameters were retrieved.

Table 3-9 *Parameters for the common guillemot population model. In case values deviate from those used within KEC 5 (Soudijn et al. 2025), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.61 (0.622)	0.046 (0.045)	beta	(SMP 2025)
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	0.931 (0.949)	0.037 (0.0447)	beta	(Laurenson et al. 2025; Wanless et al. 2023)

3.6.2.5 Razorbill

Razorbill breeding success parameters were updated based on the breeding productivity at the Flamborough and Filey Coast SPA (Bempton Cliffs) in the period 2019 – 2024. This resulted in a mean breeding success of 0.56 with a standard deviation of 0.097 (Table 3-10).

Razorbill adult survival rate was derived for KEC 5 as the weighted average value of data from five different colonies, including Skomer and Skokholm islands (Wales) and Hornøya (north Norway). The proportion of colour-ringed adults returning after winter on the Isle of May are reported by NatureScot (2024). The five year average across 2019 – 2023 was 87%, and the most recent value reported for 2024 was only 50%. However, these values are not properly estimated survival rates and the most recent value will likely further increase if more birds are resighted over time. Semple & Harker (2024) use a mean survival 0.895 based on SMP data, which is the UK nationally derived value reported by Horswill and Robinson (2015). This value seems to be in accordance with the average resighting rate at Isle of May in recent years (2019 – 2023) and was adopted as the new mean survival estimate. The associated standard deviation is 0.067.

No new original values for juvenile/immature survival and floater probability were found.

Table 3-10 *Parameters for the razorbill population model. In case values deviate from those used within KEC 5 (Soudijn et al. 2025), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.56 (0.615)	0.097 (0.085)	beta	(SMP 2025)
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	0.895 (0.911)	0.067 (0.0663)	beta	Semple & Harker (2024)

3.6.2.6 Atlantic puffin

Hin and Soudijn (2021) present an overview of the availability of demographic data on the Atlantic puffin, but only report breeding success estimates up to 2015. These data were derived from various colonies around the UK and Wales and include data that date back to the 1980s.

More recent (>2019) data on breeding productivity of Atlantic puffins from colonies bordering the (Southern) North Sea are available for Coquet Island, Farne Island, Isle of May and Fair Island from the Seabird Monitoring Programme (SMP 2025). The Isle of May is one of the largest breeding colonies for puffins (puffinry) in Britain with an estimated 52,104 (confidence interval 44,846 – 62,123) Apparently Occupied Burrows (AOB) in 2024 (NatureScot 2025). This is an increase of 33% from 39,000 AOB in 2017, but it is thought that the breeding population is currently declining (NatureScot 2025). Breeding success on Isle of May was estimated at 0.77 fledged chicks per nest in 2024. The Farne Islands SPA colony has an estimated number of 42,378 breeding pairs in 2021, with a fledging success of 59% (Hendry et al. 2022). In 2024, there were an estimated 50,103 breeding pairs on the Farne Islands SPA, with a breeding productivity of 0.51 fledged chicks per nest. The colonies at Coquet Island and Fair Island are much smaller with, respectively ~18,000 pairs in 2025 and 6,666 pairs in 2015. The breeding success parameter was calculated from data since 2019 by taking the weighted average of breeding success and colony size. This resulted in an average value of 0.68, which is slightly below the value used in KEC 4 (0.70) that was based on longer time periods and included data from colonies non-adjacent to the North Sea (Table 3-11)

For KEC 4 the immature survival rates for the Atlantic puffin were based on Breton et al. (2006). Similar rates were also used by Semple & Harke (2024) in the NE PVA tool. No new values for immature / juvenile survival have become available. The adult survival rate of 0.93 as used in KEC 4 was based on an average of values reported by Harris et al. (2005; 2013), weighted by number of years. This includes data from Norway (Røst and Hornøya colonies) and Skomer Island (Wales). More recently, Landsem et al. (2023) report a value of 0.935 (confidence interval: 0.926 – 0.942) for the Isle of May during the period 1990 – 2020. For the same colony, Layton-Matthews et al. (2023) report a value of 0.92 (confidence interval: 0.9 – 0.94) for the period 1984 – 2019. We adopt the value of Landsem et al. (0.935) because it includes more recent data. The associated standard deviation was derived at 0.0227.

Johnson et al. (2024), citing Hansen et al. (2023), report age-dependent breeding probability for puffins breeding in Iceland. This probability is low for 3 and 4 year olds (6.7%), 70% for 5 year olds and 100% for age 6+. We have adjusted our probabilities of skipped breeding based on these values (Table 3-11).

Table 3-11 *Parameters for the Atlantic puffin population model. In case values deviate from those used within KEC 4 (Soudijn et al. 2022), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.68 (0.70)	0.12 (0.11)	Beta	(SMP 2025; NatureScot 2025; Hendry et al. 2022)
Skipped breeding probability 4 year olds	P_{F4}	0.933 (0.6)	(0.01)	Beta	(Johnson et al. 2024; Hansen 2023)
Skipped breeding probability 5 year olds	P_{F5}	0.3 (0.3)	(0.01)	Beta	(Johnson et al. 2024; Hansen 2023)
Skipped breeding probability adults	P_F	0.0 (0.078)	0.0 (0.01)	Beta	(Johnson et al. 2024; Hansen 2023)
Annual survival age 0 – 3	S_{03}	(0.71)	(0.11)	Beta	KEC 4
Annual survival age 4	S_4	(0.78)	(0.092)	Beta	KEC 4
Annual survival age 5	S_5	(0.80)	(0.083)	Beta	KEC 4
Annual survival adults	S_A	0.935 (0.93)	0.0227 (0.057)	Beta	(Landsem et al. 2023)

3.6.2.7 Northern fulmar

Mean breeding success (F_A) for the northern fulmar was derived as 0.42 for KEC 4 (Soudijn et al. 2022) by averaging productivity values from colonies at Orkney Islands, Isle of May, Skomer Island (Wales) and Fair Island. This included a productivity time series from Eynhallow dating back to 1958 (Lewis et al. 2009). We collated values of breeding productivity between 2019 and 2025 from 12 UK colonies bordering the North Sea and Helgoland. From these values an average breeding success of 0.42 was derived (sd = 0.23), weighted by colony size. Interestingly, this value is the same as was used for KEC 4 (Table 3-12).

Most survival estimates for the northern fulmar date back to the previous century, especially from the long-term study on Eynhallow, Orkney Islands Scotland (Buckland 1982; Dunnet et al. 1979; Orzack et al. 2011; Grosbois and Thompson 2005). We therefore chose to adopt the more recent value of Fayet et al. (2025) of 0.927 (sd = 0.086) for Jan Mayen, Norway over the period 2014 – 2024. This value is slightly lower than the value used for KEC 4 (0.936), which was taken from Grosbois & Thompson (2005).

Other parameters were kept at their original values.

Table 3-12 *Parameters for the northern fulmar population model. In case values updated from those used within KEC 4 (Soudijn et al. 2022), the previous values are indicated within parentheses.*

Parameter	Symbol	Mean	Standard deviation	Distribution	Source
Breeding success (productivity)	F_A	0.42 (0.42)	0.23 (0.13)	Beta	(SMP 2025)
Probability floater	P_F	(0.304)	(0.113)	Beta	KEC 4
Immature survival	S_J	(0.884)	(0.054)	Beta	KEC 4
Annual survival adults	S_A	0.927 (0.936)	0.086 (0.055)	Beta	(Fayet et al. 2025)
Prob. Recruitment age 5	P_5	(0.125)	(0.01)	Beta	KEC 4
Prob. Recruitment age 6	P_6	(0.25)	(0.01)	Beta	KEC 4
Prob. Recruitment age 7	P_7	(0.375)	(0.01)	Beta	KEC 4
Prob. Recruitment age 8	P_8	(0.5)	(0.01)	Beta	KEC 4
Prob. Recruitment age 9	P_9	(0.625)	(0.01)	Beta	KEC 4
Prob. Recruitment age 10	P_{10}	(0.75)	(0.01)	Beta	KEC 4
Prob. Recruitment age 11	P_{11}	(0.875)	(0.01)	Beta	KEC 4

3.6.3 Acceptable Levels of Impact

3.6.3.1 Summary of ALI methodology

The KEC 5 assessment used the revised 'Acceptable Level of Impact' (ALI) methodology described by Hin et al. (2024). The ALI methodology contains two threshold values, *X* and *Y,* of which the values were determined by policy-makers (LVVN) based on an expert review of the species-specific conservation status (Schekkerman 2024). The *X* threshold is a threshold value for the relative difference (in percentage) in final population abundance between an unimpacted and an impacted population projection. The *X*-threshold is violated if this relative difference is larger than *X*. The *Y*-threshold is a threshold for the probability (between zero and one) of an *X*-threshold violation. The ALI is violated if the probability of an *X*–threshold violation exceeds *Y.*

As described in section 3.6.1 – Population models, the population projections are done in such a way that each unimpacted, baseline projection has an impacted counterpart that uses the same variation in baseline demographic rates. For each of *n* replicate pairwise projections, the relative difference in final population abundance was calculated as:

$$\Delta = \frac{N_t^0 - N_t^I}{N_t^0}$$

With N_t^0 and N_t^I being, respectively, the baseline and impacted population abundance after *t* simulated years. This results in a distribution of Δ with an amount of variation that is mainly driven by the variation in the estimated mortality impact of OWFs. The probability that the *X*-threshold is violated is equals to $P_{VI} = P(\Delta > X_{40})$. The ALI is violated if this fraction exceeds the ALI *Y*-threshold, i.e. if $P_{VI} > Y$. For KEC 5, 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.

3.6.3.2 ALI threshold values

The X-threshold value denotes the maximum acceptable percentage reduction in population abundance between the unimpacted and impacted population trajectories (Soudijn et al. 2025). The value of *X* is species-specific and is based a species' conservation status (Schekkerman 2024). Species with an unfavourable status received an X value of 5%, while species with a favourable status got an X value of 15%. These X values were defined per reference time of the maximum of three generations or 10 years. The X values used in the KEC assessment were recalculated based on estimates of species-specific generation times (see Table 3-13). The Y value was set at the commonly used threshold for statistical significance of 5%.

Table 3-13 *Threshold values for the Acceptable Level of Impact methodology as supplied by LVVN. X-threshold for Atlantic puffin and northern fulmar were based on Schekkerman (2024). 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	4.7	0.05
Sandwich tern	5%	13.7	30	4.86	0.05
common guillemot	15%	16.9	44	12.03	0.05
razorbill	15%	12.8	40	15.57	0.05
Atlantic puffin	5%	14.2	31	4.70	0.05
northern fulmar	5%	19.8	60	3.40	0.05

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5 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

Bemmelen, R S A van, and Ruben C Fijn. 2024. Avoidance of Offshore Wind Farms and Movements by Sandwich Terns from Pre- to Post-Breeding. Nos. 24–454. Waardenburg Ecology.

Bemmelen, Rob S A van, Jacco J Leemans, Mark P Collier, et al. 2024. “Avoidance of Offshore Wind Farms by Sandwich Terns Increases with Turbine Density.” Ornithological Applications 126 (1): duad055. https://doi.org/10.1093/ornithapp/duad055.

Bemmelen, R.S.A. van, W Courtens, MP Collier, and R. C. Fijn. 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. Bureau Waardenburg, Culemborg.

Billerman, Shawn M., Brooke K. Keeney, Paul G. Rodewald, and Thomas S. Schulenberg, eds. 2022. Birds of the World. Cornell Laboratory of Ornithology. https://birdsoftheworld.org/bow/home.

Breton, André R., Antony W. Diamond, and Stephen W. Kress. 2006. “Encounter, Survival, and Movement Probabilities from an Atlantic Puffin (Fratercula Arctica) Metapopulation.” Ecological Monographs 76 (1): 1. https://doi.org/10.1890/05-0704.

Buckland, S. T. 1982. “A Mark-Recapture Survival Analysis.” The Journal of Animal Ecology 51 (3): 3. https://doi.org/10.2307/4008.

Caswell, H. 2001. Matrix Population Models: Construction Analysis and Interpretation. 2nd ed. Sinauer Associates.

Dunnet, G. M., Janet C. Ollason, and A. Anderson. 1979. “A 28-Year Study of Breeding Fulmars Fulmarus Glacialis in Orkney.” Ibis 121 (3): 3. https://doi.org/10.1111/j.1474-919X.1979.tb06846.x.

Fayet, Annette L, Tycho Anker-Nilssen, Arild Breistøl, et al. 2025. Key-Site Monitoring in Norway 2024, Including Svalbard and Jan Mayen. Short Report Nos. 1–2025.

Fijn, Ruben C., Pim Wolf, Wouter Courtens, et al. 2014. “Post-Breeding Prospecting Trips of Adult Sandwich TernsThalasseus Sandvicensis.” Bird Study 61 (4): 566–71. https://doi.org/10.1080/00063657.2014.950942.

Garthe, Stefan, Dr Verena Peschko, Henriette Schwemmer, and Dr Moritz Mercker. 2022. “Auswirkungen des Offshore-Windkraft-Ausbaus auf Seevögel in der Nordsee.” Meeresumwelt Symposium 2022. https://www.bsh.de/DE/PRESSE/Veranstaltungen/Termine/MUS/MUS_Nachklapp/_Anlagen/Downloads/MUS-2022/Vortraege/Vortrag_Garthe.pdf?__blob=publicationFile&v=2.

Grosbois, Vladimir, and Paul M. Thompson. 2005. “North Atlantic Climate Variation Influences Survival in Adult Fulmars.” Oikos 109 (2): 2. https://doi.org/10.1111/j.0030-1299.2005.13774.x.

Grundlehner, Anne, and Mardik Leopold. 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. https://doi.org/10.18174/643585.

Grundlehner, Anne, Mardik F. Leopold, and Anna Kersten. 2025. “This Is EPIC: Extensive Periphery for Impact and Control to Study Seabird Habitat Loss in and around Offshore Wind Farms Combining a Peripheral Control Area and Bayesian Statistics.” Ecological Informatics 85 (March): 102981. https://doi.org/10.1016/j.ecoinf.2024.102981.

Hansen, Erpur S. 2023. Stofnvöktun Lunda 2020-2022 - Lokaskýrsla Til Umhverfisstofnunar. Puffin Population Monitoring Report 2020-2022. Náttúrustofa Suðurlands. https://doi.org/10.13140/RG.2.2.22993.75360.

Harris, M P, T Anker-Nilssen, R H McCleery, K E Erikstad, D N Shaw, and V Grosbois. 2005. “Effect of Wintering Area and Climate on the Survival of Adult Atlantic Puffins Fratercula Arctica in the Eastern Atlantic.” Mar Ecol Prog Ser 297: 283–96.

Harris, Michael P., Francis Daunt, Maria I. Bogdanova, et al. 2013. “Inter-Year Differences in Survival of Atlantic Puffins Fratercula Arctica Are Not Associated with Winter Distribution.” Marine Biology 160 (11): 11. https://doi.org/10.1007/s00227-013-2278-5.

Hendry, Tom, Harriet Reid, and Chris Redfern. 2022. “Breeding Birds on the Farne Islands 2021.” Transactions of the Natural History Society of Northumbria, August, 30–43.

Heße, Eileen, and Eleni Melis. in prep. Using Sensitivity Scoring to Identify Marine Bird Species for KEC Assessments. Wageningen Marine Research.

Hin, V, G IJntema, T van Kooten, and A Potiek. 2024. A Revised Methodology for Quantifying 'Acceptable Level of Impact' from Offshore Wind Farms on Seabird Populations. C034/24. Wageningen Marine Research.

Hin, Vincent, and Janneke Ransijn. 2025. Evaluating the Sensitivity of the KEC 5 Habitat Loss Assessment to Different Displacement-Effect Curves for the Common Guillemot. C069/25. Wageningen Marine Research.

Hin, Vincent, and Floor Soudijn. 2021. Feasibility Study of the Development of Population Models for the Northern Fulmar (Fulmarus Glacialis) and the Atlantic Puffin (Fratercula Arctica). C046/21. Wageningen Marine Research. https://doi.org/10.18174/546670.

Horswill, Catharine, and Robert A Robinson. 2015. Review of Seabird Demographic Rates and Density Dependence. JNCC Report No. 552. no. JNCC Report No. 552. Peterborough.

IJntema, G J, N Heida, J J Leemans, A Gyimesi, and A Potiek. 2025. Collision Effects of North Sea Wind Turbines on Bird Species within the "Kader Ecologie & Cumulatie (KEC) 5.0." Report 24-163. Waardenburg Ecology.

IJntema, Gerben, Astrid Potiek, Vincent Hin, Eleni Melis, and Martin Poot. 2025. Memo Study Area and Discounting in "Kader Ecologie & Cumulatie (KEC) 5.0." Nos. 23-0667/AstPo/KEC_tussenjaar_studiegebied. Waardenburg Ecology, Culemborg & Wageningen Marine Research, IJmuiden.

JNCC. 2015. Seabird Displacement Impacts from Offshore Wind Farms: Report of the MROG Workshop, 6-7th May 2015. JNCC Report No 568. JNCC.

Johnson, Fred A., Carl J. Walters, and Erpur Snær Hansen. 2024. Icelandic Puffin Stock Assessments 2024. Technical report. Version 1. Unpublished. https://rgdoi.net/10.13140/RG.2.2.12927.42405/1.

Landsem, Terje L., Nigel G. Yoccoz, Kate Layton-Matthews, et al. 2023. "Raising Offspring Increases Ageing: Differences in Senescence among Three Populations of a Long-Lived Seabird, the Atlantic Puffin." Journal of Animal Ecology 92 (3): 774–85. https://doi.org/10.1111/1365-2656.13884.

Lane, Jude V., Jana W.E. Jeglinski, Stephanie Avery-Gomm, et al. 2024. "High Pathogenicity Avian Influenza (H5N1) in Northern Gannets (Morus Bassanus): Global Spread, Clinical Signs and Demographic Consequences." Ibis 166 (2): 633–50. https://doi.org/10.1111/ibi.13275.

Laurenson, Kirsty, Matt J. Wood, Tim R. Birkhead, et al. 2025. "Long-term Multi-species Demographic Studies Reveal Divergent Negative Impacts of Winter Storms on Seabird Survival." Journal of Animal Ecology 94 (1): 139–53. https://doi.org/10.1111/1365-2656.14227.

Layton-Matthews, Kate, Tone K. Reiertsen, Kjell-Einar Erikstad, et al. 2023. "Consequences of Cross-Season Demographic Correlations for Population Viability." Ecology and Evolution 13 (7): e10312. https://doi.org/10.1002/ece3.10312.

Leopold, Mardik F, Martijn Boonman, Mark P. Collier, et al. 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.

Lewis, Sue, David A. Elston, Francis Daunt, Barbara Cheney, and Paul M. Thompson. 2009. "Effects of Extrinsic and Intrinsic Factors on Breeding Success in a Long Lived Seabird." Oikos 118 (4): 4. https://doi.org/10.1111/j.1600-0706.2008.17308.x.

Lilipaly, S J, W Janse, M S J Hoekstein, M M Pattikawa, M Sluijter, and P A Wolf. 2024. Broedsucces van Kustbroedvogels in Het Deltagebied in 2024. Nos. 2025–03. Deltamilieu Projecten. https://deltamilieuprojecten.nl/wp-content/uploads/2025/05/Broedsucces-kustbroedvogels-Deltagebied-2024-3.pdf.

Lilipaly, S. J., W. Janse, M.S.J. Hoekstein, M. Pattikawa, and P.A. Wolf. 2024. Broedsucces van Kustbroedvogels in Het Deltagebied in 2024. Deltamilieu Projecten Rapportnr. 2025-03. DMP.

Lilipaly, S. J., W. M. Janse, M. Pattikawa, D. Van Straalen, and M. Sluijter. 2025. Kustbroedvogels in Het Deltagebied in 2024. Rijkswaterstaat, Centrale informatievoorziening Rapport BM 25.06 Deltamilieu Projecten Rapportnr. 2025-03. Vlissingen.

NatureScot. 2025. "Isle of May National Nature Reserve - Annual Report 2024 | NatureScot." May 30. https://www.nature.scot/doc/isle-may-national-nature-reserve-annual-report-2024.

Orzack, Steven Hecht, Ulrich K. Steiner, Shripad Tuljapurkar, and Paul Thompson. 2011. "Static and Dynamic Expression of Life History Traits in the Northern Fulmar Fulmarus Glacialis." Oikos 120 (3): 3. https://doi.org/10.1111/j.1600-0706.2010.17996.x.

Peschko, Verena, Henriette Schwemmer, Moritz Mercker, Nele Markones, Kai Borkenhagen, and Stefan Garthe. 2024. "Cumulative Effects of Offshore Wind Farms on Common Guillemots (Uria Aalge) in the Southern North Sea - Climate versus Biodiversity?" Biodiversity and Conservation 33 (3): 949–70. https://doi.org/10.1007/s10531-023-02759-9.

Rijkswaterstaat. 2025. Framework for Assessing Ecological and Cumulative Effects (KEC) 5.0 (Roadmap 21GW). Report A: Scope. Rijkswaterstaat.

Schekkerman, Hans. 2024. "Invulling van Parameterwaarden Voor Herziene Acceptable Levels of Impact van Sterfte van Vogels Door Windparken Op Zee." Sovon Vogelonderzoek Nederland, August 31.

Searle, K, D Mobbs, A Butler, et al. 2014. Population Consequences of Displacement from Proposed Offshore Wind Energy Developments for Seabirds Breeding at Scottish SPAs (CR/2012/03). CR/2012/03. Centre for Ecology and Hydrology.

Searle, K R, D C Mobbs, A Butler, R W Furness, M N Trinder, and F Daunt. 2018. "Finding out the Fate of Displaced Birds (FCR/2015/19): Scottish Marine and Freshwater Science Vol 9 No 8." Scottish Marine and Freshwater Science 9 (08). https://doi.org/10.7489/12118-1.

Searle, Kate, Adam Butler, Deena Mobbs, et al. 2022. "Study to Examine How Seabird Collision Risk, Displacement and Barrier Effects Could Be Integrated for Assessment of Offshore Wind Developments." Marine Scotland Science.

Searle, Kate, Francis Daunt, Christopher John Pollock, and Adam Butler. 2025. Quantification of Mortality Rates Associated with Displacement (QuMR). Final Report. UK Centre for Ecology and Hydrology & BioSS.

Semple, Mairi, and Jaz Harker. 2024. Salamander Offshore Wind Farm Offshore EIA Report. ER.A.4.12.4 Volume ER.A.4, Annex 12.4: Population Viability Analysis (PVA).

SMP. 2025. "Seabird Monitoring Programme." Https://app.bto.org/seabirds/public/index.jsp.

SNCB. 2022. Joint SNCB Interim Displacement Advice Note. SNCB.

Soudijn, F, S. van Donk, M F Leopold, J.T. van der Wal, and V Hin. 2022. Cumulative Population-Level Effects of Habitat Loss on Seabirds 'Kader Ecologie En Cumulatie 4.0.' C007/22. Wageningen Marine Research. https://doi.org/10.18174/553775.

Soudijn, F.H., V. Hin, E. Melis, et al. 2025. Population Level Effects of Displacement of Marine Birds Due to Offshore Wind Energy Developments, KEC 5. C094/24 A. Wageningen Marine Research. https://doi.org/10.18174/683553.

Szostek, Lesley, Raul Vilela, Christina Bauch, et al. 2024. Auks in the German North Sea: Effects of Offshore Wind Farms. IBL Umweltplanung & BioConsult SH.

Tree, A. J. 2011. "Origins, Occurrence and Movements of Sandwich Tern Thalasseus Sandvicensis in Southern Africa." Marine Ornithology 39 (2). https://doi.org/10.5038/2074-1235.39.2.933.

Trinder, Mark, Susan O'Brien, and Joseph Deimel. 2024. "A New Method for Quantifying Redistribution of Seabirds within Operational Offshore Wind Farms Finds No Evidence of Within-Wind Farm Displacement." Frontiers in Marine Science 11 (April): 1235061. https://doi.org/10.3389/fmars.2024.1235061.

Van Donk, Susanne. 2024. Knowledge-Update KEC5 Density Maps Seabirds. C044/24. Wageningen Marine Research. https://doi.org/10.18174/661977.

Van Donk, Susanne, Rob Van Bemmelen, Chun Chen, Ingrid Tulp, and Ellen Melis. 2024. Seabird Maps of the North Sea : A Short Description of Methodology. Wageningen Marine Research. https://doi.org/10.18174/657263.

Vanermen, Nicolas, Thierry Onkelinx, Wouter Courtens, Marc Van de walle, Hilbran Verstraete, and Eric W. M. Stienen. 2015. "Seabird Avoidance and Attraction at an Offshore Wind Farm in the Belgian Part of the North Sea." Hydrobiologia 756 (1): 51–61. https://doi.org/10.1007/s10750-014-2088-x.

Waggitt, James J., Peter G. H. Evans, Joana Andrade, et al. 2020. "Distribution Maps of Cetacean and Seabird Populations in the North-East Atlantic." Journal of Applied Ecology 57 (2): 2. https://doi.org/10.1111/1365-2664.13525.

Wanless, Sarah, Steve D. Albon, Francis Daunt, et al. 2023. "Increased Parental Effort Fails to Buffer the Cascading Effects of Warmer Seas on Common Guillemot Demographic Rates." Journal of Animal Ecology 92 (8): 1622–38. https://doi.org/10.1111/1365-2656.13944.

Justification

Report: C094/25
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:	Dr. F.H. Soudijn Researcher
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Annex 1 Supplemental Tables

Table A1 *Displacement effect distance and displacement probability values are presented per species. For each parameter, the species-specific values as used in KEC 5 are listed. The values found in the knowledge update are presented by season as referred to in the respective publications (winter, spring, summer, autumn) where possible. Where available, the months included in each season are indicated in brackets after the values. If no months were specified in the publications, this is indicated with an asterisk instead. Mean values, value ranges, standard deviation (SD), 95% confidence intervals (CI) or 99% credibility intervals (CrI) are provided where available. Values without an indication are general values without further specification provided in the respective sources. Sources for each value are indicated by the letters a–n behind each value.*

<table>
<tr><th colspan="2">Parameter</th><th colspan="2">Northern gannet</th></tr>
<tr><td rowspan="6">Effect distance around OWFs</td><td></td><td>Value(s) used in KEC 5</td><td>1500 (n)*</td></tr>
<tr><td rowspan="5">New value(s)</td><td>Season not specified</td><td>range = 500–3000 (n)*;
mean = 1560 (g)*,
range 0–2000 (g)*</td></tr>
<tr><td>Winter</td><td>range = 0–3000 (a)*</td></tr>
<tr><td>Spring</td><td>-</td></tr>
<tr><td>Summer</td><td>range = 0–3000 (a)*</td></tr>
<tr><td>Autumn</td><td>-</td></tr>
<tr><td rowspan="6">Displacement probability</td><td></td><td>Value(s) used in KEC 5</td><td>mean = 0.85 (n)*,
SD = 0.69 (from Sandwich tern (k))</td></tr>
<tr><td rowspan="5">New value(s)</td><td>Season not specified</td><td>0.70 (g)*</td></tr>
<tr><td>Winter</td><td>-</td></tr>
<tr><td>Spring</td><td>-</td></tr>
<tr><td>Summer</td><td>May–Aug: mean = 0.67 (i);
0.75 (a)* (at 1km from OWF);
0.32 (a)* (at 5km from OWF)</td></tr>
<tr><td>Autumn</td><td>-</td></tr>
<tr><td colspan="2">Sources</td><td colspan="2">(a) Garthe et al. (2022), (g) Searle et al. (2025), (i) Trinder et al. (2024),
(k) van Bemmelen et al. (2024), (n) Vanermen et al. (2015)</td></tr>
</table>

Parameter		Sandwich tern	
Effect distance around OWFs		Values used in KEC 5	breeding season: 1500 (k)*
	New value(s)	Winter	-
		Spring	-
		Summer	-
		Autumn	-
Displacement probability		Values used in KEC 5	breeding season: mean = 0.54 (k), CI = 0.36–0.70 (k), SD = 0.69 (k)
	New value(s)	Season not specified	0.70 (g)*
		Winter	-
		Spring	-
		Summer	Apr–Oct: mean = 0.53 (m), CI = 0.40–0.62 (m); Apr–May: mean = 0.67 (m), CI = 0.51–0.78 (m); Jun–Jul: mean = 0.52 (m), CI = 0.41–0.62 (m); Aug–Oct: mean = 0.49 (m), CI = 0.00–0.74 (m)
		Autumn	-
Sources		(k) van Bemmelen et al. (2024), (m) van Bemmelen & Fijn (2024)	

Parameter		Common guillemot	
Effect distance around OWFs		Value(s) used in KEC 5	Oct–Feb: mean = 19500 (f)
	New value(s)	Season not specified	range = 1000–2000 (g)*
		Winter	Oct–Feb: mean = 16500 (f), range = 15000–18000 (f); range = 6000–12000 (h); Oct–Mar: >10000 (d)
		Spring	-
		Summer	-
		Autumn	Mid-Jul–Sep: range = 18000–21000 (f); Jul–Sep: range = 400–2000 (h)
Displacement probability		Value(s) used in KEC 5	Mid-Jul–Sep: mean = 0.79 (f)
	New value(s)	Season not specified	0.30 (g)*
		Winter	Oct–Feb: mean = 0.51 (f), CI = 0.42–0.58 (f)
		Spring	-
		Summer	-
		Autumn	Mid-Jul–Sep: CI = 0.74–0.83 (f)
Sources		(d) Grundlehner et al. (2025), (f) Peschko et al. (2024), (g) Searle et al. (2025), (h) Szostek et al. (2024)	

Parameter		Razorbill	
Effect distance around OWFs		Value(s) used in KEC 5	Oct–Mar: 2000 (c)
	New value(s)	Season not specified	1000 (g)*
		Winter	Oct–Mar: 2000 (d); Oct–Feb: no significant effect (h)
		Spring	-
		Summer	-
		Autumn	Jul–Sep: range = 6000–11000 (h)
Displacement probability		Value(s) used in KEC 5	Oct–Mar: mean = 0.43 (c), CrI = 0.40–0.45 (c), SD = 0.021 (c)
	New value(s)	Season not specified	0.40 (g)*
		Winter	0.55 (a)* (at 1km from OWF), 0.47 (a)* (at 5km from OWF); Oct–Mar: 0.402 (d) (inside OWFs);
		Spring	-
		Summer	-
		Autumn	-
Sources		(a) Garthe et al. (2022), (c) Grundlehner & Leopold (2024), (d) Grundlehner et al. (2025), (g) Searle et al. (2025), (h) Szostek et al. (2024)	

Parameter		Atlantic puffin	
Effect distance around OWFs		Value(s) used in KEC 5	-
	New value(s)	Season not specified	mean = 1500 (g)*, range = 1000–2000 (g)*
		Winter	-
		Spring	-
		Summer	-
		Autumn	-
Displacement probability		Value(s) used in KEC 5	-
	New value(s)	Season not specified	0.60 (e)*; 0.40 (g)*
		Winter	-
		Spring	-
		Summer	-
		Autumn	-
Sources		(e) NatureScot (2023), (g) Searle et al. (2025)	

Parameter			Northern fulmar
Effect distance around OWFs		Value(s) used in KEC 5	-
	New value(s)	Season not specified	mean = 1333 (g)*, range = 0–2000 (g)*
		Winter	range = 0–3000 (a)*
		Spring	range = 3000–6000 (a)*
		Summer	range = 0–3000 (a)*
		Autumn	-
Displacement probability		Value(s) used in KEC 5	-
	New value(s)	Season not specified	0.30 (g)*; 0.01–0.1 (g)*
		Winter	-
		Spring	0.91 (a)* (at 1km from OWF), 0.84 (a)* (at 5km from OWF)
		Summer	0.64 (a)* (at 1km from OWF), 0.43 (a)* (at 5km from OWF)
		Autumn	-
Sources			(a) Garthe et al. (2022), (g) Searle et al. (2025)

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With knowledge, independent scientific research and advice, **Wageningen Marine Research** substantially contributes to more sustainable and more careful management, use and protection of natural riches in marine, coastal and freshwater areas.

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