Effects of turbine collision mortality on population dynamics of 13 bird species

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Sovon

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Preface

In Environmental Impact Assessments and Appropriate Assessments for future offshore wind farms the effects of these initiatives on birds are assessed with relatively simple criteria and thresholds like the ORNIS 1% criterion and the Potential Biological Removal (PBR). These methods are easy to apply and understand but come with limitations. Population models can predict the population dynamics of species based on the most recent scientific species-specific knowledge and are therefore expected to be better predictors of the effects of additional mortality due to wind farms. Therefore, Rijkswaterstaat asked Bureau Waardenburg together with SOVON to construct population models for 13 potentially critical species to create a tool to better assess the impacts of future offshore wind farms on bird populations.

Many different people from a large variety of universities, institutes and consultancies contributed to this project with data, technical advice, and species-specific knowledge. All are thanked tremendously for their effort to do so.

This project was executed parallel to a similar project done by Wageningen Marine Research and Floor Soudijn and Tobias van Kooten are thanked for their input and help in this project.

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Summary

The planned large-scale development of offshore wind farms in the North Sea has potential consequences for many marine organisms, including seabirds. Seabirds may suffer from collisions with offshore wind farms during migratory or foraging flights at sea and this additional mortality may in turn negatively affect the populations of seabirds using the Dutch continental shelf.

In Environmental Impact Assessments and Appropriate Assessments for future offshore wind farms the effects of these initiatives on birds are assessed with relatively simple criteria and thresholds like the ORNIS 1% criterion and the Potential Biological Removal (PBR). These methods are easy to apply and understand but come with many limitations. On the other hand, population models can predict population dynamics of species based on the latest, species-specific scientific knowledge, and will probably give better estimates of bird mortality associated with wind farms.

In this study species-specific (Leslie-Matrix) population models were developed and applied to create a tool to better assess the impacts of future offshore wind farms on bird populations. These population models take parameter uncertainty into account, resulting in a different outcome for each simulation. The impact assessment is based on a comparison of the distribution of outcomes for different scenarios.

This study looked at collision victims from all offshore wind farms planned pre-2030 in the North Sea. Species of interest were selected based on the Framework for the Assessment of Ecological and Cumulative Effects (KEC 1.0 - KEC 3.0; Rijkswaterstaat 2015; 2019), and include five gulls, two skuas, two terns, three species of wildfowl and one wader. Based on best available data, population models were constructed for the current situation (without additional mortality).

Sufficient data were available to establish population models for all species except little gull. These population models give a projected population trend based on the current demographic rates. For the scenario with additional mortality, the population models assume the number of victims as estimated in the Framework for the Assessment of Ecological and Cumulative Effects 3.0 (KEC 3.0; Rijkswaterstaat 2019).

For each species, the outcome of this impacted scenario is compared to the outcome of the scenario without additional mortality. This comparison is made based on the probability of a 10% decline, the change in median projected growth rate, and the overlap between the scenarios.

The largest estimated impacts were found for the lesser black-backed gull, black tern, common shelduck and Eurasian curlew. However, the overall impacts associated with the KEC scenario seem to be relatively minor for most species (Table 1). In seven species the inclusion of additional cumulative wind farm mortality results in a final

population size within 5% of the final population size projected without additional mortality. The assessed impact on the other five species is stronger. The difference in median outcome of the population size between impacted and unimpacted scenarios is between 5 and 10% for great black-backed gull, between 15 and 20% for lesser black-backed gull and common shelduck, and over 20% for black tern and curlew.

Table 1   *Summary of results of the population models. Impacted scenario refers to the KEC-scenario.*

	Median relative population size after 30 years ($N_{30\_impacted} / N_{30\_null}$)	Probability of 10% decline within 30 years	
		null scenario	impacted scenario
Lesser black-backed gull	0.83	42	49
Great black-backed gull	0.93	57	58
Herring gull	0.95	62	63
Kittiwake	0.98	56	57
Great skua	1	36	36
Arctic skua	0.99	64	65
Common tern	0.97	65	65
Black tern	0.79	39	47
Bewick's swan	0.98	96	96
Brent goose	0.98	43	44
Common shelduck	0.82	26	30
Eurasian curlew	0.76	97	98

In order to assess the method of impact assessment using the Potential Biological Removal, an additional scenario was run with the full annual Potential Biological Removal as a measure of additional mortality. This scenario shows how the currently used threshold of additional mortality would impact the population of the selected species. This scenario shows for most species a relatively strong impact, which suggests that this level of additional mortality is too high for maintaining a stable population.

This study only calculated effect sizes of additional mortality directly due to collisions with turbines of offshore wind farms. There are many more causes of direct and indirect anthropogenic mortality for the species at hand, some of which may even turn out to be indirect effects of offshore wind farms (e.g. knock-on effects caused by ecosystem effects of shifts in hydromorphology and/or foodweb relationships). In this report we attempted to quantify this additional mortality but available data were not of such quality that they were considered valid enough to be included in this study.

This report shows how population models can be used for a population-level impact assessment of mortality due to collisions of birds with turbines. Population models provide a better picture than other methods of the possible effects of offshore wind farms on these species. However, before the models can be used to inform the permit

process, thresholds are needed for the metrics they can produce. This is a policy decision rather than a scientific one.

1 Introduction

Over the past decade, the extent of offshore wind farms in the North Sea has increased and more capacity is planned for the near as well as for the more distant future. Wind farms, including those offshore, may have various effects on birds, including displacement, barrier effects and direct mortality due to collisions with turbines. In the North Sea, this will include species that are present at some point during the year, such as during foraging trips, as well as during migration.

Under the National Energy Agreement, concluded in 2013, new offshore wind farms with a total capacity of 3.5 GW will be built in the Dutch sector of the North Sea until 2023, in addition to the five existing wind farms, which have a total capacity of 1 GW. Recently, the Dutch Government has unveiled an Offshore Wind Energy Roadmap setting out plans for the further development of offshore wind energy, including an additional 7 GW between 2023 and 2030.

In general, the effects of individual wind farms are studied separately in impact assessments and during post-construction monitoring. The rapid growth in the number of offshore wind farms increasingly calls for studies into the cumulative impacts of all these developments together. The impact of an individual wind farm on the population level of a certain species may be limited, but in accumulation with other wind farms the impact may be more profound. Regarding both the impact of habitat loss and the impact of turbine collision, research into cumulative effects is warranted (Welcker & Nehls 2016; Furness *et al.* 2013) and a few attempts were undertaken in the past (e.g. Poot *et al.* 2011).

Relatively recently the Dutch Government commissioned and published a large-scale study into the cumulative effects of 106 operational, planned and proposed (i.e. considered operational in 2023) wind farms across the southern North Sea under the research framework '*Kader Ecologie en Cumulatie*' (KEC) (Rijkswaterstaat 2019). In this study displacement and collisions were modelled based on assumptions of Bradbury *et al.* (2014) and the number of casualties due to collisions was also calculated using the extended Band model (Band 2012). The calculations were carried out for seabirds as well as for terrestrial- and waterbirds. The predicted number of casualties was compared with a threshold calculated by a Potential Biological Removal (PBR) model (cf. Wade 1998; Dillingham & Fletcher 2008). This methodology was developed earlier by Rijkswaterstaat (2015) and was updated based on new data on turbine types, development plans, bird behaviour and bird densities (Rijkswaterstaat 2019).

The previous analysis by Leopold *et al.* (2014) found that for some species the predicted mortality exceeded the PBR threshold or reached a relatively high fraction of it. The updated results of Rijkswaterstaat (2019) show lower levels of predicted mortality, resulting in most species remaining well below the species-specific PBR threshold.

Based on the results from Rijkswaterstaat (2019), including wind farm Prinses Amalia and OWEZ, the seabirds for which the predicted mortality exceeded 10% of the PBR for the southern North Sea were herring gull (35%), lesser black-backed gull (22%) and black-legged kittiwake (15%). For black tern, Eurasian curlew and common shelduck, the predicted mortality was 98%, 64% and 10% respectively, of the species-specific PBR (Rijkswaterstaat 2019).

Because of this, significant impacts of offshore wind farms at the population level could not be ruled out for all species (Rijkswaterstaat 2019). Based on the results of Rijkswaterstaat (2015), a list of bird species was identified, that either spend a substantial part of their lifecycle in the Dutch North Sea or that migrate over Dutch waters, and for which the predicted additional mortality due to wind turbines was close to the PBR threshold. Note that the updated calculations in Rijkswaterstaat (2019) show clearly lower estimated mortality for some species. Some of these species are mainly affected by habitat loss, whereas others are vulnerable to collisions with turbine rotor blades. Within this report, the impact of collisions with turbines is assessed. The species selected by Rijkswaterstaat (2015) as vulnerable are listed in Table 1.1. Three additional species have been selected by Rijkswaterstaat as potentially vulnerable (pers. comm Rijkswaterstaat, Table 1.2). The impact assessment of collision mortality on these three additional species (arctic skua, common tern and little gull) will be reported in a separate report. The impact of habitat loss on species classified as vulnerable by Rijkswaterstaat (2015) is assessed by van Kooten *et al.* (2018).

Table 1.1 *Species selected within the KEC framework (Rijkswaterstaat 2015) based on the predicted additional mortality in relation to the determined Potential Biological Removal (PBR).*

English name	Scientific name
Lesser black-backed gull	*Larus fuscus*
Great black-backed gull	*Larus marinus*
Herring gull	*Larus argentatus*
Black-legged kittiwake	*Rissa tridactyla*
Great skua	*Stercorarius skua*
Black tern	*Chlidonias niger*
Bewick's swan	*Cygnus bewickii*
Brent goose	*Branta bernicla*
Common shelduck	*Tadorna tadorna*
Eurasian curlew	*Numenius arquata*

Table 1.2 *Species additionally selected within the KEC framework as potentially vulnerable to turbine collisions.*

English name	Scientific name
Little gull	*Larus minutus*
Arctic skua	*Stercorarius parasiticus*
Common tern	*Sterna hirundo*

The model predictions of the KEC studies were based on the comparison of modelled mortality estimates with estimates of population sizes by the PBR approach. For seabirds, both mortality estimates and population size estimates in their turn are based on offshore monitoring data (aerial- & ship-based surveys, radar- and observational studies). The PBR approach has certain clear drawbacks, such as e.g. that it provides a fixed and very static figure that does not take any environmental variability into account. Moreover, it implicitly assumes a fixed level of undemonstrated density dependence in population development (O'Brien *et al.* 2017). Therefore, for at least the species listed in Table 1.1 more detailed population modelling is needed to obtain a better and more profound understanding of the (cumulative) effects of increasing numbers of offshore wind farms in the southern North Sea.

Since 2016, Rijkswaterstaat initiated a large-scale research programme called Wozep ('Wind Op Zee Ecologisch Programma', translated as Offshore Wind Ecological Programme), aimed at obtaining more knowledge about the ecological impacts of offshore wind developments in the North Sea. One of the projects within Wozep involves studying the cumulative effects of collisions with turbines on the long-term population dynamics of bird species identified in the KEC framework (Table 1.1 and Table 1.2), using population modelling. The research question of this study is the following:

Research question:
What are the cumulative long-term effects of mortality due to collisions between birds and turbines in the southern North Sea on the population level of seabirds and migratory birds?

The focus of this study is about the effects of collisions between birds and turbines at the population level. A parallel project within Wozep, carried out by Wageningen Marine Research (WMR), studies the effects of habitat loss at the population level. Within these two projects, the effects of additional mortality on population level are assessed using similar methodology. Collaboration between Bureau Waardenburg and WMR resulted in the development of a basic population model, which is used in both projects. Based on this basic population model, species-specific population models are created.

In order to be able to look at population level effects, the 'population' concerned has to be defined. This is described in Chapter 2. Note that additional scenarios can be run for other 'population' definitions. After defining the population, the population model can be filled with input parameters. For each species, we assessed the data availability, which is documented in Chapter 3. Moreover, we assessed data quality and representativeness of each data source (for description of methods, see Paragraph 2.1).

2 Methods

2.1 Population models

2.1.1 Why use population models?

The number of collision victims from all Dutch offshore wind farms in the North Sea, as estimated in the KEC calculations, provides an indication of the effect size, measured as the number of individual birds. However, these numbers of victims cannot be compared among species, because the impact of a given number of collision victims strongly depends on the species (Dierschke & Bernotat 2012).

Species differ in life history strategy on a range that can be called 'fast' to 'slow' strategies (Sæther *et al.* 1996). *Fast* species, such as most passerines, are relatively short-lived and start reproduction at young age. On the other hand, *slow* species are long-lived and only start to reproduce when older. Due to these characteristics, the effect of a 1% change in survival or reproduction on for example the population growth rate varies among species of different life history strategies.

As a consequence, the impact of a certain number of collision victims on a population may vary among species even in populations of the same size. Because individuals of *slow* species have to survive for a number of years before starting to reproduce, they only contribute to the population after some time. If mortality increases in those species, for example due to turbine collisions, this strongly affects the number of individuals eventually contributing to the population (Sæther & Bakke 2000). For this reason, the effect of additional mortality is particularly strong in species with a long lifespan and high maturation age (*slow* species). Moreover, after a period of higher mortality, species with a *slow* life history strategy generally need a longer time to recover (Koons *et al.* 2005). In order to understand the impact of the estimated additional mortality, it is vital to look at effects of this mortality at the population level.

Until now, predictions from Collision Rate Models were compared to a static PBR threshold, which is thought to be a rather crude method to assess impact on population level since not only the population size, but also the population structure and life history strategy of the species determine the impact of additional mortality. O'Brien *et al.* (2017) demonstrated that the assessment of the potential impact of additional mortality using the PBR method may result in false conclusions. The main criticism is the reliance on implicit assumptions, in particular regarding density dependence and the population trajectory. O'Brien *et al.* (2017) recommend the use of matrix population models, in which assumptions are explicitly defined.

Using population models, the future population trend can be projected. As a first step, population models will be constructed assuming no effect of wind farms (null models). In a later stage, the expected additional mortality due to turbine collisions (including a

range of possible values) will be quantified, and these matrix models will be used to assess the expected effect of this decreased survival.

Various scenarios, such as incorporating additional mortality, for example due to turbine collisions, can then be modelled to assess the impact on the population. Different types of population models exist. The most simplistic model types only calculate the population growth rate from known population sizes during subsequent years (equation 1).

$$N(t + 1) = \lambda * N(t) \qquad \text{equation 1}$$

However, more recent studies have shown that only knowing the changes in population size over the years is often not sufficient to reliably forecast the future population trend (Caswell 2001). This is mainly because variation in demographic rates is not taken into account. Demographic rates, however, commonly vary within a population; for example, in many species, survival in early life is lower than in older individuals.

'Structured' demographic models therefore give better results. These models can be structured by age, age class or stage in the lifecycle (Caswell 2001). Such structured demographic models not only provide more reliable projections, but also show how the individual life stages influence the population dynamics (Caswell & Fujiwara 2004). The effect of a change in a demographic rate varies among the stages, especially in long-lived, *slow*, species. In those species, additional mortality during the adult stage has a greater impact on the population growth rate than additional mortality early in life. Moreover, the impact, here collision-related mortality, may vary between the life stages, resulting in stage-specific effects. In structured demographic models, this can be taken into account by varying the additional mortality between age classes.

2.1.2 Leslie matrix models

We used population models based on stage-structured Leslie matrices (Leslie 1945; Caswell 2001). The model structure is defined depending on the number of sub-adult age classes. Additionally, in many species adults may skip breeding in a certain year, for example due to low food availability or strong competition with other adults (Gyimesi & Lensink 2012). These individuals are called 'non-breeding adults' or 'floaters'. If a population contains floaters, this forms a separate stage in the model. An example of a stage-based model structure is illustrated in Figure 2.1.

Within this project all population models have a time step of one year. This means that the numbers of individuals in each stage are counted once per year. A population can be modelled using pre-breeding census data or post-breeding census data. In a pre-breeding census, the numbers of individuals per stage are counted before the start of a breeding season. Hence, adult breeders are counted just before breeding, and their offspring are (almost) 1 year old when counted for the first time. In contrast, in a model using post-breeding data, the offspring are counted as chicks. We use pre-breeding population models for all species.

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In the following example we illustrate how a population is modelled using pre-breeding census data. Within this example, there are three immature stages and an adult stage. These stages are illustrated in Figure 2.1 with closed circles. Arrows indicate transitions between stages.
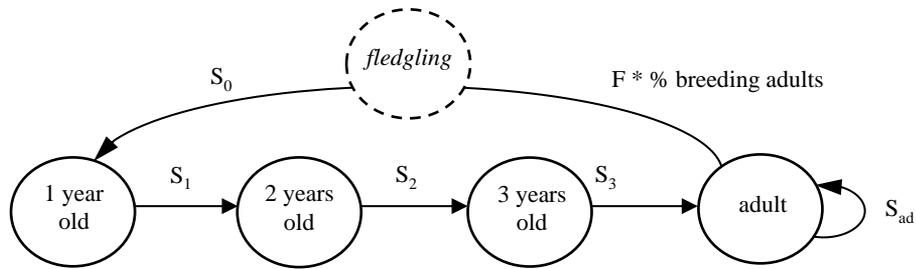
Figure 2.1   Example of a stage-based model structure with pre-breeding census data. In this case, individuals reach reproduction age at the age of 4. Adults may breed, or skip breeding (floater). S_i represents the survival of individuals in stage i; F represents the number of fledglings produced by one breeding pair. The time step in the model is 1 year.

For example, the transition from the 1-year old stage to the 2-year old stage in the next year is given by the survival of a 1-year old individual. In Figure 2.1, the average survival of a 1-year old individual is denoted as S_1. Note that this is an average survival and survival may vary between years, for example due to weather or food availability. This will be discussed in the following section about *stochasticity*. Similarly, the average survival of 2-year old individuals is given by S_2 and the average survival probability of 3-year old individuals is given by S_3. Average adult survival is S_{ad}.

An adult can undertake a breeding attempt or not, in which case it is classified as a 'floater'. This can vary between individuals and between years. A bird may breed one year and then skip an attempt the following year. Conversely, a bird may remain as a breeding adult, whereas another individual may never breed. The survival of an adult floater may differ from the survival of a breeding adult. On one hand, a floater may be in poor condition, therefore not breeding (either because of its own 'decision' or because of not obtaining a partner). Alternatively, a floater may have a lower mortality than a breeding adult, because of energetic costs of breeding and/or being less exposed to breeding related pressure factors such as predation or suboptimal feeding range, limited by reproductive duties (Gyimesi & Lensink 2012; Coulson 2002). However, this information is often not available and we have no indication of differential adult survival between breeding and non-breeding adults for our species of interest.

Breeding adults can produce offspring, which are typically measured as the number of fledglings per breeding pair (fecundity, F). Some of these fledglings survive until the next breeding season (S_0), when the numbers of individuals per stage are counted again. The contribution from breeding adults to the 1 year old stage is therefore the number of fledglings surviving until the following breeding season. This is calculated as F * S_0. Logically, the non-breeding adults do not contribute to the 1-year old stage in

the following year. For that reason, the percentage of breeding adults is taken into account as well.

The accompanying matrix structure is shown in Figure 2.2. Each 'cell' shows the contribution from one stage to another in the next year. The cell in column i and row j shows the contribution from individuals in stage i to stage j. In other words, the cell in the first column, second row, shows the contribution from stage 1 to stage 2. In this case, this is from fledgling to 1 year old individual, and is given by the survival during the first year. Similarly, the cell in column 4 and row 1 shows the contribution of the adult stage to the 1 year old stage, which is the probability of breeding (1-P_{Fl}) multiplied by the fecundity (F) multiplied by the survival during the first year (S_0).

$$A = \begin{bmatrix} 0 & 0 & 0 & (1-P_{Fl})*F*S_0 \\ S_1 & 0 & 0 & 0 \\ 0 & S_2 & 0 & 0 \\ 0 & 0 & S_3 & S_{ad} \end{bmatrix}$$

Figure 2.2   Leslie matrix with 3 immature age classes and an adult stage. $S_{i,t}$ denotes the survival in stage i in year t; F denotes the fecundity; P_{Fl} denotes the probability of non-breeding; the probability of breeding follows as 1-P_{Fl}.

Using this matrix, the population size and structure in the following year can be projected:

$$n_{t+1} = A_t * n_t \qquad \text{equation 2}$$

where:

n_t is a vector of population sizes per age class at the start of year t;

A_t is the Leslie matrix for year t;

2.1.3   Underlying assumptions

In general, it is assumed when using matrix models that populations are closed, in other words that no immigration or emigration occurs, or that these are effectively equal. Demographic data are collected from a 'study-population', which is often not a closed 'biological' population. The reasoning behind the (arbitrary) choice of this study population is given in Paragraph 2.3. In Appendix I we give more background on the assumption of balanced immigration/emigration, and implications for the population model.

Another main assumption is that demographic rates are density-independent. In other words, we assume that survival and fecundity do not vary with changing population size, unless sufficient data are available which indicate a significant effect of density dependence. For most species, data on the effect of density dependence on these parameters are, at best, limited. Incorporating density dependence would require making a number of assumptions that cannot be supported by data and bring unknown

variability into the models. See Appendix II for more information and possible consequences for the model outcome.

Input parameters of the population model, variation in parameters, and quality and representativeness
In order to assess the impact of, in this case collision-related, mortality on a population, we first construct a population model. After the model structure has been defined, data on current (stage-specific) survival and fecundity are needed.

Population structure
In order to define the stages that form the population structure, information on the number of sub-adult age-classes (in the example in Figure 2.1, there are three sub-adult age-classes), and whether non-breeding plays a role (in the example in Figure 2.1, the last stage represents the non-breeding adults), is needed. Moreover, in some cases, the use of a two-sex matrix model may be necessary. This is the case if survival or collision risk differ greatly between the sexes.

Parameters null model
As input for the population model, data on stage-specific survival and fecundity are necessary. In this report, we summarise the demographic data available for each species of interest (Tables 3.1 to 3.13). Tables 3.1 to 3.13 give an indication of available data for each species and show variation in demographic rates between populations. The most appropriate data for use in the population models will be selected from these tables. This selection will consider the quality of data (length of time series, sample size), study location and study period. Where available and of sufficient quality, studies referring to the relevant populations or the North Sea area will be used. This will reduce any potential effect of geographical variation and variation over time, for example due to changes in hunting pressure or prey availability.

Environmental variation in demographic rates versus parameter uncertainty
Population sizes as well as demographic rates vary in time. These fluctuations often seem random, also called stochastic. Three basic forms of stochasticity can be distinguished (Lande *et al.* 2003):
- Measurement error; in estimates of population size, density or demographic rates. Usually, not all individuals within a certain area can be counted. Therefore, the estimated population size or density is often based on the sampling of a smaller area (Seber 1986). This results in measurement error, which explains part of the fluctuations in demographic rates and population size or density (Lande *et al.* 2003).
- Demographic stochasticity; By chance, independent of environmental effects, demographic rates vary between years. With an average adult survival rate of 90%, it may be that by chance the survival is 91% in one year, and 88% in the following. This type of variation is purely random, and cannot be explained by any external factors (Lande *et al.* 2003).
- Environmental stochasticity; Variation in environmental conditions affects demographic rates and influences the fluctuation of population size or density.

Environmental stochasticity is caused by variation in abiotic factors such as temperature or precipitation and biotic factors such as predator density or prey availability (Fujiwara & Takada 2009). If environmental factors (strongly) affecting demographic rates are studied, it is possible to account for variation due to these factors. However, although sometimes some of the main factors affecting demographic rates are known, usually not all environmental stochasticity is explained.

Appendix III gives more background information about different forms of stochasticity, and their implementation into population models.

As it is impossible to measure all variables affecting demographic rates, it is impossible to determine exactly how much each of these three sources of variation contributes to the observed variation in demographic rates.

Within a side-project in cooperation with Wageningen Marine Research (WMR), we analysed the impact of the source of variation on the outcome of the population model. In other words, we analysed the population model for three different scenarios considering the assumptions underlying variation:
1. All variation in demographic rates was due to measurement error.
2. All variation in demographic rates was due to environmental and demographic stochasticity
3. All variation in demographic rates was due to environmental and demographic stochasticity, and demographic rates are correlated within a year. This simulates a scenario in which a good year for survival is also a good year for reproduction (depending on the correlation coefficient).

Complete results of this side project can be found in Appendix IV. Based on the results of these three scenarios, the most cautious approach is to assume that all variation in demographic rates is due to measurement error. For that reason, we used this approach.

Following this approach, we assume that demographic rates do not vary between years, but only between runs, which simulates that all variation is due to measurement error. We run the model for 50,000 iterations, each with different parameter values.
On the individual level, survival is either alive (1) or dead (0). This type of data follows a binomial distribution, which can be simulated with a beta-distribution. Similarly, an individual either attempts breeding or skips breeding. Although a breeding attempt may result in different numbers of fledglings, this can also be simulated with a beta-distribution, with either a failed brood (0 fledglings) or at least 1 fledgling. Based on data available from literature (see Chapter 3), these beta-distributions are defined.
With demographic rates varying between iterations, the output varies as well. This gives an indication of the range within which the outcome is expected to be.

2.1.4 Data quality and representativeness

For each species we searched in literature for information on the origin of individuals using the area around wind turbines in the Dutch North Sea. As documented in Chapter 3, we assessed the availability of information on demographic rates from these populations. Moreover, we assessed the data quality and representativeness for each data source, using the same approach as in Horswill & Robinson (2015). This approach of Horswill & Robinson (2015) is based on the following criteria to assess data quality:

- Q1) the number of years (>10),
- Q2) the number of individuals and
- Q3) whether or not an indication of variation between years or areas (standard deviation), or a range of error (standard error) has been reported.

Each of these criteria is scored with 0, 1, or 2: 0 for 'poor', 1 for 'intermediate/unknown' and 2 for 'good'.

In a similar way, we assess the representativeness of each data source. This representativeness is scored based on:

- R1) how recent the data are (score 2 for data of less than 10 years old; threshold between score 1 and 0 depends on the species and data availability),
- R2) how representative the area/site is for the Dutch part of the North Sea, and
- R3) how representative the data are for the current local trend in the Dutch part of the North Sea. In our study we used data on population trends since 1990 from Boele *et al.* (2017) to assess the current local trend of each species.

As described in Paragraph 2.2, we will use these demographic rates to parameterise the population model.

Example

To illustrate this we show the assessment of the quality and representativeness for selection of data sources for lesser black-backed gull. Data from Camphuysen (2013) and Harris (1970) are compared, both giving information of juvenile survival. Scores for quality and representativeness of both data sources are reported in Table 2.1.

Quality; Camphuysen (2013) uses a dataset of more than ten years (score 2 for Q1) and a high number of individuals (score 2 for Q2). Moreover, an indication of the range of variation (standard deviation) has been reported for this study (score 2 for Q3). This results in a quality score of 6. In contrast, Harris (1970) uses fewer years and individuals (score 1 for both Q1 and Q2). It is unknown whether a standard deviation or standard error has been calculated. In cases where information on one of the criteria is unavailable, a score of 1 is given for this criterion. Hence, for Harris (1970), the score for Q3 is 1. This adds up to a quality score of 3 for Harris (1970), compared to a score of 6 for Camphuysen (2013).

Representativeness; Camphuysen (2013) uses data from up to 2011, hence scoring a 2 for the first score of representativeness (R1). As these data are obtained in the Netherlands, and these birds are present at the North Sea, the score for R2 (representativeness for the area) is 2 as well. The third score compares the trend during

the study with the actual trend at the Dutch North Sea. This gives a score of 2 for R3 as well. Hence, the score for representativeness of Camphuysen (2013) is 6. In contrast, Harris (1970) is based on older data from the UK (scores for R1 and R2 of 1). The reported trend by this study is comparable to the situation at the Dutch North Sea, giving a score of 2 for R3. This adds up to a representativeness of 4 for Harris (1970), indicating that these data are less representative for the Dutch North Sea than Camphuysen (2013) with a score of 6.

Depending on data availability per demographic rate, it is decided whether the estimate with the highest scores on data quality and representativeness is used, or a weighted estimate is calculated. In case of a weighted estimate, this estimate is weighted by the scores on data quality and data representativeness.

Table 2.1   Assessment of quality and representativeness of two data sources, using the approach of Horswill & Robinson (2015).

	Quality				Representativeness			
Reference	Q1	Q2	Q3	**sum**	R1	R2	R3	**sum**
Camphuysen (2013)	2	2	2	**6**	2	2	2	**6**
Harris (1970)	1	1	1	**3**	1	1	2	**4**

2.1.5   Output population models

Examples of output can be the population growth rate, the population size after a given period, or the probability that a population decreases below a certain population size. Within this report, we use the following four metrics.

1. Median population growth rate, and the 5% and 95% quantile for each scenario (unimpacted and impacted). Figure 2.3 shows an example of a frequency distribution of simulated population growth rates. Due to variation in survival and fecundity, the population growth rates differ between simulations, resulting in a distribution around a median population growth rate. A population growth rate (also referred to as lambda) of 1.0 indicates a stable population, whereas a value of 1.03 indicates an annual population growth of 3%, and a value of 0.99 indicates an annual population decline by 1% (Equation 1). The 5% and 95% quantiles give an indication of the range of outcomes for each scenario.
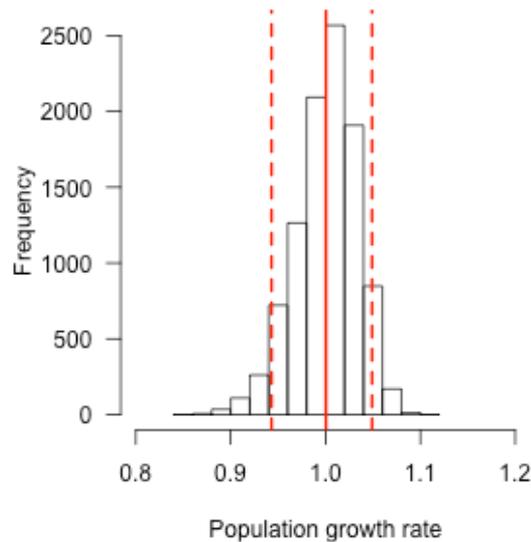
Figure 2.3   Example of a frequency distribution of population growth rates based on 1000 iterations. Continuous red line indicates the median population growth rate, dashed lines the values below which 5% and 95% of the simulations are found.

2. Relative median population size after 30 years. To calculate this metric, the median final population size of the impacted scenario is divided by the median final population size of the unimpacted scenario. This gives an idea of the impact of the additional mortality on the median population size.

3. Probability of a population decline of 10% or more within 30 years. For this metric, we assess the proportion of runs that result in a 10% decline in population size compared to the starting population size. This measure takes into account the variation in population growth rate between iterations. 30 years is the time for which a permit is given, and corresponds closely to the estimated life expectancy of an offshore wind farm.

4. Overlap in the distributions of final population sizes between the impacted and unimpacted scenarios. Here we compare the scenario without wind farm impact with the impacted scenario. For this, we use the following metric: the probability of the scenario without additional mortality to reach the median population size of the scenario with additional mortality. To clarify, this is illustrated in Figure 2.4. In this hypothetical example, the median final population size (i.e. 50th percentile) of the null scenario is 100 individuals, and the median final population size of the impacted scenario is 80 individuals. For this metric, we report the probability of getting a final population size of 80 or less for the scenario without wind farms (null scenario). In other words, what is the probability to reach the outcome of the impacted scenario without having additional mortality due to turbine collisions? For example, if in 5% of the simulations of the null scenario (unimpacted), the final population size is below the median of the impacted scenario (80 in the example illustrated in fig. 2.4), the probability to reach the outcome of the impacted scenario without having additional mortality due to turbine collisions is 5%, i.e. 0.05.
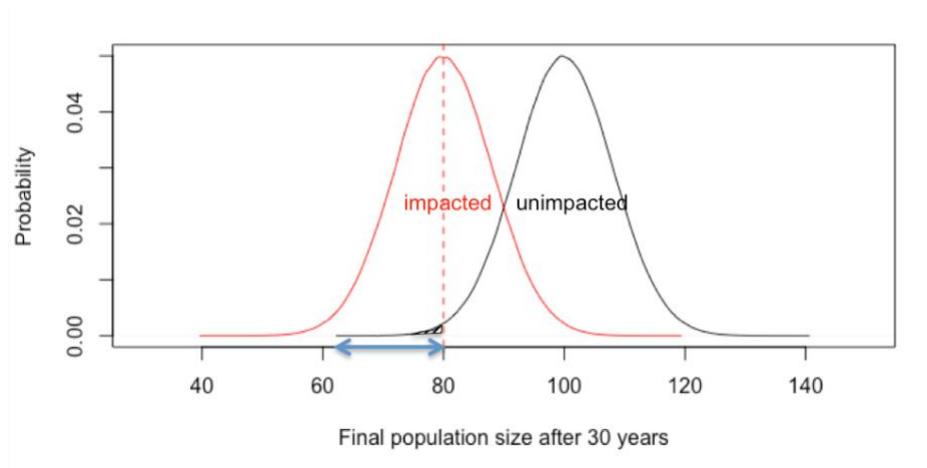
Figure 2.4   Example of distribution of final population sizes of a scenario without impact (black line) and a scenario with impacted (red line). The dashed red line represents the median final population size of the impacted scenario. The dashed area to the left of the vertical dashed line represents the proportion of runs of the scenario without impact of OWFs, which have a final population size below the median of the impacted scenario.

2.1.6   Perturbation analysis

In a so-called perturbation analysis, the impact of changing one or more input parameters on the model output can be assessed (see Caswell 2001: Chapter 9). A change in any of the input parameters of the population model affects the model output, for example population growth rate. However, depending on the demographic rates (survival of different age classes, fecundity), the population growth rate (or any other model outcome) may be particularly impacted by changes in a certain parameter. Two main approaches of perturbation analyses are 'sensitivity' and 'elasticity' analyses. For a description of the methods, see Chapter 9 in Caswell (2001). In short, a sensitivity analysis shows the effect of an absolute change in a parameter on a model output, whereas an elasticity analysis assesses the effect of a relative change. See Appendix V for more information about both types of perturbation analyses.

Within this project, we analyse for each species the relative impact of changes in the fecundity, and survival rates of chicks, immatures and adults on the population growth rate.

2.2   Collision victims

2.2.1   Additional mortality due to collisions

Having structured the null-models (Paragraph 2.1) for each of the species of interest (Chapter 1), the next step is to add additional mortality due to collisions with turbines. The potential number of collision victims for each species of interest has been estimated within the KEC 3.0 framework using the extended SOSS Band model (Band 2012 applied by Gyimesi *et al.* (2019) in Rijkswaterstaat 2019). For species not included in

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the KEC 3.0 framework (little gull, common tern, arctic skua), estimates from KEC 1.0 have been used.

In this version of the SOSS Band model, the estimated number of collision victims does not come with an uncertainty estimate or a range of likely outcomes. In order to assess the effect of a change in these numbers of collision victims, we run the population models for the estimates from the SOSS Band model, for 1%, 5% and 10% fewer collision victims, and for 1%, 5% and 10% more collision victims. This gives insight into the effect of uncertainty of numbers of collision victims on the outcome of the population model.

Only very recently, the opportunity exists to calculate collision estimates using a stochastic collision rate model, which was published in 2018 by Marine Scotland[1]. This collision rate model is based on the SOSS Band model (Band 2012) used in this study, but provides a standard deviation around the estimates. Naturally this also implies consequences for the outcomes of the population models. In addition to the population models with deterministic additional mortality in the main text, models with stochastic collision rates were also performed in this study. Since stochastic collision rate modelling is not widely used we have not incorporated the results in the main text but a detailed comparison of the results with and without stochasticity in collision rates is given in Appendix VIII.

The species of interest reported in Table 1.1 can be divided into seabirds and migrating landbirds. This categorisation is made because seabirds are present in the southern North Sea for a longer period of time, resulting in a prolonged period of potential collision risk, whereas migrating landbirds are only at risk of turbine collisions while passing through during migration. For that reason, the approach used in the collision rate models (Rijkswaterstaat 2019) differs between seabirds and migrating landbirds.

Seabirds
The bird density within the proposed wind farms is based on bimonthly seabird surveys from the period 1991 - 2017. The bird densities are converted to fluxes in the SOSS Band model in order to estimate the number of collision victims based on turbine characteristics and species characteristics (including avoidance rates). The results of these analyses were published by Rijkswaterstaat (2019). In the present study, the numbers of victims in the bimonthly periods (Rijkswaterstaat 2019) are divided by the total number of individuals in the southern North Sea in the same period (based on the same bimonthly surveys) to calculate a fraction of victims for each bi-monthly period (Figure 2.5). For each species the resulting bimonthly fractions of additional mortality due to collisions are presented in Table 2.2. Based on these bimonthly collision risks, an annual percentage of victims was calculated (Table 2.3).

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[1] sCRM, available at https://www2.gov.scot/Topics/marine/marineenergy/mre/current/StochasticCRM
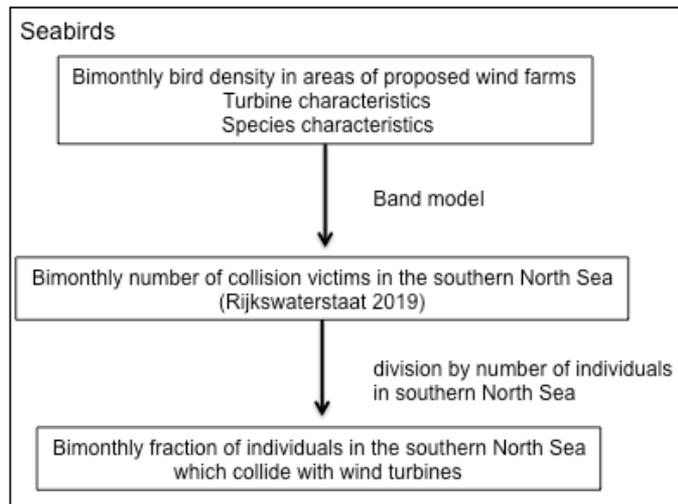
Figure 2.5   Schematic overview of the calculation of the monthly fraction collision victims for seabirds in the southern North Sea.

Table 2.2   Bimonthly percentages of collision victims per seabird species. Values indicate the percentage of collision victims among all individuals present in the southern North Sea. Annual percentage of victims is a weighted average, weighted by the bimonthly counts (MWTL, ESAS). Numbers of victims and bimonthly counts are based on Rijkswaterstaat (2019). Due to difficulty in distinguishing common tern and arctic tern in aerial surveys, these species are merged into 'Commic tern'. The majority of these individuals are common terns.

Species	Aug, Sep	Oct, Nov	Dec, Jan	Feb, Mar	Apr, May	Jun, Jul
Lesser black-backed gull	0.52%	1.01%	0.47%	0.58%	0.46%	0.57%
Great black-backed gull	0.25%	0.24%	0.15%	0.14%	0.19%	0.20%
Herring gull	0.23%	0.12%	0.12%	0.16%	0.18%	0.36%
Black-legged kittiwake	0.04%	0.05%	0.05%	0.04%	0.03%	0.04%
Little gull	0.07%	0.12%	0.17%	0.20%	0.12%	0.07%
Arctic skua	0.02%	0.02%	0.01%	0.01%	0.01%	0.03%
Great skua	0.00%	0.01%	0.01%	0.01%	0.01%	0.00%
'Commic tern'	0.09%	0.02%	0.16%	0.00%	0.10%	0.04%

Table 2.3   Annual percentages of collision victims per seabird species. Values indicate the percentage of collision victims among all individuals present in the southern North Sea. Annual percentage of victims is a weighted average, weighted by the bimonthly counts presented in Table 2.2 (MWTL, ESAS). Numbers of victims and bimonthly counts are based on Rijkswaterstaat (2019).

Species	Percentage annual victims
Lesser black-backed gull	0.557%
Great black-backed gull	0.185%
Herring gull	0.164%
Black-legged kittiwake	0.043%
Little gull	0.126%
Arctic skua	0.018%
Great skua	0.005%
'Commic tern'	0.077%

Migrating landbirds

The SOSS Band model does not incorporate an approach specifically for migrating landbirds, only for 'migrant species'. This approach requires flux as input parameter rather than a density. For most migrant species, particularly landbirds, this relates to two passages over the North Sea each year. For migrant landbirds, the actual flyway population sizes crossing the North Sea were determined based on population estimates of BirdLife International (Rijkwaterstaat 2019). All individuals of the flyway population are assumed to cross the southern North Sea twice a year, although it can be argued that for some species such as common shelduck more crossings are made annually. Based on one crossing per year, an annual number of collision victims has been calculated (Rijkswaterstaat 2019). Similar to the approach for seabirds, this number of collision victims is divided by the total number of individuals 'at risk', which is in this case the flyway population. This results in one annual proportion of additional mortality, in contrast to the approach for seabirds, where monthly proportions of victims are calculated (Figure 2.6). The resulting proportions of annual collision victims per species are presented in Table 2.4.

Table 2.4   Proportion annual collision victims per migrating landbird species.

	Percentage annual victims (proportion of flyway population)
Brent goose	0.06%
Common shelduck	0.51%
Eurasian curlew	0.93%
Black tern	0.81%
Bewick's swan	0.06%
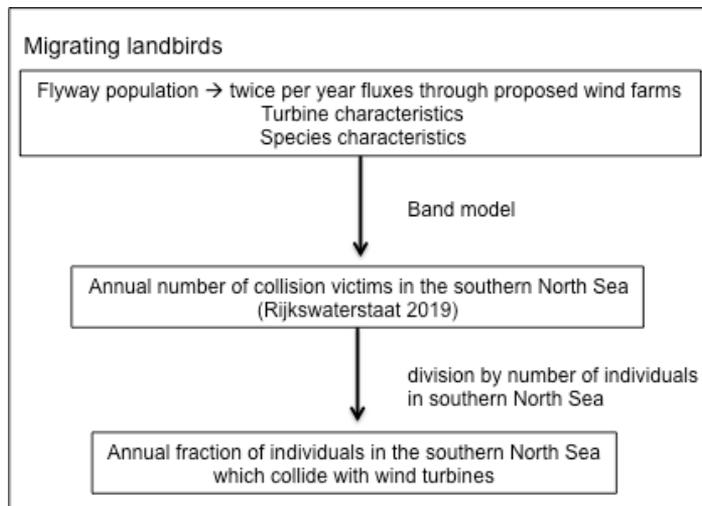
Figure 2.6   Schematic overview of the calculation of the annual fraction of collision victims for migrating landbirds in the southern North Sea.

Stage-specific collision risks

The additional mortality resulting from collisions, as estimated in KEC 3.0 (Rijkswaterstaat 2019), can be applied to different life stages within the given population model. If data are available as to the life stages at risk, these can be incorporated by apportioning the additional mortality accordingly. As described in Paragraph 2.1, for a long-lived species the impact of additional mortality is greater for adults than for immatures.

Data may be available on stage-specific collision risk through assessing the presence of different stages in offshore areas. For example, immatures may spend more time at sea during the breeding season compared to adults, which are more restricted as they have to go back to their breeding site. Alternatively, breeding adults may experience a high collision risk in cases where wind farms are close to a breeding colony (Fox *et al.* 2006). When simulating populations from which there is an indication of such differences in collision risk, this should be incorporated into the population model.

Collision risk may also differ between sexes due to differences in behaviour (Masden & Cook 2016; Marques *et al.* 2014; Camphuysen 2013). For example, differences in foraging behaviour and area use between sexes may result in one sex having a higher collision risk than the other. If this is the case, the use of a two-sex population model may be necessary.

In cases of limited data on stage- or sex-specific collision risk, assumptions will be made regarding the apportioning of victims among age classes and sexes. Approaches often used include:

1)   All stages have the same collision risk. The stage distribution among the victims is assumed to equal the stable stage distribution in the population and the mortality is defined as a certain fraction of each age class.

2) Differences in collision risk are assumed to be driven by differences in presence in the area. If life stages differ in use of offshore areas, their collision risk can be assumed to differ accordingly. Differences in area use can be based on for example surveys or logger data.

3) Only adults are assumed to collide. Especially for long-lived species, the loss of an adult has a much larger impact on the population than the loss of a juvenile. This approach is often used as a worst-case scenario.

Data on stage- and sex-specific collision risk are used where available (approach 2). If no data are available, the first approach is used and a constant collision risk between stages is assumed.

2.2.2 Populations of interest

The aim of this project is to study the effects of collisions between birds and turbines in the southern North Sea. Boundaries of the study area are shown in Figure 2.7.
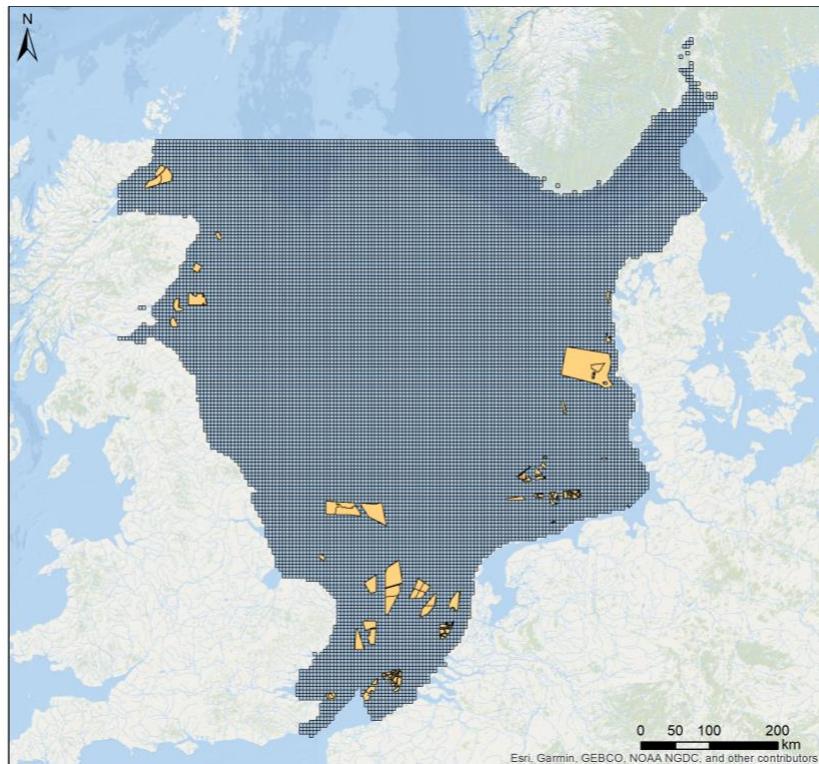
Fig. 2.7   Study area (grid cells) and wind farms included in the analysis of collision mortality (yellow).

As re-sightings of ringed birds further offshore, and in general for the species of interest, are scarce, only very few data on the origin of potential collision victims are available. This makes it difficult to assess from which geographical breeding populations the collision victims originate.

In this case the research question aims to assess the impact of additional mortality on the number of individual birds in the southern North Sea. It is therefore important that the input data of the population model correspond to the same population.

For seabirds, the monthly proportion of victims is based on the bird density in wind farms in the southern North Sea, and the bird density in the entire southern North Sea (inside + outside wind farms). In contrast, the proportion of victims of migrating landbirds is based on the flyway population.

This means that for all species, these proportions of victims can be used to assess the impact in the corresponding populations (southern North Sea and flyway population). If the impact on a different population is to be assessed, the proportion of victims needs to be calculated for this specific population.

In the population models, the initial population size needs to be defined. In case of seabirds, we assume the maximum number of individuals in the southern North Sea at any point in time. For migrating landbirds, we use the same flyway population as used in Rijkswaterstaat (2019). These population sizes were determined based on population estimates from BirdLife International (2018).

Uncertainty exists in how these populations relate to birds at risk from wind turbines in the North Sea, and therefore the appropriate figures to use for 'population' size. In consultation with Rijkswaterstaat and collaborators from the parallel project on population level effects of habitat loss (WMR), it was therefore decided to focus on the relative change in population size instead of focusing on absolute numbers. In other words, instead of for example reporting the final population size after 30 years, we report the relative final population size, defined as follows:

$$\text{relative final population size} = \frac{final\ population\ size\ with\ additional\ mortality}{final\ population\ size\ without\ additional\ mortality}.$$

Similarly, we report the probability of a 10% decline for each scenario, i.e.:

the probability of the final population size being 90% or less of the initial population size (of the same scenario).

With the collision mortality being a proportion, the initial population size does not affect the outcome of the population model and impact assessment (as long as the population is large enough).

3 Model characteristics per species

For each species, we describe input parameters and model assumptions in this chapter. This is based on species characteristics and available input parameters reported in Appendix VI.

As described in Paragraph 2.2, the 'population' definition used in this project differs between seabirds and migrating landbirds. For seabirds, we assume the maximum number of individuals in the southern North Sea at any point in time, and for migrating landbirds, we use the same flyway population as used in Rijkswaterstaat (2019).

Regarding the other parameters of the model, if multiple data sources were available for the same parameter, we always chose parameter values based on source quality and representativeness. In case one data source clearly has a better representativeness and/or quality, only this source was used. In several cases, different data sources are representative and have sufficient quality. In those cases, we calculated weighted averages, for which data sources of low quality and/or representativeness have a lower weight than data sources of higher quality and/or representativeness.

We ran the population model for different scenarios of collision mortality. The scenario with the number of collision victims as estimated in KEC 3.0 including PAWP and OWEZ (Rijkwaterstaat 2018a,b) is referred to as the "KEC-scenario". To illustrate the impact of a difference in number of victims, we also simulated scenarios with a 1%, 5% and 10% lower number of victims (referred to as "KEC-1%-scenario", "KEC-5%-scenario" and 'KEC-10%-scenario"), as well as a 1%, 5% and 10% higher number of victims ("KEC+1%-scenario", "KEC+5%-scenario" and 'KEC+10%-scenario"). In addition, we ran the population model for the "PBR scenario", which assumes the number of victims to be equal to the PBR threshold. This gives an indication of the population-level impact of the number of victims according to the previously used PBR threshold.

3.1 Lesser black-backed gull

Basic demographic rates
Several data sources are available for survival of lesser black-backed gulls. In order to get survival rates that reflect the defined population as closely as possible, we calculated a weighted average for each parameter. This value is based on the values reported in Table VI.1 (Appendix VI), and weighted according to data quality and representativeness. Juvenile survival is based on Harris (1970), Camphuysen & Gronert (2012) and Camphuysen (2011). Immature survival is based on Camphuysen (2013) and Camphuysen (2011). Adult survival is based on data from Camphuysen & Gronert (2012), Camphuysen (2011), Wanless *et al.* (1996), Horswill & Robinson (2015). This resulted in the parameter values for stage-specific survival reported in Table 3.1.1.

Similarly, fecundity is based on the values reported in Table VI.1 (Appendix VI), and weighted according to data quality and representativeness. Used sources for fecundity are Wanless *et al.* (1996), Gyimesi *et al.* (2011), Camphuysen in Koffijberg *et al.* (2017), Spaans *et al.* (1994), Perrins & Smith (2000) and Mavor *et al.* (2008).

We calculated the proportion of floaters among adult lesser black-backed gulls (i.e. incidence of non-breeding) to be 43.5%. This is calculated as the average of two available estimates as reported in Appendix VI, Table VI.1 (Camphuysen 2013; Calladine & Harris 1997).

Table 3.1.1 Parameters used in population model of lesser black-backed gull. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.1 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.52	weighted estimate
Immature survival	0.856	weighted estimate
Adult survival	0.914	weighted estimate
Fecundity	0.792	weighted estimate
Age of first breeding	5	
Incidence of nonbreeding (percentage floaters)	43.5%	average of available estimates

Collision victims
The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.2 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.56%. We assume that the numbers of collision victims per stage depend on the age distribution in the southern North Sea. In other words, if adults use the southern North Sea more intensively than immatures, a relatively high proportion of victims are assumed to be adults. The age distribution of lesser black-backed gulls in the southern North Sea has been analysed by Camphuysen & Leopold (1994). They found that on

average 82.9% of the lesser black-backed gulls were adults, 10.3% were first-year individuals, and the remaining 6.8% immatures (Table 3.1.3). We used the same distribution among the collision victims. Although male lesser black-backed gulls use marine habitats more than females (Camphuysen *et al.* 2015), no empirical evidence exists on differences in collision risk between sexes. For that reason, we assume no sex-specific collision risk, and use a one-sex population model.

Table 3.1.2 Parameters related to collision mortality.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.56%	based on Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	stage-specific, based on presence in Southern North Sea (Camphuysen & Leopold 1994); first year: 0.09%; immatures 0.12%; adults 0.7%	Camphuysen & Leopold (1994)

Table 3.1.3 Distribution of age classes of lesser black-backed gulls in the Southern North Sea. Source: Camphuysen & Leopold (1994).

Month	Adults	Immatures	First year	% adult	Sample
Jan	103	5	16	83.1	124
Feb	78	9	22	71.6	109
Mar	1214	78	34	91.6	1326
Apr	2371	183	22	92	2576
May	3291	430	216	83.6	3937
Jun	2001	183	28	90.5	2212
Jul	3327	293	63	90.3	3683
Aug	2602	78	497	81.9	3177
Sep	1565	119	907	60.4	2591
Oct	910	72	369	67.4	1351
Nov	333	19	39	85.2	391
Dec	38	1	6	84.4	45
Totals	17833	1470	2219	82.9	21522

3.2 Great black-backed gull

Basic demographic rates

Data on demographic rates of great black-backed gull are limited. For adult survival and fecundity, available data are weighted by quality of the data source, giving the estimates provided in Table 3.2.1. In the calculation of the weighted estimate for fecundity, the data source from the USA (Butler & Trivelpiece 1981) is left out because of low quality and representativeness. Age of first breeding is assumed to be 4 (Robinson 2018). Data on the incidence of non-breeding are lacking, and we assumed this to be 25%. To our knowledge, data on juvenile and immature survival are lacking. Survival rates are expected to be similar between great black-backed gull and herring gull due to similar wintering areas. Therefore, we use estimates of subadult survival of herring gull.

Table 3.2.1 Parameters used in population model of great black-backed gull. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.2 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.436	weighted estimate herring gull
Immature survival	0.8	weighted estimate for herring gull
Adult survival	0.859	weighted estimate
Fecundity	0.968	weighted estimate selected sources
Age of first breeding	4	Robinson (2018)
Incidence of nonbreeding (percentage floaters)	25%	estimate

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.2 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.185%. We assume equal collision risk for males and females. Subadult individuals may be at higher risk because they use the area around offshore wind farms during spring and summer as well (www.vogeltrekatlas.nl). However, no clear empirical data on differences in collision risk are available. A higher proportion of subadult individuals would result in a smaller impact on population level. Therefore, as a precautionary approach, we assume that age classes do not differ in collision risk.

Table 3.2.2 Parameters related to collision mortality of great black-backed gull.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.185%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	according to stable stage distribution population model	assumption

3.3 Herring gull

Basic demographic rates
We calculated weighted averages based on available data, as reported in Table VI.3 (Appendix VI). Resulting demographic rates are reported in Table 3.3.1. The estimates by Camphuysen & Gronert (2012) and Koffijberg *et al.* (2017) are left out in the calculation of the weighted average of fecundity, as the data overlap with Camphuysen in Koffijberg *et al.* (2017). Hence, fecundity is based on Camphuysen in Koffijberg *et al.* (2017), Mavor *et al.* (2008), Sellers & Shackleton (2011) and Wanless *et al.* (1996). Adult survival is based on Wanless *et al.* (1996) and Camphuysen & Gronert (2012), as these sources are clearly of higher quality. Similarly, available estimates as reported in Table VI.3 (Appendix VI) are used to calculate a weighted estimate of juvenile survival (Camphuysen 2013; Chabrzyk and Coulson 1976; Wanless *et al.* 1996) and immature survival (Camphuysen 2013; Wanless *et al.* 1996).

Table 3.3.1 Parameters used in population model of herring gull. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.3 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.436 (first year)	weighted estimate
Immature survival	0.8	weighted estimate
Adult survival	0.8425	weighted estimate selected sources
Fecundity	0.852	weighted estimate selected sources
Age of first breeding	4	Wanless *et al.* (1996)
Incidence of nonbreeding (percentage floaters)	10%	estimate

Collision victims
The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.2 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.164%. Individuals spending more time at sea are assumed to experience higher collision risk. Therefore, we used data from Camphuysen & Leopold (1994) to assess the distribution of age classes at sea (Table 3.3.2). Based on this data source, we assumed 67% adults, 14% immatures, and 19% first year individuals. This results in a stage-specific additional annual mortality.

Table 3.3.2 Parameters related to collision mortality of herring gull.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.164%	based on Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	stage-specific; among victims we assume 67% adults, 14% immatures, and 19% first year individuals	Camphuysen & Leopold (1994)

Table 3.3.3 Distribution of age classes of herring gulls in the Southern North Sea. Source: Camphuysen & Leopold (1994).

Month	Adults	Immatures	First year	% adult	Sample
Jan	6345	959	1469	72.3	8773
Feb	4102	727	802	72.8	5631
Mar	3143	876	505	69.5	4524
Apr	1642	822	139	63.1	2603
May	1399	529	553	56.4	2481
Jun	1482	146	14	90.3	1642
Jul	1174	44	12	95.4	1230
Aug	370	31	151	67.0	552
Sep	109	117	431	16.6	657
Oct	877	345	1354	34.0	2576
Nov	1554	202	566	66.9	2322
Dec	699	198	207	63.3	1104
Totals	22896	4996	2219	67.2	34095
Totals winter (Oct-March)	16720	3307	4903	67.1	24930

3.4 Black-legged kittiwake

Basic demographic rates
The values used in the kittiwake population model are reported in Table 3.4.1. Data on adult survival and fecundity rates in areas around the North Sea are available from several studies (Table VI.1 in Appendix VI)). For these demographic rates, we calculated weighted averages, which were weighted by data quality. The weighted estimate for adult survival is based on Horswill & Robinson (2015), Sandvik *et al.* (2005) and Reiertsen *et al.* (2014), which are of higher quality than other available sources. Similarly, the weighted average for fecundity is based on relatively higher quality data sources, which are Mavor *et al.* (2008), Frederiksen *et al.* (2004), and the estimate by the JNCC Seabird Monitoring Programme Database. For juvenile survival, only one estimate was available (Coulson & White 1959).

Table 3.4.1 Parameters used in population model of kittiwake. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.4 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.79	Coulson & White (1959); Horswill & Robinson (2015)
Immature survival	0.9	Coulson & White (1959); Thomas & Coulson (1988)
Adult survival	0.8487	weighted estimate selected sources
Fecundity	0.558	weighted estimate selected sources
Age of first breeding	4	Coulson (2011); Mavor *et al.* (2008); Cam *et al.* (2002)
Incidence of nonbreeding (percentage floaters)	25%	estimate

Collision victims
The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.2 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.043%. We assume equal collision risk for males and females, and therefore assume that 50% of victims are females. Although Furness (2015) suggests that the proportion of adults may be lower further offshore, clear empirical data showing this pattern are lacking. For that reason, we assume that the victims follow the same distribution as the stable stage distribution of the defined population. Compared to the

suggested lower abundance of adults further offshore (Furness 2015), our approach is a more cautious approach with a higher proportion of adults among the victims.

Table 3.4.2 Parameters related to collision mortality of kittiwake.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.043%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	according to stable stage distribution population model	assumption

3.5   Little gull

Basic demographic rates
For little gull, data availability is very limited. Available data are reported in Table VI.5.

Data on juvenile and immature survival are not available from the literature. Adult survival is estimated at 0.8 (Garthe & Hüppop 2004), based on similar species (not specified).

For fecundity, several sources are available (Koks 1998; Putkonen 1939; Cramp & Simmons 1983; Haverschmidt 1942; see Table VI.5 in Appendix VI). In our population models, the used measure for fecundity is number of fledglings per breeding pair. Several studies report only clutch size (i.e. number of eggs per nest). Clutch sizes vary between 2.17 and 2.71 (see Table VI.5 in Appendix VI). Only two studies studied numbers of fledglings (Veen 1978; Veen 1980), both based on data from the Lauwerszee (Netherlands). Both studies report very low numbers of fledglings per breeding pair (0 - 0.1; Veen 1978; Veen 1980), which is likely due to these birds being outside their core breeding area and possibly concerning pioneering individuals.

Due to the lack of suitable data, no population model for little gull is constructed. Further research is necessary in order to get estimates of survival and breeding success. Without these data, no reliable population model can be constructed.

3.6 Great skua

Basic demographic rates

Values used in the great skua population model are reported in Table 3.6.1. Data on adult and juvenile survival and fecundity rates in areas around the North Sea are available from several studies (Table VI.6 in Appendix VI). For these demographic rates, we calculated weighted averages, which were weighted by data quality (for description, see in Chapter 2.1 and 3.1). The weighted average for fecundity is based on the estimate from the JNCC Seabird Monitoring Programme Database, and Mavor *et al.* (2008). Juvenile survival is based on Furness (1978) and Balmer & Peach (1997). Only one estimate was available for immature survival (Furness 1978). Adult survival is based on Furness (1978), Balmer & Peach (1997), Ratcliffe *et al.* (2002), Catry *et al.* (1998) and del Hoyo *et al.* (1996).

Table 3.6.1 Parameters used in population model of great skua. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.6 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.82	weighted estimate
Immature survival	0.93	Furness (1978)
Adult survival	0.89	weighted estimate
Fecundity	0.536	weighted estimate selected sources
Age of first breeding	7	Robinson (2018)
Incidence of nonbreeding (percentage floaters)	8.9%	Catry *et al.* (1998)

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.2 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.005%. We assume equal collision risk for males and females, and therefore assume that 50% of victims are females. We assume no difference between age classes in collision risk.

Table 3.6.2 Parameters related to collision mortality of great skua.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.005%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	according to stable stage distribution population model	assumption

3.7 Arctic skua

Basic demographic rates

All values used in the arctic skua population model are reported in Table 3.7.1. Data are available for each demographic rate.

The most recent estimate for juvenile survival is given by Cook & Robinson (2010), which is indirectly derived from the observed population trend, breeding success and adult survival. This estimate is close to the other available estimate by O'Donald (1983). In addition, O'Donald (1983) reports a subadult survival of 0.346. In other words, 34.6% of juveniles survives until the age of 4 (age of first breeding). With a first-year survival of 0.74 (Cook & Robinson 2010), this gives an annual survival during the second, third and fourth year of 0.77.

Adult survival is estimated by Phillips & Furness (1998) and O'Donald (1983). As Phillips & Furness (1998) has a higher quality and representativeness, this estimate has been used for adult survival in the population model.

Several studies have analysed breeding success of arctic skua. We calculated a weighted average for fecundity, for which we weighted by data quality (for description, see in Chapter 2.1 and 3.1). The weighted average for fecundity is based on studies with at least five years of data. In addition, O'Donald *et al.* (1974) is excluded, as data from this time period are expected to be less representative than more recent data. Hence, the weighted estimate of fecundity is based on Phillips *et al.* (1996), Dawson *et al.* (2011), Perkins *et al.* (2018), Mavor *et al.* (2008) and Cook & Robinson (2010).

For arctic skua, the frequency of breeding is assumed to be lower in years with poor conditions (concerning food availability and/or predation). Catry *et al.* (1998) found 8% and 3% non-breeders among experienced breeders in 1993 and 1994. However, for this species, this is expected to be an underestimate for the annual proportion of floaters. These proportions are based on experienced breeders, and individuals without breeding experience are not included in the analysis. The percentage of floaters is assumed to be between 15 and 25%. Within the population models, we assumed 25% floaters.

Table 3.7.1 Parameters used in population model of arctic skua. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.7 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.74	Cook & Robinson (2010)
Immature survival	0.77	based on O'Donald (1983)
Adult survival	0.9	Phillips & Furness (1998)
Fecundity	0.488	Weighted estimate
Age of first breeding	4	Phillips & Furness (1998)
Incidence of nonbreeding (percentage floaters)	25%	Estimate

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.4 (based on KEC 1.0; Rijkswaterstaat 2015). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.018 (based on KEC 1.0; Rijkswaterstaat 2015). We assume equal collision risk for males and females, and therefore assume that 50% of victims are females. We assume no difference between age classes in collision risk.

Table 3.7.2 Parameters related to collision mortality of arctic skua.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.018%	Rijkswaterstaat (2015)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	stable stage structure	assumption

3.8 Common tern

Basic demographic rates
All values used in the common tern population model are reported in Table 3.8.1. Data are available for each demographic rate.

Survival estimates are available from the Wadden Sea (van der Jeugd *et al.* 2014) and from Germany (Becker & Ludwigs 2004). For immature survival, estimates from Germany and from the Dutch Wadden Sea are available (Becker & Ludwigs 2004). For common terns in the Wadden Sea, van der Jeugd *et al.* (2014) report an average first-year survival of 0.616 (sd 0.03) including the period until fledging, and a survival of 0.685 for the period from fledging to the following spring. The same study reports a second-year survival of 0.646 (sd 0.028), and an annual adult survival of 0.885. This is very similar to the findings of Becker & Ludwigs (2004), reporting 47% survival in the first 2 years after fledging (based on van der Jeugd *et al.* (2014), this is 0.685*0.646 = 44%). Becker & Ludwigs (2004) estimated the survival during the third year at 0.85, and the adult survival (from fourth year onwards) at 0.9. In contrast, van der Jeugd *et al.* (2014) found a better model fit when including individuals in their third year in the adult stage, and estimated adult survival (from third year onwards) at 0.885. Due to higher quality and representativeness, survival estimates from van der Jeugd *et al.* (2014) are being used in the population model.

Age of first breeding for common tern is three (Becker & Ludwigs 2004; Robinson 2018). In the year after hatching, all individuals remain in the south. In the spring following that (almost two years after hatching), some of the individuals return, although it is unclear which proportion of those individuals returns. For our population models, we assume this to be 50% of the individuals. In the spring after that, at the age of three, all individuals are assumed to have returned to breed.

The breeding frequency is relatively high for common tern. Becker & Ludwigs (2004) showed that only 9% of experienced breeders skips breeding in a given year. As this analysis only includes experienced breeders, the true percentage of floaters is expected to be somewhat higher, although most individuals are assumed to attempt breeding. For our population models, we assumed 10% floaters.

Several studies have analysed breeding success of common tern. We calculated a weighted average for fecundity, for which we weighted by data quality (for description, see in Chapter 2.1 and 3.1). The weighted average for fecundity is 0.56. This is based on all references reported in Table VI.8 in appendix VI. Used data sources for the calculation of weighted average of fecundity are Schekkerman *et al.* (2017), van der Jeugd *et al.* (2014), Becker *et al.* (2001) and Stienen *et al.* (2019).

Collision victims
The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.3 (based on KEC 1.0; Rijkswaterstaat 2015). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.077% over all individuals (all age classes) (Table 2.4). We assume equal collision

risk for males and females, and therefore assume that 50% of victims are females. Individuals staying in southern Europe or Africa are not at risk of collisions with turbines in the North Sea. In the year after hatching, all individuals remain in the south. In the spring following that (almost two years after hatching), some of the individuals return, although it is unclear which proportion of those individuals returns. For our population models, we assume this to be 50% of the individuals. In the spring after that, at the age of three, all individuals are assumed to have returned to breed.

Table 3.8.1 Parameters used in population model of common tern. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.8 (Appendix VI) and text for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.685	van der Jeugd *et al.* (2014)
Immature survival	0.646	van der Jeugd *et al.* (2014); second year
Adult survival	0.885	van der Jeugd *et al.* (2014)
Fecundity	0.56	Weighted estimate
Age of first breeding	3	Becker & Ludwigs (2004); Robinson (2018)
Incidence of nonbreeding (percentage floaters)	10%	Based on Becker & Ludwigs (2004); Becker *et al.* (2001), which report 9% floaters among experienced breeders

Table 3.8.2 Parameters related to collision mortality of common tern.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.077 %	Rijkswaterstaat (2015)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	stage-specific collision risk, see text	assumption

3.9 Black tern

Basic demographic rates

The values used in the black tern population model are reported in Table 3.9.1. We used best available estimates (see VI.9 (Appendix VI) for more available estimates). For survival, all estimates are indirectly derived from Monte Carlo estimation based on breeding success and population trend (van der Winden & van Horssen 2008). For fecundity, we calculated a weighted average based on Tinbergen & Heemskerk (2016), van der Winden & van Horssen (2008) and van der Winden (2005).

Table 3.9.1 Parameters used in population model of black tern. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.9 (Appendix VI) and text for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.595	van der Winden & van Horssen (2008)
Immature survival	0.595	van der Winden & van Horssen (2008)
Adult survival	0.849	van der Winden & van Horssen (2008)
Fecundity	0.86	weighted estimate selected sources
Age of first breeding	2 or 3	van der Winden & van Horssen (2008)
Incidence of nonbreeding (percentage floaters)	80% at age 2, 10% from age 3 onwards	Estimate

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in table 2.4 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.81%. Collision risk is assumed to be equal for males and females. Individuals in their first year stay in West Africa, and only come back to the breeding grounds at the age of two or three years, when first breeding takes place (Servello 2000; van der Winden & van Horssen 2008). Therefore, we assume that these individuals are not at risk of colliding with turbines in the North Sea. For other age classes, we assume equal collision risk. Hence, we determined the stage distribution of victims as the stable stage structure of the population model, but excluding individuals in their second year.

Table 3.9.2 Parameters related to collision mortality of black tern.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.81%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	Among victims no 2nd year individuals; 1st year and adult according to stable stage structure	assumption

3.10 Bewick's swan

Basic demographic rates
Values used in the Bewick's swan population model are reported in Table 3.10.1. For all stage-specific survival rates, we used legring data from Wood *et al.* (2017). Two (very similar) estimates of fecundity were available (Table 3.10). We used the most recent of these estimates (Wood *et al.* 2016).

Table 3.10.1 Parameters used in population model of Bewick's swan. See Table 3.10 for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.78	Wood *et al.* (2017)
Immature survival	0.83	Wood *et al.* (2017)
Adult survival	0.83	Wood *et al.* (2017)
Fecundity	15 first year individuals per adult on wintering grounds	Wood *et al.* (2016)
Age of first breeding	3	Rees (2006)
Incidence of nonbreeding (percentage floaters)	[Not applicable, incorporated into measure of fecundity]	

Collision victims
The percentage collision victims per bimonthly period in the southern North Sea is reported in table 2.4 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.06% (based on KEC 3.0; Rijkswaterstaat 2019). We assume that collision risk is equal for males and females, and age classes do not differ in collision risk.

Table 3.10.2 Parameters related to collision mortality of Bewick's swan.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.06%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	according to stable stage distribution population model	assumption

3.11   Brent goose

Basic demographic rates

The values used in the brent goose population model are reported in Table 3.11.1. To our knowledge, no data are available on juvenile survival of brent goose (Robinson 2018). Instead, in our population model, we use the estimated juvenile survival of black brent *Branta (bernicla) nigricans* as reported by Sedinger *et al.* (2007). It should be noted that this (sub)species may differ in juvenile survival from *Branta (bernicla) bernicla* which is the taxon under consideration here, but better data are not available. Data on adult survival and immature survival are available from several studies (Table VI.6 in Appendix VI). For these demographic rates, we calculated weighted averages, which were weighed by data quality (for description, see in Chapter 2.1 and 3.1). Immature survival is based on Sedinger *et al.* (2007), Ebbinge *et al.* (2002), Boyd (1962) and Balmer & Peach (1997). Adult survival is based on Sedinger *et al.* (2007), Robinson (2018), Ebbinge *et al.* (2002), Sedinger *et al.* (2002), Cramp (1986), Desholm (2009). Fecundity was based on Nolet *et al.* (2013).

Table 3.11.1   Parameters used in population model of brent goose. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.11 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.51	Sedinger *et al.* (2007); note that this is an estimate for black brant
Immature survival	0.849	weighted estimate
Adult survival	0.859	weighted estimate
Fecundity	0.17	Nolet *et al.* (2013); (based on number of first winter ind. per adult between 2006-2016)
Age of first breeding	2	Robinson (2018); Sedinger *et al.* (2006)
Incidence of nonbreeding (percentage floaters)	[Not applicable, incorporated into measure of fecundity]	

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.4 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.06%. We assume equal collision risk for males and females. We found no indication for age-specific collision risks, and therefore assume that the distribution of age-classes among victims follows the stable stage structure of the population.

Table 3.11.2   Parameters related to collision mortality of brent goose.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.06%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	according to stable stage distribution population model	assumption

3.12 Common shelduck

Basic demographic rates

The values used in the common shelduck population model are reported in Table 3.12.1. Data on adult survival and fecundity rates are available from several studies (Table VI.12 in Appendix VI). For adult survival, we used the estimate by Patterson *et al.* (1983), as this estimate is based on a longer time series than the other available estimate. For fecundity, we used a weighted estimate based on Lensink (2001) and Pienkowski & Evans (1982). We found no data on the proportion of floaters. We assume that 10% of the adults are floaters. For juvenile and immature survival, only one estimate was available (Patterson *et al.* 1983; also reported by Robinson 2018).

Table 3.12.1   Parameters used in population model of common shelduck. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.12 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.25	Estimate based on Patterson *et al.* (1983) and Robinson (2018)
Immature survival	0.67	Estimate based on Patterson *et al.* (1983) and Robinson (2018)
Adult survival	0.886	Patterson et al. (1983); Robinson (2018)
Fecundity	0.92	weighted estimate
Age of first breeding	2	Patterson *et al.* (1983); Robinson (2018)
Incidence of nonbreeding (percentage floaters)	10%	estimate

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in Table 2.4 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.51%. We assume equal collision risk for males and females, and therefore assume that 50% of victims are females. We assume no juveniles among the victims, as these do not perform moult migration (Eltringham & Boyd 1963; Wernham *et al.* 2002). For individuals above 1 year old, we assume no difference between age classes in collision risk.

Table 3.12.2   Parameters related to collision mortality of common shelduck.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.51%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	Among victims no 1st year individuals; 2nd year and adult according to stable stage structure	assumption

3.13 Eurasian curlew

Basic demographic rates

The values used in the Eurasian curlew population model are reported in Table 3.13.1. Data on adult and juvenile survival and fecundity rates in areas around the North Sea are available from several studies (Table VI.13 in Appendix VI). For adult survival, we calculated weighted averages, which were weighed by data quality (for description, see in Chapter 2.1 and 3.1) (Berg 1991; Grant *et al.* 1999). Bainbridge & Minton (1978) report estimates of juvenile survival and immature survival. For fecundity, we used data from 1981-2006 as reported in Roodbergen *et al.* (2012).

Table 3.13.1   Parameters used in population model of Eurasian curlew. 'Weighted estimates' are weighted by data quality and representativeness; see Table VI.13 (Appendix VI) for used sources. Juvenile survival is defined as the survival from fledging to the following spring.

Demographic rate	Mean value used in population model	Source
Juvenile survival	0.47	Bainbridge & Minton (1978)
Immature survival	0.63	Bainbridge & Minton (1978)
Adult survival	0.84	weighted estimate, data after 1980
Fecundity	0.355	Roodbergen *et al.* (2012), data 1981-2006
Age of first breeding	3	Robinson (2018); Bainbridge & Minton (1978)
Incidence of nonbreeding (percentage floaters)	10%	estimate

Collision victims

The percentage collision victims per bimonthly period in the southern North Sea is reported in table 2.4 (based on KEC 3.0; Rijkswaterstaat 2019). Taking into account variation in species abundance through the seasons, this results in an annual mortality of 0.93%. We assume equal collision risk for males and females, and that victims are divided among the age classes according to the stable stage structure based on the population model.

Table 3.13.2   Parameters related to collision mortality of Eurasian curlew.

Demographic rate	Values used in population model	Source
1) Percentage annual collision victims southern North Sea	0.93%	Rijkswaterstaat (2019)
2) Sex ratio (perc. females)	50%	assumption
3) Age distribution among victims	according to stable stage distribution population model	assumption

4 Model outcomes per species

In this chapter, we describe the results for the null-scenario, KEC-scenario and KEC+XX%-scenarios, and PBR-scenario. The null-scenario describes the current situation, without additional wind farms. The KEC-scenario describes the situation with estimated additional mortality (Rijkswaterstaat 2019). KEC+XX% describes the situation with XX% more or XX% less additional mortality than estimated in Rijkswaterstaat (2019). In addition, we show the results of the PBR scenario, which describes the scenario in which the number of victims equals the PBR.

The presentation and interpretation of the results are described in Paragraph 2.1.5. The distributions of projected annual growth rates are presented for each scenario. This illustrates the overlap in distributions between the scenarios, which gives an impression of the impact of the additional mortality. In addition, distributions of median population growth rates, relative population size and the probability of the null scenario to be below the median of the affected scenario are shown for each individual scenario. The results of an elasticity analysis conclude each species-paragraph to illustrate the sensitivity of the model output to a change in different input parameters.

In this chapter, we show the outcome of the population models assuming the collision risk as calculated in the KEC framework (Rijkswaterstaat 2019). This assumes a constant collision risk. For a selection of seabird species assessed in this study, stochastic collision rates have been calculated using the recently published stochastic collision risk model by Marine Scotland (2018). A comparison of the results with and without stochastic collision rates is presented in Appendix VIII. For each of the species for which stochastic collision rates have been modelled, the impact on the output of the population model is shortly described within this chapter as well.

In this report we only present the outcomes of the various population models and therefore present the implication for the studied species. In this report we do not assess any of these effects with respect to the current European legislation nor do we provide any suggestions on which level of additional mortality is legally acceptable. Our results only draw conclusions based on a certain proposed threshold, but the final judgement on which threshold(s) should be used is up to the Competent Authority.
Examples of such thresholds can be:
- *"The probability of a 10% decline (from the current population size) within 30 years may not exceed XX%."* This compares the current population size with the population size after 30 years of additional mortality.
- *"The population size after 30 years with additional mortality should be at least XX% of the population size after 30 years without additional mortality."* This compares the outcome of the scenario with additional mortality with the scenario without additional mortality.
- Alternatively, the probability distribution of the null scenario and impacted scenario can be compared. For example, one can assess the probability of reaching the

outcome of the impacted scenario in a situation without additional mortality. This would give an indication of how much the outcomes of both scenarios overlap. An example of such a threshold can be: "The probability of reaching the median outcome of the impacted scenario if there is no additional mortality, should not be lower than XX%." This takes into account the variation in outcome of both the null scenario and the impacted scenario.

Possible thresholds are also described in among others Cook & Robinson (2017) and May *et al.* (2019). Within this results section, we present results in such a way that they can be used for legislation. After the government has decided which thresholds should be used, these population models can be used to assess whether the expected impact results in violation of the law.

4.1 Lesser black-backed gull

Without additional mortality, the population model projects a stable population, with a median lambda of 1.003 (Table 4.1). The 5th and 95th percentile of the projected lambdas are 0.9412 and 1.049, indicating that 90% of the runs result in a trend between a decline of 5.9% (lambda 0.941 = 1 – 0.059) and an increase of 4.9% (lambda 1.049). This corresponds with the observed stable population trend (sovon.nl, Meetnet Watervogels).

The KEC-scenario projects a very similar lambda of 0.9969 (Table 4.1), indicating a median projected annual population decline of 0.3%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.936 and 1.043, i.e. between an annual decline of 6.4% and increase of 4.3%.

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.1.
For the null scenario, 42% of the runs show a decline in the population size of 10% or more over 30 years. For the scenario with additional mortality based on KEC 3.0 (Rijkswaterstaat 2019), such a decline is projected in 49% of the runs.

The median final population size is for the KEC-scenario 83.3% of the final population size for the null scenario (Figure 4.1.2; Table 4.1).
There is a strong overlap between distributions of final population sizes of the null-scenario and KEC-scenario (Figure 4.1.1, left side). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 43% (Table 4.1). In other words, without additional mortality the probability of getting the median outcome of the KEC scenario is 43%.

The distribution of annual growth rates of the PBR-scenario deviates more from the distribution of the null scenario (Figure 4.1.1, right side). In comparison to the KEC-scenario, the PBR-scenario projects a lower median population growth rate of 0.976 (annual decline of 2.4%), with a 5th and 95th percentile of 0.916 and 1.021). The PBR-scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would increase considerably from 42% of the simulations for the null scenario to 76% of the simulations for the PBR-scenario. Moreover, the median final population size for the PBR-scenario is 56.3% lower than for the null scenario (relative final population size is 0.437; Table 4.1). This results in a clearly lower final population size than for the KEC scenario.

Table 4.1   *Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0030	0.9412	1.0493	42%		
scenario KEC	0.9969	0.9360	1.0427	49%	0.833	43%
scenario KEC-1%	0.9970	0.9358	1.0428	49%	0.835	43%
scenario KEC-5%	0.9974	0.9370	1.0431	49%	0.844	43%
scenario KEC-10%	0.9974	0.9365	1.0431	49%	0.844	43%
scenario KEC+1%	0.9969	0.9360	1.0426	50%	0.832	43%
scenario KEC+5%	0.9968	0.9358	1.0428	50%	0.830	42%
scenario KEC+10%	0.9969	0.9344	1.0428	50%	0.831	43%
scenario PBR	0.9757	0.9162	1.0210	76%	0.437	22%
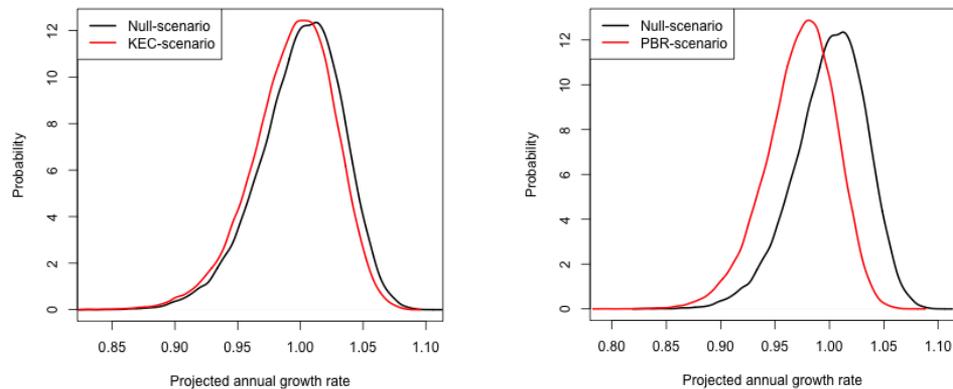
Fig. 4.1.1   *Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).*
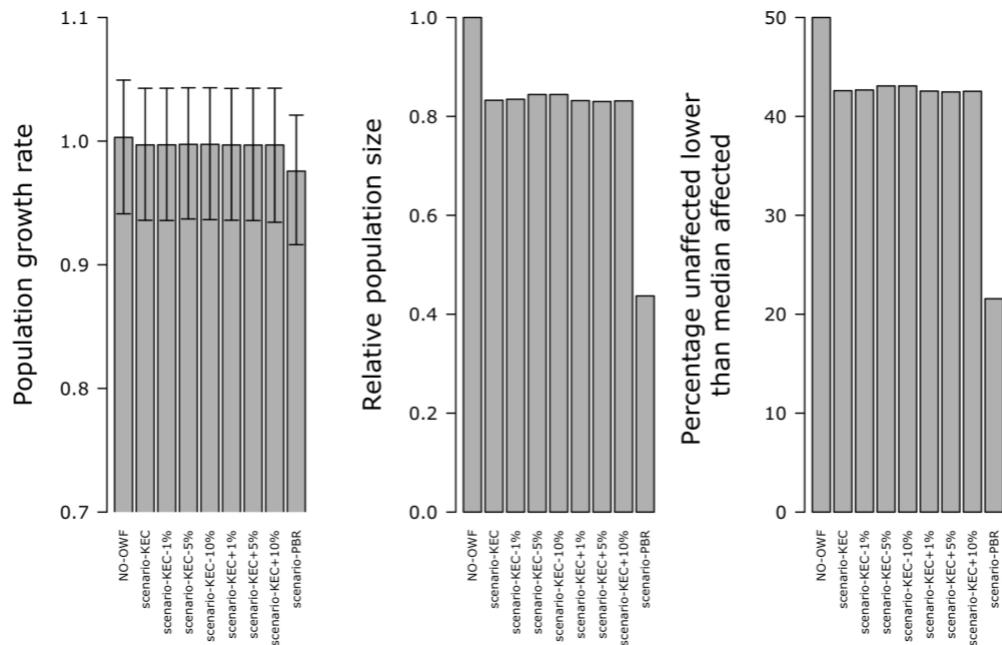
Fig. 4.1.2   *Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.*

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.1.3 and 4.1.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.

Stochastic number of collision victims

In addition to the population models with deterministic additional mortality, models with stochastic collision rates have been performed. A more detailed comparison of the results with and without stochasticity in collision rates is given in Appendix VIII. For lesser black-backed gull, variation in body length, wingspan and flight speed does not affect the projected population growth rate. Stochasticity in flight height distribution gives a lower estimated additional mortality (0.3% compared to 0.56% for the deterministic CRM), resulting in a smaller impact on the population growth rate, relative final population size (relative to null scenario) and percentage unaffected lower than median affected.
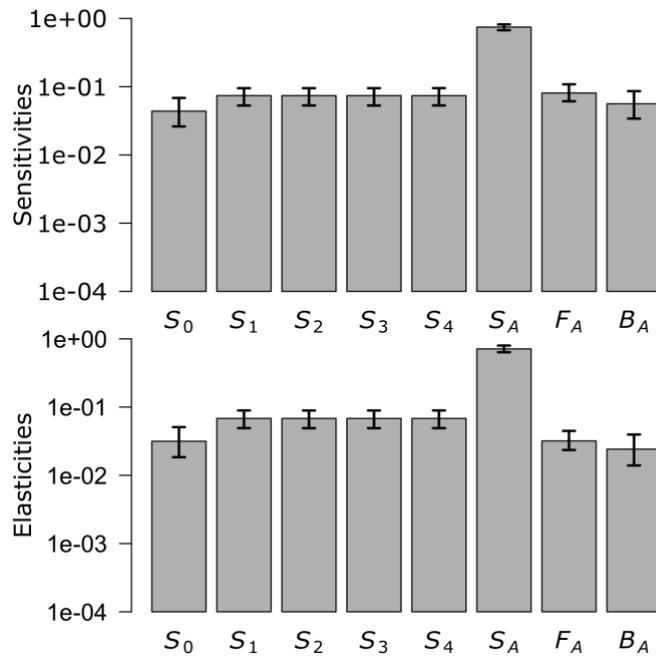
Fig. 4.1.3   *Elasticity and sensitivity of model output to input parameters.* S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
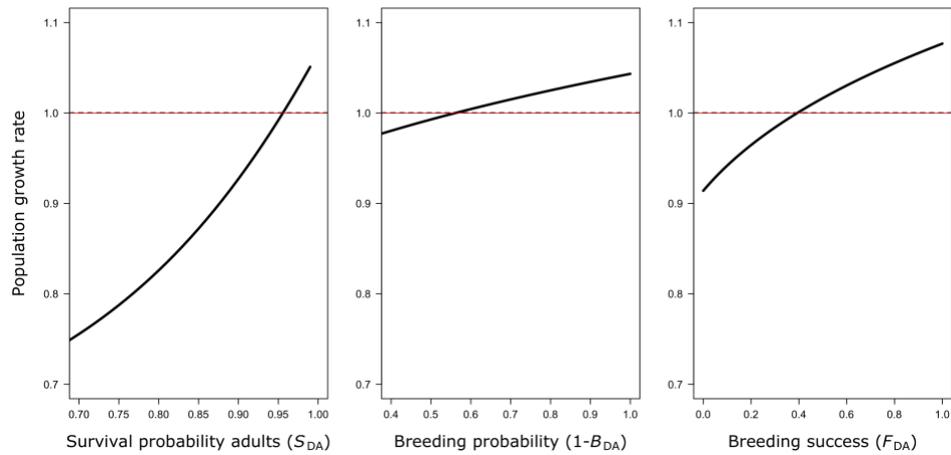
Fig. 4.1.4   *Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population).*

4.2   Great black-backed gull

Without additional mortality, the population model projects a declining population, with a median lambda of 0.988. This indicates an annual population decline by 1.2%. The 5th and 95th percentile of the projected lambdas are 0.891 and 1.065, indicating that 90% of the runs result in a trend between a decline of 10.9% and an increase of 6.5%. This corresponds with the observed population decline (sovon.nl, Meetnet Watervogels).

In contrast, the KEC-scenario projects a slightly lower lambda of 0.9856, indicating a median projected annual population decline of 1.4%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.891 and 1.063, i.e. between an annual decline by 10.9% and increase of 6.3%.

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.2.
For the null scenario, 57% of the runs show a 10% or stronger decline in population size over 30 years. For the scenario with additional mortality based on KEC 3.0 (Rijkswaterstaat 2019), such a decline is projected in 58% of the runs.

The median final population size is for the KEC-scenario 92.8% of the median final population size for the null scenario (Fig. 4.2.2; Table 4.2).
There is a strong overlap between distributions of final population sizes of the null-scenario and KEC-scenario (Figure 4.2.1, left side). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 48%. In other words, without additional mortality the probability of getting the median outcome of the KEC scenario is 48%.

The PBR-scenario projects a clearly lower median population growth rate of 0.9617 (annual decline of 3.8%), with a 5th and 95th percentile of 0.867 and 1.038. The PBR-scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would considerably increase from 57% of the simulations for the null scenario to 76% of the simulations for the PBR-scenario. Moreover, the median final population size for the PBR-scenario is 55.5% lower than for the null scenario (relative final population size is 0.445; Table 4.2). This results in a clearly lower final population size than for the KEC scenario.

Table 4.2   *Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9881	0.8906	1.0648	57%		
scenario KEC	0.9856	0.8905	1.0626	58%	0.928	48%
scenario KEC-1%	0.9854	0.8900	1.0628	58%	0.923	48%
scenario KEC-5%	0.9854	0.8894	1.0633	58%	0.923	48%
scenario KEC-10%	0.9856	0.8900	1.0630	58%	0.927	48%
scenario KEC+1%	0.9854	0.8890	1.0623	58%	0.923	48%
scenario KEC+5%	0.9851	0.8904	1.0622	59%	0.914	48%
scenario KEC+10%	0.9855	0.8907	1.0633	58%	0.926	48%
scenario PBR	0.9617	0.8673	1.0375	76%	0.445	32%
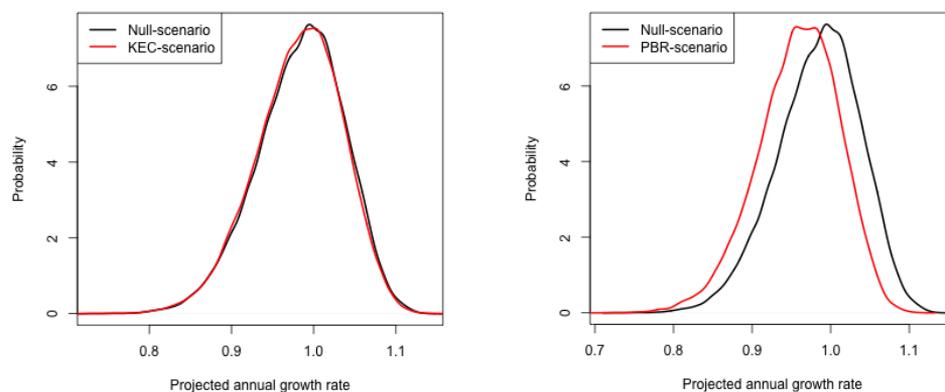
Fig. 4.2.1   *Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).*
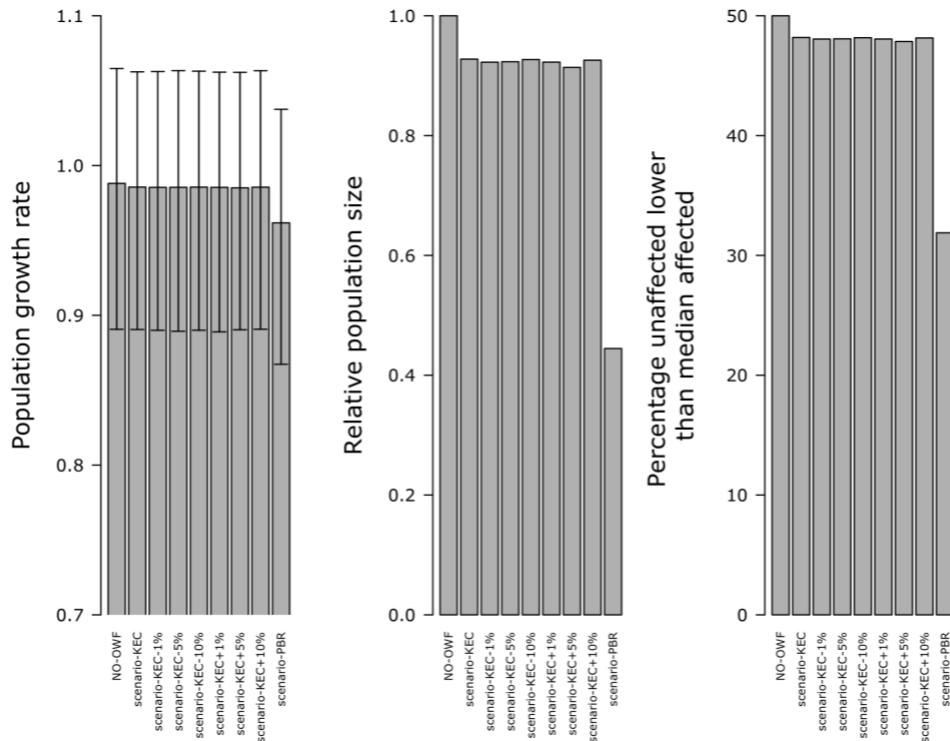
Fig. 4.2.2   *Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.*

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, first year survival or breeding probability (Figure 4.2.3 and 4.2.4). The impact of a change in sub-adult survival is not as strong as for adult survival, but stronger than the impact of changes in other parameters. These results show that a 1% change in adult (or subadult) survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult (and subadult) survival than for other parameters.

Stochastic number of collision victims

In addition to the population models with deterministic additional mortality, models with stochastic collision rates have been performed. A more detailed comparison of the results with and without stochasticity in collision rates is given in Appendix VIII. For great black-backed gull, variation in body length, wingspan and flight speed does not affect the projected population growth rate. In contrast, sampling from different flight height distributions results in this particular case in a higher proportion of individuals at rotor height compared to the number of victims estimated in KEC 3.0 using the deterministic model (Rijkswaterstaat 2019). Hence, stochasticity in flight height distribution gives a higher estimated additional mortality (0.55% compared to 0.18% for

the deterministic CRM), resulting in a stronger impact on the population growth rate, relative final population size (relative to null scenario) and percentage unaffected lower than median affected.
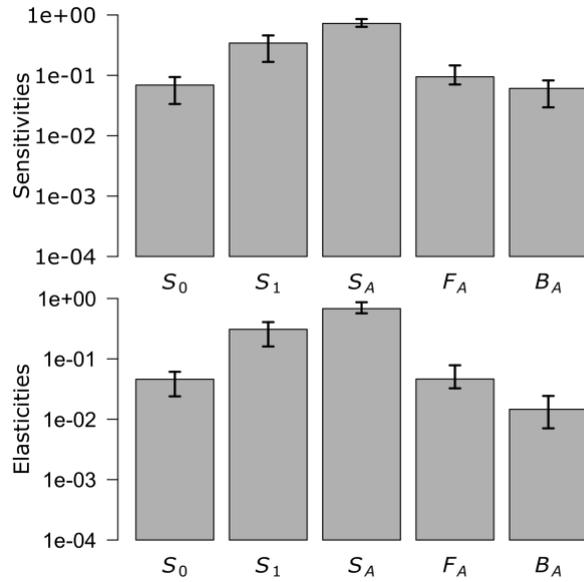
Fig. 4.2.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
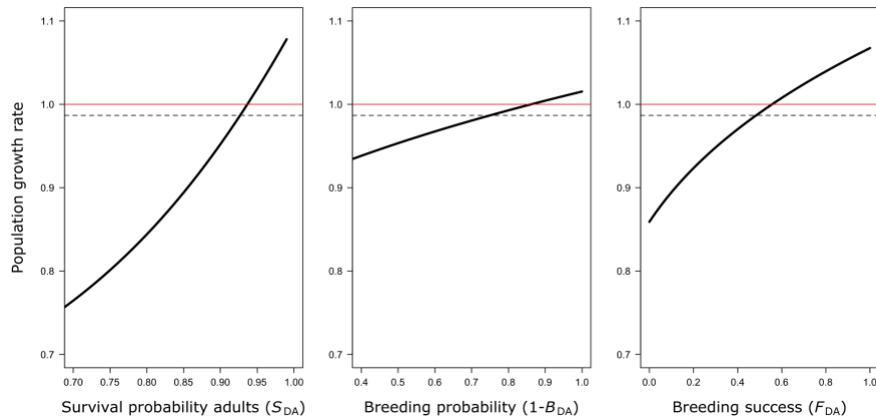
Fig. 4.2.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.3 Herring gull

Without additional mortality, the population model projects a declining population, with a median lambda of 0.9830 (1.7% decline; Table 4.3). The 5th and 95th percentile of the projected lambdas are 0.8995 and 1.0509, indicating that 90% of the runs result in a trend between an annual decline of 10% and an increase of 5.1%. This corresponds with the observed population decline (sovon.nl, Meetnet Watervogels).

The KEC-scenario projects a very similar median lambda of 0.9812 (Table 4.3), indicating a median projected annual population decline of 1.9%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.8991 and 1.0483, i.e. between an annual decline of 10.1% and increase of 4.8%. This is not only very similar to the null scenario, but also to the projected population growth rate for the PBR-scenario (median 0.9779; 5% and 95% quantile 0.8954 and 1.0457).

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.3 and Figure 4.3.2.
For the null scenario, 62% of the runs show a 10% or stronger decline in population size over 30 years. For the scenario with additional mortality based on KEC 3.0 (Rijkswaterstaat 2019), such a decline is projected in 63% of the runs.

The median final population size for the KEC-scenario is 94.7% of the median final population size for the null scenario (Figure 4.3.2; Table 4.3).
There is a very strong overlap between distributions of final population sizes of the null scenario and KEC scenario (Figure 4.3.1). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 48% (Table 4.3). In other words, the probability of getting the median outcome of the KEC scenario is very similar for the null scenario (50%) and the KEC scenario (48%).

For herring gull, the PBR is relatively small. The PBR-scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would increase from 62% to 66%. The median final projected population size is 14.4% lower than the final population size for the null scenario (relative final population size is 0.856; Table 4.3).

Table 4.3   *Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9830	0.8995	1.0509	62%		
scenario KEC	0.9812	0.8991	1.0483	63%	0.947	48%
scenario KEC-1%	0.9809	0.8986	1.0494	63%	0.939	48%
scenario KEC-5%	0.9817	0.8995	1.0494	63%	0.960	49%
scenario KEC-10%	0.9816	0.8976	1.0491	63%	0.959	49%
scenario KEC+1%	0.9811	0.8992	1.0494	63%	0.943	48%
scenario KEC+5%	0.9813	0.8984	1.0491	63%	0.949	49%
scenario KEC+10%	0.9818	0.8992	1.0489	63%	0.963	49%
scenario PBR	0.9779	0.8954	1.0457	66%	0.856	46%
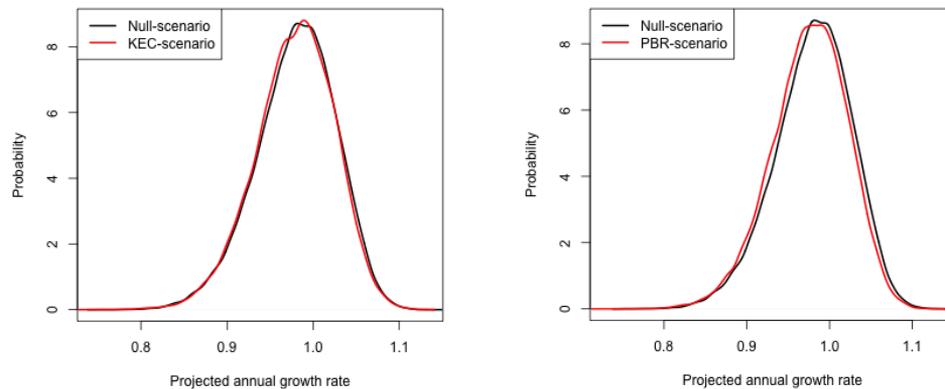
Fig. 4.3.1   *Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).*
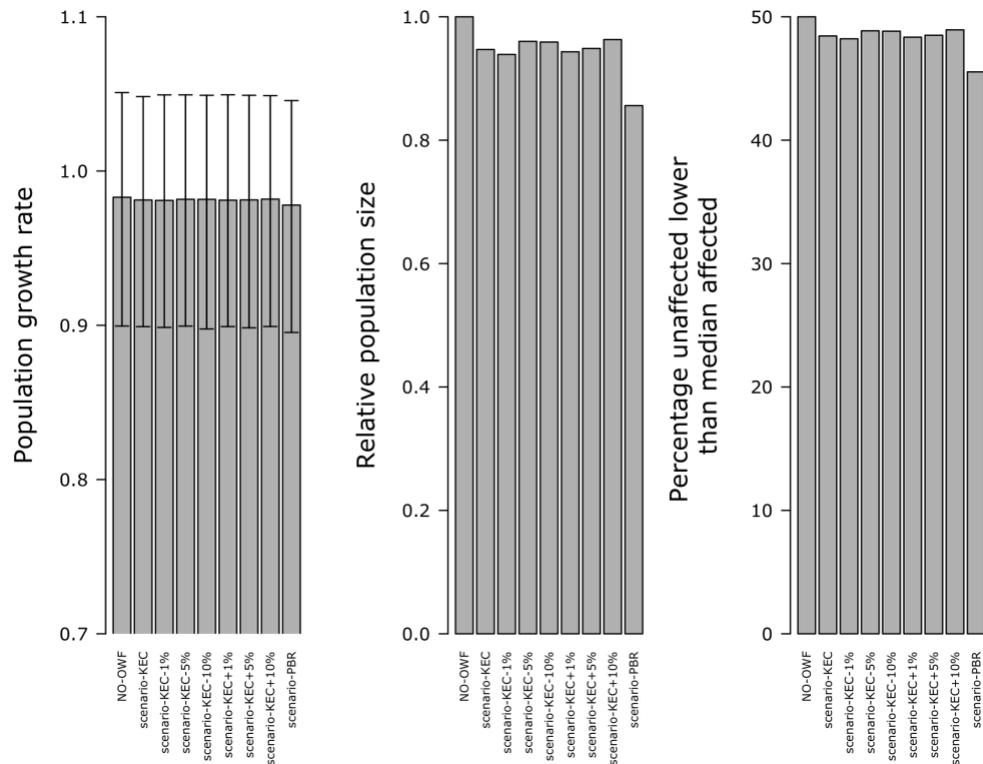
Fig. 4.3.2 Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.3.3 and 4.3.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.

Stochastic number of collision victims

In addition to the population models with deterministic additional mortality, models with stochastic collision rates have been performed. A more detailed comparison of the results with and without stochasticity in collision rates is given in Appendix VIIII. For herring gull, variation in body length, wingspan and flight speed does not affect the projected population growth rate (0.16% additional mortality). Sampling from different flight height distributions results in this particular case in a slightly lower proportion of individuals at rotor height compared to the number of victims estimated in KEC 3.0 using the deterministic model (Rijkswaterstaat 2019). Hence, stochasticity in flight height distribution gives a slightly lower estimated additional mortality (0.12% compared

to 0.16% for the deterministic CRM). The impact on the population growth rate, relative final population size (relative to null scenario) and percentage unaffected lower than median affected is limited.
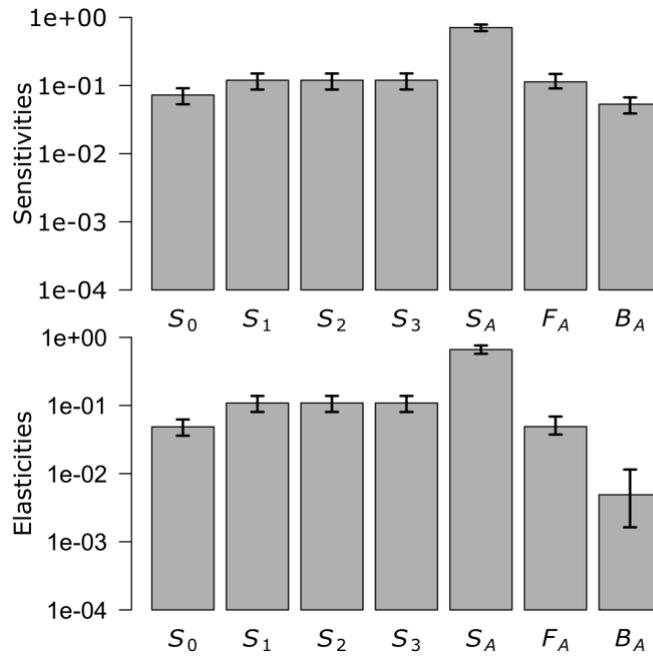
Fig. 4.3.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
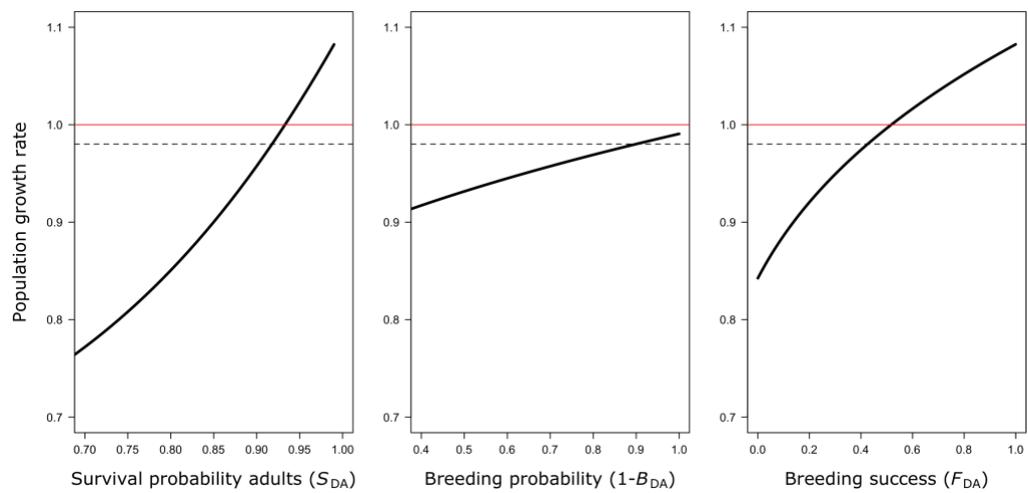
Fig. 4.3.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.4   Black-legged kittiwake

Without additional mortality, the population model projects a declining population, with a median lambda of 0.9888. This indicates an annual population decline by 1.12%. The 5th and 95th percentile of the projected lambdas are 0.9031 and 1.0653, indicating that 90% of the runs result in a trend between a decline of 9.69% and an increase of 6.53%. This corresponds with the observed population decline (sovon.nl, Meetnet Watervogels).

The KEC scenario projects a very similar lambda of 0.9882, indicating a median projected annual population decline of 1.18%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.9027 and 1.0637, i.e. between an annual decline of 9.73% and increase of 6.37%. For the PBR scenario, the projected annual growth rate is slightly lower with 0.9858 (1.42% annual decline).

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.4.
For the null scenario, 56% of the runs show a 10% or stronger decline (Table 4.4). The probability of such a decline is 57% for the KEC scenario, and 58% for the PBR scenario.

The median final population size for the KEC scenario is 98.2% of the median final population size of the null scenario (Figure 4.4.2; Table 4.4). For the PBR scenario, this is 91.3% of the median final population size of the null scenario.

There is a very strong overlap between distributions of final population sizes of the null scenario and the KEC scenario (Figure 4.4.1). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 49%. In other words, the probability of getting the median outcome of the KEC scenario is roughly the same for the null scenario (49%) and the KEC scenario (50%).

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would slightly increase from 56% to 59%. The median final population size after 30 years would be 8.7% lower than for the null scenario (relative population size is 0.913).

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9888	0.9031	1.0653	56%		
scenario KEC	0.9882	0.9027	1.0637	57%	0.982	49%
scenario KEC-1%	0.9874	0.9023	1.0638	57%	0.957	49%
scenario KEC-5%	0.9882	0.9029	1.0644	57%	0.980	49%
scenario KEC-10%	0.9881	0.9027	1.0642	57%	0.979	49%
scenario KEC+1%	0.9883	0.9035	1.0639	57%	0.986	50%
scenario KEC+5%	0.9884	0.9021	1.0635	57%	0.988	50%
scenario KEC+10%	0.9879	0.9034	1.0642	57%	0.972	49%
scenario PBR	0.9858	0.9001	1.0616	59%	0.913	48%
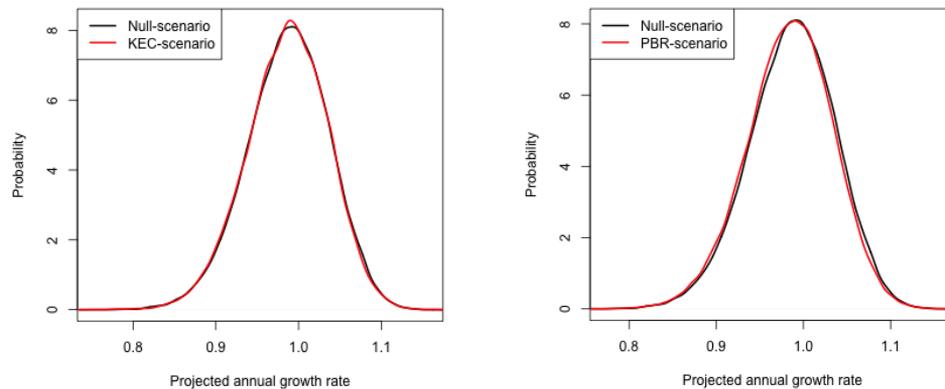
Fig. 4.4.1   *Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).*
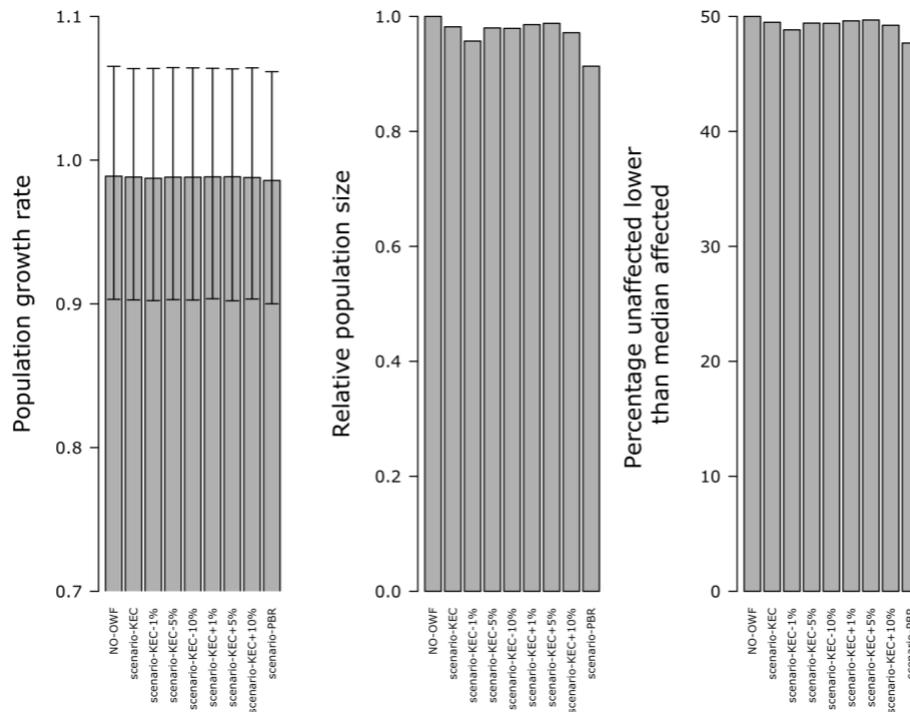
Fig. 4.4.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.4.3 and 4.4.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.

Stochastic number of collision victims

In addition to the population models with deterministic additional mortality, models with stochastic collision rates have been performed. A more detailed comparison of the results with and without stochasticity in collision rates is given in Appendix VIII. For black-legged kittiwake, variation in body length, wingspan, flight speed and flight height distribution does not strongly affect the proportion additional mortality (0.04% additional mortality for deterministic as well as stochastic CRM). No impact of stochasticity in the CRM is found on the population growth rate, relative final population size (relative to null scenario) and percentage unaffected lower than median affected.
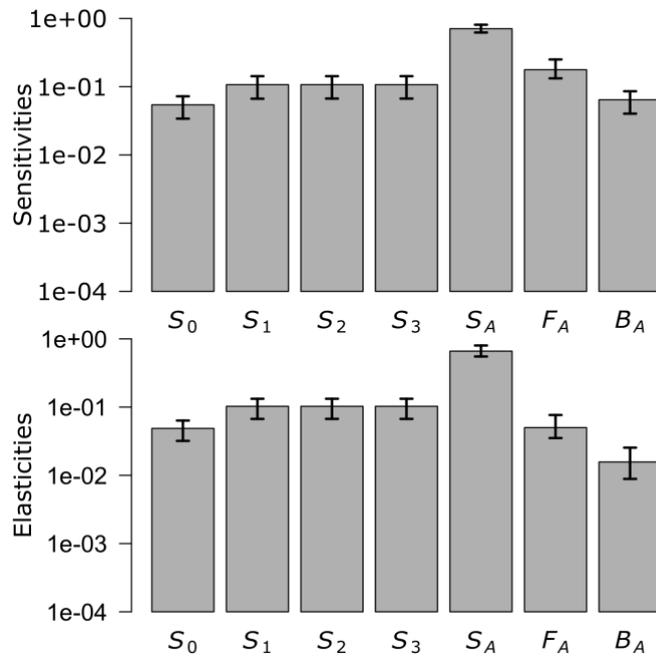
Fig. 4.4.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
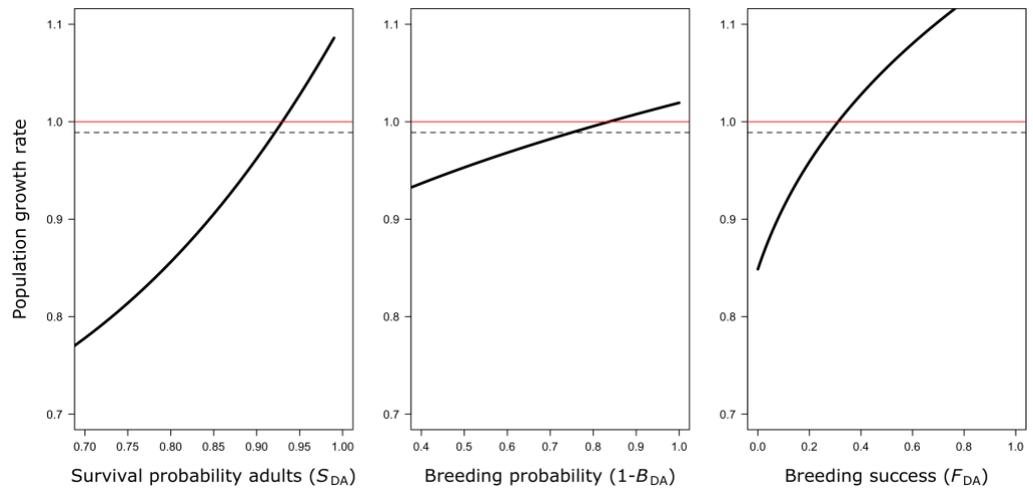
Fig. 4.4.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.5   Little gull

Due to a lack of data on little gull population dynamics, no population model could be
constructed for little gull. See Appendix VI.5 and Paragraph 3.5.

4.6 Great skua

Without additional mortality, the population model projects a slightly increasing population, with a median lambda of 1.0152 (Table 4.6, null scenario). This indicates an annual population growth rate of 1.52%. The 5th and 95th percentile of the projected lambdas are 0.9196 and 1.0924, indicating that 90% of the runs result in a trend between a decline of 8% and an increase of 9.2%. This corresponds with the observed increasing population (sovon.nl, Meetnet Watervogels).

The impact of the number of collision victims on the defined population is so small that no impact on the median population growth rate was found. The median annual growth rate of the 50,000 simulations is even somewhat higher for the impacted scenario, compared with the null scenario (see Table 4.6). This illustrates the small impact of the additional mortality on the defined population. The median final population size of the KEC scenario is similar to the median final population size of the null scenario. In contrast, the median final population size for the PBR scenario is only 62.3% of median final population size for the null scenario, and hence clearly lower. The median projected annual growth rate for the PBR-scenario is 0.9993. Hence, the projected population growth rate is 1.5 percentage point lower than for the null and KEC scenarios.

As pointed out, the projected annual growth rates of the null scenario and the impacted scenario strongly overlap (Figure 4.6.1; Figure 4.6.2; Table 4.6). The probability of finding a population growth rate below 1.0154 (median population growth rate of the KEC scenario) is for the KEC scenario as likely as for the null scenario (for both scenarios 50%).

For the null scenario as well as for the KEC scenario, 36% of the runs show a 10% or stronger decline. The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would increase to 48% of the simulations. Moreover, the median final population size for the PBR scenario is 38% lower than for the null scenario (relative final population size is 0.623; Table 4.6). This results in a clearly lower final population size than for the KEC scenario.

Table 4.6   Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0152	0.9196	1.0924	36%		
scenario KEC	1.0154	0.9201	1.0922	36%	1.008	50%
scenario KEC-1%	1.0158	0.9209	1.0926	36%	1.019	50%
scenario KEC-5%	1.0159	0.9192	1.0926	36%	1.022	51%
scenario KEC-10%	1.0156	0.9194	1.0919	36%	1.014	50%
scenario KEC+1%	1.0160	0.9207	1.0924	36%	1.026	51%
scenario KEC+5%	1.0158	0.9202	1.0923	36%	1.018	50%
scenario KEC+10%	1.0155	0.9200	1.0924	36%	1.010	50%
scenario PBR	0.9993	0.9057	1.0748	48%	0.623	38%
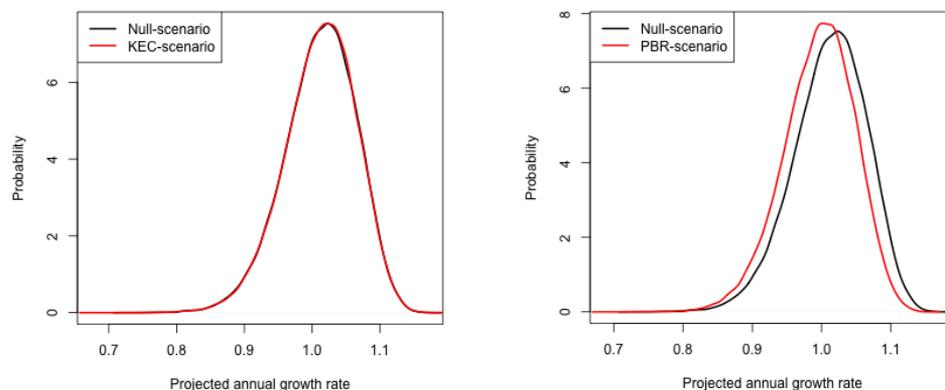
Fig. 4.6.1   Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).
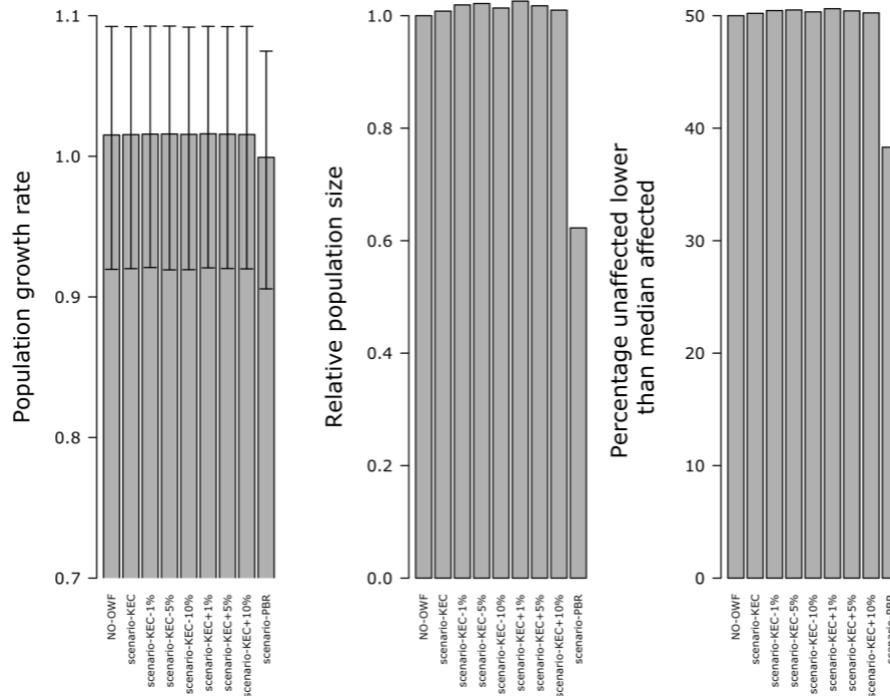
Fig. 4.6.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.6.3 and 4.6.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.

Stochastic number of collision victims

In addition to the population models with deterministic additional mortality, models with stochastic collision rates have been performed. A more detailed comparison of the results with and without stochasticity in collision rates is given in Appendix VIII. For great skua, variation in body length, wingspan and flight speed does not affect the mean proportion additional mortality (0.0043% additional mortality for stochastic CRM, versus 0.0048% additional mortality for deterministic CRM). Stochasticity in flight height distribution gives a higher estimated additional mortality (0.0186%). With the proportion of additional mortality being relatively small, the impact on the population growth rate, relative final population size (relative to null scenario) and percentage unaffected lower than median affected is limited for both the scenarios based on deterministic and stochastic CRMs.
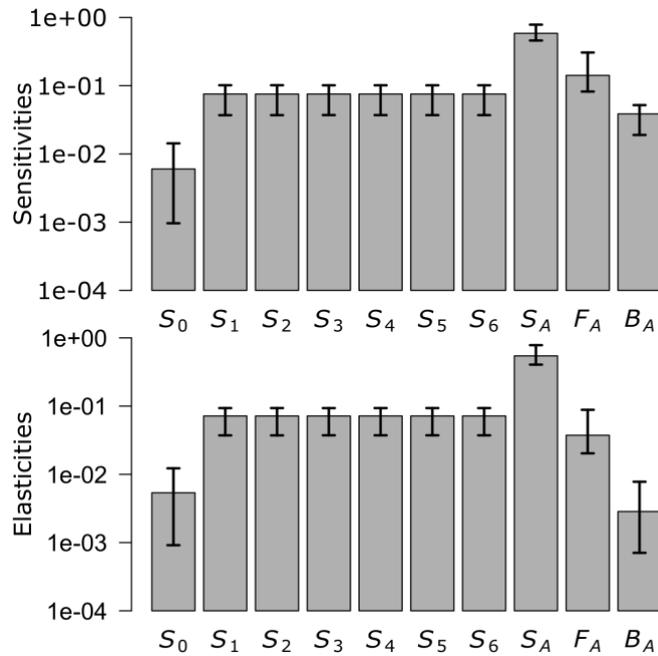
Fig. 4.6.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
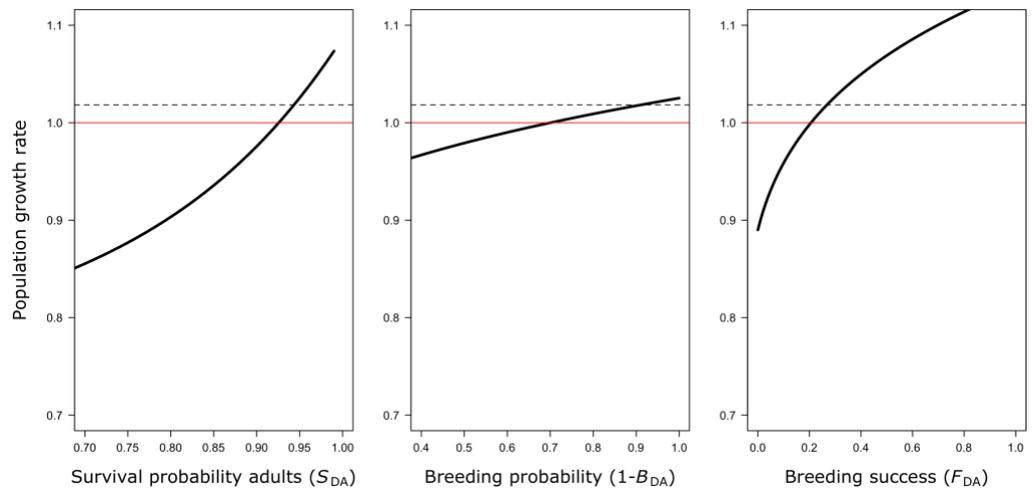
Fig. 4.6.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.7 Arctic skua

Without additional mortality, the population model projects a slightly decreasing population, with a median lambda of 0.9827 (Table 4.7, null scenario). This indicates an annual population decline of 1.73%. This corresponds with the observed population decline (sovon.nl, Meetnet Watervogels). The 5th and 95th percentile of the projected lambdas are 0.9042 and 1.0355, indicating that 90% of the runs result in a trend between a decline of 9.6% and an increase of 3.55%.

The KEC scenario projects a very similar lambda of 0.9823, indicating a median projected annual population decline of 1.77%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.9042 and 1.0354, i.e. between an annual decline of 9.6% and increase of 3.54%. Hence, the projected distribution of population growth rates of the null scenario and the KEC scenario are very similar. For the PBR-scenario, the projected annual growth rate is clearly lower with 0.9168 (8.3% annual decline).

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.7.
For the null scenario, 64% of the runs show a 10% or stronger decline. The probability of such a decline is 65% for the KEC scenario. For the PBR scenario, all iterations project a decline by at least 10% in 30 years.

There is a very strong overlap between distributions of final population sizes of the null scenario and the KEC scenario (Figure 4.7.1). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 50%. In other words, the probability of getting the median outcome of the KEC scenario is roughly the same for the null scenario and the KEC scenario.

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would strongly increase from 64% to 100%. The median final population size after 30 years would be 87.5% lower than for the null scenario (relative population size is 0.125).

Table 4.7   *Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9827	0.9042	1.0355	64%		
scenario KEC	0.9823	0.9042	1.0354	65%	0.986	50%
scenario KEC-1%	0.9818	0.9045	1.0352	65%	0.973	49%
scenario KEC-5%	0.9822	0.9041	1.0351	65%	0.984	49%
scenario KEC-10%	0.9826	0.9050	1.0358	64%	0.996	50%
scenario KEC+1%	0.9825	0.9039	1.0358	65%	0.993	50%
scenario KEC+5%	0.9823	0.9045	1.0351	64%	0.988	50%
scenario KEC+10%	0.9825	0.9048	1.0354	65%	0.994	50%
scenario PBR	0.9168	0.8436	0.9665	100%	0.125	8%
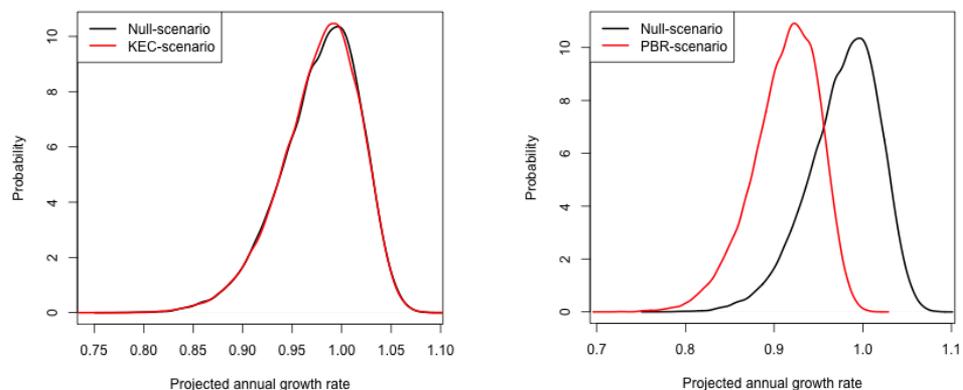
Fig. 4.7.1   *Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).*
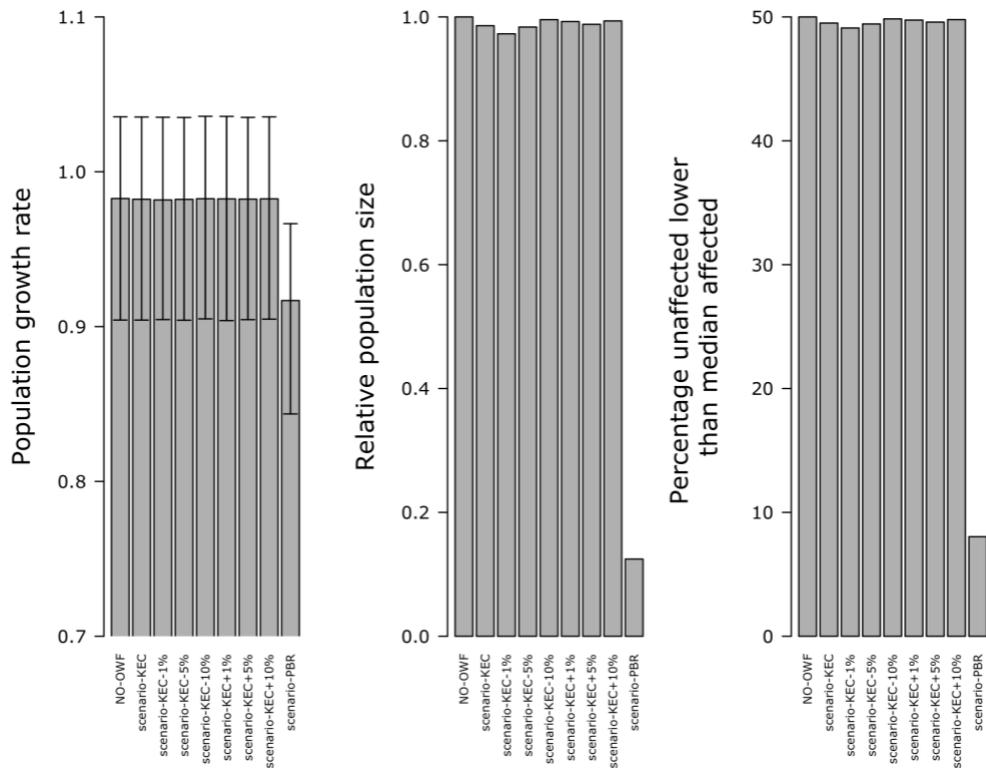
Fig. 4.7.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.7.3 and 4.7.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.
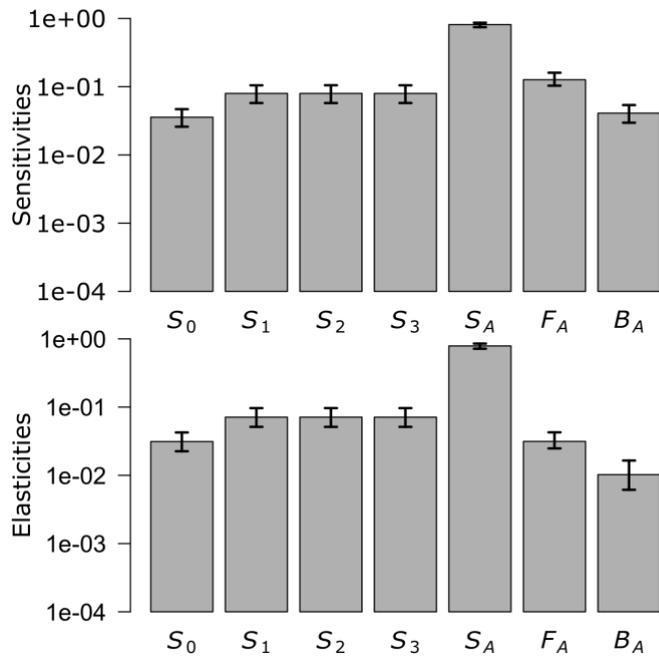
Fig. 4.7.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
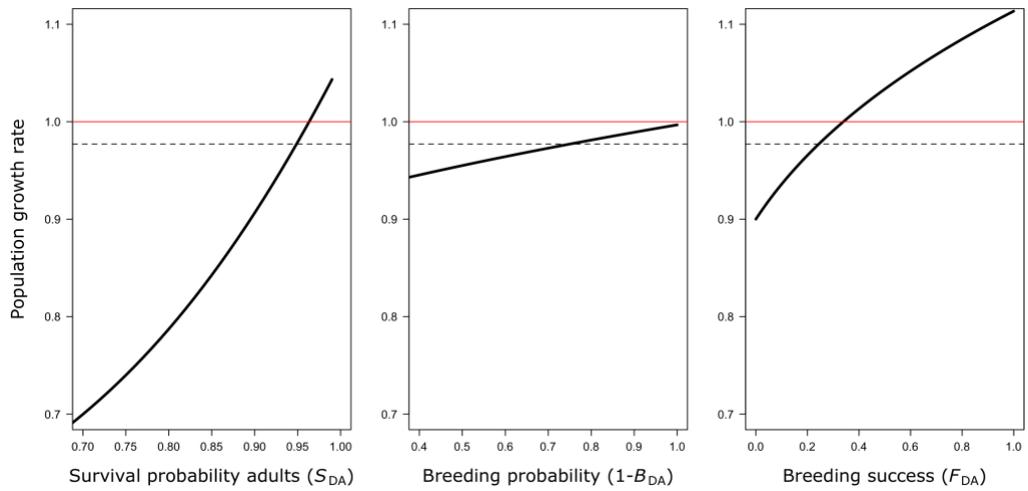
Fig. 4.7.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.8 Common tern

Without additional mortality, the population model projects a slightly decreasing population, with a median lambda of 0.9788 (Table 4.8, null scenario). This indicates an annual population decline of 2.12%. This corresponds with the observed population decline (sovon.nl, Meetnet Watervogels). The 5th and 95th percentile of the projected lambdas are 0.8913 and 1.0495, indicating that 90% of the runs result in a trend between a decline of 10.9% and an increase of 4.95%.

The KEC scenario projects a very similar lambda of 0.9780, indicating a median projected annual population decline of 2.2%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.8904 and 1.0491, i.e. between an annual decline of 11.0% and increase of 4.91%. Hence, the projected distribution of population growth rates of the null scenario and the KEC scenario are very similar. For the PBR scenario, the projected annual growth rate is clearly lower with 0.9512 (4.9% annual decline).

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.8.
For the null scenario, 65% of the runs show a 10% or stronger decline. The probability of such a decline is the same for the KEC scenario. For the PBR scenario, 85% of the iterations project a decline by at least 10% in 30 years.

There is a very strong overlap between distributions of final population sizes of the null scenario and KEC scenario (Figure 4.8.1). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 49%. In other words, the probability of getting the median outcome of the KEC scenario is roughly the same for the null scenario and the KEC scenario.

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would strongly increase from 65% to 85%. The median final population size after 30 years would be 57.6% lower than for the null scenario (relative population size is 0.424).

*Table 4.8   Summary of model output for the scenario without additional mortality (null scenario)
and the scenario with additional mortality (KEC-scenario). Numbers of victims for
KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the
population growth rate (lambda). 5% and 95% quantiles represent the value below
which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this
range). The following column shows the proportion of runs with a population decline
of 10% or more after 30 years compared to the initial population size. The relative
population size after 30 years is calculated as the final population size of the
impacted scenario divided by the final population size of the unimpacted scenario.
The last column shows the percentage of the results of the null scenario, which have
a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9788	0.8913	1.0495	65%		
scenario KEC	0.9780	0.8904	1.0491	65%	0.974	49%
scenario KEC-1%	0.9784	0.8903	1.0486	65%	0.986	50%
scenario KEC-5%	0.9785	0.8902	1.0497	65%	0.990	50%
scenario KEC-10%	0.9782	0.8909	1.0492	65%	0.979	49%
scenario KEC+1%	0.9783	0.8896	1.0485	65%	0.985	50%
scenario KEC+5%	0.9781	0.8907	1.0492	65%	0.979	49%
scenario KEC+10%	0.9782	0.8895	1.0488	65%	0.982	49%
scenario PBR	0.9512	0.8658	1.0214	85%	0.424	29%
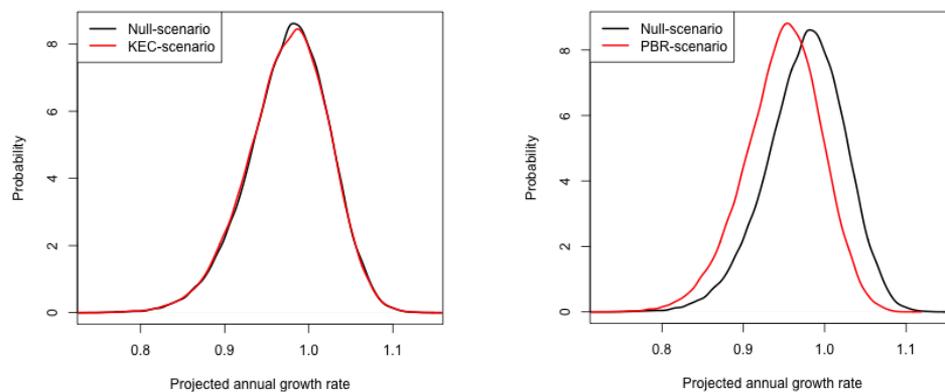
*Fig. 4.8.1   Left: Distribution of annual population growth rates for null-scenario (black line) and
KEC-scenario (red line). Right: Distribution of annual population growth rates for
null-scenario (black line) and PBR-scenario (red line).*
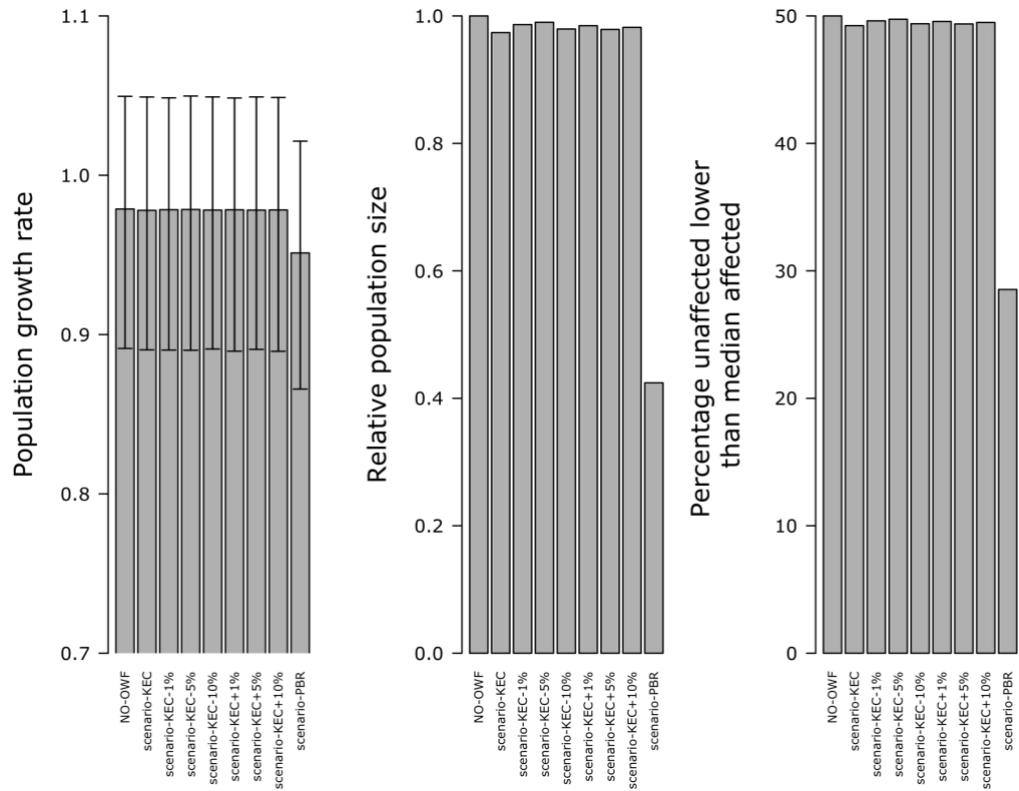
Fig. 4.8.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.8.3 and 4.8.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.
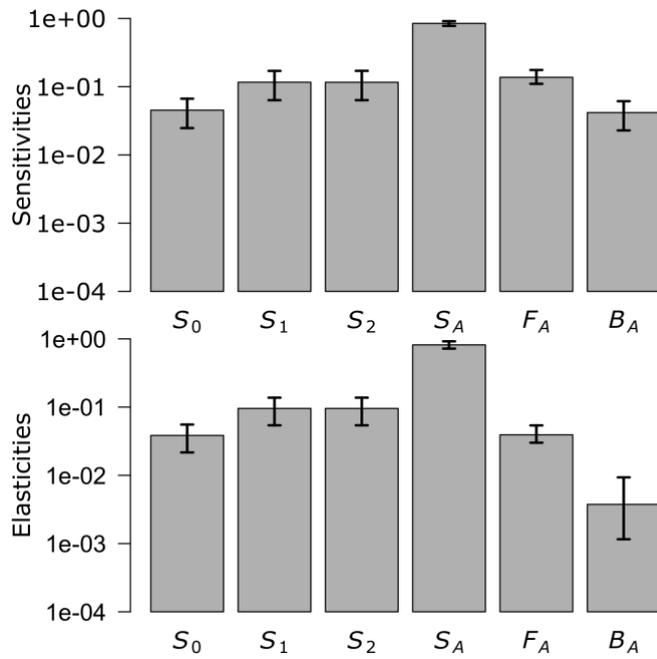
Fig. 4.8.3   Elasticity and sensitivity of model output to input parameters. S_j represents the
survival of stage j, F_A represents the fecundity and B_A the breeding probability.
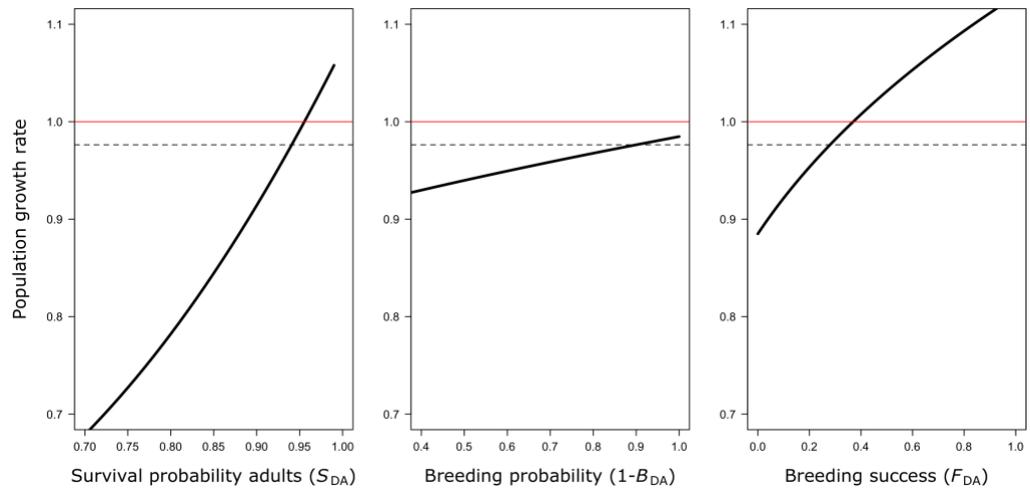
Fig. 4.8.4   Results perturbation analysis of adult survival probability, breeding probability and
breeding success. The point where the black line crosses the red line is the value
for which the population growth rate is 1.0 (stable population). The dashed line
represents the current population growth rate.

4.9 Black tern

Without additional mortality, the population model projects a stable population, with a median lambda of 1.0075, which indicates a 0.75% annual population growth. The 5th and 95th percentile of the projected lambdas are 0.9372 and 1.0660, indicating that 90% of the runs result in a trend between a decline of 6.28% and an increase of 6.07%. The observed population trend is stable, or slightly negative (sovon.nl, Meetnet Watervogels). The projected population trend corresponds with the observed trend.

The KEC scenario projects a smaller median lambda of 0.9994, indicating a median projected annual population decline of 0.06% (i.e. stable / slight decline). The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.9293 and 1.0571, i.e. between an annual decline of 7.0% and increase of 5.7%. Figure 4.9.1 shows the distribution of projected population growth rates for the null scenario and the KEC scenario. For the PBR scenario, the projected annual growth rate is similar to the KEC scenario with 0.9994. This is not surprising, as the estimated additional mortality is very close to the PBR threshold (98% of the PBR, Rijkswaterstaat 2019; see Rijkswaterstaat 2019 for calculation of PBR threshold).

The proportion of simulations for the null scenario with a decline of 10% or stronger is 39%. For the KEC scenario and PBR scenarios, 47% of the simulations show such a decline.
For both the KEC and PBR scenarios, the median final population size is 21.5% lower than for the null scenario (relative population size after 30 years is 0.785).

*Table 4.9   Summary of model output for the scenario without additional mortality (null scenario)
and the scenario with additional mortality (KEC-scenario). Numbers of victims for
KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the
population growth rate (lambda). 5% and 95% quantiles represent the value below
which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this
range). The following column shows the proportion of runs with a population decline
of 10% or more after 30 years compared to the initial population size. The relative
population size after 30 years is calculated as the final population size of the
impacted scenario divided by the final population size of the unimpacted scenario.
The last column shows the percentage of the results of the null scenario, which have
a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0075	0.9372	1.0660	39%		
scenario KEC	0.9994	0.9293	1.0571	47%	0.785	42%
scenario KEC-1%	0.9996	0.9289	1.0579	47%	0.790	42%
scenario KEC-5%	0.9999	0.9293	1.0579	47%	0.795	42%
scenario KEC-10%	1.0005	0.9301	1.0587	46%	0.810	43%
scenario KEC+1%	0.9995	0.9291	1.0572	47%	0.786	42%
scenario KEC+5%	0.9991	0.9300	1.0571	47%	0.778	42%
scenario KEC+10%	0.9986	0.9292	1.0568	48%	0.765	41%
scenario PBR	0.9994	0.9295	1.0582	47%	0.785	42%
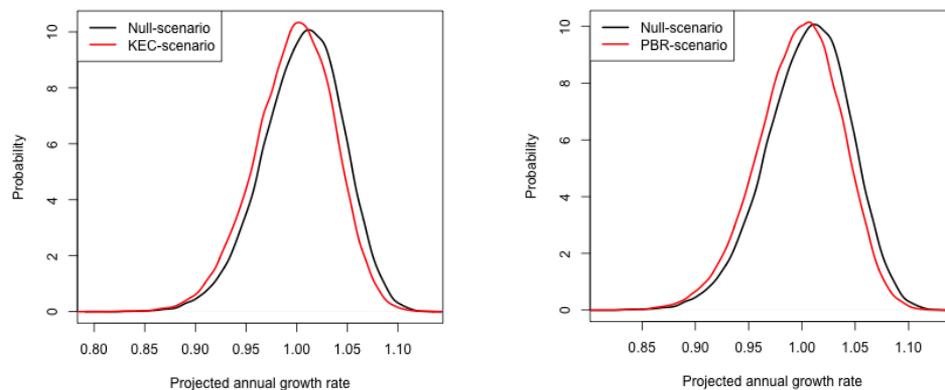
*Fig. 4.9.1   Left: Distribution of annual population growth rates for null-scenario (black line) and
KEC-scenario (red line). Right: Distribution of annual population growth rates for
null-scenario (black line) and PBR-scenario (red line).*
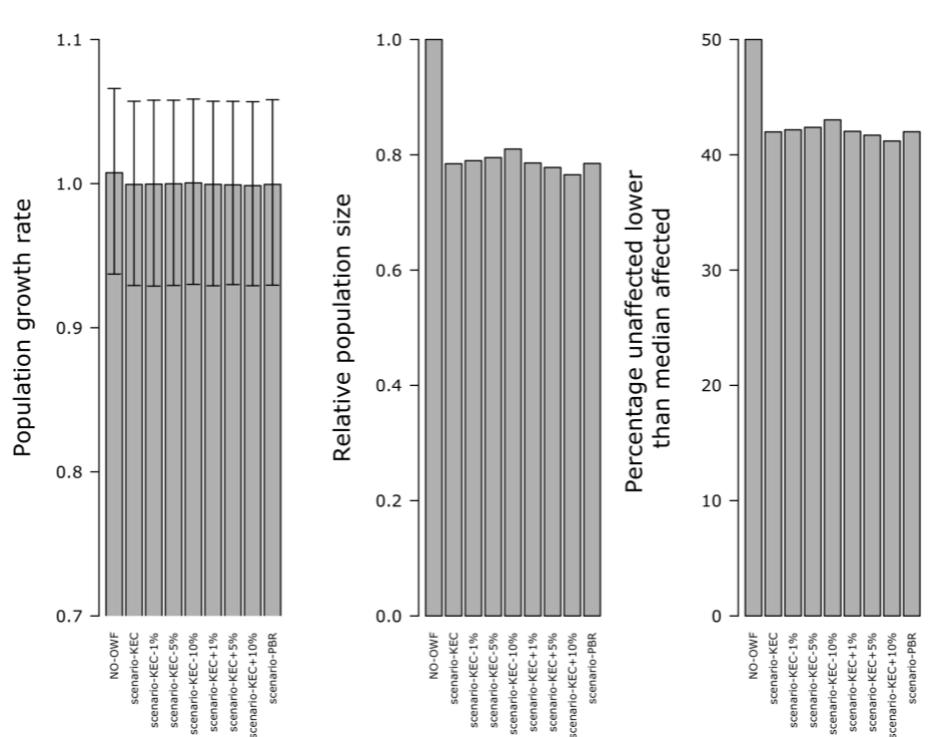
Fig. 4.9.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.9.3 and 4.9.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters. However, in comparison to other species, differences between elasticity and sensitivity of demographic rates are not very large for black tern.
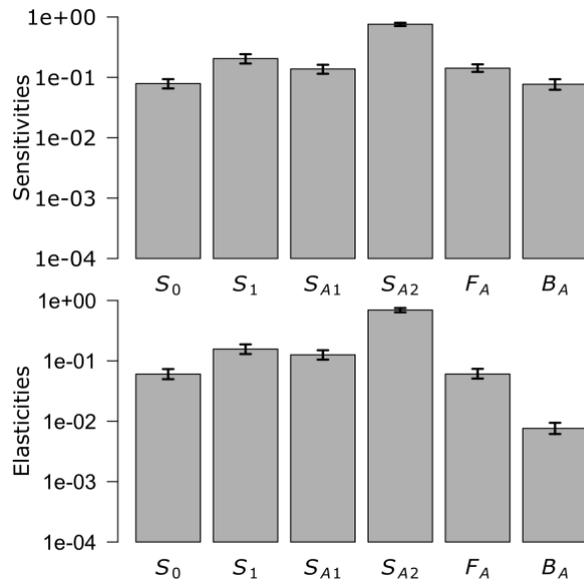
Fig. 4.9.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
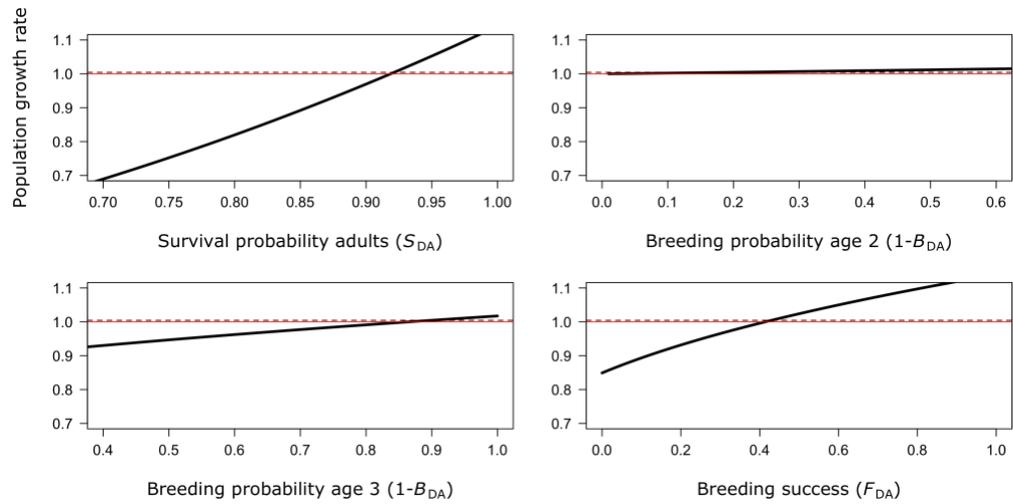
Fig. 4.9.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.10   Bewick's swan

Without additional mortality, the population model projects a declining population, with a median lambda of 0.8989. This indicates an annual population decline by 10.1%. The 5th and 95th percentile of the projected lambdas are 0.8056 and 0.9903, indicating that 90% of the runs result in a trend between an annual decline by 19.6% and an annual decline by 1%. This projected population decline corresponds with the observed population decline (Rees & Beekman 2010; sovon.nl, Meetnet Watervogels).

The KEC scenario projects a slightly lower but very similar lambda of 0.8983, indicating a median projected annual population decline of 10.2%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.8053 and 0.9896, i.e. between an annual decline by 19.5% and an annual decline by 1.0%. In comparison, the median projected population growth rate for the PBR scenario is 0.8922 (5% and 9% percentiles 0.7995 - 0.9839).

For the null scenario, 96% of the runs show a 10% or stronger decline. Due to the strong overlap between the simulations for the impacted and null scenarios, the proportion of simulations with a 10% decline after 30 years is the same for the KEC scenario (Table 4.10).

The median final population size for the KEC scenario is 97.8% of the median final population size of the null scenario (i.e. 2.2% lower than the null scenario) (Figure 4.10.2; Table 4.10).

There is a very strong overlap between distributions of final population sizes of the null scenario and KEC scenario (Figure 4.10). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 50%. The probability of finding a population growth rate below 0.8983 (median population growth rate of the KEC scenario) is for the KEC scenario as likely as for the null scenario (for both scenarios 50%).

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would only slightly increase from 96% to 97%. The median final population size after 30 years would be 20% lower than the median final population size for the null scenario. This results in a clearly lower final population size than for the KEC scenario.

Table 4.10 Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.8989	0.8056	0.9903	96%		
scenario KEC	0.8983	0.8053	0.9896	96%	0.978	50%
scenario KEC-1%	0.8984	0.8056	0.9897	96%	0.982	50%
scenario KEC-5%	0.8983	0.8060	0.9895	96%	0.980	50%
scenario KEC-10%	0.8988	0.8064	0.9906	96%	0.995	50%
scenario KEC+1%	0.8978	0.8051	0.9897	96%	0.963	49%
scenario KEC+5%	0.8978	0.8060	0.9898	96%	0.962	49%
scenario KEC+10%	0.8980	0.8061	0.9899	96%	0.969	49%
scenario PBR	0.8922	0.7995	0.9839	97%	0.799	45%
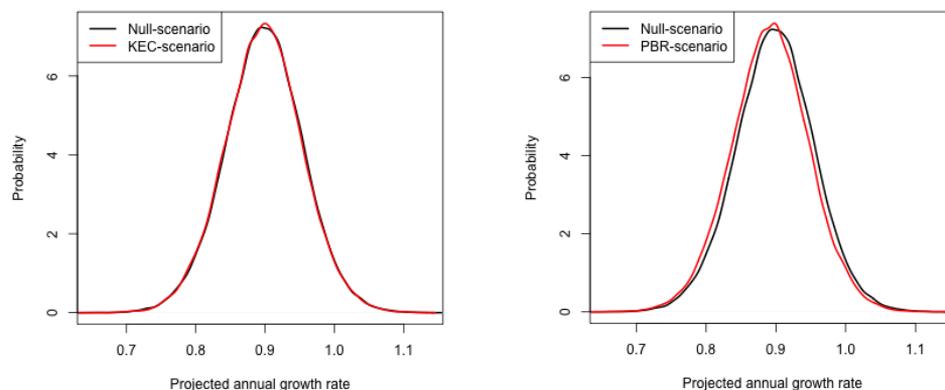
Fig. 4.10.1 Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).
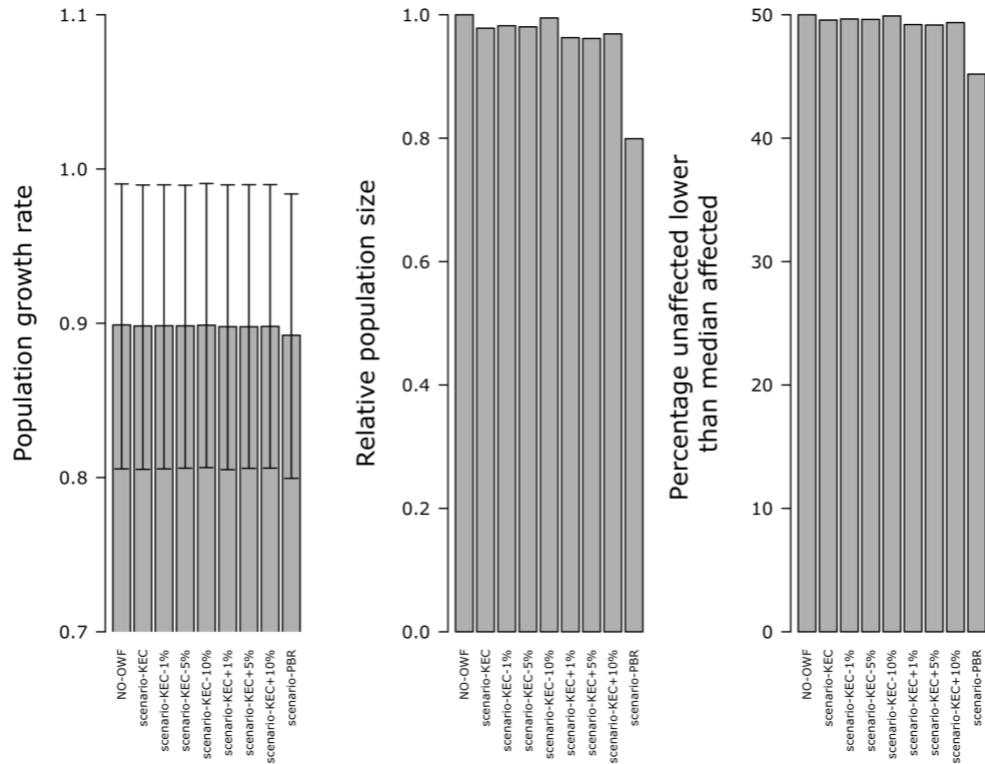
Fig. 4.10.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity or sub-adult survival (Figure 4.10.3 and 4.10.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.
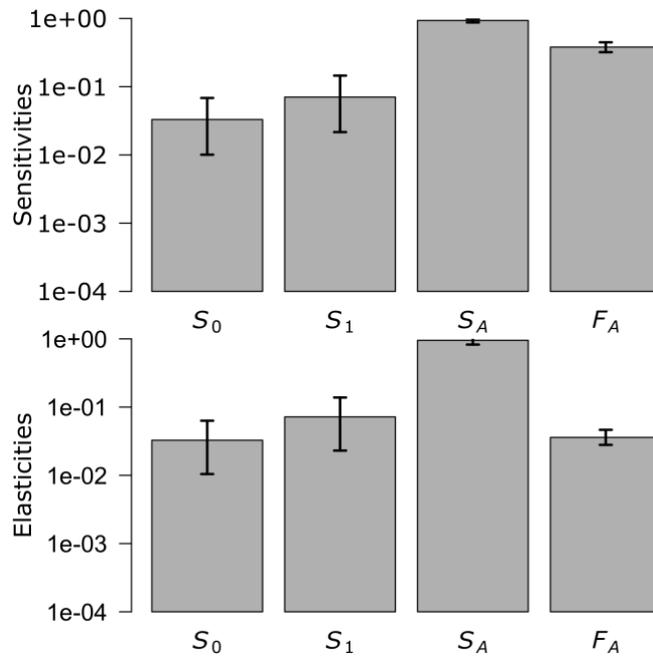
Fig. 4.10.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j and F_A represents the fecundity.
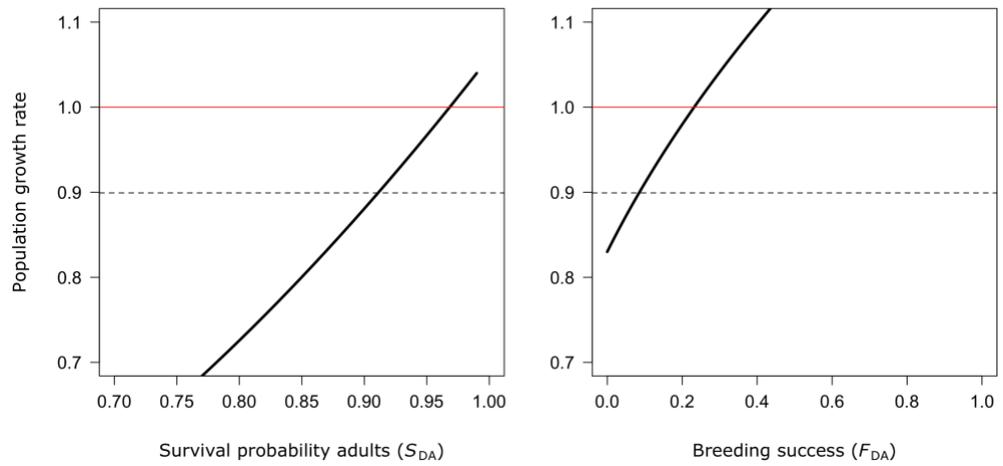
Fig. 4.10.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.11 Brent goose

Without additional mortality, the population model projects a stable population, with a median lambda of 1.0031. This indicates a median annual population growth of 0.29%. The 5th and 95th percentile of the projected lambdas are 0.9422 and 1.0628, indicating that 90% of the runs result in a trend between a decline of 5.8% and an increase of 6.3%. This corresponds with the observed stable population (sovon.nl, Meetnet Watervogels).

The KEC scenario projects a very similar lambda of 1.0023, indicating a median projected annual population growth of 0.23%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.9415 and 1.0623, i.e. between an annual decline of 5.85% and increase of 6.2%. The median projected population growth rate of the PBR-scenario is 0.9640 (annual decline by 3.6%), hence clearly lower.

For the null scenario, 43% of the runs show a 10% or stronger decline. For the scenario with additional mortality based on KEC 3.0 (Rijkswaterstaat 2019), such a decline is projected in 44% of the runs.

The median final population size for the KEC scenario is 97.7% of the median final population size of the null scenario (Figure 4.11.2; Table 4.11). The similarity in population projections for the null and the KEC scenarios can be seen in the strong overlap between distributions of final population sizes (Figure 4.11.1). The probability of finding a population growth rate below 1.0031 (median population growth rate of the KEC scenario) is for the KEC scenario as likely as for the null scenario (for both scenarios 50%).

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would considerably increase from 43% of the simulations for the null scenario to 82% of the simulations for the PBR scenario. The median final population size for the PBR scenario is 70% lower than for the null scenario (relative final population size is 0.303; Table 4.11). This results in a clearly lower final population size than for the KEC scenario. Moreover, only 15% of the runs without additional mortality result in a final population size at or below the median of the PBR scenario.

Table 4.11 *Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0031	0.9422	1.0628	43%		
scenario KEC	1.0023	0.9415	1.0623	44%	0.977	49%
scenario KEC-1%	1.0018	0.9413	1.0615	44%	0.962	49%
scenario KEC-5%	1.0022	0.9412	1.0623	44%	0.973	49%
scenario KEC-10%	1.0024	0.9414	1.0624	43%	0.980	49%
scenario KEC+1%	1.0025	0.9412	1.0623	43%	0.982	49%
scenario KEC+5%	1.0026	0.9408	1.0618	43%	0.984	49%
scenario KEC+10%	1.0022	0.9415	1.0613	44%	0.974	49%
scenario PBR	0.9640	0.9048	1.0215	82%	0.303	15%
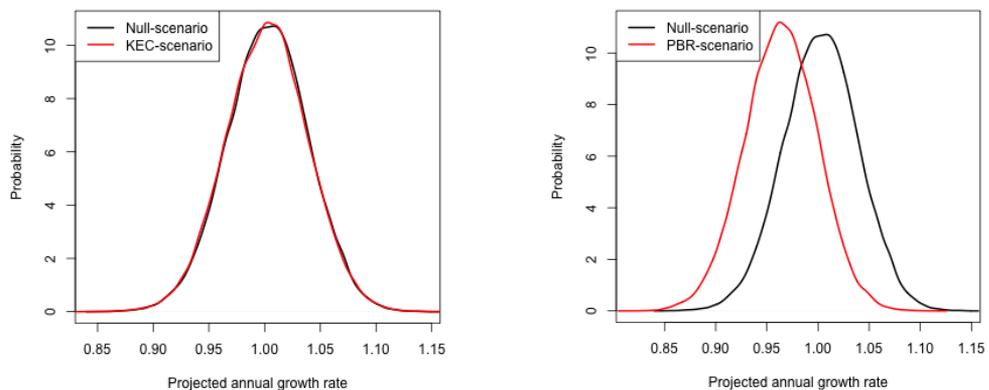
Fig. 4.11.1 Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).
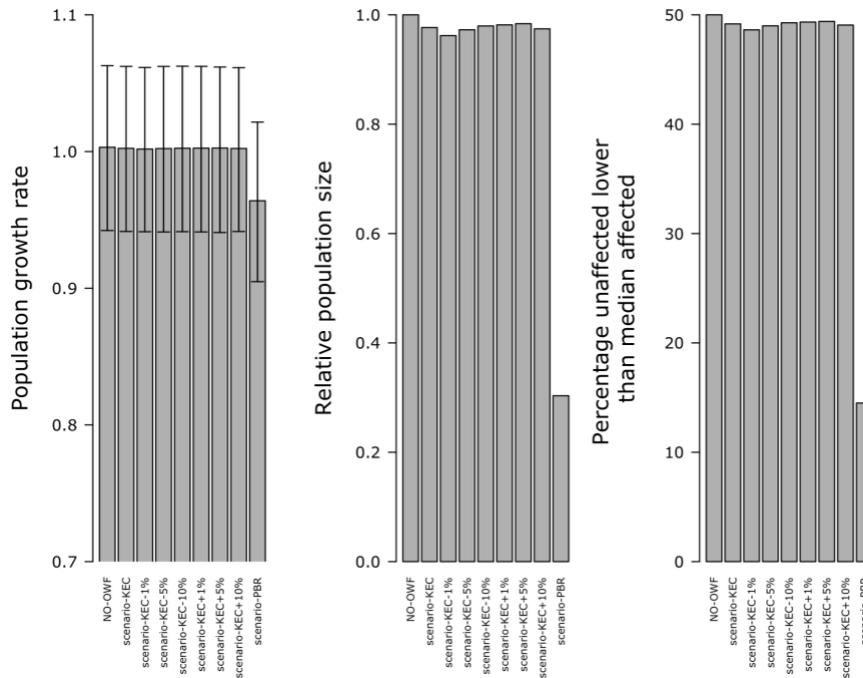
Fig. 4.11.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation shows that a relative change in adult survival has a stronger impact than changes in fecundity or sub-adult survival (Figure 4.11.3 and 4.11.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.
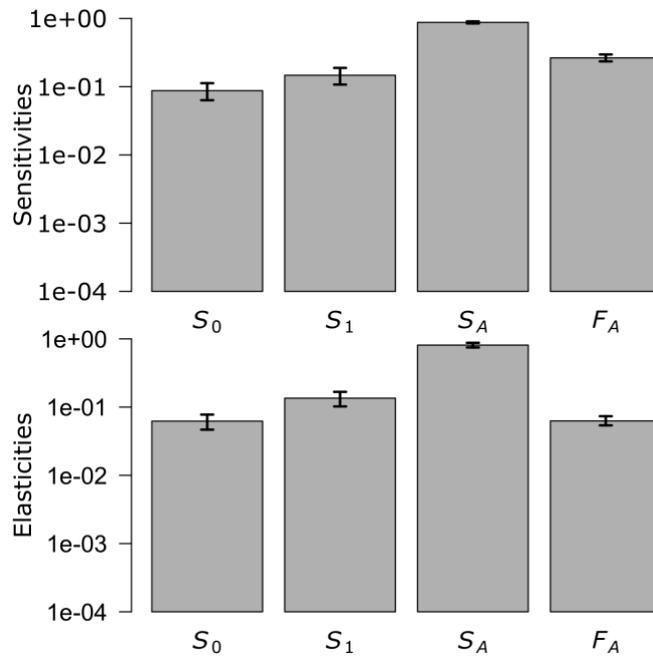
Fig. 4.11.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
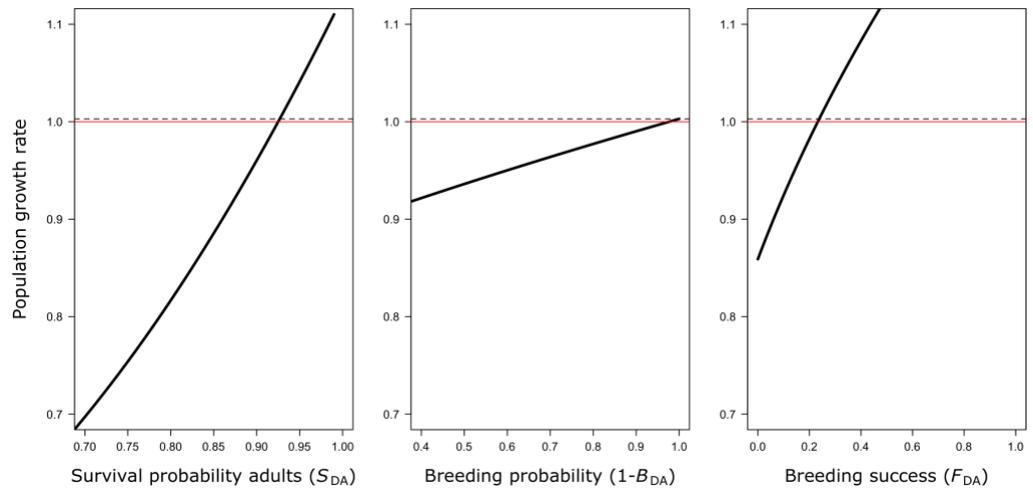
Fig. 4.11.4   Results perturbation analysis of adult survival probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.12   Common shelduck

Without additional mortality, the population model projects a stable population, with a median lambda of 1.0268. This indicates a median annual population growth rate of 2.68%. The 5th and 95th percentile of the projected lambdas are 0.9422 and 1.0881, indicating that 90% of the runs result in a trend between a decline of 5.8% and an increase of 8.8%. This corresponds with the observed population increase (sovon.nl, Meetnet Watervogels).

In contrast, the KEC scenario projects a slightly lower lambda of 1.0202, indicating a median projected annual population increase by 2.0%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.937 and 1.081, i.e. between an annual decline by 6.3% and increase by 8.1%. The PBR scenario projects a median population growth rate of 0.9603 (3.97% annual decline).

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.12.
For the null scenario, 26% of the runs show a 10% or stronger decline. For the scenario with additional mortality based on KEC 3.0 (Rijkswaterstaat 2019), such a decline is projected in 30% of the runs.

The median final population size is for the KEC scenario is 82.3% of the median final population size of the null scenario (17.7% lower than the null scenario) (Figure 4.12.2; Table 4.12).

There is a strong overlap between distributions of final population sizes of the null scenario and KEC scenario (Figure 4.12.1). The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 44%. In other words, without additional mortality the probability of getting the median outcome of the KEC scenario is 44%.

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would increase from 26% of the simulations for the null scenario to 83% of the simulations for the PBR scenario. Moreover, the median final population size of the PBR scenario is 86.6% lower than for the null scenario (relative final population size is 0.134; Table 4.12). This results in a clearly lower final population size than for the KEC scenario.

Table 4.12 *Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0268	0.9422	1.0881	26%		
scenario KEC	1.0202	0.9371	1.0810	30%	0.823	44%
scenario KEC-1%	1.0198	0.9365	1.0814	31%	0.813	44%
scenario KEC-5%	1.0202	0.9371	1.0813	30%	0.822	44%
scenario KEC-10%	1.0206	0.9370	1.0820	30%	0.833	44%
scenario KEC+1%	1.0199	0.9367	1.0814	30%	0.817	44%
scenario KEC+5%	1.0200	0.9369	1.0808	30%	0.819	44%
scenario KEC+10%	1.0192	0.9361	1.0798	31%	0.799	43%
scenario PBR	0.9603	0.8816	1.0175	83%	0.134	9%
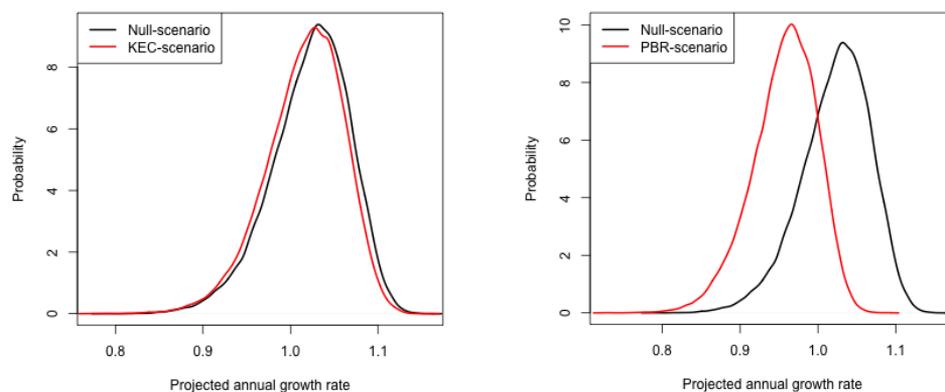
Fig. 4.12.1 *Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).*
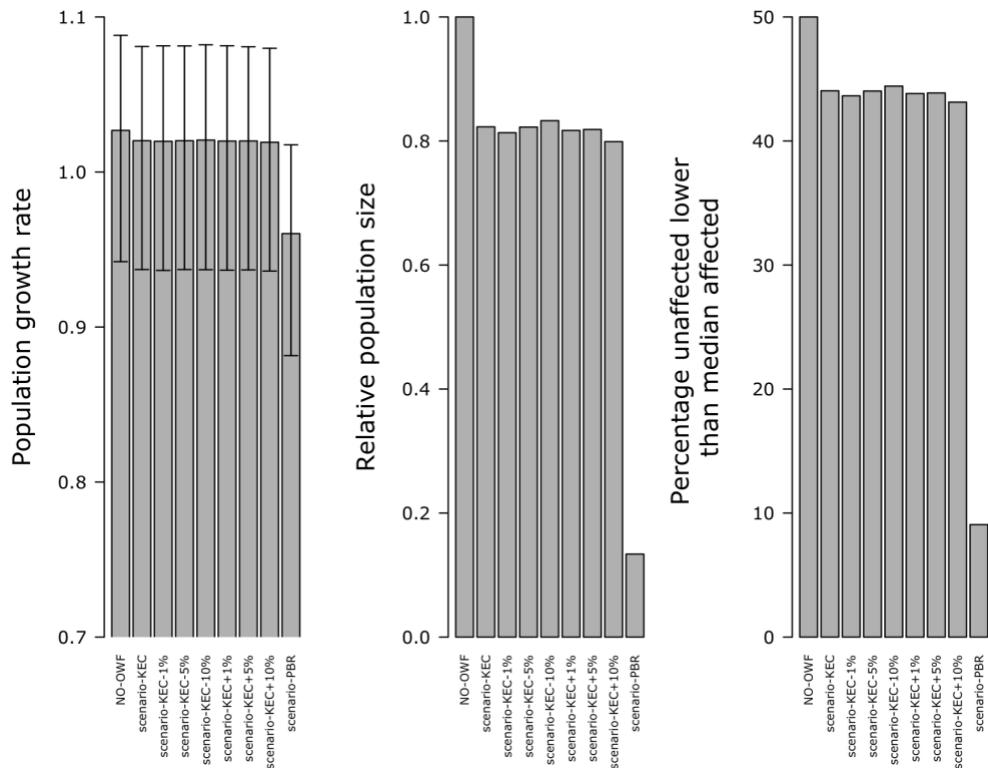
Fig. 4.12.2 Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation shows that a relative change in adult survival has a stronger impact than changes in fecundity or sub-adult survival (Figure 4.12.3 and 4.12.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters. However, in comparison to other species, differences between elasticity and sensitivity of demographic rates are not very large for common shelduck.
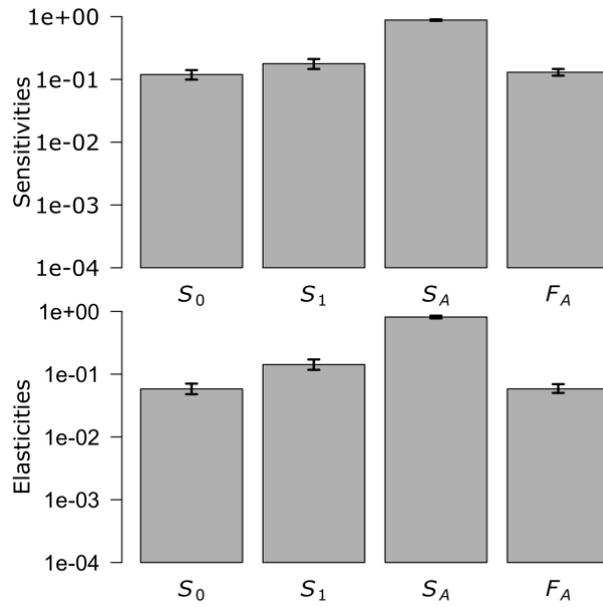
Fig. 4.12.3 Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j and F_A represents the fecundity.
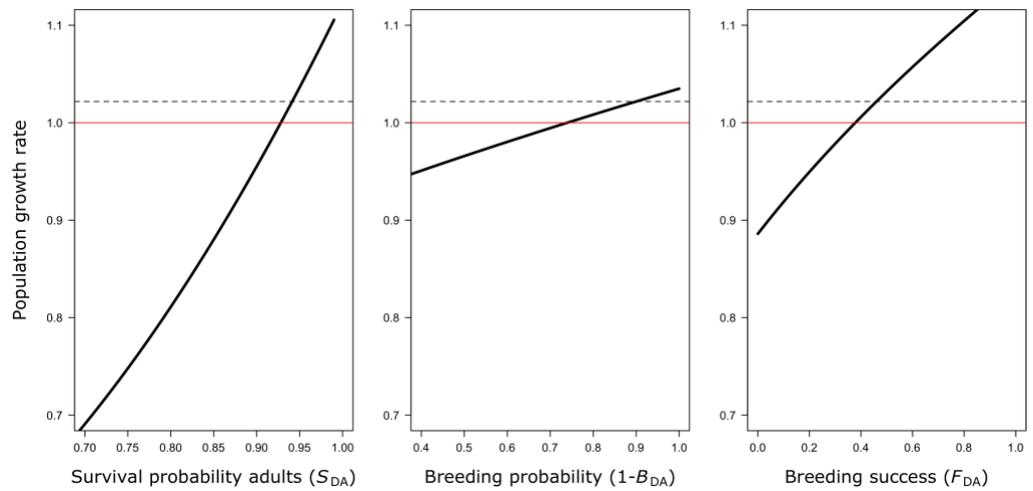
Fig. 4.12.4 Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

4.13   Eurasian curlew

Without additional mortality, the population model projects a declining population, with a median lambda of 0.9183, which indicates a median population decline of 8.17%. The 5th and 95th percentile of the projected lambdas are 0.8348 and 0.9879, indicating that 90% of the runs result in a trend between a decline of 16.5% and a decline of 1.2%. This corresponds with the observed population decline (Delany *et al.* 2009; sovon.nl, Meetnet Watervogels).

The KEC scenario projects a lambda of 0.9098, indicating a median projected annual population decline of 9.0%. The 5th and 95th percentile indicate that 90% of the runs result in a lambda between 0.8270 and 0.9791, i.e. between an annual decline of 17.3% and a decline of 2.1%. The PBR scenario projects a median population growth rate of 0.9056.

The impact of the additional mortality is assessed by comparing the output of each scenario with additional mortality with the output of the null scenario (without additional mortality). This is summarized in Table 4.13.
For the null scenario, 97% of the runs show a 10% or stronger decline. Such a decline is found for the KEC scenario in 98% of the runs, and for the PBR scenario in 99% of the runs.

The median final population size is for the KEC scenario is 75.8% of the median final population size of the null scenario (i.e 24.2% lower than the null scenario) (Figure 4.13.2; Table 4.13).

There is a relatively strong overlap between distributions of final population sizes of the null scenario and KEC scenario. The proportion of the results of the null scenario with a lower population growth rate than the median growth rate of the KEC scenario is 43%. In other words, without additional mortality the probability of getting the median outcome of the KEC scenario is 43%.

The PBR scenario shows that if the number of victims equals the PBR, the probability of a decline of 10% or more in 30 years would remain similar to the null and KECscenarios (99% vs. 98% vs. 97%). However, the median final population size after 30 years would be 34.2% lower than the median final population size for the null scenario (relative final population size is 0.658; Table 4.13). This results in a clearly lower median final population size than for the KEC scenario.

Table 4.13   Summary of model output for the scenario without additional mortality (null scenario) and the scenario with additional mortality (KEC-scenario). Numbers of victims for KEC-scenario and PBR-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found (i.e. 90% of the outcomes fall within this range). The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9183	0.8348	0.9879	97%		
scenario KEC	0.9098	0.8270	0.9791	98%	0.758	43%
scenario KEC-1%	0.9099	0.8272	0.9786	98%	0.758	43%
scenario KEC-5%	0.9106	0.8267	0.9798	98%	0.776	43%
scenario KEC-10%	0.9115	0.8280	0.9797	98%	0.800	44%
scenario KEC+1%	0.9099	0.8273	0.9788	98%	0.759	43%
scenario KEC+5%	0.9099	0.8260	0.9791	98%	0.758	43%
scenario KEC+10%	0.9090	0.8266	0.9776	98%	0.737	42%
scenario PBR	0.9056	0.8224	0.9743	99%	0.658	40%
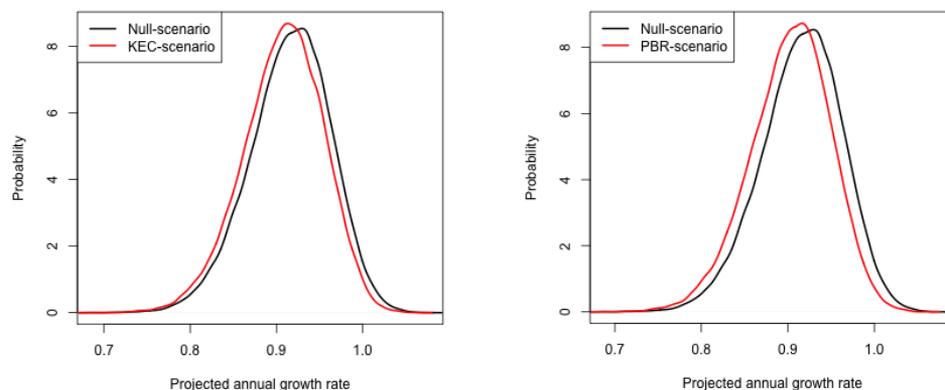
Fig. 4.13.1   Left: Distribution of annual population growth rates for null-scenario (black line) and KEC-scenario (red line). Right: Distribution of annual population growth rates for null-scenario (black line) and PBR-scenario (red line).
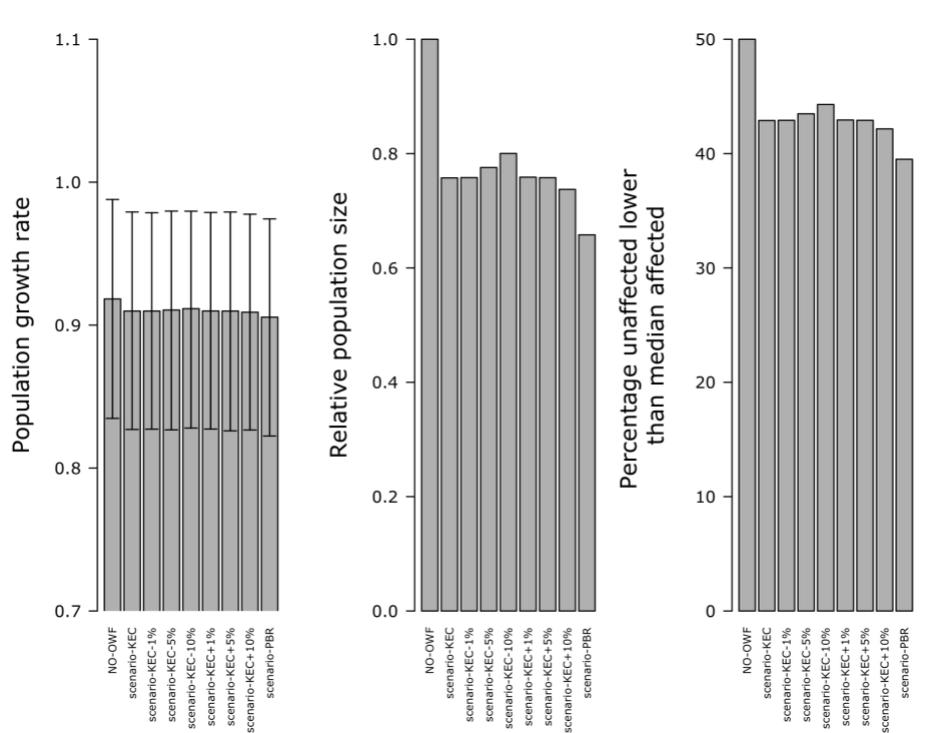
Fig. 4.13.2   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Perturbation analysis

The perturbation analysis shows that a relative change in adult survival has a stronger impact than changes in fecundity, sub-adult survival or breeding probability (Figure 4.13.3 and 4.13.4). These results show that a 1% change in adult survival has a stronger impact on the outcome of the population model than a 1% change in the other parameters. This also indicates that the impact of a measurement error is also stronger for adult survival than for other parameters.
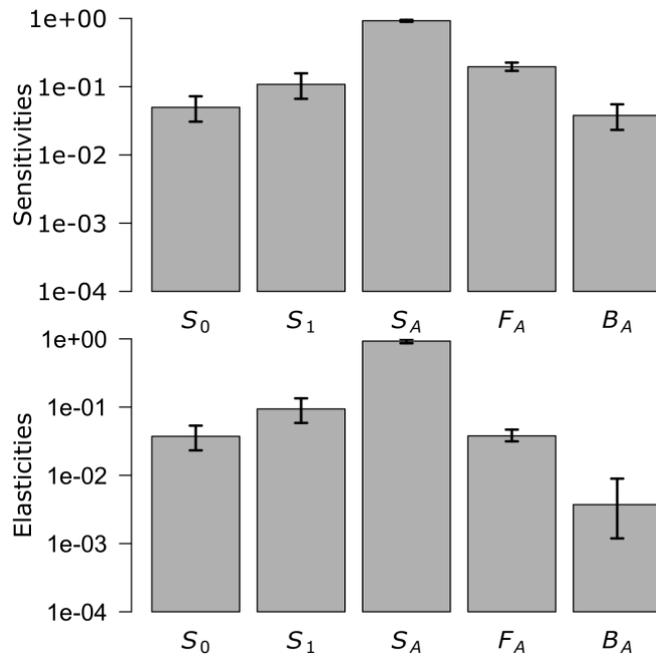
Fig. 4.13.3   Elasticity and sensitivity of model output to input parameters. S_j represents the survival of stage j, F_A represents the fecundity and B_A the breeding probability.
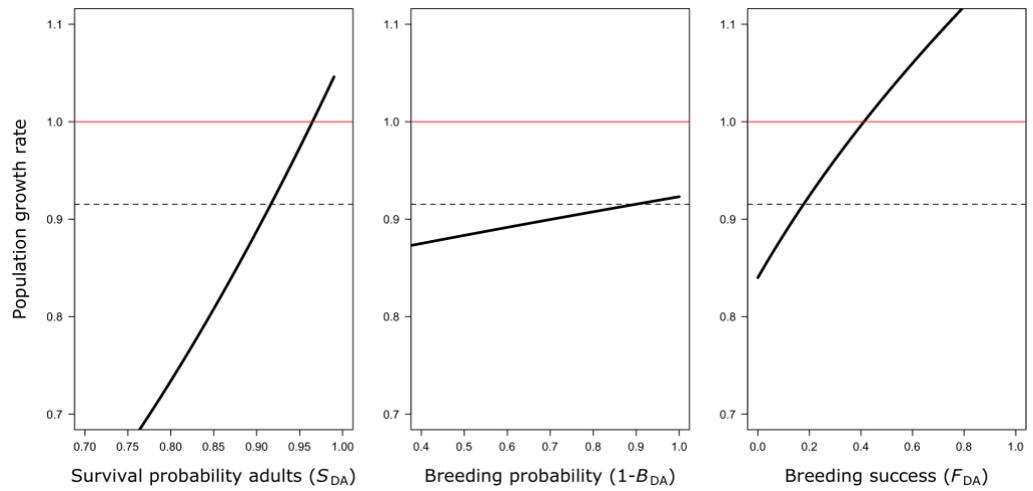
Fig. 4.13.4   Results perturbation analysis of adult survival probability, breeding probability and breeding success. The point where the black line crosses the red line is the value for which the population growth rate is 1.0 (stable population). The dashed line represents the current population growth rate.

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5 Other anthropogenic sources of mortality

In addition to collisions with wind turbines, other anthropogenic sources of mortality can impact the bird populations studied here. As part of the project we aimed to place the mortality due to turbine collisions in perspective to other sources of anthropogenic mortality. In other words, how does the mortality due to collisions with wind turbines relate to the mortality due to other sources of anthropogenic mortality?

First it is important to note that human activity may result either directly or indirectly in bird mortality. Here, we focused on direct mortality only and indirect mortality from through processes such as habitat loss, food depletion and climate change are not taken into account since these are very difficult to quantify and dependent on a large number of variables and factors. Although the assessment of the impact of indirect mortality was not among the aims of this study, it should be noted that these processes may have a strong impact.

Loss *et al.* (2015) compared sources of *direct* anthropogenic mortality based on a review on data from the USA, and concluded that the total number of bird victims due to turbine collisions seems lower than due to other anthropogenic sources. It should be noted that in cumulation with all the other sources of direct anthropogenic mortality, the additional mortality caused by collisions with wind turbines the total impact on population level might exceed the legally accepted thresholds.

The relative impact of different sources of anthropogenic mortality strongly differs between species (groups). Predation by domestic cats for example, which is an important source of anthropogenic mortality as pointed out by Loss *et al.* (2015), is unlikely to play a large role for the species selected in this study although note that potential high additional mortality can occur when (feral) cats obtain access to gull colonies (Towns *et al.* 2011). The species selected in this study may be impacted by collisions with vehicles, buildings and man-made structures, or poisoning. We reviewed available data on mortality due to these factors for the selected species, mainly based on ring recoveries.

Results are presented in Appendix IX. This gives an impression of the expected impact of this source of anthropogenic mortality for each species (group). However, it is concluded that obtaining comparable data on estimated additional mortality as a result of each anthropogenic source is difficult and at the moment not feasible. This is mainly due to the following reasons:

- studies often focus on the impact of one specific anthropogenic source. Due to the lack of a standardized approach, it is hard to compare between sources of mortality based on different studies. Moreover, the relative impact of different sources of anthropogenic mortality is location-specific.
- reporting rates can vary greatly among different sources of mortality. For example, the probabilities of finding a victim, knowing the cause of death and reporting this correctly differs greatly between sources of anthropogenic mortality. Certain causes of mortality may be more obvious whereas some

causes may not be apparent to the reporter. For example, a bird found along a road will frequently be reported as a traffic victim, whereas for a dead bird found elsewhere the cause of death may not be initially apparent.

- the majority of victims remain unreported.

For these reasons, it is (at the moment) not possible to assess per species (group) how many victims are expected from each source of anthropogenic mortality. Therefore it is currently not feasible to incorporate other sources of mortality into our population models.In order to be able to compare levels of additional mortality due to different sources of anthropogenic mortality, standardized studies are necessary. As pointed out, the main problem is the difference in detection probability of victims from different sources of additional mortality. It may be possible to get insight in the occurrence of non-natural mortality is through (colour-)ringing of birds. For a few bird species (particularly large gulls, geese and swans) datasets are probably large enough to get a quantitative insight in non-natural mortality sources and the incident rates of the various sources. However, datasets of most birds are likely too small to do any quantitative analyses. Another way to obtain sources of mortality based on birds with data loggers. However, relatively large numbers of individuals need to be equipped with data loggers to obtain reliable estimates of additional mortality due to different anthropogenic sources.

6 Discussion and conclusions

In this report population models are used for the cumulative assessment of population-level impact of additional mortality due to turbine collisions in the North Sea. One of the main aims of this project was to find out whether impact assessment can be done using population models, instead of the PBR method, which is currently being used. Although the PBR method gives an indication of the maximum level of additional mortality, population models give a more detailed insight in dynamic consequences of mortality and more reliable results (O'Brien *et al.* 2017). Population models can be used if input data of sufficient quality are available. Moreover, these input data should be representative for the study population. We conclude based on the results of this study that for 12 out of the 13 selected species (all except for little gull), sufficient data were available to create and use population models for the impact assessment.

Summarized (in Table 6.1) we found that for most species (7/12), the inclusion of additional mortality based on the KEC-scenario results in a final population size within 5% of the final population size projected without additional mortality (first column is 0.95 or higher). The assessed impact on the other five species is stronger. The difference in median outcome of the population size between impacted and unimpacted scenarios is between 5 and 10% for great black-backed gull, between 15 and 20% for lesser black-backed gull and common shelduck, and over 20% for black tern and curlew.

Table 6.1   *Summary of results of the population models. Impacted scenario refers to the KEC-scenario.*

	Median relative population size after 30 years ($N_{30\_impacted}$ / N_{30_null})	Probability of 10% decline within 30 years	
		null scenario	impacted scenario
Lesser black-backed gull	0.83	42	49
Great black-backed gull	0.93	57	58
Herring gull	0.95	62	63
Kittiwake	0.98	56	57
Great skua	1	36	36
Arctic skua	0.99	64	65
Common tern	0.97	65	65
Black tern	0.79	39	47
Bewick's swan	0.98	96	96
Brent goose	0.98	43	44
Common shelduck	0.82	26	30
Curlew	0.76	97	98

A different way to assess impact is to determine the probability of a certain decline. For illustrative purposes, we also present the probability of a 10% decline within 30 years (Table 6.1). Note that for some species, the probability of a 10% decline is already

relatively high for the null scenario. In other words, even without additional mortality, such a decline is likely. Comparison of the probability of a 10% decline for the null scenario and the impacted scenario shows that the increased probability of such a decline is relatively small in most cases, with some exceptions.

The PBR threshold is often used as maximum acceptable additional mortality. The PBR approach has certain clear drawbacks, such as e.g. that it provides a fixed and very static figure that does not take any environmental variability into account. Moreover, it implicitly assumes a fixed level of undemonstrated density dependence in population development (O'Brien *et al.* 2017). Within this study, we also simulated the population trend with additional mortality according to the PBR threshold (PBR-scenario). This gives an impression of the impact of this amount of additional mortality. In general, the PBR-scenarios show a clearly stronger impact than the scenarios according to KEC 1.0 and 3.0 (Rijkswaterstaat 2015; 2019). For black tern, the outcome of the PBR-scenario is very similar to the outcome of the KEC-scenario. This is not surprising, with the estimated additional mortality based on KEC 3.0 being 98% of the PBR threshold. PBR thresholds are determined in KEC 1.0 and KEC 3.0 (Rijkswaterstaat 2015; 2019), and are based on the minimal population size, maximum growth rate and a recovery factor (*rf*) (Wade 1998). Within the KEC framework (Rijkswaterstaat 2015; 2019), the recovery factor is assumed to depend on the IUCN criteria (IUCN 2018). Note that the outcome of the PBR calculation strongly depends on the recovery factor. Within these calculations, uncertainty was not taken into account.

Most impact assessments focus on a single wind farm. In contrast, we assessed the cumulative impact of all planned wind farms in the North Sea. This type of cumulative impact assessment is more realistic, and therefore recommended (Masden *et al.* 2010; Drewitt & Langston, 2006). Note that drawing conclusions on which level of mortality or population effect is considered to be 'acceptable' was not among the aims of this study, and depends on legal decisions.

6.1 Assumptions and limitations

In this study we used Leslie matrix models that project numbers of individuals per stage based on stage-specific survival and fecundity. The main assumption of this method is that the population is closed (i.e. changes in population trend are caused by changes in survival and/or fecundity). Given the large scale of the population studied, the impact of immigration and emigration plays a smaller role. This is because for example movements from Texel to IJmuiden are not seen as emigration, as individuals stay within the same 'population' (North Sea).

Demographic rates can be independent of the population density, but may also be density-dependent (Dhondt *et al.* 1992; Sillet *et al.* 2004). For example, stronger competition at higher population densities may result in reduced survival or fecundity. Although functions of density dependence can be incorporated into the model, the

nature and strength to which density dependence can influence demographic rates and population dynamics are largely unknown for many bird populations.

Although positive density dependence can occur, in most cases negative density dependence is reported. For the species discussed in this report, if density dependence plays a role, it is likely to be negative. In case of negative density dependence, the population growth is limited at higher density. That means that theoretically the population size of an increasing population increases up to a certain level, referred to as the 'carrying capacity'. At that level, the population size remains relatively constant. This level is defined by the resource availability in the area (food, breeding habitat, etc.). In a declining population, negative density dependence results in the growth rate becoming less negative as the population declines. In contrast, without density dependence, the growth rate is not limited at higher densities.

However, data availability to investigate density dependence is often limited, as many other factors affect the population size. This makes the choice of the carrying capacity rather arbitrary. This choice strongly influences the outcome of the population model. Incorporating density dependence would require making a number of assumptions that cannot be supported by data and bring unknown variability into the models.

6.2 Impact of other population definitions

Within this study, the impact assessment was focused on the North Sea. Input parameters of the population model (stage-specific survival and fecundity, and additional mortality in the impacted scenarios) are aimed to be representative for this population definition.

If a different population definition is used, the outcome of the impact assessment may be different due to local differences in demographic rates, and different fractions of collision victims. Note that it is important to know how many of the expected collision victims originate from this defined 'population'. For example, if the impact of bird collisions with offshore wind turbines in the North Sea on the Dutch breeding population of lesser black-backed gull is to be assessed, it should be taken into account that not all of the victims in the southern North Sea stem from this defined population (i.e. Dutch breeding population). Note that although the impact of victims in the North Sea is assessed, individuals from the Dutch breeding population may collide with wind farms outside this particular area.

Due to these differences in fraction of collision mortality and possible differences in demographic rates between populations (or population definitions), the use of a different population definition will yield different results of the population models.

6.3 Impact of data quality and representativeness

Within this study, data quality and representativeness is assessed for each of the input parameters of the species-specific population models. Assignment of scores for quality and representativeness are based on Horswill & Robinson (2015).

Table 6.2 presents an overview of the quality and representativeness of the available data for each species. The main knowledge gaps are survival rates of various species. High quality data on survival are absent for little gull and limited for all age classes for shelduck and curlew. In addition, data availability on juvenile survival is poor for great black-backed gull, brent goose, great skua and kittiwake. For kittiwake, great skua and great black-backed gull, data on immature survival are also lacking.

Calculating survival rates for these species can be done based on ring recoveries. These data need to be collected from the various ringing programmes across Europe. For most species substantial numbers of birds have been ringed (except for species such as little gull), but the problem with most seabirds is that recovery rates are fairly low due to the fact that most birds die at sea and are not found. Specifically juveniles and immatures are particularly sea-bound and are generally not resighted in the colonies, hence the lack of good quality survival data of these age-classes. However, for most species survival analyses of ringing data will yield better estimates of survival rates specifically for adult survival.

*Table 6.2   Data availability and quality of species of interest. Data availability for each species is classified as 'good' when at least one data source with a high score for quality and representativeness is available. See paragraph 2.1 for description of data quality and representativeness. * data available, but still needs to be analysed.*

Species	Survival	Fecundity	Origin of birds in North Sea	Distribution of age classes in North Sea
Lesser black-backed gull	Good	Good	Good	Good
Great black-backed gull	Juveniles: Poor Immatures: Poor Adults: Good	Good	Poor*	Poor*
Herring gull	Good	Good	Good	Good
Kittiwake	Juveniles: Poor Immatures: Poor Adults: Good	Good	Good	Poor
LIttle gull	Juveniles: Poor Immatures: Poor Adults: Poor	Poor	Poor	Poor
Great skua	Juveniles: Poor Immatures: Poor Adults: Moderate	Good	Good	Poor
Arctic skua	Juveniles: Poor Immatures: Poor Adults: Moderate	Good	Poor	Poor
Common tern	Juveniles: Poor Immatures: Poor Adults: Good	Good	Good	Moderate
Black tern	Moderate; indirect estimates	Good	Moderate	Moderate
Bewick's Swan	Good	Good	Good	Good
Brent goose	Juveniles: Poor Immatures: Good Adults: Good	Good	Good	Good
Shelduck	Poor	Moderate / good	Poor	Good
Curlew	Poor	Moderate	Poor	Good

6.4   Impact and improvement of calculation of collision victims

For each of the study species, the numbers of collision victims are based on Rijkswaterstaat (2019). For seabirds, the species-specific numbers of victims are calculated from bird densities as determined by aerial and ship-based surveys. Within the calculation of the numbers of collision victims, it is implicitly assumed that each bimonthly period all individuals are replaced by new individuals. This can be made more

realistic by assuming a species-specific estimated residence time. This residence time can be estimated by GPS data and/or ring recoveries (if sufficient data are available).

For a selection of species (lesser black-backed gull, great black-backed gull, herring gull, kittiwake, great skua and gannet), we performed additional analyses using a recently developed stochastic colllision risk model (sCRM, Marine Scotland 2018) in which variation in input parameters results in a range around the estimated number of victims. Note that within these analyses, only flight height, flight speed, body length and wingspan are varied. Variation in bird density is expected to have a relatively strong impact on the outcome of the collision risk model. For that reason, including variation in bird density would strongly improve the estimated range of the number of collision victims. In addition, such an analysis is also recommended for the other study species. This gives insight into the range of expected numbers of victims.

The results of the sCRM show that a change in flight height distribution can strongly affect the estimated number of collision victims. Introducing stochasticity in flight heights is done by sampling flight heights from multiple data sources of altitude distribution rather than choosing only one figure for flight height. This can have a profound influence on the number of collision victims (see f.e. the different consequences of introducing stochasticity in flight height in lesser and great black-backed gull (§4.1 and 4.2)). Hence, knowledge of flight height distribution is important to get a reliable estimate of the number of victims. This knowledge is in particular necessary for great black-backed gull, curlew and black tern, as for these species the estimated number of victims is relatively high, and no empirical data on flight height distribution are available. For great black-backed gull and curlew, flight height data have been collected during dedicated GPS-logger studies in Germany, but these data are not yet publicly available. Analysis of available data for these species is recommended, as a better flight height distribution would strongly improve the estimated number of collision victims. Data on flight height distribution for black tern are not yet available. Data collection and analysis of flight height distribution are recommended for black tern. Unfortunately GPS or air-pressure loggers are currently not small enough to study these parameters in black terns. However, it is possible to collect this type of data with field studies, by using a laser rangefinder to determine flight height.

6.5 Recommendations for improving quality of impact assessment

Improved quality of input data will result in more reliable impact assessments. Sensitivity analyses for each species show that the selected species are most sensitive to changes in adult survival. This also means that uncertainty in adult survival most strongly affects the reliability of the outcome of the population model. For that reason, additional research is recommended to fill knowledge gaps for in particular (adult) survival estimates.

For a few species, additional data are potentially available, but still need to be analysed. This is for example the case for survival of great black-backed gulls, and survival of herring gull and lesser black-backed gull from the Dutch delta, but also survival of brent geese.

For other species, existing survival data are only sparsely available. For example shelduck and curlew are relatively understudied and ringing data are not analysed in detail yet.

6.6 Conclusion and follow-up

This report shows that for 12 of the 13 study species, it is possible to assess the impact of additional mortality due to collisions of birds with wind turbines using population models. Several metrics are presented which can be used for an impact assessment. In this report we do not assess any of these effects with respect to the current European legislation nor do we provide any suggestions on which level of additional mortality is legally acceptable. Our results only draw conclusions based on a certain proposed threshold, but the final judgement on which threshold(s) should be used is up to the Competent Authority.

Examples of such thresholds can be:
- *"The probability of a 10% decline (from the current population size) within 30 years may not exceed XX%."* This compares the current population size with the population size after 30 years of additional mortality.
- *"The population size after 30 years with additional mortality should be at least XX% of the population size after 30 years without additional mortality."* This compares the outcome of the scenario with additional mortality with the scenario without additional mortality.
- Alternatively, the probability distribution of the null scenario and impacted scenario can be compared. For example, one can assess the probability of reaching the outcome of the impacted scenario in a situation without additional mortality. This would give an indication of how much the outcomes of both scenarios overlap. An example of such a threshold can be: "The probability of reaching the median outcome of the impacted scenario if there is no additional mortality, should not be lower than XX%." This takes into account the variation in outcome of both the null scenario and the impacted scenario.

Possible thresholds are also described in among others Cook & Robinson (2017) and May *et al.* (2019). Within the results section, we present results in such a way that they can be used for legislation. After the government has decided which thresholds should be used, these population models can be used to assess whether the expected impact results in violation of the law.

7 Literature

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Appendices

I Immigration and emigration in population models

When using matrix population models, it is generally assumed that populations are closed (Caswell 2001). In other words, immigration and emigration are assumed to either not occur, or to effectively be equal. This means that if a population model shows a growing population, this growth is caused by survival rates or fecundity, and not by net immigration into the population. Demographic data, however, are collected from a 'study population', which is often not a closed 'biological' population. This means that the assumption of a closed population is not always realistic. For that reason, we have to ask ourselves the following questions:

How do immigration and emigration rates vary between subpopulations and within subpopulations?

What is the potential impact of the assumption of a 'closed' population on the outcome of the population model, and on the assessed impact of additional mortality?

Variation between subpopulations: Source vs. sink populations
A population in which mortality is compensated for by net immigration is not sustainable. This can be illustrated with the concept of different subpopulations within the same total population (generally referred to as metapopulation), differing in immigration and emigration rates, and in demographic rates. For example, due to low quality habitat (for example poor prey availability), individuals in subpopulation 1 may have low survival and fecundity, which would without immigration result in a declining subpopulation. However, due to net immigration from elsewhere into this subpopulation, the size of this subpopulation may remain stable. This type of subpopulation is referred to as a 'sink' (Pulliam 1988). Such subpopulations are not sustainable and have a negative effect on the total metapopulation. On the other hand, a subpopulation in which the number of emigrating individuals exceeds the number of immigrants is called a 'source' population. A source population has a positive effect on the metapopulation.

For that reason, even though a certain subpopulation shows a stable population size, it is important to understand whether this population size is maintained by demographic rates, or whether this subpopulation may act as a sink. In absence of data on immigration and emigration, this can be assessed by comparing the observed population growth rate with the expected growth rate given the demographic rates. The expected growth rate based on current demographic rates indicates how sustainable the population is, without immigration or emigration.

Subpopulation 1:
Sink

Subpopulation 2:
Source

Low fecundity
Low survival
→ Negative expected growth rate without immigration or emigration

But net immigration increases observed growth rate

High fecundity
High survival
→ Positive expected growth rate without immigration or emigration

But net emigration results in lower observed growth rate
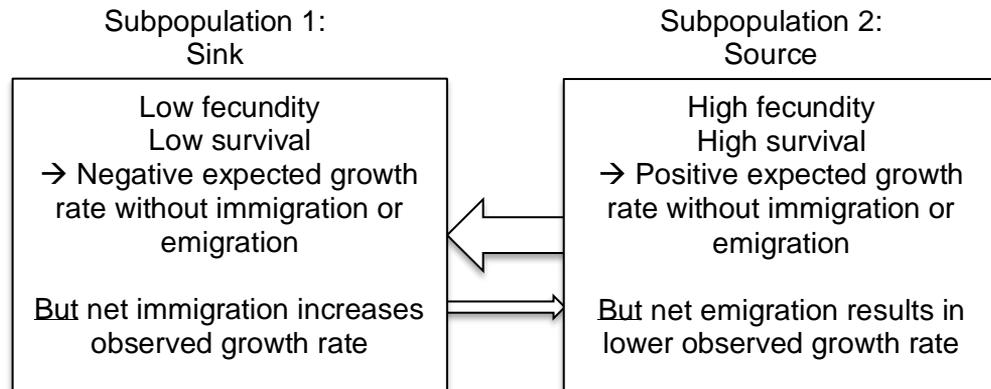
Figure I.1   Example of two subpopulations illustrating source-sink dynamics.

Variation in immigration and emigration within a subpopulation

Even when looking at one specific subpopulation, the extent of immigration and emigration may differ between years. This strongly depends on how neighbouring populations are developing. In addition, immigration may be density-dependent. In case of a subpopulation for which the population size is restricted by available nesting habitat, the death of established breeders presents an opportunity for immigrants to take over this nesting location. Such a pattern has been found for American redstart (*Setophaga ruticilla*) (Wilson *et al.* 2017). As a result of this negative density-dependent immigration, it is possible that additional mortality does not affect the population trend of this subpopulation, because increased immigration compensates for collision mortality. However, as these immigrants stem from neighbouring populations, these populations may be affected.

Implications for the assessment of the impact of additional mortality

The aim of this project is not to analyse whether a (sub)population is a source or sink, but instead to assess the sustainability of that (sub)population with and without additional mortality due to collisions between birds and turbines. Whether a population is sustainable (by itself) is determined by the (age-specific) fecundity and survival, and this is analysed in absence of immigration and/or emigration. In the example illustrated in Figure I.1, both the sink and the source population may be stable. However, the sink population is not sustainable 'by itself' based on the fecundity and survival; in contrast, it is depending upon the inflow of other subpopulations ('sources') within the same metapopulation. Without this input from other subpopulations, the population would decline.

To get an indication of how sustainable a population is 'by itself', we studied the expected population growth rates based on current (age-specific) fecundity and survival. This gives an indication of how the growth rate of the (sub)population of interest, without net inflow from other (sub)populations of the same metapopulation (net immigration), or net outflow (net emigration). In other words, immigration and emigration are not incorporated into the models. Assuming no net immigration or emigration is not

only more feasible, due to lacking information on immigration and emigration, but in our case also more suitable for answering the research question.

Moreover, as we study the population dynamics on a large scale (southern North Sea), immigration and emigration play a smaller role. This is because for example movements from Texel to IJmuiden are not seen as emigration, as individuals stay within the same 'population' (southern North Sea).

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II Density dependence

Demographic rates can be independent of the population density, but may also be density-dependent (Dhondt *et al.* 1992; Sillet *et al.* 2004). For example, stronger competition at higher population densities may result in reduced survival or fecundity. Although functions of density dependence can be incorporated into the model, the nature and strength to which density dependence can influence demographic rates and population dynamics is largely unknown for many bird populations.

Although positive density dependence can occur, in most cases negative density dependence is reported. For the species discussed in this report, if density dependence plays a role, it is likely to be negative. In case of negative density dependence, the population growth is limited at higher density. That means that theoretically the population size of an increasing population increases up to a certain level, referred to as the 'carrying capacity' (Y and Z in Figure II.1). At that level, the population size remains relatively constant. This level is defined by the resource availability in the area (food, breeding habitat, etc.). In a declining population, negative density dependence results in the growth rate becoming less negative as the population declines. In contrast, without density dependence, the growth rate is not limited at higher densities.
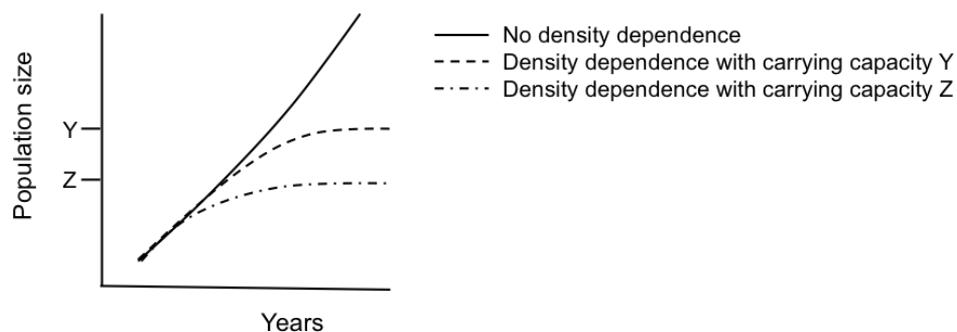
Figure II.1   Exemplary population trend with and without density dependence.

The population models used in this study do not incorporate density dependence of input parameters. Instead we assume that survival and fecundity do not vary with changing population size, unless sufficient data are available which indicate a significant effect of density dependence. For most species, data on the effect of density dependence on these parameters are, at best, limited. Incorporating density dependence would require making a number of assumptions that cannot be supported by data and bring unknown variability into the models. See text box for more information and possible consequences for the model outcome.

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What if density dependence plays a role, but we do not account for it?

Ideally, the strength of density dependence can be analysed using long time series for which population density or population size as well as demographic rates are measured. However, the strength of density dependence is relatively difficult to measure, and varies between subpopulations and between years. This is partly due to the effect of density independent factors such as weather, which mask the effect of density dependence. Moreover, the strength of density dependence depends on how close a subpopulation is to its carrying capacity. As shown in Figure II.1 in this Appendix, the effect of density dependence becomes more clear at densities closer to carrying capacity.

If information on density dependence is not available, we assume no density dependence. As pointed out in the text, incorporation of density dependence requires an arbitrarily chosen level of carrying capacity, which strongly influences the outcome of the population model. If density dependence does play a role, but is not accounted for in the population models, the impact assessment can be seen as a worst case scenario. At population sizes close to carrying capacity, survival and/or fecundity would be reduced if density dependence plays a role. Additional mortality would in this case be, at least partly, compensatory. For example, when density dependent survival but not fecundity plays a role, a certain number of individuals would have died without wind turbines due to competition at higher densities. Mortality due to turbine collisions reduces competition, and acts therefore in a compensatory way. This means that up to a certain level, which is the number of individuals which would die due to increased competition, the population sizes would not decline as a result of additional mortality, but would stay around carrying capacity.

Note that this impact of density dependence is theoretical and simplified. The existence of a carrying capacity is heavily discussed in science. It seems more realistic that the carrying capacity fluctuates between years, depending on for example environmental factors. It should be noted that in a population model without density dependence, and assuming average demographic rates do not change over time, a population which increases based on initial demographic rates will increase to infinity. Similarly, a declining population will decline to extinction. One should therefore realize that this projection is based on current knowledge, and changes in the future environment can not be known and can therefore not be modelled.

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III Environmental variation versus measurement error

Population sizes as well as demographic rates vary in time. These fluctuations often seem random, also called stochastic. Three basic forms of stochasticity can be distinguished (Lande *et al.* 2003):

- <u>Measurement error</u> in estimates of population size, density or demographic rates. Usually, not all individuals within a certain area can be counted. Therefore, the estimated population size or density is often based upon the sampling of a smaller area (Seber 1982). This results in measurement error, which explains part of the fluctuations in demographic rates and population size or density (Lande *et al.* 2003).
- <u>Demographic stochasticity</u>. By chance, independent of environmental effects, demographic rates vary between years. With an average adult survival rate of 90%, it may be that by chance the survival is 91% in one year, and 88% in the following. This type of variation is purely random, and cannot be explained by any external factors (Lande *et al.* 2003).
- <u>Environmental stochasticity</u>. Variation in environmental conditions affects demographic rates and influences the fluctuation of population size or density. Environmental stochasticity is caused by variation in abiotic factors such as temperature or precipitation and biotic factors such as predator density or prey availability (Fujiwara & Takada 2009). If environmental factors (strongly) affecting demographic rates are studied, it is possible to account for variation due to these factors. However, although sometimes some of the main factors affecting demographic rates are known, usually not all environmental stochasticity is explained.

To account for demographic variation, stochasticity can be incorporated in matrix models. This means that instead of using the same survival and fecundity rates each year, these values vary around a mean. The magnitude of demographic variation is specified by the standard deviation or any other parameter of a suitable probability distribution. This is illustrated in Figure 2.5. In this case, the average survival is 0.9 (i.e. 90%), and the standard deviation is 0.02 (i.e. 2%). In most years, the survival will be around 0.9. In 95% of the years, the value will be between 2 standard deviations of the mean, as illustrated with the dashed lines in Figure III.1. In other words, in this example, the survival rate will be between 0.86 and 0.94 for 95% of the years. Hence, stochasticity is incorporated by varying demographic rates between years within a run.
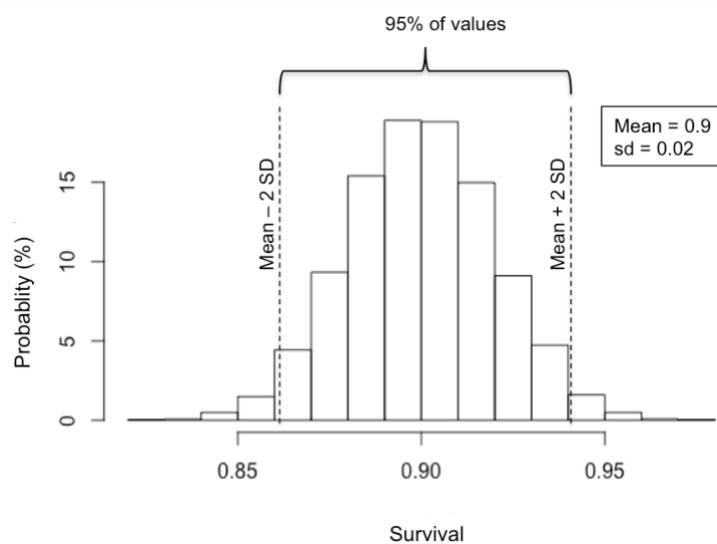
Figure III.1   Illustration of stochasticity, i.e. variation between years. The shown distribution has a mean of 0.9 and a standard deviation of 0.02; hence, 95% of the values are between 0.86 (mean – 2 SD) and 0.94 (mean + 2 SD).

Measurement error

Measurement error can also be incorporated into the population models. This measurement error gives an indication of to what extent the true value of a parameter may deviate from the reported value, as quantified by the standard error of the estimate. In studies with large sample sizes, the estimated value will be close to the true parameter value. However, in cases with limited data, the uncertainty can be more profound. For example, even though an average survival of 0.90 is estimated, the 'true' survival may actually be 0.905 (Figure III.2). The standard error of the mean is an indication of the uncertainty of the estimate. Less variation between years and/or long time series result in smaller uncertainty and therefore smaller standard errors. Note that the standard deviation and standard error are not the same. The standard deviation is a measure for variation around the mean (this can for example be between years, locations or individuals), whereas the standard error is an indicator for the uncertainty of the estimate, which depends on the standard deviation and the sample size. The standard error is calculated as the standard deviation divided by the square root of the sample size. In other words, the standard error depends on the variation around the mean (standard deviation) and the sample size. A standard error is smaller for estimates with a smaller standard deviation, and/or a larger sample size.

It is possible to account for this uncertainty by varying the parameter values between runs. To find out to what extent this difference between estimated and true survival affects the outcome of the population model, the model can be additionally run with different values for survival. For example, if the estimated survival is 0.9, in addition to running the population model with a mean survival of 0.9, running the model with a mean survival of 0.91 and 0.89 gives an indication of the impact of this error of measurement.
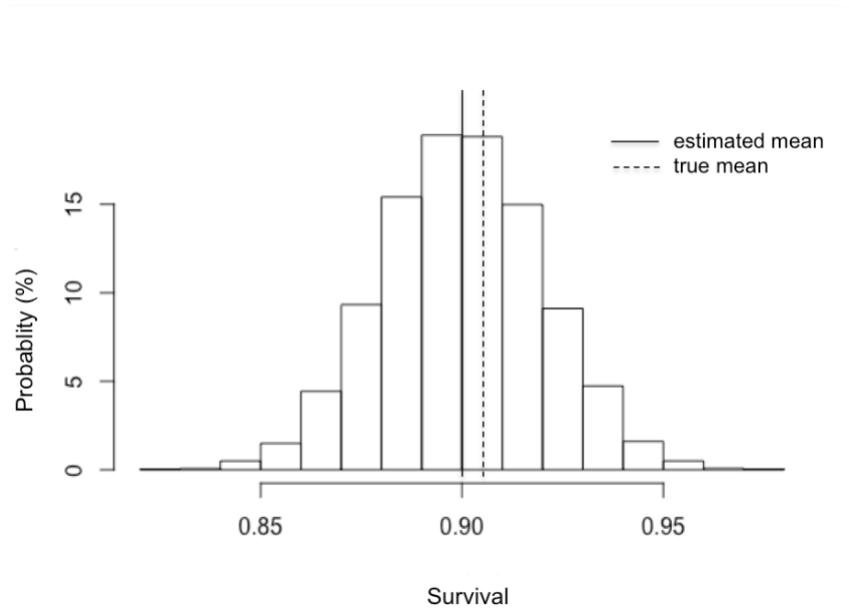
Figure III.2   Illustration of uncertainty in parameter estimates. The solid line represents the average survival as estimated by the data; the dashed line shows the true average survival. Hence, in this case, the estimate is an underestimation.

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IV Simulating the effect of the source of variation on the outcome of the population model

Annex: Implementation methods for analysis parameter uncertainty

A. Potiek, F.H. Soudijn & T. van Kooten, October 31, 2018

Introduction

Parameter estimates are usually reported with a measure of variance, often in the form of a standard deviation. Variability in parameter values may stem from measurement error and/or natural variability in parameter values. Measurement error is an inherent part of any measurement and the measurement process. For example, if the measured adult survival rate is 90%, it is possible that the 'true' survival is actually 91%. A large measurement error indicates that the estimate is relatively uncertain. Natural variability in parameter values is caused by fluctuations in environmental conditions such as food availability and the weather. In a year with favourable environmental conditions, survival and fecundity are likely to be higher than in other years. It is often not possible to separate the causes for variability in the parameter values. However, the impact of the two on the outcome of a population model may strongly differ.

Here, we study to what extent the source of parameter uncertainty (measurement error or natural variability) matters for the outcome of our main study, the effect of OWF-related displacement on the population dynamics of seabirds.

In theory, if all variance is explained by measurement error, the demographic rate is (slightly) under- or overestimated every single year in the population model. Hence, the effect adds up over time. In contrast, variation between years results in some years with above-average parameter values, and some years with below-average parameter values. Over time, the average will move towards the estimated value. Several demographic rates vary between years. In our population models, input parameters are juvenile survival, immature survival, adult survival, fecundity and probability of non-breeding. These demographic rates may vary independently, or the variation may be correlated. Such correlations would occur if natural variability was caused by 'good' and 'bad' years, in which all parameters would be positively or negatively affected. In addition to the source of the variation, we study the effect of correlated variability in parameter values.

This is essentially an analysis of model assumptions. The effect of the various assumptions will be qualitatively similar for any of our studied species. Therefore, we do the analysis only for the common guillemot, for which we have the best data on

variance of parameter values. The results can, in a qualitative sense, be extrapolated to the other species.

These results will allow us to choose the most appropriate way to incorporate parameter value uncertainty in our main analysis. An important guiding principle in this choice will be the precautionary principle. If one assumption will lead to larger effects of OWFs and we have no data to choose between the different possibilities, the precautionary principle requires us to choose the most conservative method.

Aim

This Appendix covers four different subprojects:

Measurement error
1. Test the impact of measurement error on the uncertainty of the outcomes of the population model.

Natural variability
2. Test the effect of stochastic fluctuations in the parameter values through time on the uncertainty of the outcomes of the population model. In contrast to the analysis on the impact of measurement error (1), the parameter values are here (2) assumed to vary between years.

3. Test the effect of within-year correlations between values of the parameters (survival and fecundity) on the uncertainty of the outcomes of the population model. In subproject 2, we assumed that demographic rates varied independently, whereas in subproject 3 we run the model for different strengths of correlations.

Empirical natural variability: evidence for correlated demographic rates
4. Analyse multi-annual parameter datasets of some of our model species (Wozep habitat loss and collision victims) to determine how parameter values naturally vary through time and to what extent demographic rates are correlated.

Methods

We study the effect of the source of measurement error (subprojects 1-3) using the population model of the common guillemot. The model details are described in the main section of this report.

Measurement error

Subproject 1
The uncertainty in the model output due to measurement error can be estimated using Monte Carlo methods. This method implies random sampling from the probability distributions of all parameter estimates. For subproject 1, we follow the assumption that

the deviation from the mean in parameter values is independent for each parameter. In addition, this method assumes that the uncertainty stems from inaccurate measurements of parameter values rather than from temporal changes. Hence, parameter values do not vary in time. The choice for the probability distributions of the parameter estimates depends on the availability of the data for the species that is under study. If very little data are available for a certain species, the range from which a parameter value is drawn will be wider than when good data are available.

We calculate all model output (e.g. population growth rates, sensitivity analysis and population sizes) for a large number of random draws from the parameter probability distributions. Based on the results of these simulations we determine confidence intervals around the model output.

Natural variability

Subproject 2
A stochastic matrix model allows for variability of the parameter values through time. At each time step, parameter values are drawn randomly from a probability distribution of the parameter values. The shape of the probability distributions is based on the information that is available for the species under study. Using this method, we run 10,000 simulations, in which demographic rates are independently, randomly drawn from the probability distribution. Based on the results of the simulations we determine confidence intervals around the model output.

Subproject 3
We also investigate the effect of covariance between the parameter values on the model outcome. If the correlation between survival and reproduction is strong, this means that a good year for survival is also a good year for reproduction. In contrast, if they are uncorrelated, survival and fecundity vary independently. To analyse the impact of such a correlation, we use a similar approach as described for subproject 2. However, we define an additional parameter C, which is the degree of correlation between demographic rates. This parameter C varies between 0 and 1, corresponding to uncorrelated to completely correlated demographic rates.

Empirical natural variability: correlation in measured parameters

Subproject 4
Using datasets of life history parameters that consist of multiple years of data from the same colony, we analyse the effect of "natural" environmental stochasticity on the model output. The previous step in the analysis will help us understand how correlated parameter values may affect the model outcomes. A dataset with "natural" environmental stochasticity shows us how strong the correlation between parameter values and also the frequency of "good" and "bad" years is in reality. The previous step in the analysis is needed because there is not a lot of multi-year data of life history parameters available. As the level of correlation between parameter values varies

between species, we perform this analysis for several species. However, this analysis requires high-quality and long-term data, the availability of which will be limited.

We searched the literature for papers reporting survival rates as well as fecundity. An overview of the available data is shown in Table IV.1. Correlations between demographic rates in the same year were tested on significance, as well as delayed effects of survival in the previous year on breeding success in the present year.

Table IV.1   Overview of data used for testing correlations

Source	Species	Parameters	Area	Time period
Meade *et al.* (2013)	Guillemot	immature survival, adult survival, breeding success	Skomer (UK)	1992-2004
Crespin *et al.* (2006a,b)	Guillemot	immature survival, adult survival, breeding success	Isle of May (UK)	1983-1994
Ebbinge *et al.* (2002)	Brent goose	adult survival, breeding success (% juveniles in winter)	Western Europe (winterin g area)	1956-1998
Centre for Ecology and Hydrogoly (CEH), monitoring programme	Guillemot, kittiwake, razorbill, puffin	return rate, breeding success	Isle of May (UK)	2007-2018

In addition to the data found in published literature, more recent data from Isle of May was collated based on annual newsletters from the CEH long-term monitoring programme, in which among others guillemot, kittiwake, puffin and razorbill are monitored (https://www.ceh.ac.uk/our-science/projects/isle-may-long-term-study). These newsletters report breeding success (fledglings per breeding pair) and return rates for adults. Return rates can be used as a proximate for annual survival. Although return rates underestimate annual survival, it can be assumed that this is the case for all years within the monitoring program (2007-2018).

Input data population models (table IV.2), basis for subproject 1-3

Species: common guillemot
We assume that the data underlying all parameter values follow a binomial distribution. Therefore, uncertainty around a mean parameter value $\overline{Sa}$ with variance σ_{Sa}^2 can be described by a beta distribution with:

$$\alpha = \overline{Sa}^2 (\frac{1-\overline{Sa}}{\sigma_{Sa}^2} - \frac{1}{\overline{Sa}}) \, , \, \beta = (\alpha - \overline{Sa}\,\alpha)/\overline{Sa} \, .$$

We do not have the data and we do not know n_i, we simulate data based on the beta distributions as described above. We repeat this for all parameter values. The variance

is taken as the standard deviation cubed. Note that although the used parameter distribution is the same for subprojects 1-3, the assumed source of variation differs. In the first subproject, we assume that the variation is due to measurement error. The parameter values are drawn at the start of a simulation, and used for each year within that simulation. For a new simulation, new parameter values are drawn. Hence, each simulation uses different demographic rates, which are constant over time. In the second subproject, demographic rates vary over time. Within each simulation, new values for each demographic rate are drawn every year. For the third subproject, the simulation starts with the definition of the degree of correlation. Subsequently, a parameter defining year quality is drawn. Depending on the degree of correlation, the effect of year quality on demographic rates can be strong (high degree of correlation) or weak (low degree of correlation).

Table IV.2: Parameter values and sources used in the population model for common guillemot.

Symbol	Mean value	Variance	unit	Description	Remark	Source
F_a	0.627	0.147 (sd)	Year-1	Fledged offspring		1
D	0.08	0.03 (sd)	-	Skipped breeding probability, all adult stages	No source for SD, the value is arbitrary	1
S_0	0.56	0.013	-	Annual Survival probability age 0		1
S_1	0.792	0.034	-	Annual Survival probability age 1		1
S_2	0.917	0.022	-	Annual Survival probability age 2		
S_a	0.939	0.067 (sd)	-	Annual Survival probability immatures & adults, age 3+		1
a_m	6		Years	Age at recruitment		1

[1]Horswill & Robinson 2015

Results

For each of the scenarios, we show a frequency distribution of the population growth rates, calculated over 10,000 runs (Figure IV.1-3).

1. All variation due to measurement error
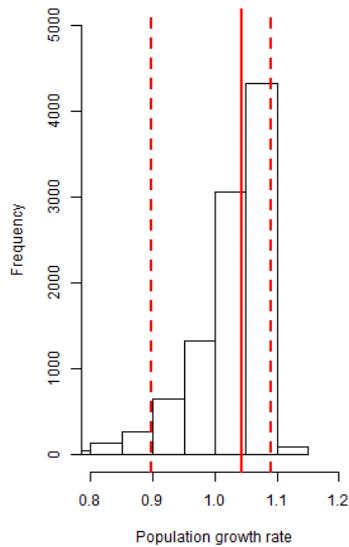
Figure 1   Population growth rates (with mean, red solid line and 5% and 95% edge of results, red dashed lines) and inner 90% of population projections (with mean of all projections, red solid line) for Monte Carlo simulations. Parameter values are as given in Table 2.

2. All variation due to natural variability
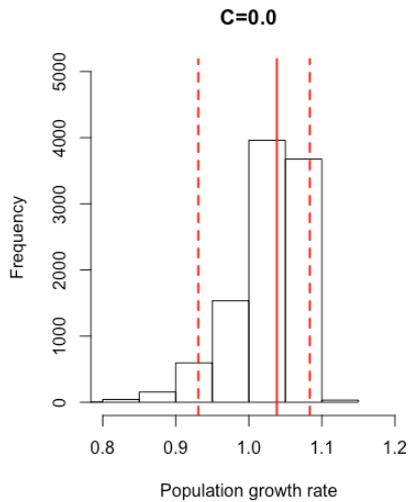
Figure 2   Frequency distribution of population growth rates assuming all variation is due to natural variability. Demographic rates within years are assumed to be independent (not correlated, C=0). Continuous line represents median population growth rate, and dashed lines the 90% confidence interval.

3. All variation due to natural variability; Impact of correlated demographic rates
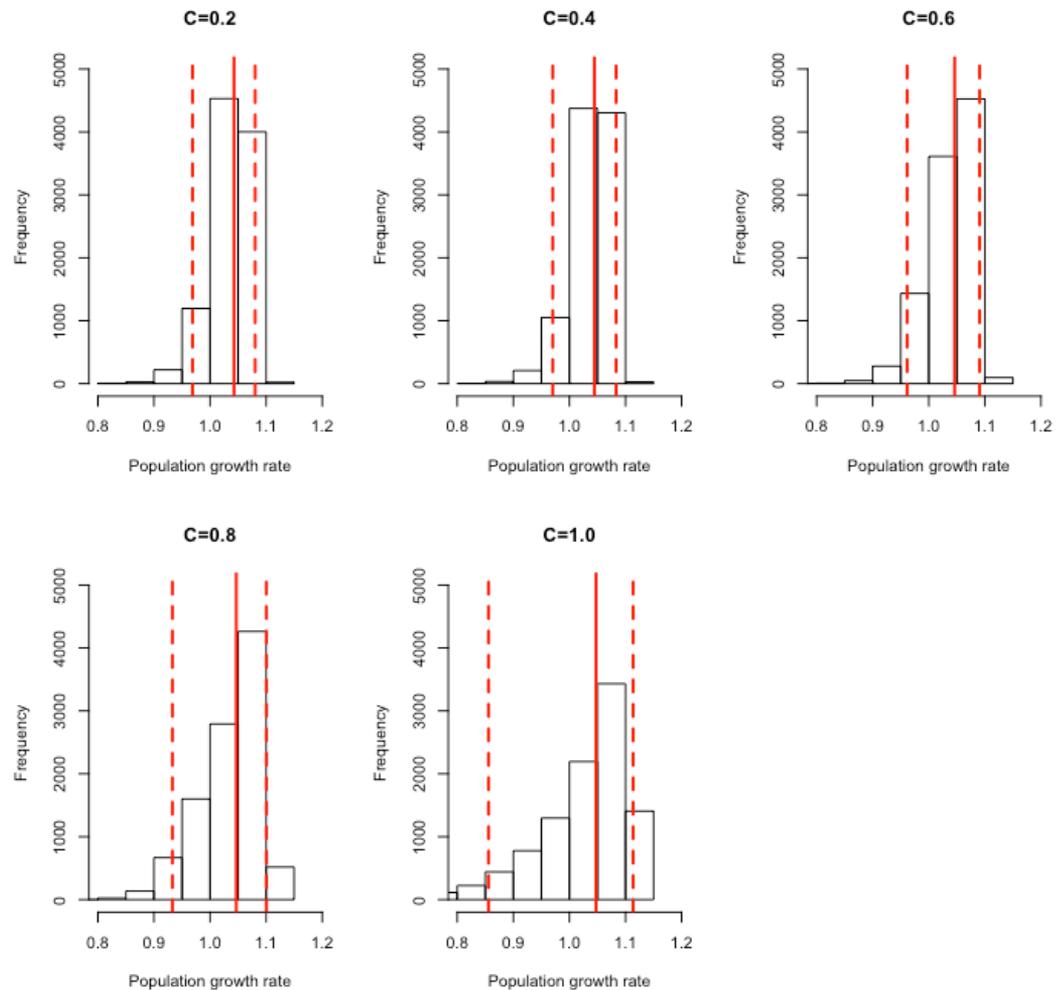
Figure 3   *Frequency distribution of population growth rates with varying degree of correlation between parameters (Low C: weak correlation, C=1: fully correlated values). Continuous line represents median population growth rate, and dashed lines the 90% confidence interval.*

4. Empirical natural variability: correlation in measured parameters

Demographic rates for guillemots reported by Meade *et al.* (2013) and Crespin *et al.* (2006a and b) were not significantly correlated.

For brent goose, we analysed data from Ebbinge *et al.* (2002) to test for correlations between demographic rates. In this study, breeding success in year i is defined as the percentage of first-winter birds in the following winter. We tested for correlations between breeding success and adult survival in the year prior to the breeding season, as well as to the year following the breeding season. We found no correlation between the breeding success and the adult survival in the following year. However, we found a

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significant negative correlation between adult survival prior to the breeding season and breeding success (Figure IV.4). This significant negative correlation suggests that a year with low adult survival is likely followed by high breeding success. This was contrasting with our expectation that individuals are in poorer body condition after a strong winter with high mortality, and therefore have lower breeding success. Potentially, density dependence may explain this pattern: less competition for resources after a winter with high mortality may result in higher breeding success. However, this might be an artefact of the type of data collection. With both measures depending on the number of adults, a relatively low number of adults in a certain year suggests low adult survival compared to other years. In the calculation of breeding success, the number of first-year individuals is divided by a relatively low number of adults. In case the number of first-years is comparable between years, this gives a relatively high proportion of first-years.

Brent goose
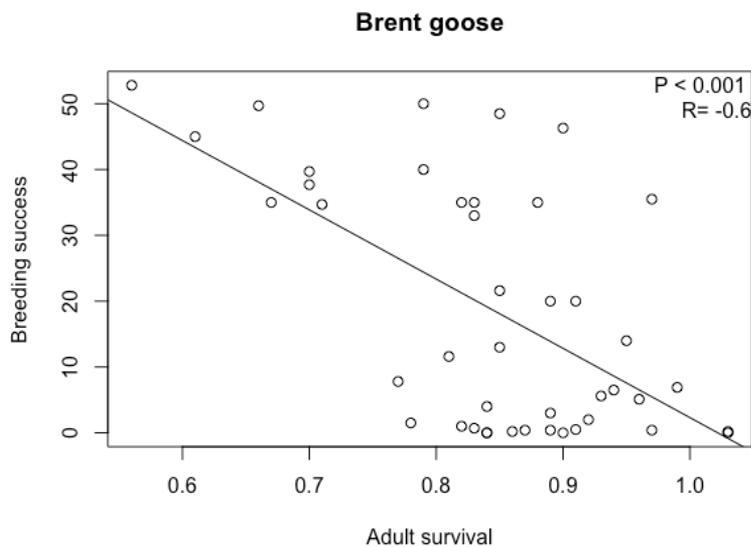
Figure IV.4 Relation between adult survival from year i to i+1 and breeding success in year i+1.

Data on the breeding success and return rates of guillemot, kittiwake, razorbill and puffin from the CEH long-term monitoring programme did not show significant correlations for kittiwake and razorbill. For both guillemot and puffin, there seems to be a significant positive relationship between breeding success and return rate in the year before as well as after the breeding season (Figure IV.5). However, the cause of these correlations is a single data point of extremely low breeding success. In all cases, the exclusion of this data point results in a non-significant correlation.
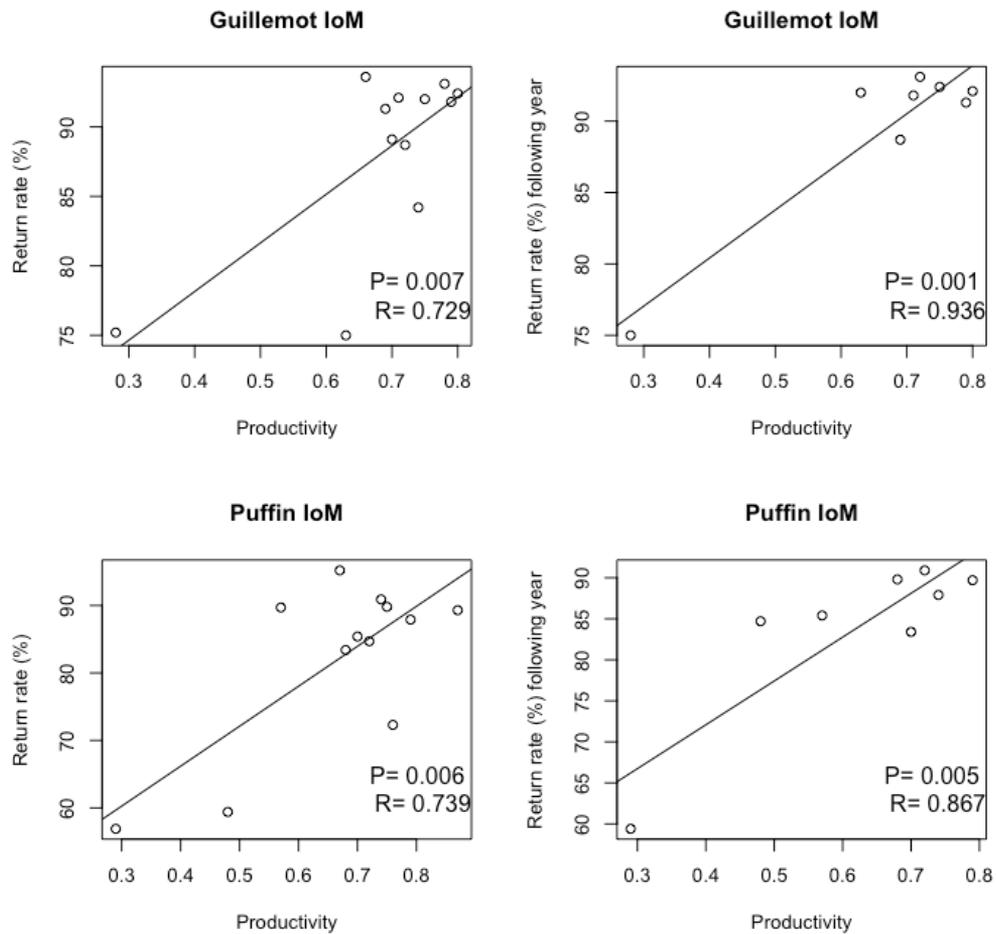
Figure IV.5 Correlations between breeding success and the return rate of the year prior to the breeding season (left panel), and between breeding success and the return rate of the year following the breeding season (right panel).

Conclusions

- Stochastic simulations with complete correlation give the widest variance in population growth rate distribution, but we see no support for such strong correlations in any of the data examined.
- Uncorrelated and weakly correlated temporal variation of parameter values leads to the narrowest population growth rate distributions
- Time-invariant measurement error (Monte Carlo approach) leads to higher variation in population growth rate distribution.
- There is no empirical basis to distinguish between temporal variations in parameter values and measurement error.
- Applying the precautionary approach leads us to choose the Monte Carlo approach, because it results in the highest uncertainty among assumptions with equal plausibility.

- Long-term studies and repeated measures of parameters are needed to empirically distinguish between the assumptions tested here.

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V Perturbation analysis

In a so-called perturbation analysis, the impact of changing one or more input parameters on the model output can be assessed (see Caswell 2001: Chapter 9). A change in any of the input parameters of the population model affects the model output, for example population growth rate. However, depending on the demographic rates (survival of different age classes, fecundity), the population growth rate (or any other model outcome) may be particularly impacted by changes in a certain parameter.

Two main approaches of perturbation analyses are 'sensitivity' and 'elasticity' analyses (see Chapter 9 of Caswell (2001)). A sensitivity analysis shows the effect of an absolute change in a parameter on a model output, for example on the population growth rate. In other words, a sensitivity analysis can be used to assess how the population growth rate is impacted by a 0.01 (absolute) change in a parameter, for example a change of fecundity from 1.2 to 1.21. In contrast, an elasticity analysis assesses the impact of a relative change in a vital rate on the model output. Using the same example, this would be the impact of a 1% change in fecundity on the population growth rate, i.e. a change from 1.2 to 1.188. Similarly, when testing the effect of changes in survival on the population growth rate, a sensitivity analysis shows the impact of an absolute change of 1% (for example a change in juvenile survival from 60% to 59%), whereas in an elasticity analysis the impact of a 1% relative change is assessed (for example from 60% to 59.4%).
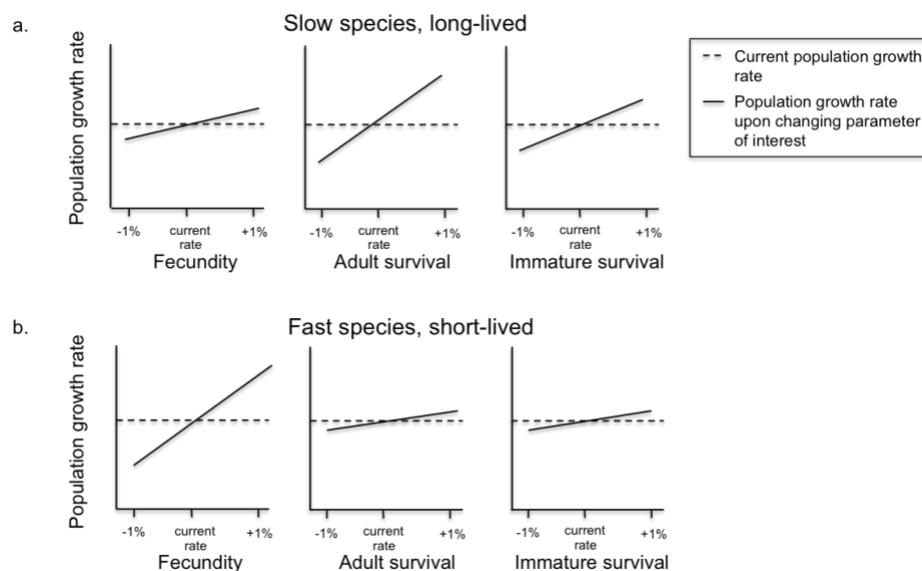
Figure V.1 Example of the impact of a 1% change in fecundity, adult survival and immature survival on the population growth rate of a long-lived species (a) and a short-lived species (b). Dashed lines show the current projected growth rates. Continuous lines show how this projected growth rate changes when only a specific parameter (shown on the x-axis) is adjusted. This type of results can be obtained by performing an elasticity analysis.

Figure V.1 shows for two species how a 1% increase or decrease in fecundity, adult survival or immature survival affects the expected population growth rate. By looking at

the slope of those lines, it can be concluded that for the species in Figure V.1a, a 1% change in adult survival has a stronger impact on the population growth rate than a 1% change in fecundity or immature survival. This fits to a species with a slow life history strategy, i.e. a long-lived species. In comparison, Figure V.1b shows a species with a fast life history strategy, i.e. a short-lived species. In this type of species, the effect of a 1%-change in fecundity on the population growth rate is stronger than the effect of a 1%-change in survival.

Hence, this type of analysis shows which parameters have the strongest influence on the outcome of the model. Uncertainty in parameters with a large impact results in lower confidence in the model outcome. Therefore, these input parameters need to be as precise as possible. Within this project, we analyse the relative impact of changes in demographic rates on the population growth, by performing an elasticity analysis.

References

Caswell, H., 2001. Matrix population models. John Wiley & Sons, Ltd.

VI Inventory of input parameters for population models

VI.1 Lesser black-backed gull

Occurrence in and around the Dutch North Sea

Lesser black-backed gulls breed in all countries around the North Sea (Camphuysen 2011). They mainly forage in the North Sea, often making foraging trips of more than 200 km. The distribution of lesser black-backed gulls in the southern North Sea during the breeding season is concentrated around the breeding colonies, with higher numbers also towards the Frisian Front area (Camphuysen 2013). The breeding colonies are located along the coast, but with increasing frequency birds are also observed breeding as well as feeding more inland. After the breeding season, in August, large concentrations are observed in the southern North Sea and the southern part of the central North Sea. In autumn, lesser black-backed gulls migrate southwestwards towards the Iberian Peninsula and the western coast of Africa (Camphuysen 2013, www.vogeltrekatlas.nl). Increasingly more individuals stay in Northwest Europe during winter (Ross-Smith *et al.* 2014). This is likely to be related to milder winters in recent decades.

The population model for lesser black-backed gull is based on the population model created by Poot *et al.* (2011) with updated parameters. While the model in Poot *et al.* (2011) only used survival and fecundity estimates from the UK, we have incorporated estimates from the Netherlands as well.

Available demographic data

Demographic rates of lesser black-backed gulls are relatively well studied. Estimates are available from both the Netherlands (Lake Volkerak, Texel, Terschelling) and the UK. In particular the breeding colony on Texel has been studied intensively.

Within the UK, populations vary in trend; for example, on Skomer Island, the population decreases (Perrins & Smith 2000), whereas the population in Barrow Town increases (Sellers & Shackleton 2011). Adult survival rates are similar throughout the UK (around 90%), but productivity strongly varies. In Barrow Town, productivity estimated in 2009 was 2.05 chicks (almost at fledging) per breeding pair (Sellers & Shackleton 2011), whereas productivity on Skomer Island averaged less than 0.2 fledglings per breeding pair between 1987 and 2000 (Perrins & Smith 2000).

Griffiths (1992) demonstrated that mortality during nestling stage seems to be sex-biased, with higher mortality in males. However, it is unclear whether this difference during the nestling stage persists throughout the first year, or is compensated for later during that year. In contrast, Camphuysen & Gronert (2012) showed that including sex in models explaining adult survival does not improve the model performance, indicating that adult survival is not sex-biased.

Camphuysen & Gronert (2012) mention that their fecundity estimate is based on individuals that laid eggs, while they gained the impression in the field that 'a substantial number' of lesser black-backed gulls that visited the Texel colony did not eventually breed in that season (lack of subsequent observations, numerous empty nests in the colony). Calladine & Harris (1997) mention that 34% and 40% of adult lesser black-backed gulls, all with some previous breeding experience, failed to breed in two years on the Isle of May.

Implications for population model
Estimates of adult survival and fecundity are available for several populations around the North Sea (Table VI.1). Information on juvenile survival is more limited, but recent estimates of high quality are available from one study (Camphuysen 2013).
Collision risk may differ between stages. Camphuysen & Leopold (1994) studied the age composition of lesser black-backed gulls on the North Sea. Using this information, we deduce the expected age composition among the victims.
For colonies where males spend more time at the North Sea than females, males are more at risk of colliding. This is the case on Texel (Camphuysen *et al.* 2015) and possibly in other coastal colonies as well. However, due to the lack of clear empirical data on differences in collision risk between males and females, we assume a one-sex population model.
Note that the time spent at sea seems to vary among colonies, and especially between coastal and inland colonies. As a result, the collision risk might also be colony-specific. The impact of additional mortality may therefore be stronger at coastal colonies, compared to others. Even if scenarios for the population definition used here (see Paragraph 2.1) shows that the estimated additional mortality has no large impact, it should be noted that the impact may be stronger on the smaller scale of a specific colony.

Table VI.1 *Age-specific survival rates (VI.1a) and fecundity rates (VI.1b) of lesser black-backed gulls from different populations. Empty cells indicate no available information for this population. *: ringed as chicks. Local population trend: ++: strongly increasing; =: stable; -: declining. Data type: [a] Colour-ring resightings, [b] Review. Reference: [1] Camphuysen (2013); [2] Harris (1970); [3] Camphuysen & Gronert (2012); [4] Camphuysen (2011); [5] Wanless et al. (1996); [6] Horswill & Robinson (2015); [7] Gyimesi et al. (2011); [8] Camphuysen in Koffijberg et al. (2017); [9] Spaans et al. (1994); [10] Sellers & Shackleton (2011); [11] Perrins & Smith (2000); [12] Mavor et al. (2008); [13] Calladine & Harris (1997). Data quality and representation are assessed based on the criteria described in Chapter 1. * several projects are currently being carried out to determine additional estimates of especially survival rates (colour-ring programmes in Europe) but also fecundity rates, so this overview is not a complete inventory and additional analyses might yield better estimates.*

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival	0.306	554*	0.075		2006-2011	++	Texel, NL	[a]	[1], [3]	6	6
	0.82					++	UK	[a]	[2]	3	4
	0.57	343*			1984-1996	++	NL (Delta + Wadden Isles)	[a]	[4]	6	4
Immature survival	0.825	554*	0.052		2006-2009	++	Texel, NL	[a]	[1]	5	6
	0.89	343*	0.02		1984-1996	++	NL (Delta + Wadden Isles)	[a]	[4]	6	4
Adult survival	0.91	554*		0.036	2006-2011	++	Texel, NL	[a]	[3]	6	6
	0.95					++	Texel, NL	[a]	[4]	6	6
	0.91			0.017	1983-1994	++	UK	[a]	[5]	5	5
	0.885	Based on 2 studies	0.022		[review]	[review]	[UK, review]	[b]	[6]	6	6

_b._

Demographic Rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
Lake Volkerak, NL: inside enclosure	1.62		0.96		2010		[7]	2	5
Lake Volkerak, NL: outside enclosure	0.8		0.4		2010		[7]	2	5
Texel, NL	0.49		0.18		2006-2016	++	[8]	6	6
Texel, NL	0.47		0.19		2006-2011	++	[3]	6	6
Terschelling, NL	0.85 (1992), 0.71 (1993)	1992: n=26; 1993: n=21	0.92		1992-1993	=	[9]	1	6
Several colonies UK	2.14 fledg per successful brood	96 broods			2009	-	[10]	1	4
UK	0.46			[strong variation]	1982-1998		[11]	3	1
Several colonies UK	UK average: 0.530; means per colony range from 0.17 to 0.88	6 colonies	0.325	variable, around 0.05 (0.17 +- 0.02; 0.88 +- 0.07)	1986-2005	variable	[12]	3	1
Isle of May, UK	0.813	5329 nests; 6 years	0		1989-1994	++	[5]	6	4
Age of first breeding	5 / 6 years						[3]		
Incidence of missed breeding	50%						[1]		
	37%	109 of 292 attempts			1993-1994		[13]		

_b._

VI.2   Great black-backed gull

Occurrence in and around North Sea
The breeding range of the great black-backed gull covers the north Atlantic and adjacent seas (Mitchell *et al.* 2004). Of the 100,000-110,000 breeding pairs in Europe, the majority breeds in Norway (40,000), followed by Britain (17,000), Sweden (15,000) and Iceland (15,000). Fewer individuals breed in the Netherlands and Germany, with an estimated 61-65 breeding pairs in the Netherlands (Boele *et al.* 2017), and around 4,000 breeding pairs in France (Mitchell *et al.* 2004).

Many great black-backed gulls breeding in the UK are residents and spend the winter close to their nesting grounds, where they mainly forage in the North Sea. Individuals breeding in Norway and Denmark move southwards outside the breeding season. Individuals breeding in France, Belgium, the Netherlands, UK, Denmark, and the whole of Fennoscandinavia east to the Kola Peninsula use the North Sea in winter for foraging (Taylor 2014, www.vogeltrekatlas.nl) and are therefore at risk of collisions with wind turbines located in this area.

Available demographic data
Data on sub-adult survival of great black-backed gulls are lacking and we found only three estimates of adult survival (Table VI.2). Barrett *et al.* (2015) provide a high-quality estimate on adult survival. Data on fecundity are available from different populations, but relatively old. Furness (2016) noted that the adult survival estimate is rather low for a seabird of this size and suggested that it may have been depressed by some surviving individuals not returning to breed in all years.

Implications for population model
Data on juvenile and immature survival are lacking. For adult survival, we found one high quality estimate. Data on movement patterns are scarce.

We have used available information on fecundity and adult survival. With data on subadult survival of great black-backed gull lacking, we use data on herring gull. Survival rates are expected to be similar between those species due to similar wintering areas.

Table VI.2   Age-specific survival rates (VI.2a) and fecundity rates (VI.2b) of great black-backed gulls from different populations. Empty cells indicate no available information for this population. Local population trend: +: increasing. References: [1] Glutz von Blotzheim et al. (1984); [2] Barrett et al. (2015); [3] Mavor et al. (2008); [4] Verbeek (1979); [5] Schekkerman et al. (2017); [6] Butler & Trivelpiece (1981); [7] Robinson (2018). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Location	Data type	Reference	Data quality	Data Representativeness
Juvenile survival											
Immature survival											
Adult survival											
	0.93								[1]	3	3
	0.82	208	0.017		2001-2014		Norway		[2]	6	5

b.

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Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
UK	1.109	10 colonies, 2-72 nests per colony	0.54	average SE: 0.156, per colony: 0.06-0.29	different colonies, per colony up to 13yrs (1986-2005)	variable	[3]	6	3
UK, Walney Island	0.645	56			1973-1974	+	[4]	2	3
NL, Deltagebied	1.0	several locations, 56 location-years			2006-2015		[5]	5	6
USA, Maine (little duck island)	Low density: 0.21; High density: 0.83				1979		[6]	1	1
Age of first breeding	4 years						[7]		
Incidence of missed breeding									

VI.3 Herring gull

Occurrence in and around North Sea
Herring gulls can be found year-round in the North Sea (Camphuysen 2013). Highest concentrations are found within 12 miles from the coast, where breeding colonies are located (Fijn *et al.* 2015). Breeding occurs at the coast of all countries around the North Sea. In winter, individuals spread throughout the southern North Sea and the English Channel, and some move inland (Fijn *et al.* 2015).

Herring gulls use both the sea and inland (for example rubbish dumps) for foraging. Feeding ranges of individuals from a Dutch colony are reported to be 35 km (Spaans 1971). A more recent estimate is that 95% of the herring gulls breeding on Terschelling foraged within 54 km from the colony (Camphuysen 1995). In the last decades, populations of herring gulls declined (www.sovon.nl, JNCC Seabird Monitoring Programme Database: www.jncc.gov.uk/smp).

Available demographic data
Data on demographic parameters are available for Texel, Germany and several colonies in the UK (Table VI.3). Wanless *et al.* (1996) constructed a simple population model for herring gull. Using data on the number of breeding adults and reproductive rates from a population on Isle of May, they estimated survival rates. Survival is also estimated in a breeding colony on Texel (NL) using colour rings, which gave slightly lower estimates than those provided by Wanless *et al.* (1996) (Camphuysen & Gronert 2012).

Adult survival is sex-specific, with higher survival for males, and estimates of adult survival are slightly lower for individuals from Texel (Camphuysen & Gronert 2012) compared to estimates from other colonies.
The age of first breeding usually lies between 4 and 6 years of age (Camphuysen & Gronert 2012; Wanless *et al.* 1996). Although Coulson *et al.* (1982) and Migot (1992) suggested that some herring gull adults with previous breeding experience might skip breeding, Wanless *et al.* (1996) did not find evidence for this phenomenon, but Calladine & Harris (1997) found that 33% and 37% of adult herring gulls, all with some previous breeding experience, failed to breed in two years on the Isle of May.
Non-breeding is incorporated in the model, by specifying a breeding probability for adults. By running a perturbation analysis (see Chapter 2; Appendix V), the effect of including a fraction of breeders on the long-term population dynamics can be assessed.

Implications for population model
The population model was based on the model used in Poot *et al.* (2011), using updated demographic parameters. We suggest a population model with 3 juvenile age classes, and sexual maturity from the age of 4 onwards. As some individuals start breeding only at the age of 5, 6, or even 7 (Camphuysen & Gronert 2012; Chabrzyk & Coulson 1976), we suggest incorporating a proportion of non-breeders in the model.

We suggest using female-based population models. Although survival, and also offshore distribution, seems to differ between the sexes (Klein 2001, Camphuysen *et al.* 2011, Camphuysen 2013), no information about sex-specific collision risk is available. Based on information of a population on Texel, female survival seems to be lower than male survival. Hence, this can be considered a cautious approach. It is unclear whether certain age classes experience higher collision risk. However, individuals from age classes which spend more time at sea can be assumed to have a higher collision risk. Therefore, we have used data from Camphuysen & Leopold (1994) to determine the age-specific collision risk.

Table VI.3 *Age-specific survival rates (VI.3a) and fecundity rates (VI.3b) of herring gull from different populations. Empty cells indicate no available information for this population. *: ringed as chick. Data type: [a] Colour-ring resightings. References: [1] Camphuysen (2013); [2] Chabrzyk & Coulson (1976); [3] Wanless et al. (1996); [4] Camphuysen & Gronert (2012); [5] Glutz von Blotzheim et al. (1984); [6] Camphuysen in Koffijberg et al. (2017); [7] Koffijberg et al. (2017); [8] Mavor et al. (2008); [9] Sellers & Shackleton (2011). Data quality and representativeness are assessed based on the criteria described in Chapter 2.*

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival	0.25		0.06		2006-2011	-	Texel, NL	[a]	[1]	5	6
	0.63 from fledging to age 4						UK		[2]	0	2
	0.45 from fledging to age 4			0.07	1989-1994	strong fluctuations	Isle of May, UK		[3]	4	4
Immature survival	0.89					-	Texel, NL	[a]	[1]	4	6
	0.7	119 *		0.06	2006-2012	-	Texel, NL	[a]	[4]	5	6
	0.45 from fledging to age 4			0.07	1989-1994	strong fluctuations	Isle of May, UK		[3]	4	4
Adult survival	♀0.79 ♂ 0.86	119 *		♀ 0.049 ♂ 0.038	2006-2012	-	Texel, NL	[a]	[4]	5	6
	0.93								[5]	0	2
	0.935								[2]	0	2
	♂ 0.88; ♀0.858			♂ 0.013; ♀ 0.02	1989-1994		Isle of May, UK	[a]	[3]	4	3

b.

Demgroaphic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
Texel, NL	0.86		0.31	-	2006-2011	-	[4]	5	6
Texel (update), NL	0.68		0.29		2006-2016	=	[6]	6	6
Texel (Westerduinen), NL	0.41		0.15		2005-2015	=	[7]	6	6
Vlieland (Vliehors), NL	0.82				2009-2016	=	[7]	3	6
Schiermonnikoog	0.69		0.29		2006-2016		[7]	6	5
UK, several colonies	0.914	up to 17 years per colony, 7 colonies	0.207	Between 0.08 and 0.16	1986-2005	variable	[8]	5	4
UK, Isle of May	1.378	10270 nests	0.303		1989-1994	strong fluctuations	[3]	4	4
UK, two colonies	2.02	98			2009	-	[9]	0	4
Age of first breeding	4-6						[4]		
	4						[3]		
	5.25						[2]		
Incidence of missed breeding	Breeding frequency once every 1.5 (Texel) to 1.6 years (IJmuiden)						[1]		

VI.4 Black-legged kittiwake

Occurrence in and around North Sea:
The black-legged kittiwake is the most common gull on the Dutch continental shelf during winter, mainly using areas further offshore. Only small numbers of kittiwakes breed on the oil platforms on the Dutch Continental Shelf (Camphuysen & Leopold 2007; Geelhoed *et al.* 2011).

Large breeding colonies around the North Sea are located in Northeast-England, East-Scotland, and on the Orkneys and Shetland Isles. Colonies in S Norway, Denmark and Germany are smaller, totalling less than 10,000 pairs (Mitchell *et al.* 2004). Individuals from the entire North-Atlantic breeding range are present in the Dutch North Sea (Frederiksen *et al.* 2012). The European population has been declining since the 1980s, mainly due to decreased food availability. The current decline is estimated at >40% over three generations (39 years) leading to a 'vulnerable' status on the global 'IUCN Red List of Threatened Species' (BirdLife International 2018), in addition to a similar status on the European, British and Norwegian Red Lists ('Endangered').

Kittiwakes spend the winter at sea in the Northwest Atlantic (Frederiksen *et al.* 2012) but also in the North Sea and Norwegian Sea. Immature birds stay at sea until maturation (Coulson & Neve de Mevergnies 1992). In summer, kittiwakes stay much closer to the colonies (mean feeding range of 25 km (Thaxter *et al.* 2012)). However, this range is strongly dependent on colony and habitat characteristics (Soanes *et al.* 2013). For example, adults from a colony in Northeast England have a feeding range of approximately 50 km during the breeding season (Robertson *et al.* 2014) and much larger foraging ranges have been found for various colonies.

Available demographic data
Data on juvenile survival are limited (Table VI.4). We found only one estimate of juvenile survival from the area around the North Sea, which is relatively old (Coulson & White 1959). Population sizes and dynamics of kittiwakes have changed substantially over the past decades, so the accuracy of this figure is unclear at this moment. McKnight (2017) found a somewhat lower survival to breeding age in kittiwakes in Alaska. However, this is most likely due to site-specificity and especially a higher age of first reproduction in Alaska compared to Atlantic kittiwakes.

Cam *et al.* (2002) found that the age of first breeding was on average 4.0 years. However, Coulson (2011) reported substantial individual variation, with age of first breeding ranging from 2 to 10 years old. Moreover, Coulson (2011) showed that males tend to breed earlier than females (means 4.0 vs. 4.7).
In addition to data from literature, data have been extracted from the JNCC Seabird Monitoring Programme Database (www.jncc.gov.uk/smp).

Implications for population model

For kittiwakes, data on adult survival and fecundity rates in areas around the North Sea are available from several studies (Table VI.4). In contrast, information on juvenile survival is scarce, with only one estimate. Estimates of survival for different populations are comparable, whereas fecundity is relatively variable.

Similar to an existing population model for kittiwake (Cork Ecology & Bureau Waardenburg 2018), we suggest constructing a matrix model with three immature age classes, followed by an adult stage class. In other words, we suggest assuming that sexual maturity is reached at age 4 (first breeding in 5th calendar year), which is the average age of first reproduction reported by several studies (Cam *et al.* 2002; Mavor *et al.* 2008).

Furness (2016) pointed out that, at least in summer, juveniles may be overrepresented in areas further away from the coast. Collision risk may therefore differ between juveniles and older individuals. As juveniles spend more time at sea, we expect an increased collision risk for the younger age classes. In case the collision risk is higher for younger age classes than for adults, the impact of the additional mortality is stronger. As a worst case scenario, we suggest assuming the victims per stage class based on the stable stage structure. If requested, we can run additional scenarios adjusting the number of juveniles among the victims.

Table VI.4 Age-specific survival rates (VI.1a) and fecundity rates (VI.1b) of kittiwakes from different populations. Empty cells indicate no available information for this population. Local population trend: +: increasing; -: declining; =: stable. Data type: [a] Ringing programme; [b] Review. References: [1] Coulson & White (1959); [2] Horswill & Robinson (2015); [3] Thomas & Coulson (1988); [4] Harris et al. (2000); [5] Frederiksen et al. (2004); [6] Cam et al. (2002); [7] Sandvik et al. (2005); [8] Coulson & Wooller (1976); [9] Reiertsen et al. (2014); [10] del Hoyo et al. (1996); [11] Mavor et al. (2008); [12] JNCC Seabird Monitoring Programme Database, www.jncc.gov.uk/smp; [13] Coulson (2011). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativenes
Juvenile survival	1st year: 0.79				1954-1958	+	UK	[a]	[1], [2]	0	2
Immature survival	0.9				1954-1958	+	UK	[a]	[1], [3]	0	2
Adult survival	0.9				1954-1958	+	UK	[a]	[1], [3]	0	2
	0.882	359 ringed		0.017	1986-1996		UK	[a]	[4]	6	3
	0.854		0.051		[review]	[review]	[review]	[b]	[2]	6	5
	0.908				1986-2002	Until '90 +, then -	UK	[a]	[5]	5	4
	0.81			0.017	1987-1999		France	[a]	[6]	6	2
	0.88		0.09		data 1990-2002		Norway	[a]	[7]	6	4
	0.842	based on 8 studies		0.7	variable	-	variable	[b]	[7]	6	5
	♀ 0.86, ♂ 0.81			♀0.008, ♂0.010	1954-1974	=	UK	[a]	[8]	5	2
	0.85		0.66-	0.04-	1990−2011	--	Norway	[a]	[9]	6	6
	0.81								[10]	2	2

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
	1.16					+	[1], [3]	0	2
UK (several colonies)	0.68 (colony-specific means range from 0.21-1.25)			0.03 (SE of mean 0.68)	1986-2005	variable	[11]	6	4
UK (Saltburn cliffs)	0.62	9 years, on average >200 breeding pairs per year	0.2		2000-2008	=	[12]	5	3
UK (Isle of May)	1990-1999: 0.3 2000-2002: 0.68			1990-1999: SE=0.04 2000-2002: SE=0.11	1986-2002	+ until1990, then -	[5]	6	4
Age of first breeding	4						[11]		
	4						[6]		
	Males: 3.97; females: 4.7 (range 2-10)						[13]		
Incidence of missed breeding									

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VI.5   Little gull

Occurrence in and around North Sea
Peak numbers of little gull occur during autumn migration with fewer during spring migration. Small numbers winter in the North Sea with most birds continuing south or across the UK to the Irish Sea (Wernham *et al.* 2002). The main breeding range includes northwest Russia and eastern Siberia. Breeding occasionally occurs outside this range with small numbers now regularly appearing on European coasts.

Available demographic data
Based on a literature review and Robinson (2018), estimates of juvenile and immature survival are not available. For adult survival, one estimate is available. This estimate of 80% annual survival is based on a closely related species (Garthe & Hüppop 2004, species not specified).
Data on fecundity are available for several sources, as reported in table VI.5. However, data quality is very limited.

Age of first breeding is assumed to be 2-3 years (Cramp and Simmons 1983; Berg 1937; Veen 1980). To our knowledge, no information is available on the incidence of non-breeding.

Table VI.5 *Age-specific survival rates (VI.5a) and fecundity rates (VI.5b) of little gulls from different populations. Empty cells indicate no available information for this population. Local population trend: +: increasing. References: [1] Garthe & Hüppop (2004); [2] Koks (1998); [3] Putkonen (1939); [4] Cramp and Simmons (1983); [5] Haverschmidt (1942) ;[6] Veen (1978); [7] Veen (1980); [8] Berg (1937).*

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival										0	0
Immature survival										0	0
Adult survival	0.8							[estimate based on closely related species]	[1]	0	1

b.

Demgroaphic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
NL	Clutch size: 2,6 eggs/nest	163 nests	0.6		1942-1996	fluctuations	[2]	6	4
Finland	Clutch size: 2,71 eggs/nest	214 nests					[3]; [4]	4	3
NL	Clutch size: 2,17 eggs/nest	29 nests					[4]; [5]	0	3
NL, Lauwerszee	0	3 years (resp. 25, 44, 30 breeding pairs per year)			1975-1977		[6]; [7]	2	3
NL, Lauwerszee	0.1	5 colonies, 59 nests			1978		[7]	1	3
Age of first breeding	2-3						[4]; [8]		
	>2 calendar yr						[7]		
Incidence of missed breeding									

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VI.6 Great skua

Occurrence in and around North Sea

The main breeding areas of great skua are located in Great Britain and Iceland, with resp. 9,600 and 5,400 breeding pairs (Mitchell *et al.* 2004). Smaller numbers breed in Svalbard (600-1500 bp), Faroes (500) and along the Norwegian and N-Russian coasts (200-250; Birdlife International 2018), where the breeding range has recently expanded along the Barents Sea shores.

Wintering grounds of great skuas are located off northwest Africa and southern Europe (Magnusdottir *et al.* 2012). Although most individuals winter further south, small numbers of great skuas are present in the Dutch North Sea from July to February, with a peak during migration in September and October (trektellen.nl; Camphuysen & Leopold 1994; Furness 2015). Most of these individuals are located relatively far off the coast (Fijn *et al.* 2016).

Available demographic data

Great skuas are relatively well studied. Table VI.6 gives a summary of demographic rates from different studies. Survival rates are comparable between studies, whereas there is large variation in reproductive success. For example, Mavor *et al.* (2008) compared the reproductive success at nine different sites in Scotland for the time period between 1986 and 2005. For each site, the average reproductive success was calculated. These site-specific averages varied between 0.33 (SE 0.33) in Northern Scotland (Hoy, Orkney) and 0.88 (SE 0.2) in West Scotland (Handa).

Although many studies report estimates of breeding success and adult survival, estimates of juvenile survival are limited. The only available estimate stems from Furness (1978), who used data on ring recoveries to get an indication of juvenile survival. The survival during the first 6 months (first calendar year) is estimated to be 84%, followed by 92% survival in the second calendar year. This estimate of second-year survival is close to the adult mortality estimated in this study (93%). Based on this study by Furness (1978), the BTO reports an estimated first-year survival of 0.8 (BTO birdfacts, Robinson 2018), hence assuming that 4% of the fledglings die during the second half of the first year.

Determination of exact age is often not possible for adults. For that reason, most studies calculate a general estimate for adult survival, not specified for different age classes. One exception is Ratcliffe *et al.* (2002), who showed age-dependent survival, with adult survival initially increasing with age, followed by a decline at higher age (above age of 22). This change in survival rate within the adult stage can be incorporated by constructing population models with several adult stages, representing for example 'young' adults, 'more experienced' adults and 'older' adults. However, the merit of incorporating age-dependency is very small. Adult survival is usually estimated for all adults, hence including younger and older individuals. Moreover, the calculation of age-specific estimates results in smaller sample sizes (because the groups from which the

averages are calculated are smaller), and therefore a higher uncertainty (larger standard error) compared to a general class of adults. There is no indication of age-specific collision risk. Due to the small merit, unnecessary complication of the population models, and increased uncertainty of age-specific estimates (compared with a general estimate of adult survival), we will not incorporate age-specific adult survival in the population model in line with the other species.

Catry *et al.* (1998) studied the frequency of non-breeding in adult great skuas with previous breeding experience on Shetland Islands and found relatively low levels of non-breeding (8.9%, range 4-14%). Non-breeding was mainly caused by loss of mate due to death or divorce, or loss of territory in case of male birds.

Implications for population model
Data on all necessary vital rates are available, as reported in Table VI.6. However, we found only one estimate for subadult survival (individuals of 1 year of age), which is relatively old. Using a perturbation analysis (see Chapter 2, elasticity and/or sensitivity analysis), we assess the effect of a change in this estimate on the model outcome. If the parameter has a large effect, an error in the vital rate has a relatively strong effect on the outcome. If requested, we can run an additional scenario with a lower subadult survival (as a worst case scenario).

Data on adult survival from different populations show similar estimates (Table VI.6). As Catry *et al.* (1998) found that 8.9% of adults did not breed, we suggest incorporating this level of non-breeding in the population model.

Moreover, Wade *et al.* (2014) showed that non-breeding adults spend more time at sea than breeding adults. This may result in higher collision risk of non-breeding adults during the breeding season. However, to our knowledge, no studies directly show this difference in collision risk between breeding adults and floaters. We therefore suggest running the model assuming no difference in collision risk between adult breeders and non-breeders, and between stages. If requested, we can run an additional worst-case scenario in which a larger proportion of victims are assumed to be adults.

We suggest constructing a population model with 6 immature age-classes, followed by an adult stage. In other words, first breeding is assumed at the age of 7. After Catry *et al.* (1998) we suggest a breeding probability of 91%. Fecundity is relatively variable between populations. Depending on the population definition, fecundity (and other demographic rates) should be defined based on relevant populations. For our population models, we calculated a weighted estimate of fecundity based on data quality and representativeness.

Table VI.6   Age-specific survival rates (VI.6a) and fecundity rates (VI.6b) of great skua from different populations. Empty cells indicate no available information for this population. Data type [a]: ring recoveries. References: [1] Furness (1978); [2] Balmer & Peach (1997); [3] Ratcliffe et al. (2002); [4] Catry et al. (1998); [5] del Hoyo et al. (1996); [6] JNCC Seabird Monitoring Programme Database, www.jncc.gov.uk/smp; Fair Isle; [7] Jones et al. (2008); [8] Phillips et al. (1999); [9] Mavor et al. (2008); [10] Robinson (2018). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival	0.84 in first 6 months	4 years			?, <1978		UK	[a]	[1]	0	3
	0.8	4 years			?, <1978		UK	[a]	[1], [2]	0	3
Immature survival	0.93				?, <1978			[a]	[1]	0	3
Adult survival	0.89; but age effect				1988-1999	-	UK	[a]	[3]	5	3
	0.93	236 recoveries		0.02	?, <1978		UK		[1], [2]	0	3
	0.87; ranging from 0.8-0.93	8 years, 1224 recoveries	0.055		1989-1996		UK	[a]	[4]	5	3
	0.9								[5]	2	2

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b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
UK	0.45	29466 nests	0.29		2000-2017	+	[6]	6	6
UK	0.49	433 nests		0.06	2003-2006	=	[7]	4	4
UK	St. Kilda: 0.96 (1996); Foula: 1.09 (1994-1996)	St. Kilda: 184 nests; Foula: 383 nests			1994-1996	+	[8]	2	4
UK	0.69 chicks per pair	767 nests	0.34		1989-1995		[4]	5	3
UK, 9 different sites	0.64 (site-specific estimates 0.33-0.88)	9 sites			1986-2005	variable	[9]	6	4
Age of first breeding	7 years						[10]		
Incidence of missed breeding	8.9 %	1020 birds		1.4 %	1989-1996		[4]		

VI.7 Arctic skua

Occurrence in and around North Sea
Arctic skuas breed in coastal (mostly the islands of) Scotland, Iceland, Norway and Sweden. Outside of the breeding season the species is widespread throughout the North Sea with numbers peaking during autumn migration. Fewer birds pass through the North Sea during spring migration (Furness 2015). Most birds present in the North Sea during autumn originate from breeding populations in the High-Arctic, Fennoscandia and Iceland with the remainder from Scottish populations (Furness 2015). Main wintering areas are in the southern hemisphere off Australia, South Africa and South America (Wernham *et al.* 2002). Breeding birds usually breed annually, although may fail to lay in years with poor food availability (Mitchel *et al.* 2004).

Available demographic data
Juvenile survival is estimated by O'Donald (1983) and Cook & Robinson (2010). These estimated juvenile survival rates based on these sources are similar (0.68 and 0.74 respectively).
Immature survival is only estimated by O'Donald (1983). The author reports 34.6% survival until breeding age. With a first-year survival of 0.74 (Cook & Robinson 2010) and an age of first breeding of 4, this gives an annual survival during the second, third and fourth year of 0.77.
Adult survival is estimated by Phillips & Furness (1998) and O'Donald (1983). Reported estimates by these sources are similar (0.9 and 0.886 respectively).

Data on fecundity are available for several sources, as reported in Table VI.7.

Phillips & Furness (1998) calculated an average age of first breeding of 4.385. With some individuals skipping breeding, we assume the age of first breeding to be 4 years. For arctic skua, the frequency of breeding is assumed to be lower in years with poor conditions (concerning food availability and/or predation). Catry *et al.* (1998) found 8% and 3% non-breeders among experienced breeders in 1993 and 1994. However, for this species, this is expected to be an underestimate for the annual proportion floaters. These proportions are based on experienced breeders, and individuals without breeding experience are not included in the analysis. The percentage of floaters is assumed to be between 15 and 25%. Within the population models, we assumed 25% floaters.

Implications for population model
We constructed a population model with 4 juvenile age classes, and sexual maturity from the age of 4 onwards. We assume a frequency of non-breeding of 25%.
No information is available to determine whether collision risk differs between age classes. Therefore, we assume equal collision risk for all age classes. In other words, we assume the distribution of age classes among victims to be according to the stable stage distribution of the population.

Table VI.7   Age-specific survival rates (VI.7a) and fecundity rates (VI.7b) of arctic skuas from different populations. Empty cells indicate no available information for this population. Local population trend: +: increasing. Data type: [a] Ringing programme. References: [1] O'Donald (1983); [2] Robinson (2018); [3] Cook & Robinson (2010); [4] Horswill & Robinson (2015); [5] Phillips & Furness (1998); [6] O'Donald et al. (1974); [7] Phillips et al. (1996); [8] Dawson et al. (2011); [9] Perkins et al. (2018); [10] Mavor et al. (2008); [11] Jones (2003); [12] Baber (1989); [13] Baber (1990); [14] Furness & Aitken (1992); [15] Catry et al. (1998).

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival	0,68								[1]; [2]	3	2
	0,74								[3]	3	3
Immature survival	0,346								[1]; [4]	3	2
Adult survival	0,9	324 individuals	0,009		1993-1995		UK (Foula)	[a]	[5]	5	3
	0,886								[1]; [2]	3	2

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. Trend	Reference	Data quality	Data representation
UK (Fair Isle)	1,4385	488 nests	0,6538		1949-1963	++	[6]	6	2
UK (Foula)	1,17	352 nests	0,1065		1993-1995		[5]	5	3
UK (Foula)	0,54 (range: 0,09 to 0,97)	9 years			1986-1994	-	[7]	5	4
UK (Fetlar)	0,399 (range: 0 to 1,6)	22 years (n=8-31)			1986-2007	untll 2001: -- , since 2001: stable	[8]	6	6
UK (several colonies)	0,91-0,29				1992-2015	-	[9]	5	6
UK (Shetland/Orkney)	0,4868	120 nests			1986-2005		[10]	5	4
UK	0,52				1986-2008		[3]	5	5
UK (Handa Island)	1,22	32 nests	0,1		2003		[11]	2	3
UK (Handa Island)	1,28	3 years (n=20-28)			1989-1991		[12]; [13]; [14]; [11]	2	3
Age of first breeding	4						[2]		
	4.396	101 individuals			1948-1959; 1970-1976		[1]		
Incidence of missed breeding	in 1993 8% of experienced breeders skipped breeding; in 1994 3% of experienced breeders skipped breeding	196 individuals			1993-1994		[15]		

VI.8 Common tern

Occurrence in and around North Sea
Common terns breed throughout Europe, including countries surrounding the North Sea. The majority of first-year birds remain off the west coast of Africa (between Senegal and Nigeria) with most second year birds returning north, albeit later than breeding birds (Wernham *et al.* 2002).

Available demographic data
For common terns in the Wadden Sea, van der Jeugd *et al.* (2014) report an average first-year survival of 0.616 (sd 0.03) including the period until fledging, and a survival of 0.685 for the period from fledging to the following spring.
The same study reports a second-year survival of 0.646 (sd 0.028), and an annual adult survival of 0.885. This is very similar to the findings of Becker & Ludwigs (2004), reporting 47% survival in the first 2 years after fledging (based on van der Jeugd *et al.* (2014), this is 0.685*0.646 = 44%). Becker & Ludwigs (2004) estimated the survival during the third year at 0.85, and the adult survival (from fourth year onwards) at 0.9. In contrast, van der Jeugd *et al.* (2014) found a better model fit when including individuals in their third year in the adult stage, and estimated adult survival (from third year onwards) at 0.885.

Data on fecundity are available for several sources, as reported in Table VI.8. Based on these sources, we calculated a weighted average (weighted by data quality and representativeness, for description, see in Chapter 2.1 and 3.1).

Common terns start breeding at the age of 3 (Becker & Ludwigs 2004; Robinson 2018). Becker & Ludwigs (2004) estimated the proportion of individuals with no breeding attempt to be 9%. However, note that this is based on experienced breeders. The percentage of floaters in the population model should include inexperienced breeders as well, and therefore be higher. In the current study, the proportion floaters is assumed to be 10%.

Implications for population model
We constructed a population model with 3 subadult age classes, and sexual maturity from the age of 3 onwards. We assume a frequency of non-breeding of 10%.

Table VI.8   Age-specific survival rates (VI.8a) and fecundity rates (VI.8b) of common terns from different populations. Empty cells indicate no available information for this population. Local population trend: -: decreasing. Data type: [a] ring recoveries. References: [1] Van der Jeugd et al. (2014); [2] Becker and Ludwigs (2004); [3] Becker et al. (2001); [4] Schekkerman et al. (2017); [5] Stienen et al. (2009), based on reports Griend study area.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival	0.685 from fledging to following spring	5743 ringed individuals			1991-2010	- (since 2000)	Wadden Sea	[a]	[1]	5	6
Immature survival	0.646 during second year	5743 ringed individuals			1991-2010	- (since 2000)	Wadden Sea	[a]	[1]	5	6
	0.47 survival in first 2 years after fledging						Germany		[2]; [3]	3	2
	0.85 survival during the third year						Germany		[2]; [3]	3	2
Adult survival	0.885, vanaf derde kalenderjaar	5743 ringed individuals			1991-2010	- (since 2000)	Wadden Sea	[a]	[1]	5	6
	0.9						Germany		[2]; [3]	3	2

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. Trend	Reference	Data quality	Data representation
Delta area (NL + BE), coastal	0.60	501 site-year combinations			1994-2016	variable, overall stable	[4]	5	6
Dutch Wadden Sea, coastal	0.33				1991-2010	- (since 2000)	[1]	5	6
Germany	1.30						[3]	3	2
Griend (NL), island	0.41 (min-max 0.00-1.00)	16 years	0.35		1992-2007	variable, overall stable	[5]	6	5

Age of first breeding	3 (first breeding in 4th calendar year)
Incidence of missed breeding	9% floaters among experienced breeders

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VI.9 Black tern

Occurrence in and around North Sea

Although the black tern has a wide range, the Dutch and adjacent German black tern breeding populations are relatively isolated within Europe and are of major importance for the West European population (van der Winden & Viksne 1997). The number of breeding pairs in the Netherlands and Germany strongly declined in the last century, most likely due to poor breeding success, and stabilized since 1990 (van der Winden & van Horssen 2008).

Black terns are present in the Netherlands from the second half of April until October (sovon.nl) and spend the winter in West Africa (van der Winden *et al.* 2014). Individuals in their first year stay in West Africa, and only come back to the breeding grounds at the age of two or three years, when first breeding takes place (Servello 2000; van der Winden & van Horssen 2008). Migration occurs both over land and over sea (del Hoyo *et al.* 1996; van der Winden *et al.* 2014).

In winter black terns are predominantly coastal and use inland waters in the vicinity of the coast, as well as marine waters up to 400-600 km offshore (BirdLife International 2018).

Available demographic data

Empirical data on survival rates of black tern are largely lacking (Table VI.9; Tinbergen & Heemskerk 2016). Van der Winden & van Horssen (2008) estimated survival rates based on empirical data on breeding success and the trend in numbers of breeding pairs. All available estimates are for breeding colonies in the Netherlands (Table VI.9).

After their first migration to the wintering grounds, individuals stay in this area during the second calendar year, and return for breeding only at the age of 2 or 3. Although some individuals start breeding at the age of 2 (Servello 2000), these individuals are rarely successful in their first breeding attempt (van der Winden & van Horssen 2008). Both van der Winden (2008) and Tinbergen & Heemskerk (2016) showed that reproductive rates of black tern in the Netherlands are not density-dependent at the local scale. Information about density dependence of survival rates is lacking.

Implications for population model

Data on survival of black terns are very limited (Table VI.9). The only available estimates are based on empirical data on reproduction and population trend (van der Winden & van Horssen 2008). As there is no indication of density dependent survival, we suggest assuming density independence.

For fecundity estimates from several colonies within the Netherlands are available (Table VI.9). Fecundity estimates vary substantially among these colonies. As a result, the impact may also strongly vary among different colonies and therefore we suggest simulating the dynamics using a range of fecundity rates. We suggest assuming a density independent fecundity rate based on literature (Table VI.9).

We suggest the use of a population model with a first-year stage, one sub-adult age class and an adult age class. Individuals start breeding at the age of 2, even though individuals at this age are rarely successful (van der Winden 2008). In field studies estimating breeding success, these unsuccessful breeding attempts are included. As the age of the adults is often unknown, it is not possible to identify an age-specific estimate of fecundity. In this estimate, 2-year old breeders (i.e. first breeders) are included. We therefore use a general estimate for fecundity, without age specificity. However, we assume that individuals of 2-year old individuals have a much lower probability of breeding than older individuals (80% vs. 10%).

As juveniles stay for 1.5-2.5 years in western Africa after their first migration, they will not be at risk of colliding with turbines in the North Sea. We suggest that the stage distribution of victims will be determined based on the stable stage structure of the population model, but excluding individuals in their second year in order to account for this stay-over in wintering grounds.

Table VI.9 Age-specific survival rates (VI.9a) and fecundity rates (VI.9b) of black terns from different populations. Empty cells indicate no available information for this population. Data type: [a] fit to observed population trend given measured fecundity data; Monte Carlo estimation. References: [1] van der Winden & Horssen (2008); [2] Tinbergen & Heemskerk (2016); [3] van der Winden (2008); [4] van der Winden (2005). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival [year 1, 2, 3; indirect estimate]	0.595				Based on population sizes 1993-1999	=	Netherlands	[a]	[1]	3	6
Immature survival											
[same as above] [year 1, 2, 3; indirect estimate]	0.595				Based on population sizes 1993-1999	=	Netherlands	[a]	[1]	3	6
Adult survival [>= year 4; indirect estimate]	0.849				Based on population sizes 1993-1999	=	Netherlands	[a]	[1]	3	6

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
Netherlands	0.99	7 colonies	0.2		2010-2015	=	[2]	6	6
Netherlands	0.66 (overall, also habitat-specific estimates)	721 bp	0.55		1993-1999	- or =, depending on habitat type	[1]	5	4
Netherlands	1.2 (in 2006, good year)	83 bp			2006	"= (+) stable, slight recovery	[3]	2	5
Netherlands	0.9 (before 1999 lower: <0.4 in 1995 and 1996)				1996-2003	=	[4]	2	5

Age of first breeding	3						[1]		
Incidence of missed breeding									

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VI.10   Bewick's swan

Occurrence in and around North Sea
The Bewick's swan's breeding range spans from the arctic to the Siberian tundra. Birds which spend the winter in Northwest Europe breed along the Barentsz Sea eastward to Vaigach and Nova Zembla (Nagy *et al.* 2012). Between 1996 and 2005, the majority of the wintering birds were recorded in the Netherlands (48-82%), followed by Great Britain (17-32%) (Rees & Beekman 2010).

The Northwest European winter population of Bewick's swans has declined by over 40%, since it reached peak size in 1995 (Rees & Beekman 2010; Nagy *et al.* 2012). As a consequence the species currently classifies as endangered on the European Red List of bird species (BirdLife International 2018). In recent years, numbers wintering in Greece have increased to over 8,000 and these include some birds that previously wintered around the North Sea (Litvin & Vangeluwe 2016, Wood *et al.* 2017), but this increase does not fully compensate for the decline in W-Europe.

Collision risk with wind turbines in the North Sea mainly arises when swans cross the (southern) North Sea during autumn and spring migration (and sometimes during cold winter weather on the continent). These crossings usually occur in an E-W direction between SE England (East Anglia) and (the northern part of) the Netherlands, but at least in spring also directly across the German Bight towards Schleswig-Holstein and Denmark (Griffin *et al.* 2016).

Available demographic data
Bewick's swans have high annual survival rates, and are long-lived (Rees & Beekman 2010). Age of first breeding usually is between 2 and 4 years of age (Evans 1979), with an average of 3.1 (Rees 2006).

Using a large dataset of almost 4000 individually marked and resighted Bewick's swans, Wood *et al.* (2017) calculated apparent survival with respect to age and sex class, time period and marker type. We suggest using the estimates based on legrings instead of neckrings, as these data have been collected throughout the study periods. Moreover, they seem more reliable as neckring-based mortality calculations may depress survival estimates. Wood *et al.* (2017) found no significant differences between the sexes. Survival estimates from mark-resight data are biased towards underestimates if permanent emigration from the study population occurs. Although Wood *et al.* (2017) found no clear support for an effect of increasing numbers of swans wintering in Greece on apparent survival in a statistical test across their entire time series, it is noteworthy that survival estimates dropped from 2008/09 onwards, exactly matching the onset of the increase in Greece. An emigration rate of 7% per year from a W European winter population of about 18,000 birds would result in a total number of c. 7600 emigrants over six years, close to the number now wintering in Greece. Also because Wood *et al.* (2017) could not identify environmental changes elsewhere that explain the drop in apparent survival, the possibility remains that it is (partly) an artefact

caused by swans emigrating to wintering sites in SE Europe rather than a true increase in mortality. However, only few marked birds previously wintering in Western Europe have been observed there so far and further studies are needed to explain this trend.

Wood *et al.* (2016) reported a brood size of 2.05 juveniles per family on UK wintering grounds between 1964 and 2014. For this measure, only successful broods are included. Instead, we use the number of first-year individuals on wintering grounds as an estimate for fecundity. Using the estimated survival during the first 6 months, the number of fledglings per adult can be calculated. This way, non-breeding adults are incorporated in this measure (as these adults are counted as well during survey, lowering the proportion of first year individuals). Hence, floaters are not separately defined in the population model. Annual proportions of juveniles in winter flocks are reported by Wood *et al.* (2016). These juvenile proportions (4-24%) have not shown a significant change over time.

Implications for population model
The demographic rates of Bewick's swans are relatively well-studied. Using the extensive dataset from Wood *et al.* (2017) and Wood *et al.* (2016), reliable estimates of stage-specific survival and fecundity are obtained. With respect to the uncertainty surrounding the lower survival rates reported for recent years, it seems wise to explore the effect of adopting either the most recent values or those for 2000-2010 when modelling potential wind farm effects. As Wood *et al.* (2017) found no sex differences in survival, and collision risk is not very likely to be sex-specific either, because the sexes usually migrate together, it is not necessary to construct a two-sex population model.

We suggest the use of a one-sex population model with a first-year stage, two subadult age classes, and an adult age class. Breeding takes place in the adult age class, from the age of 3 onwards, with a constant fecundity rate based on literature (Table VI.10). Survival differs between age classes.

It is unclear whether certain age classes experience higher collision risk, but this is perhaps not likely as swan families remain together over winter and migration generally occurs in larger flocks. Therefore, we suggest assuming constant collision risk, and divide the victims according to the stable stage structure.

Table VI.10 Age-specific survival rates (VI.10a) and fecundity rates (VI.10b) of Bewick's swan from different populations. Empty cells indicate no available information for this population. Data type: [a] neck collar resightings (alive and dead); Cormack-Jolly-Seber Mark Recapture; [b] leg ring and neck collar resightings. References: [1a] neck collar resightings, Wood et al. (2017); [1b] leg ring resightings, Wood et al. (2017); [2] Beer & Ogilvie (1972); [3] Nichols et al. (1992); [4] Wood et al. (2016); [5] Evans (1979); [6] Rees (2006). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativenes
Juvenile survival	♂ 0.72 ♀ 0.73	94 neck collars (1988-2014)		0.04	1970-2015; used data from 2000-2015	- [increase until 1995, then decline]	wintering grounds NW Europe	[a]	[1a]	6	6
	♂ 0.77 ♀ 0.78	3900 rings (1970-2014)		0.04	1970-2015; used data from 2000-2015	- [increase until 1995, then decline]	wintering grounds NW Europe	[a]	[1b]	6	6
Immature survival	♂ 0.79 ♀ 0.8	35 neck collars (1988-2014)		0.04	1970-2015; used data from 2000-2015	- [increase until 1995, then decline]	wintering grounds NW Europe	[a]	[1a]	6	6
	♂ 0.83 ♀ 0.83	568 leg rings (1970-2014)		0.04	1970-2015; used data from 2000-2015	- [increase until 1995, then decline]	wintering grounds NW Europe	[a]	[1b]	6	6
Adult survival	♂ 0.77 ♀ 0.79	475 neck collars (1988-2014)		0.02	1970-2015; used data from 2000-2015	- [increase until 1995, then decline]	wintering grounds NW Europe	[a]	[1a]	6	6
	♂ 0.81 ♀ 0.83	1890 ringed adults (1970-2014)		0.02	1970-2015; used data from 2000-2015	- [increase until 1995, then decline]	wintering grounds NW Europe	[a]	[1b]	6	6
	0.85	-							[2]	2	2
	0.92	5963 marked individuals		0.035	1966-1990	+	USA (North Carolina, Alaska)	[b]	[3]	6	0

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
NW Europe	2.05 juv/bp (wintering grounds)		0.4		1988-2013	-	[4]	6	6
UK	2.1 juv/bp (wintering grounds)				1963-1978		[5]	4	3
Age of first breeding	2 - 4 yrs						[5]		
	♂ 3.18 yrs ♀ 3.05 yrs		♂ 1.35 ♀ 1.12				[6]		
Incidence of missed breeding									

VI.11   Brent goose

Occurrence in and around North Sea

Brent geese are divided into three (sub)species: dark-bellied brent *Branta (bernicla) bernicla*, pale-bellied brent *B. (b.) hrota*, and black brent goose *B. (b.) nigricans*. Of these three (sub)species, the dark-bellied brent goose is by far the most abundant in the Netherlands. This population numbers about 246,000 individuals (Fox *et al.* 2010). Combining the population sizes of migrating birds in the Netherlands (max. 100,000, sovon.nl) and those that cross the North Sea from the Netherlands to the UK (50,000, Frost *et al.* 2017), we will define 150,000 individuals as the 'population' to be modelled.

Brent geese are present in the Netherlands from September through May, either wintering (up to 63,000 individuals) or on the way to wintering areas further south (c. 95,000 individuals) (Hornman *et al.* 2012). Nearly all of these birds are from the western Siberian breeding population (between Taimyr and Yamal peninsulas). Numbers of staging birds peak around the end of April and beginning of May, when individuals wintering further south and west migrate back to their breeding grounds (Voslamber & van Winden 2004).

The main wintering areas in the Netherlands are the Wadden Sea (up to 90%) and Delta regions (Hornman & van Winden 2013). Maximum numbers of brent geese in the Netherlands increased since the 1970s, and have remained stable since the 1990s (www.sovon.nl). Brent geese forage along the coast, both on saltmarshes and inland, mainly on grasslands (Hornman & van Winden 2013).

Available demographic data

Sedinger *et al.* (2007) performed a large-scale analysis of survival rates of black brent. Based on model performance, survival was estimated for two age classes: first-year individuals and individuals of at least 1 year old (second-year individuals and adults combined). However, for dark-bellied brent, Ebbinge *et al.* (2002) reported constant survival after 6 months of age, i.e. after arrival of juveniles on the wintering grounds. After 1972, when brent geese were no longer hunted in W-Europe, survival was higher (0.86) than in the period before (0.81; Summers & Underhill 1991; Ebbinge *et al.* 2002).

Breeding productivity in dark-bellied brent geese is measured on the wintering grounds as the proportion of juvenile birds in the population. Summers & Underhill (1991) reported that density dependent effects on either reproductive success or survival were not statistically significant. In contrast, Ebbinge *et al.* (2002) found significant density dependence in reproduction, but not in adult survival. This finding was corroborated by Nolet *et al.* (2013), although they showed that a larger part of the decline in productivity with population growth is explained by a declining frequency of lemming peaks allowing top breeding years. Nolet *et al.* (2013) conclude that density dependence only starts to have an effect at a population size above 200,000 individuals. It seems to be driven by the limited availability of suitable nesting habitat, although other factors such as competition for food cannot be excluded (Ebbinge *et al.* 2002; Nolet *et al.* 2013).

The numbers of fledglings per adult are calculated based on the number of first year individuals on wintering grounds per adult and the estimated survival during the first 6 months. Instead of the number of fledglings per breeding pair, this gives the number of fledglings per adult (breeding + non-breeding). For that reason, this measure incorporates non-breeding adults (floaters) as well. Hence floaters are not separately defined in the population model.

Pettifor *et al.* (2000) developed a spatially explicit, individual-based behavioural model for (dark-bellied) brent geese in order to assess the effect of habitat loss. This type of model is useful for studies on habitat loss and/or barrier effects, or for assessing differences in collision risk based on differences in spatial and/or temporal behaviour between age classes or sexes. For the assessment of population-level effects of turbine collisions, this method is not suitable, because it is rather the effects of direct mortality that have been studied than the effects of changes in behaviour. Therefore, it is in our case better to increase the mortality based on estimated collision mortality, and study the consequent changes in population dynamics.

Implications for population model
We will build a population model for the western Siberian (*B.(b.) bernicla*) population. Data on demographic rates for this population are quite robust and those from other populations are less appropriate due to variation in vital rates among populations. Estimates for adult survival in breeding populations in Alaska (referring to *Branta (bernicla) nigricans / orientalis*) are very similar to estimates from other areas, suggesting similar subadult survival as well (Table VI.11a; Sedinger *et al.* 2007; Ebbinge *et al.* 2002; Boyd 1962; Balmer & Peach 1997).

We suggest constructing a population model with a first-year stage, one sub-adult stage, and first reproduction at age 2 (third calendar year). With no indication for sex-specific vital rates or collision risk, we suggest a one-sex population model.
We assume that adult survival is constant over age and density-independent. As fecundity at the population level has shown a long-term decline, it is important to use values from a recent period. Nolet *et al.* (2013) found indications for density-dependent reproduction. If requested, this can be included in the population model by limiting fecundity at higher density.

No differences in collision risk are expected between sexes or age-classes. Therefore, we assume that the stage-distribution among the victims is similar to the stable stage distribution from the population model.

Table VI.11 Age-specific survival rates (VI.11a) and fecundity rates (VI.6b) of dark-bellied brent goose from different populations. Empty cells indicate no available information for this population. Data type: [a] ring recoveries; [b] estimated based on wintering population sizes. References: [1] Sedinger et al. (2007); [2] Robinson (2018); [3] Ebbinge et al. (2002); [4] Boyd (1962); [5] Balmer & Peach (1997); [6] Sedinger et al. (2002); [7] Cramp (1986); [8] Desholm (2009); [9] Nolet et al. (2013); [10] Nicolai (2003), Chapter 2; [11] WWT monitoring programme; https://monitoring.wwt.org.uk/our-work/goose-swan-monitoring-programme/species-accounts/dark-bellied-brent-goose; [12] Sedinger et al. (2006). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Juvenile survival	0.51, black brent	>17,000 ringed; >500 recovered		0.05	1986-2002	-	Alaska	[a]	[1]	2	2
	"no data"								[2]	0	2
Immature survival	0.83 from age 1 onwards)	>600 recovered (from age 1 onwards)		0.01	1986-2002	-	Alaska	[a]	[1]	6	2
	0.86 (survival on winter grounds)		0.05		1988-1998	<1990: + >1990: =	Several areas	[b]	[3]	6	4
	0.86 (first winter to breeding age (2))						UK		[4], [5]	0	3
Adult survival	0.83	since 1990 >600 recovered (from age 1		0.01	1986-2002	-	Alaska	[a]	[1]	6	2
	0.86 (from 1988-1998)		0.05		1988-1998	+	Several	[b]	[3]	6	4
	0.9	210 unique individuals		0.036	1991-1996	=	Individuals from arctic	[a]	[2], [6]	4	2
	0.85								[7], [8]	1	2

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
	0.21 1-year old individuals per adult	Based on counts on wintering grounds, 26 years	0.19		1990-2016	>1995: =/-, <1995: +	[9]	5	6
Alaska	1.43 goslings per brood	699 goslings, 459 broods; several locations	0.26		1999-2000		[10]	4	2
Several areas	Average family size since 1988: 2.645; is nr of offspring per family in winter				used data from 1988-1998 (total study period: 1955-1998)		[3]	5	5
UK	2.25 young per successful brood				2010-2017		[11]	3	5
	1.3						[7], [8]	0	2
Age of first breeding	2						[12]		
	2						[2]		
Incidence of missed breeding	[Not applicable, incorporated in fecundity measure]								

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VI.12 Shelduck

Occurrence in and around North Sea
Shelduck breed in all countries surrounding the North Sea (BirdLife International 2018). Breeding used to be restricted to coastal areas, but the frequency of breeding inland has increased in recent years (Linton & Fox 1991; Sovon 2002).

One-year old and older shelduck breeding in the UK have a clear moult-migration. Most individuals cross the North Sea to the Helgoland Bight (German Waddensea), where they join other birds from northern and western Europe (Wernham *et al.* 2002; Platteeuw 1980). Less extensive moult migration occurred between southern England and the Dutch Delta, though this region has lost significance as a moulting area. On the other hand, the Dutch Wadden Sea has become more important (Kleefstra *et al.* 2011). Some other shelduck remain in the UK and moult on North Sea estuaries (BirdWatch Ireland 2017). Garthe *et al.* (2007) studied seabird numbers in the German North Sea and observed high numbers of shelduck, exclusively along the coast. Juveniles do not join the moult migration (Eltringham & Boyd 1963; Wernham *et al.* 2002). In Sep-Nov, shelduck disperse from their moulting areas to wintering sites in the western Wadden Sea and estuaries in the UK and France, involving a second North Sea crossing for birds from the UK and Ireland.

Available demographic data
Only limited demographic data are available. Pienkowski & Evans (1982) compared reproductive rates and survival of colonial and non-colonial shelduck in the Firth of Forth, Scotland. In colonies, duckling survival was considerably lower (factor 6 difference) than at isolated sites, due to predation by herring gulls during competition between adults (Pienkowski & Evans 1982).
Several studies found that reproductive success in shelduck is density- dependent (Patterson 1982; Pienkowski & Evans 1982). They found a reduced number of fledglings per breeding pair at higher breeding densities. However, nowadays shelducks breed more dispersed. Therefore, older data are likely to be less representative for the current situation. For example, reproductive success is currently likely to be less density-dependent.

Implications for population model
Data on demographic rates of shelduck are rare (Table VI.12). Data for fecundity are limited, with one estimate for individuals in colonies and two estimates for individuals in isolated sites. Shelduck nowadays breed more dispersed, and hence data on fecundity in a colony are likely not representative. Due to a change from colonial to more dispersed breeding, reproduction is likely to be less dependent on density than previously found (Patterson 1982; Pienkowski & Evans 1982). Higher densities in colonies result in more interactions between individuals, which may limit reproduction (competition for food, nesting places, etc.). To our knowledge, no data are available on density dependence of reproduction in dispersed breeders. We therefore suggest no implementation of density dependence in the model.

Only one estimate of juvenile survival is available (Patterson *et al.* 1983). Although data are limited, information is available for each of the demographic rates necessary for constructing a population model.

We suggest using a one-sex population model with a first-year stage, one sub-adult age-class and an adult stage, parameterized with estimates from the literature. First reproduction is assumed at the age of 2 (Patterson *et al.* 1983).

Juveniles do not perform moult migration. Therefore, collision risk is likely lower, or even zero, for juveniles than for individuals of at least 1 year of age.

For that reason, we suggest to assume no victims among juveniles, and no difference in collision risk for the other, older age classes.

As available estimates or the population rates are relatively old, these might be less reliable. Using an elasticity analysis, which will be done for all species, we will assess the relative impact of changes in age-specific survival and fecundity on the model-outcome (see Chapter 2). Based on this analysis, it becomes clear which parameters strongly affect the outcome. If requested, we can run additional scenarios varying an uncertain demographic rate with a strong impact.

Table VI.12 Age-specific survival rates (VI.12a) and fecundity rates (VI.12b) of shelducks from different populations. Empty cells indicate no available information for this population. Data type: [a] Observations, number of ducklings vs number of fledglings; [b] ringing, combined with identification using paint; [c] colour rings. References: [1] Pienkowski & Evans (1982); [2] Patterson et al. (1983); [3] Robinson (2018); [4] Lensink (2001). Data quality and representativeness are assessed based on the criteria described in Chapter 2.

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data
Duckling survival up to fledging	isolated: 0.167; colonies: 0.73	colony: 482 ducklings; isolated: 160 ducklings			1977-1979	=	Scotland, Firth of Forth	[a]	[1]	1	3
Juvenile survival	0.166 to age 2				1962-1979		Scotland, Aberdeenshire	[b]	[2], [3]	3	3
Immature survival	-- (0.166 to age 2)				1962-1979		Scotland, Aberdeenshire	[b]	[2], [3]	3	3
Adult survival	0.886 (M 0.909, F 0.880)				1962-1979		Scotland, Aberdeenshire	[b]	[2], [3]	2	3
	Successful breeders (colony): 0.893; Summer residents: 0.826	187 successful breeders; 483 summer residents			1971-1978	=	Scotland, Firth of Forth	[c]	[1]	2	3

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
NL, river Waal	0.7-1.1 over all pairs; 2.8-4.6 per successful breeding pair	276 breeding attempts (successful + unsuccessful)			1971-1975; 1992-2001	+	[4]	5	5
Scotland, Firth of Forth	Colony: 0.207 Isolated sites: 0.943	colony: 65; isolated 24 (over 3 years)	colony 0.1 isolated 0.19		1976-1979	=	[1]	3	3

Age of first breeding	2						[2], [3]		
Incidence of missed breeding									

VI.13   Curlew

Occurrence in and around North Sea

Eurasian curlews in W-Europe previously mostly bred in heathland, peat moor and dunes, but nowadays mainly use grasslands on sandy soil or peat land (e.g. Sovon 2002). The species is widely distributed, breeding in large parts of Europe including river valleys and bogs in Scandinava and Russia. Wintering occurs mainly in estuarine areas in northwest and south Europe, Africa, the Middle East, and Asia (BirdLife International 2018). Individuals breeding in the Netherlands move to the southwest, mainly to France (juveniles) and the UK (adults, www.vogeltrekatlas.nl). Birds breeding in the UK move southwest and winter on the coasts of the UK and Ireland and down to France as far southwest as Portugal. There is little evidence that birds breeding in the UK move across the North Sea (Wernham *et al.* 2002). However, most birds wintering in the UK have crossed the southern North Sea (Wernham *et al.* 2002). Birds wintering in the Wadden Sea and estuaries around the North Sea originate from breeding areas ranging from The Netherlands and N Germany to Scandinavia and N Russia as far east as the Ural Mountains (www.vogelatlas.nl; Wernham *et al.* 2002).

Available demographic data

Data on adult survival and fecundity are available from several sites (Table VI.13; Roodbergen *et al.* 2012, Klok *et al.* 2009), though from W Europe rather than from boreal regions where most of the population breeds. Older estimates might be less reliable, as these estimates are from the period when curlews were still hunted in several countries.

Several studies from different areas report estimates of adult survival (Table VI.13). Estimates vary between 0.82 and 0.885. Estimates of fecundity are also reported for different areas (Table VI.13) and vary between 0.235 and 0.34.

There are no data on density dependence of vital rates.

Implications for population model

Estimates are available for each of the parameters necessary for the population model (Table VI.13). However, some estimates stem from relatively old studies and might therefore be less reliable. We suggest using a one-sex population model, parameterized with estimates weighted by source quality and representativeness. For pre-adult survival no recent estimates are available. The impact of changes in subadult survival can be assessed based on the elasticity analysis, which is done for each species.

Data on density-dependent survival and/or fecundity are not available. We therefore construct models without density dependence.

Regarding the implementation of additional mortality due to turbine collisions, we suggest assuming that victims are divided among the age classes according to the stable stage structure based on the population model.

Table VI.13 Age-specific survival rates (VI.13a) and fecundity rates (VI.13b) of curlews from different populations. Empty cells indicate no available information for this population. Data type: [a] review; [b] dead recoveries, Lack method; [c] ringing recoveries, Lack method, amended by Haldane; [d] live resightings. References: [1] Roodbergen et al. *(2012); [2] Glutz von Blotzheim* et al. *(1984); [3] Bainbridge & Minton (1978); [4] Kipp (1982); [5] Berg (1991); [6] Grant* et al. *(1999); [7] Evans & Pienkowski (1984); [8] Robinson (2018). Data quality and representativeness are assessed based on the criteria described in Chapter 2.*

a.

Demographic rate	Mean	n	SD	SE	Study period	Local population trend	Location	Data type	Reference	Data quality	Data representativeness
Chick survival (up to fledging)	0.49	3 studies		6.08	[review]; between 1996-2006	[review]	[review]	[a]	[1]	6	4
Juvenile survival	0.34				1969-1974		NL	[b]	[2]	1	3
	0.47 (from fledging to 1yr old)	153 recoveries			1909-1975		UK	[c]	[3]	3	2
Immature survival	0.63 (2nd cy)	50 recoveries			1909-1975		UK	[c]	[3]	3	2
Adult survival	0.885				1973-1980	-	Germany	[d]	[4]	2	4
	0.82				1985-1992	+	Sweden	[d	[5]	2	2
	0.72				1969-1974		Netherlands	[b]	[2]	2	3
	0.82-0.88				[meta-analysis]	variable	variable	[a]	[6]	3	2
	0.74 (>2nd cy)	74 recoveries		0.025	1909-1975		UK	[c]	[3]	3	2
	0.736 ± 0.025	66 years					UK	[b]	[7]	4	2

b.

Demographic rate, location	Mean (fledg/bp, unless otherwise stated)	n	SD	SE	Study period	Local pop. trend	Reference	Data quality	Data representativeness
[review]	0.34	based on 250 studies		0.02	1996-2006	[review]	[1]	6	3
Germany	0.3				?, <1982	-	[4]	2	4
Sweden	0.25				?, <1991		[5]	2	3
N-Ireland	0.29 (variation between years: 0.14-0.56)	352 breeding pairs (102 fledglings)	0.17		1993-1995	-	[6]	3	3
Age of first breeding	2 or (more likely) 3						[3]		
	2						[8]		
Incidence of missed breeding									

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VII Origin of expected victims

This appendix lists some ideas and thoughts about the populations of origin of birds that are expected as collision victims in the southern North Sea.

VII.1 Lesser black-backed gull

Based on the predictions in the EIAs, most of the lesser black-backed gull victims in Dutch waters are expected in March-July (Fijn *et al.* 2016). In May-July most of these will belong to breeding populations of southeastern UK, Belgium and the Netherlands (*L. fuscus graellsii* and *L. fuscus intermedius*), probably with a small addition of immature birds from German, Danish and Norwegian populations. In Feb-Apr and Aug-Oct, nearly all the lesser black-backed gulls from breeding areas in Germany, Denmark, S- and W-Norway and SW-Sweden also pass through the southern North Sea (www.vogeltrekatlas.nl, Bakken *et al.* 2003, Bønløkke *et al.* 2006, Fransson *et al.* 2008, Bairlein *et al.* 2014).

Movements seem to differ between individuals, which is likely to affect the collision risk. Using GPS tracking data from individuals breeding in a colony on the Dutch island of Texel, Camphuysen *et al.* (2015) showed that males spend considerably more time at the North Sea than females, which mostly fed on land or nearshore and in the Wadden Sea. The same was true in the UK where males spent more time at sea than females later in the breeding season (Thaxter *et al.* 2015). In active breeders, the amount of time spent in marine habitats (North Sea and coastal waters) during long trips amounted to 34% in females and 78% in males (Camphuysen 2011). In contrast, Gyimesi *et al.* (2011) showed that individuals breeding in a different colony in the Netherlands, Lake Volkerak, mainly use terrestrial foraging locations. As most GPS studies are carried out on adult birds, not much is known about the whereabouts of immatures. Furness (2016) points out that, especially during summer, the North Sea may hold large numbers of immatures. Decades of ship-based observations at sea have shown, however, that between March and July, the majority (c. 90%) of birds present are in adult plumage (Camphuysen 2013; Leopold 2017).

VII.2 Great black-backed gull

Only few great black-backed gulls breed in the Netherlands (Boele *et al.* 2017). Numbers present in the Dutch North Sea are much higher during winter (Fijn *et al.* 2016). Based on the predictions in the EIAs, great black-backed gull victims in Dutch waters are expected mainly between September and March (Gyimesi *et al.* 2016). Most individuals present in the Dutch North Sea probably come from breeding populations all over Fennoscandinavia and east to the Kola Peninsula in NW Russia, as well as the east coast of the United Kingdom (vogeltrekatlas.nl; Bakken *et al.* 2003, Wernham *et al.* 2002, Furness 2016). In particular, subadult individuals from northern breeding populations are present in and around the Netherlands during spring and summer as well (www.vogeltrekatlas.nl), most likely because these individuals are not yet restricted to breeding grounds.

VII.3 Herring gull

Based on the predictions in the EIAs, the vast majority of herring gull victims in Dutch waters are expected between December and February (Gyimesi *et al.* 2016). In autumn and summer, collision risk is expected to be lower due to lower densities on the North Sea (Stone *et al.* 1995; Camphuysen 2013). During the breeding season, most individuals are concentrated around the colonies, whereas during the rest of the year they spend most of their time along the coasts and at sea (Camphuysen 2013).
Herring gulls breeding in the Netherlands mostly stay during winter or move no further south than Belgium or northern France (Camphuysen 2013). Additionally, individuals breeding in northern and north-eastern Europe spend the winter in the Netherlands as well. These are probably mainly birds from Norway and the Kola Peninsula in NW-Russia. Also large numbers of herring gulls from the UK use the North Sea in winter. Herring gulls from the Baltic region also reach the southern North Sea, though most of those seem to remain in the southern Baltic and Denmark (Bakken *et al.* 2003, Fransson *et al.* 2008, Saurola *et al.* 2013). In winter, about 170.000 herring gulls are present in the North Sea, and about 160.000 individuals are present in Dutch inland areas (Hornman *et al.* 2012). Camphuysen & Leopold (1994) studied the distribution of, for example, herring gulls on the southern North Sea based on ship surveys between 1985 and 1993. On average, 67% of the observed individuals are adult. The percentage of adults varies throughout the year, with on average 94% adults in July and 17% in September.

VII.4 Kittiwake

Based on the predictions in the EIAs, most kittiwake victims in Dutch waters are expected between November and March (Gyimesi *et al.* 2016). Most individuals present in the North Sea come from breeding populations in the North Sea itself (c. 40%, mainly UK), the Barentsz Sea (c. 40%), western Norway (c. 10%), and SW Britain and France (c. 10%; Frederiksen *et al.* 2012).

The distribution of juvenile kittiwakes is poorly understood (Furness 2016). Adults are likely to remain close to breeding colonies in summer. Further away from the colonies, including in the southern North Sea, juveniles are overrepresented, particularly so during the breeding season (Furness 2016).

VII.5 Great skua

The main peak in migration over the North Sea is in September and October (Camphuysen & van Dijk 1983; Platteeuw *et al.* 1994; trektellen.nl). Hence, this is the period of maximum collision risk with turbines in this area.

Furness (2015) estimated the numbers of individuals in the UK North Sea and Channel waters, based on several data sources such as trektellen.nl, ringing data and geolocator data. Based on this review, most individuals originate from the UK. Approximately 10% of the individuals are expected to stem from breeding colonies outside UK, mostly

Iceland. In addition, lower numbers come from populations in Norway (www.vogeltrekatlas.nl, Bakken *et al.* 2003). Birds from Iceland and Svalbard winter partly in the NE-Atlantic (including the North Sea) and partly in the NW-Atlantic off Canada (Magnusdottir *et al.* 2012). Based on these sources and population sizes a tentative breakdown of the origin of birds present in the North Sea would be 90% UK/Faroes, 5% Iceland, and 5% Norway/Barentsz sea (Furness 2015).

GPS-data from the UK shows that non-breeding adults spend more time at sea further away from the colony than breeding adults (Wade *et al.* 2014).

VII.6   Brent goose

Nearly all individuals present in the Dutch North Sea originate from the western Siberian breeding population (dark-bellied brent). These constitute flocks crossing between the Wadden Sea and wintering sites in England and flocks short-cutting the concave Dutch coastline between North-Holland and Zeeland and further south towards France. This short-cutting is a common feature during both spring and autumn migration, mainly in October-November, March to mid-April and late May (e.g. Camphuysen & Van Dijk 1983; Platteeuw *et al.* 1994; www.trektellen.org). Short-cutting may also occurs between the Dutch Wadden Sea islands and Denmark (Dokter & Ebbinge 2013).
A minority of the brent geese flying over the North Sea originates from the pale-bellied population breeding in Svalbard. Significant numbers (but even then well under 1000 birds) reach the Netherlands only in severe winters, when displaced from traditional wintering sites in Denmark (Cottaar *et al.* 1999).

As brent goose pairs and their young of the year stay (and move) together throughout the winter or even during the first return migration, no age or sex differences in collision risk are expected.

VII.7   Shelduck

Shelduck victims are expected to come from areas around the North Sea. Individuals are present around the Dutch North Sea throughout the year, mostly coming from breeding populations in the Netherlands and the UK and some individuals from Denmark and France (BirdLife International 2018; www.vogeltrekatlas.nl). Among these, UK and Irish breeding birds are most likely to be found in the more offshore parts of the southern North Sea, during sea crossings.

In case movement patterns or habitat use differ between age classes, collision risk may also vary. Juveniles do not moult in their first calendar year (with consequently lower collision risk) but one-year olds do (Eltringham & Boyd 1963; Wernham *et al.* 2002). Whether flight height and habitat use on a smaller scale differ between age classes and/or between sexes is unclear.

VII.8   Curlew

Curlews are present around the Dutch North Sea throughout the year. Curlew collision victims in the Dutch North Sea are expected to originate from breeding populations ranging from the Netherlands through Scandinavia (particularly Finland) and northern Russia that cross the North Sea following breeding and again in late winter early spring (www.vogeltrekatlas.nl; Wernham *et al.* 2002).

Data on flying behaviour of different age classes further offshore are limited. In a pilot study, four curlews resting in the Wadden Sea were tagged with GPS-loggers in order to gain information on their breeding grounds and movement patterns (Schwemmer *et al.* 2016). All adults (n=3) migrated to their breeding grounds in (north-) eastern Russia in April. There is no indication of sex-specific movement patterns.

VII.9   Black tern

Black terns are present in the area around the Dutch North Sea between mid-April and October (van der Winden *et al.* 2014). As they breed inland and do not carry out feeding flights offshore, they are not at risk of colliding during the breeding season. They only potentially come within offshore wind farms during spring and autumn migration, i.e. April-May and mid July-Oct (Camphuysen & van Dijk 1983; Platteeuw *et al.* 1994, trektellen.nl). Individuals in the Dutch North Sea are expected to originate from Dutch breeding colonies as well as from Sweden, Denmark, Finland, Germany and Eastern Europe (www.vogeltrekatlas.nl).

Collision risk with turbines in the North Sea may vary between age classes. After migrating to western Africa in their first calendar year, individuals stay here for 1.5 - 2.5 years, and come back for the breeding season at the age of 2 or 3 (Servello 2000; van der Winden & van Horssen 2008). Turbines in the North Sea will not affect those individuals during this time (but note that local turbines in wintering areas possibly affect black terns during this time).

VII.10   Bewick's swan

A large proportion of Bewick's swans winters in the Netherlands and the UK. These individuals stem from populations around the Barents Sea (Rees 2006, Nagy *et al.* 2012). Some of these birds migrate over land (via northern Germany) towards the Netherlands, from where individuals wintering in the UK pass the Dutch North Sea (Gyimesi *et al.* 2017). Others cross the German bight from Denmark (Gyimesi *et al.* 2017).

Fijn *et al.* (2012) showed that Bewick's swans avoid wind turbines in flight at a terrestrial wintering site in the Netherlands, but collision victims, especially during the night, cannot be ruled out. Age classes and sexes do not differ in their distribution; indeed pairs and family groups stay together during autumn migration and most of the winter (Rees 2006). Therefore collision risk can be expected to be similar between age classes and sexes.

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VIII Stochastic Collision Rate Modelling

A. Potiek, J.J. Leemans & A. Gyimesi, April 3rd, 2019

1. Introduction

In 2018, Marine Scotland published the Stochastic Collision Risk Model (sCRM)[2]. This model is based on the SOSS Band model (Band 2012), but allows more detailed input data to be used than the SOSS Band model, specifically in relation to modelling variability around certain parameters. This translates into a range of estimates being produced as opposed to single figures. Therefore, the model has the ability to calculate standard deviations around the mean monthly numbers of expected collisions. This gives an indication of the uncertainty around the estimated collision rate.

Within the population models presented in Potiek *et al.* (2019), all input parameters except for collision rate vary between years and iterations. This simulates variation between years, and incorporates uncertainty of the estimates. As a result, the outcome of the population model is not one certain value (for example population growth rate). Incorporation of variation in input parameters gives a more realistic range of likely outcomes. In addition to variation in survival, fecundity and breeding probability, stochasticity in the collision rate further improves the outcome of the population models.

Collision rate not only varies between years, but also between individuals. For example, a given individual may spend more time at rotor height than another individual, resulting in a higher collision risk. Moreover, the quality of the estimation of a collision rate depends on the quality of the input data. If input parameters are relatively uncertain, the calculated collision rate based on these input parameters will be uncertain as well. Depending on the uncertainty of input parameters, the outcome may be relatively uncertain as well. Note that some input parameters affect the outcome more strongly than others (Chamberlain *et al.* 2006 for previous collision rate model (Band 2012)).

This appendix presents (the uncertainty in) the cumulative number of collision victims of six seabird species across wind farms in the southern North Sea for several scenarios with different input parameters, and describes the implications of introducing uncertainty in collision mortality into the population models of these species.

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[2] Available at https://www2.gov.scot/Topics/marine/marineenergy/mre/current/StochasticCRM

2. Methods

2.1 sCRM

The sCRM calculations presented in this appendix are done for the same seabird species and wind farms as used in Gyimesi *et al.* (2018), who calculated the cumulative number of collision victims across wind farms in the southern North Sea under the research framework '*Kader Ecologie en Cumulatie*' (KEC) (Rijkswaterstaat 2019). All sCRM simulations were performed in R (R core team 2017).

Different scenarios

For each species, we ran 12 different scenarios in which we used different combinations of values for avoidance rate, wind turbine size and time in operation, plus one extra scenario in which all values were equal to those used in the KEC (Gyimesi *et al.* 2018) (Table 2.1). In addition, we used different flight height distributions in scenario 1 to 12, than in the 13th scenario (i.e. the *KEC scenario*). More details are given in the paragraphs below. For each species, 1000 iterations per scenario have been run.

Table 2.1. Overview of used values for avoidance rate, size, time in operation, and flight height distribution in each scenario. Turbine sizes used in the KEC study are by 12 of the 15 turbine types equal to the 'Min-scenario' and only deviate by the largest turbines (see Table 2.4).

Scenario	Avoidance rate	Turbine size	Time in operation	Flight height distribution
1	KEC	Min	100% operational	sample from individual-based distributions
2	KEC	Min	Realistic fluctuating	sample from individual-based distributions
3	KEC	Min	Realistic average	sample from individual-based distributions
4	KEC	Max	100% operational	sample from individual-based distributions
5	KEC	Max	Realistic fluctuating	sample from individual-based distributions
6	KEC	Max	Realistic average	sample from individual-based distributions
7	Bowgen & Cook (2018)	Min	100% operational	sample from individual-based distributions
8	Bowgen & Cook (2018)	Min	Realistic fluctuating	sample from individual-based distributions
9	Bowgen & Cook (2018)	Min	Realistic average	sample from individual-based distributions
10	Bowgen & Cook (2018)	Max	100% operational	sample from individual-based distributions
11	Bowgen & Cook (2018)	Max	Realistic fluctuating	sample from individual-based distributions
12	Bowgen & Cook (2018)	Max	Realistic average	sample from individual-based distributions
KEC-scenario	KEC	KEC	KEC: 100% operational	KEC: average flight-height distribution

Bird data

For each species, we varied the values for avoidance rates between different scenarios. Avoidance rates were either equivalent to those used in the KEC (i.e. as in Gyimesi *et al.* 2018), or to those presented in Bowgen & Cook (2018) (Table 2.2).

Table 2.2. Overview of the avoidance rates of each species used in the different scenarios, originating from either the KEC or Bowgen & Cook (2018).

Species	Avoidance rate	
	KEC	Bowgen & Cook (2018)
kittiwake	0.995	0.98
great black-backed gull	0.995	0.993
great skua	0.995	0.995[a]
herring gull	0.995	0.993
lesser black-backed gull	0.995	0.993
northern gannet	0.995	0.995

[a] not presented in Bowgen & Cook (2018), thus taken as equal to KEC

We incorporated variability in the body length, wingspan and flight speed of each species by adding standard deviations (Table 2.3). Therefore, the model randomly sampled a value for these parameters in each iteration based on a normal (zero-truncated) distribution with given mean and standard deviation. Standard deviations of body length and wingspan of each species were calculated based on ranges given by Snow and Perrins (1998) and the assumptions that the middle of this range was the mean value and that all data falls within three standard deviations from the mean. Standard deviations of flight speeds of herring gull and lesser black-backed gull were calculated based on data from GPS tags placed on birds in Dutch, Belgian, and British colonies around the Southern North Sea (Gyimesi *et al.* 2017a), while we used standard deviations of flight speeds of great black-backed gull and kittiwake as reported in Alerstam *et al.* (2007) and of great skua and northern gannet as reported in Pennycuick (1990). Due to a lack of data, we did not incorporate standard deviations of nocturnal activity. Mean values of all bird parameters were equivalent to Gyimesi *et al.* (2018). Also, we used the same bird density data as in Gyimesi *et al.* (2018).

The sCRM has the ability to randomly sample a flight height distribution in each iteration from a list of different flight height distributions. Therefore, we incorporated more variability in the model by adding different flight height distributions for each species in scenario 1 to 12. Flight height distributions of lesser black-backed gull and herring gull were calculated based on data from GPS tags placed on birds in Dutch, Belgian, and British colonies around the Southern North Sea (Gyimesi *et al.* 2017a). We used a separate distribution for each individual bird with more than 1,500 data points. For great black-backed gull, we sampled from two different distributions from Swedish and Danish logger data (Gyimesi *et al.* 2017b), and one distribution as used in the KEC, which is based on Johnston *et al.* (2014). Furthermore, we generated 200 different flight height distributions of kittiwake, great skua and gannet, by sampling from a zero-truncated normal distribution, with means and standard deviations based on 95% confidence intervals presented per height class in Johnston *et al.* (2014). In the 13th scenario, we used a single flight height distribution for each species, based on Johnston *et al.* (2014) and in accordance with the KEC.

Table 2.3. Mean and standard deviations (sd) used in the sCRM for body length, wingspan, flight speed and nocturnal activity of each bird species.

species	body length (m)		wingspan (m)		flight speed (m/s)		nocturnal activity	
	mean	sd	mean	sd	mean	sd	mean	sd
kittiwake	0.39	0.003	1.08	0.04	13.10	0.40	0.50	0
great black-backed gull	0.71	0.023	1.58	0.03	13.70	1.20	0.50	0
great skua	0.56	0.008	1.36	0.01	14.90	3.80	0	0
herring gull	0.60	0.015	1.44	0.02	11.34	3.91	0.01	0
lesser black-backed gull	0.58	0.020	1.43	0.03	9.41	3.92	0.43	0
northern gannet	0.94	0.022	1.73	0.03	14.90	2.60	0.25	0

Turbine parameters

For the size of wind turbines, we used a minimum and maximum scenario per different megawatt (MW) wind turbine in scenario 1 to 12 by varying the rotor diameter and hub height, based on a study in windfarm Borssele (Fijn *et al.* 2015) (table 2.4). In the 13th scenario, we used the same values for rotor diameter and hub height as in the KEC. The turbine power used in different wind farms was in accordance with the KEC. Also, turbine parameters rotor speed, blade width and pitch were equal to the KEC in all scenarios. Furthermore, we simulated three different variants of time in operation, being 1) 100% operational in all months, as used in the KEC, 2) realistically fluctuating throughout the year according to published operational times of wind farm East Anglia 3 (APEM 2015), and 3) a constant realistic average (92.58%) throughout the year, also based on East Anglia 3.

Table 2.4. Overview of the different values used for rotor diameter and hub height per turbine power.

turbine power (MW)	rotor diameter (m)			hub height (m)		
	min	max	KEC	min	max	KEC
2	80	80	80	60	60	65
3	100	121	100	75	86	75
3.3	112	112	112	79	79	81
3.6	120	130	120	85	90	85
4	116	140	116	83	95	83
5	129	156	129	90	103	90
6	142	171	142	96	111	96
7	153	185	153	102	118	102
8	164	198	164	107	124	107
8.4	164	203	164	107	127	111
9	174	210	174	112	130	112
9.5	164	216	164	107	133	105
10	183	221	221	117	136	141
12	201	242	220	126	146	145
15	224	271	232	137	161	142

2.2   Population models

Age structured population models are used to predict dynamic changes in seabird populations due to additional mortality. The methods to do so are described in more detail in Potiek *et al.* (2019). In summary, a Leslie matrix model is defined, in which the numbers of individuals per age class are modelled based on age-specific survival and fecundity. The impact of (age-specific) mortality from turbine collisions was assessed by decreasing the age-specific survival. The additional mortality is based on the estimated proportion of collision victims, and the distribution of age classes among the collision victims.

Potiek *et al.* (2019) presented the results as a fixed percentage of collision victims. This appendix presents stochastic proportions of collision victims, and shows the impact of incorporating this stochasticity into the population models. This is done by randomly drawing the proportion of collision victims for each iteration and each year from a specified distribution. This random distribution is defined based on the average proportion of victims and the standard deviation of this estimate found in the analysis described in paragraph 2.1.

3. Results

3.1 sCRM

A summary of the mean cumulative number of collisions with standard deviations for each species across different scenarios is given in Table 3.1.

On average, great black-backed gull showed the highest number of collisions of all the species (2,795 ± 568). This is contrary to the KEC scenario in which lesser black-backed gull had more than twice as many victims as great black-backed gull. This difference is due to the input of different flight height distributions in both species, as this changed the proportion of birds flying through the rotor-swept area. For example, in one of the three flight height distributions of great black-backed gull from which we sampled, the proportion of birds flying below the average lowest blade tip height (25m) was just 19%, instead of 75% in the distribution used in KEC, meaning that a much greater proportion of birds flew at collision risk height in the KEC. Sampling from a list with separate flight height distributions of individual lesser black-backed gulls resulted in a substantial lower number of collisions (1,303 ± 330), than when using a single average flight height distribution as in KEC (1,990 ± 363). For herring gull, using such individual flight distributions resulted in lower collision numbers as well, although this difference was smaller than for lesser black-backed gull (669 ± 179 vs. 780 ± 128).

Simulations for kittiwake, great skua and northern gannet all showed on average higher collision mortality in scenario 1 to 12 than in the KEC scenario. In the case of great skua and northern gannet, this higher mortality is likely to be the consequence of slightly larger proportions of birds flying through the rotor-swept area, as a result of the input of different flight height distributions than in KEC. For kittiwake, the difference with the KEC scenario is mainly explained by the use of different avoidance rates.

Table 3.1. Summary of the mean collision numbers with standard deviation for each species in each scenario (n=1000).

Scenario	kittiwake		great black-backed gull		great skua		herring gull		lesser bl.-backed gull		northern gannet	
	mean	sd	mean	sd	mean	sd	mean	sd	mean	sd	mean	sd
1	367	5.6	2,366	245	16.1	1.6	560	114	1,119	220	361	20
2	342	5.1	2,204	229	14.7	1.5	521	107	1,026	197	334	18
3	340	5.3	2,195	232	14.9	1.5	520	108	1,035	200	335	18
4	356	4.9	2,528	264	15.8	1.3	606	114	1,179	204	365	17
5	331	4.6	2,360	252	14.5	1.2	563	106	1,074	181	337	15
6	329	4.5	2,344	246	14.7	1.2	561	106	1,086	183	337	15
7	1,469	23	3,327	366	16.2	1.6	791	165	1,566	308	362	20
8	1,369	20	3,078	332	14.8	1.5	731	149	1,430	277	335	18
9	1,360	21	3,053	323	15.0	1.5	732	149	1,447	282	335	18
10	1,422	20	3,528	386	15.9	1.3	857	157	1,645	281	365	17
11	1,326	18	3,290	341	14.5	1.2	797	150	1,504	253	338	16
12	1,317	18	3,265	335	14.7	1.2	789	147	1,527	256	338	16
Avg. (1-12)	861	518	2,795	568	15.2	1.5	669	179	1,303	330	345	22
KEC	324	4.4	763	30	3.7	0.3	780	128	1,990	363	244	10

Of the three parameters that we varied between each scenario, avoidance rate generally had the largest effect on collision numbers (Table 3.2). Chamberlain *et al.* (2006) found a similar strong impact of avoidance rate on the outcome of the SOSS Band model (Band 2012). The avoidance rate of 98% for kittiwake (scenario 7-12) resulted in more than four times as many collision victims than an avoidance rate of 99.5%. Similarly, slightly lower avoidance rates for great black-backed gull, herring gull, and lesser black-backed gull (0.993 vs. 0.995) resulted in substantially more victims. Furthermore, the effect of time in operation is proportional to the difference in its value between the scenarios. For example, the realistic average operational time of 92.58% resulted for all species in a decrease in collision numbers of about the same percentage (i.e. 92.5%). Finally, turbine size showed no strong or univocal effect on collision numbers.

Table 3.2. Comparison of the mean collision numbers of all six species between the different values used for avoidance, turbine size, and time in operation in scenario 1 to 12.

	lesser bl.-backed gull	great black-backed gull	herring gull	kittiwake	great skua	northern gannet
Avoidance						
- KEC	1,087	2,327	555	344	15.1	345
- Bowgen & Cook	1,520	3,250	783	1,377	15.2	346
Turbine size						
- min	1,271	2,704	643	875	15.3	344
- max	1,336	2,916	696	847	15.0	347
Time in operation						
- 100% operational	1,377	2,937	704	904	16.0	363
- Realistic fluctuating	1,259	2,641	653	842	14.6	336
- Realistic average	1,274	2,714	651	837	14.8	336

3.2 Selection scenarios and calculation proportion of victims

In order to assess how the use of a stochastic collision risk impacts the outcome of the population model and impact assessment, we compare the outcome of the population models with a fixed proportion of collision victims with the outcome of the population models with stochastic proportions of collision victims. This is done for lesser black-backed gull, great black-backed gull, herring gull and black-legged kittiwake. Gannet is not included since no population model incorporating collision victims was constructed for this species.

Within this paragraph, we compare the results of the models with deterministic collision rates (as reported in Chapter 4) with the results of the population models with stochastic additional mortality. Both the population model with deterministic collision rates and the model with stochastic collision rates have been run for 50,000 iterations. More detailed results of the model with deterministic collision rates are presented in Chapter 4 of Potiek *et al.* (2019).

To illustrate the impact of incorporating stochasticity in body length, wingspan, flight speed and, most importantly, flight height distribution, the output of the population model with stochastic collision rates is compared to the output of the population model with a deterministic collision rate (see Box 1). In the rest of this paragraph, the scenario with stochastic collision rates is referred to as the KEC-sCRM scenario.

Proportion of victims

Within the population models, the proportion of victims is used as input parameter instead of the number of victims. The proportion of victims is calculated as the number of estimated victims based on the collision rate model, divided by the summed bimonthly counts based on MWTL/ESAS boat and aerial surveys (see Potiek *et al.* 2019; Rijkswaterstaat 2019). This summed bimonthly count represents the highest possible number of individuals present in the North Sea in a year, assuming that every two months, all individuals are replaced by a new set of individuals. In reality, individuals are often present in several bimonthly periods. This cautious approach represents a worst-case scenario, estimating the maximum number of victims.

For example, in case of lesser black-backed gull, the summed bimonthly counts is 367,543 individuals. Based on this value, and the output of the CRM and sCRM, the proportions of victims are presented in Table 3.3 for the stochastic scenario with individual-based flight height distributions, the stochastic scenario with average flight height, and the deterministic CRM.

For lesser black-backed gull, the numbers of victims for the stochastic scenario with individual-based flight height distributions, the stochastic scenario with an average flight height distribution, and the deterministic scenario are reported in Table 3.3. The proportion of collision victims among lesser black-backed gulls is 0.304% of all individuals in the North Sea according to the results of the sCRM-scenario with stochastic flight height distribution (scenario 1 in Table 3.1; comparable to KEC scenario, but with variation in body length, wing span and flight speed, and with individual-based flight height distributions). If flight height distribution is not assumed stochastic, the outcome of the sCRM (with stochastic body length, wing span and flight speed) is 0.541%, which is very similar to the outcome of the deterministic CRM as presented in Rijkswaterstaat (2019) (0.557%). Hence, stochasticity in flight height distribution strongly affects the estimated number

of collision victims. Whereas in Rijkswaterstaat (2019) an average flight height distribution is used for all individuals, it is now assumed that individuals may vary in flight height distribution. This variation is modelled by giving each individual an individual-based flight height distribution from available data based on data loggers (see Paragraph 2.1).

Table 3.3. Comparison of the mean collision numbers of lesser black-backed gull between the deterministic collision rate model (Rijkswaterstaat 2019) and a selection of scenarios described in paragraph 2.1. For stochastic collision rates, means and standard deviations are reported. In both reported stochastic CRMs, body length, wingspan and flight speed are stochastic.

	Deterministic CRM	Stochastic CRM	
		Comparable to KEC scenario, with average flight height	Comparable to KEC scenario, but with individual-based flight height distributions
Number of victims	2,046	1,990 +- 363	1,119 +- 220
Proportion victims	0.00557	0.00541+- 0.0010	0.00304 +- 0.0006

For great black-backed gull, the numbers of victims for the stochastic scenario with individual-based flight height distributions, the stochastic scenario with an average flight height distribution, and the deterministic scenario are reported in Table 3.4. The proportion of collision victims among great black-backed gulls is 0.545% of all individuals in the North Sea according to the results of the sCRM-scenario with stochastic individual-based flight height distribution (scenario 1 in Table 3.1; comparable to KEC scenario, but with variation in body length, wing span and flight speed, and with individual-based flight height distributions). If flight height distribution is not assumed stochastic, the outcome of the sCRM (with stochastic body length, wing span and flight speed) is 0.176%, which is very similar to the outcome of the deterministic CRM as presented in Rijkswaterstaat (2019) (0.180%). Hence, stochasticity in flight height distribution strongly affects the estimated number of collision victims. Whereas in Rijkswaterstaat (2019) an average flight height distribution is used for all individuals, it is now assumed that individuals may vary in flight height distribution. This variation is modelled by giving each individual an individual-based flight height distribution from available data based on data loggers (see Paragraph 2.1).

Table 3.4. Comparison of the mean collision numbers of great black-backed gull between the deterministic collision rate model (Rijkswaterstaat 2019, including OWEZ and PAWP) and a selection of scenarios described in paragraph 2.1. For stochastic collision rates, means and standard deviations are reported. In both reported stochastic CRMs, body length, wingspan and flight speed are stochastic.

	Deterministic CRM	Stochastic CRM	
		Comparable to KEC scenario, with average flight height	Comparable to KEC scenario, but with individual-based flight height distributions
Number of victims	781	763 +- 30	2,366 +- 245
Proportion victims	0.00180	0.00176 +- 0.00007	0.00545 +- 0.00056

For herring gull, the numbers of victims for the stochastic scenario with individual-based flight height distributions, the stochastic scenario with an average flight height distribution, and the deterministic scenario are reported in Table 3.5. The proportion of collision victims among herring gulls is 0.12% of all individuals in the North Sea according to the results of the sCRM-scenario with stochastic individual-based flight height distribution (scenario 1 in Table 3.1; comparable to KEC scenario, but with variation in body length, wing span and flight speed, and with individual-based flight height distributions). If flight height distribution is not assumed stochastic, the

outcome of the sCRM (with stochastic body length, wing span and flight speed) is 0.16%, which is very similar to the outcome of the deterministic CRM as presented in Rijkswaterstaat (2019) (0.16%). Hence, stochasticity in flight height distribution has a stronger effect on the estimated number of collision victims than stochasticity in the other variables (body length, wing span and flight speed).

Table 3.5. Comparison of the mean collision numbers of herring gull between the deterministic collision rate model (Rijkswaterstaat 2019, including OWEZ and PAWP) and a selection of scenarios described in paragraph 2.1. For stochastic collision rates, means and standard deviations are reported. In both reported stochastic CRMs, body length, wingspan and flight speed are stochastic.

	Deterministic CRM	Stochastic CRM	
		Comparable to KEC scenario, with average flight height	Comparable to KEC scenario, but with individual-based flight height distributions
Number of victims	776	780 +- 128	560 +- 114
Proportion victims	0.0016	0.0016 +- 0.00027	0.0012 +- 0.00024

For kittiwake, the numbers of victims for the stochastic scenario with individual-based flight height distributions, the stochastic scenario with an average flight height distribution, and the deterministic scenario are reported in Table 3.6. The proportion of collision victims among kittiwakes is 0.044% of all individuals in the North Sea according to the results of the sCRM-scenario with stochastic individual-based flight height distribution (scenario 1 in Table 3.1; comparable to KEC scenario, but with variation in body length, wing span and flight speed, and with individual-based flight height distributions). If flight height distribution is not assumed stochastic, the outcome of the sCRM (with stochastic body length, wing span and flight speed) is 0.039%. In comparison, the outcome of the deterministic CRM as presented in Rijkswaterstaat (2019) is 0.043%. Hence, incorporation of stochasticity in flight height distribution, body length, wing span or flight speed does not strongly affect the estimated number of collision victims.

Table 3.6. Comparison of the mean collision numbers of kittiwake between the deterministic collision rate model (Rijkswaterstaat 2019, including OWEZ and PAWP) and a selection of scenarios described in paragraph 2.1. For stochastic collision rates, means and standard deviations are reported. In both reported stochastic CRMs, body length, wingspan and flight speed are stochastic.

	Deterministic CRM	Stochastic CRM	
		Comparable to KEC scenario, with average flight height	Comparable to KEC scenario, but with individual-based flight height distributions
Number of victims	355	324 +- 4.4	367 +- 5.6
Proportion victims	0.00043	0.00039 +- 0.000005	0.00044 +- 0.000006

For great skua, the numbers of victims for the stochastic scenario with individual-based flight height distributions, the stochastic scenario with an average flight height distribution, and the deterministic scenario are reported in Table 3.7. The proportion of collision victims among great skuas is 0.0186% of all individuals in the North Sea according to the results of the sCRM-scenario with stochastic individual-based flight height distribution (scenario 1 in Table 3.1; comparable to KEC scenario, but with variation in body length, wing span and flight speed, and with individual-based flight height distributions). If flight height distribution is not assumed stochastic, the outcome of the sCRM (with stochastic body length, wing span and flight speed) is 0.0043%, which

is very similar to the outcome of the deterministic CRM as presented in Rijkswaterstaat (2019) (0.0048%). Hence, incorporation of stochasticity in body length, wing span or flight speed does not strongly affect the estimated number of collision victims, but stochasticity in flight height distribution does.

Table 3.7. Comparison of the mean collision numbers of great skua between the deterministic collision rate model (Rijkswaterstaat 2019, including OWEZ and PAWP) and a selection of scenarios described in paragraph 2.1. For stochastic collision rates, means and standard deviations are reported. In both reported stochastic CRMs, body length, wingspan and flight speed are stochastic.

	Deterministic CRM	Stochastic CRM	
		Comparable to KEC scenario, with average flight height	Comparable to KEC scenario, but with individual-based flight height distributions
Number of victims	4.15	3.7 +- 0.3	16.1 +- 1.6
Proportion victims	0.000048	0.000043 +- 0.000003	0.000186 +- 0.00002

3.3 Impact of stochastic collision rates on output population models

Within this paragraph, we describe the impact of incorporating stochastic collision rates on the population level, using metrics as reported in Potiek *et al.* (2019). These include the median and distribution (5% and 95% percentile) of the growth rate, relative final population size (impacted relative to null scenario) and the overlap between the null and impacted scenario for each of the selected species. In addition, we report the proportion of runs with more than 10% decline within 30 years.

Note that for the project in which the impact of collision mortality was to be modelled (Potiek *et al.* 2019), northern gannet was not among the selected study species. Therefore, no results for northern gannet are presented within this chapter.

3.3.1 Lesser black-backed gull

Table 3.3.1 presents the outcomes of the population models for the deterministic KEC-scenario and two sCRM scenarios (with and without stochasticity in flight height distribution). The median projected population growth rate for the deterministic KEC-scenario is 0.9976, i.e. a slight annual decline by 0.2%. This is very similar to the median projected growth rate for the sCRM scenario without stochasticity in flight height distribution (0.9974). This shows that stochasticity in body length, wing span and flight speed does not strongly affect the median outcome. The 5% and 95% quantiles also show similarity in the width of the distribution of projected population growth rates for these two scenarios (see also Figure 3.3.1).

In contrast, due to lower estimated number of victims in the sCRM with stochastic individual-based flight height distribution (in addition to stochasticity in body length, wing span and flight speed), the median projected growth rate is higher (1.000). As a result, the scenario with stochasticity in body length, wing span, flight speed and flight height distribution, projects a 10% lower population size after 30 years compared to the null scenario (relative population size after 30 years is 0.905), whereas for the deterministic KEC scenario and the stochastic scenario without stochasticity in flight height distribution the final population size after 30 years was 16-17% lower than the null scenario. Hence, inclusion of stochasticity in flight height distribution results in a smaller assessed impact than using the deterministic CRM.

However, note that the distributions of the scenario based on two stochastic CRMs and the deterministic CRM strongly overlap with each other, and with the null scenario. The percentage unaffected lower than median affected in Figure 3.3.1 shows that the overlap in output of the population model is somewhat smaller between the sCRM scenario with individual-based flight height distributions and the null scenario, than between the other scenarios and the null scenario.
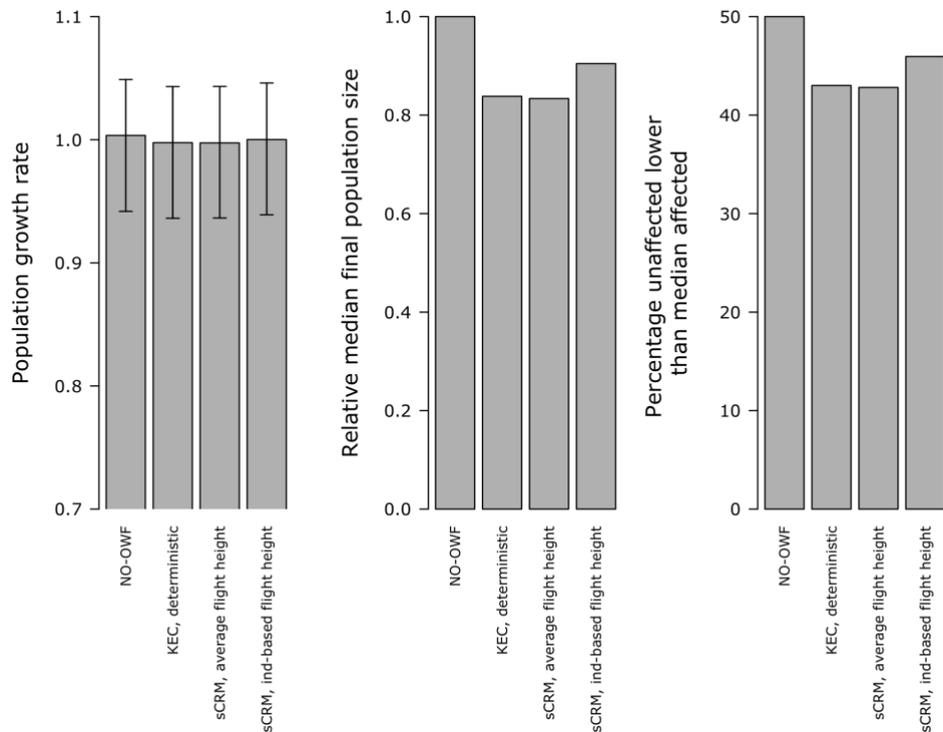
Fig. 3.3.1   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario.

Table 3.3.1 Summary of model output for the scenario without additional mortality (null scenario), the scenario with additional mortality according to the deterministic collision rate model (KEC-scenario), and the scenario with additional mortality according to the stochastic collision rate model (KEC-sCRM-scenario). Numbers of victims for KEC-scenario are based on Rijkswaterstaat (2019). PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found. The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0035	0.9418	1.0489	42%		
deterministic CRM (KEC)	0.9976	0.9362	1.0432	49%	0.838	43%
stoch. CRM; av. flight height	0.9974	0.9365	1.0433	49%	0.834	43%
stoch. CRM; ind. flight height	1.0001	0.9391	1.0461	46%	0.905	46%

3.3.2 Great black-backed gull

Table 3.2.2 summarizes the outcomes of the population models for the deterministic KEC-scenario and two sCRM scenarios (with and without stochasticity in flight height distribution). The median projected population growth rate for the deterministic KEC-scenario is 0.9854, i.e. an annual population decline by 1.5%. This is very similar to the median projected growth rate for the sCRM scenario without stochasticity in flight height distribution (0.9859). This shows that stochasticity in body length, wing span and flight speed does not strongly affect the median outcome. The 5% and 95% quantiles also show similarity in the width of the distribution of projected population growth rates for these two scenarios.

In contrast, the sCRM scenario with a stochastic individual-based flight height distribution, in addition to stochasticity in body length, wing span and flight speed, projects a lower median population growth rate of 0.9821 (i.e. an annual population decline by 1.8%). As a result, the scenario with stochasticity projects a 15% lower population size after 30 years compared to the null scenario, whereas for the deterministic KEC scenario this was only 5% lower than the null scenario.

Please note that Gyimesi *et al.* (2017) mention that flight altitude measurements of great black-backed gull with GPS loggers are very scarce and only collected during the breeding season. Whether flight altitudes differ in winter is unknown and thus the results presented here should be regarded as indicative only.

The percentage unaffected lower than median affected in Figure 3.2.2 shows that the overlap in output of the population model is somewhat smaller between the sCRM scenario with individual-based flight height distributions and the null scenario, than between the other scenarios and the null scenario. However, the left panel of Figure 3.2.2 shows that the distributions of population growth rates still strongly overlap.

Fig. 3.3.2   *Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario. In each of the three subfigures, the first bar shows the null scenario without additional mortality through collisions with turbines. The second bar shows the scenario based on the deterministic collision rate calculated in the KEC-study (Rijkswaterstaat 2019). The third bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in body length, wingspan and flight speed (no stochasticity in flight height distribution). The last bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in flight height distribution (individual-based, see paragraph 2.1), body length, wingspan and flight speed.*

Table 3.3.2   *Summary of model output for the null scenario, the scenario with additional mortality according to the deterministic collision rate model (KEC-scenario; Rijkswaterstaat 2019), and two scenarios with additional mortality according to the stochastic CRM: one without stochasticity in flight height distribution, and one with stochastic flight height distribution. PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found. The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9873	0.8919	1.0644	57%		
deterministic CRM (KEC)	0.9854	0.8897	1.0629	58%	0.945	49%
stoch. CRM; av. flight height	0.9859	0.8891	1.0631	58%	0.958	49%
stoch. CRM; ind. flight height	0.9821	0.8860	1.0587	61%	0.853	46%

3.3.3   Herring gull

Table 3.3.3 summarizes the outcomes of the population models for the deterministic KEC-scenario and two sCRM scenarios (with and without stochasticity in flight height distribution). The median projected population growth rate for the scenario based on the deterministic CRM is 0.9816, i.e. an annual population decline by 1.8%. For the scenario based on the stochastic CRMs (with and without stochasticity in flight height distribution) the median projected growth rates are very similar to the scenario based on the deterministic CRM (0.9815 and 0.9823). This shows that stochasticity in body length, wing span and flight speed does not strongly affect the median outcome.
The 5% and 95% quantiles also show similarity in the width of the distribution of projected population growth rates for these scenarios.

The scenarios based on deterministic and stochastic collision rates project a similar probability on a 10% decline after 30 years (63%), compared to a 62% probability on such a decline for the null scenario. Hence, the probability of a 10% decline is very similar for the null scenario as for each of the scenarios with additional mortality.

The projected final population size for the scenarios based on the sCRM with stochasticity in body length, wingspan and flight speed (but not flight height distribution) is 5 percent lower than for the null scenario (relative population size after 30 years is 0.95), which is similar to the scenario based on the deterministic CRM. For the scenario based on the sCRM with stochasticity in flight height distribution as well, the relative population size is 2.3% lower than for the null scenario (relative population size is 0.977).

The percentage unaffected lower than median affected in Figure 3.3.3 shows a strong overlap in output of the different population models. This overlap can be seen in the distribution of population growth rates in the left panel of Figure 3.3.3 as well.

Fig. 3.3.3   *Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario. In each of the three subfigures, the first bar shows the null scenario without additional mortality through collisions with turbines. The second bar shows the scenario based on the deterministic collision rate calculated in the KEC-study (Rijkswaterstaat 2019). The third bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in body length, wingspan and flight speed (no stochasticity in flight height distribution). The last bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in flight height distribution (individual-based, see paragraph 2.1), body length, wingspan and flight speed.*

Table 3.3.3   *Summary of model output for the null scenario, the scenario with additional mortality according to the deterministic collision rate model (KEC-scenario; Rijkswaterstaat 2019), and two scenarios with additional mortality according to the stochastic CRM: one without stochasticity in flight height distribution, and one with stochastic flight height distribution. PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found. The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9831	0.9001	1.0510	62%		
deterministic CRM (KEC)	0.9816	0.8982	1.0494	63%	0.955	49%
stoch. CRM; av. flight height	0.9815	0.8989	1.0493	63%	0.952	49%
stoch. CRM; ind. flight height	0.9823	0.8989	1.0494	63%	0.977	49%

3.3.4   Kittiwake

Table 3.3.4 further summarizes the outcomes of the population models for the deterministic KEC-scenario and two sCRM scenarios (with and without stochasticity in flight height distribution). The median projected population growth rate for the scenario based on the deterministic CRM is 0.9880, i.e. an annual population decline by 1.2%. For the scenario based on the stochastic CRMs (with and without stochasticity in flight height distribution) the median projected growth rates are 0.9880 as well. This shows that stochasticity in body length, wing span and flight speed does not strongly affect the median outcome.

The 5% and 95% quantiles also show similarity in the width of the distribution of projected population growth rates for these scenarios.

The scenarios based on deterministic as well as stochastic collision rates project a 57% probability on a 10% decline after 30 years, compared to a 56% probability on such a decline for the null scenario.

The projected final population size for the scenarios with deterministic or stochastic collision rates is 2 percent lower than for the null scenario (relative population size after 30 years is 0.98).

The percentage unaffected lower than median affected in Figure 3.3.4 shows the strong overlap in output of the population models for different sCRM scenarios and the null scenario.
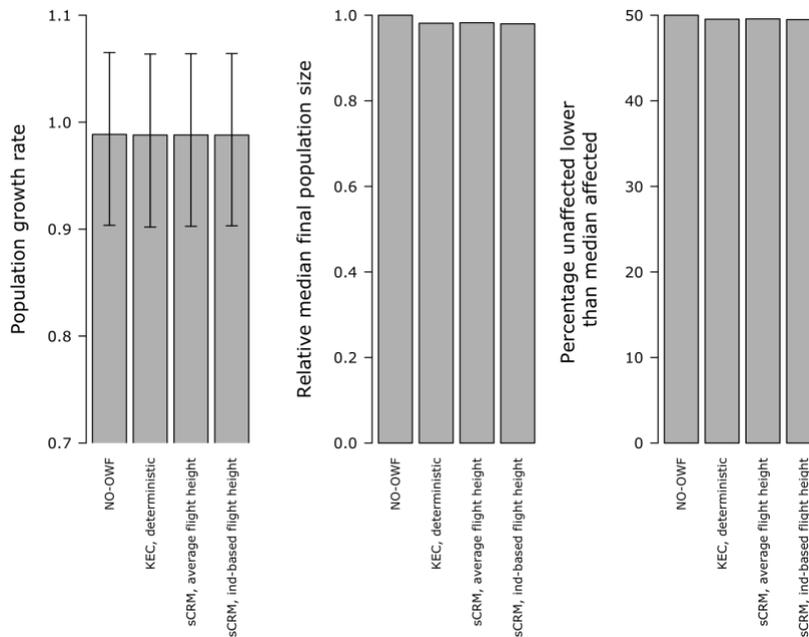
Fig. 3.3.4 *Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario. In each of the three subfigures, the first bar shows the null scenario without additional mortality through collisions with turbines. The second bar shows the scenario based on the deterministic collision rate calculated in the KEC-study (Rijkswaterstaat 2019). The third bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in body length, wingspan and flight speed (no stochasticity in flight height distribution). The last bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in flight height distribution (individual-based, see paragraph 2.1), body length, wingspan and flight speed.*

Table 3.3.4 *Summary of model output for the null scenario, the scenario with additional mortality according to the deterministic collision rate model (KEC-scenario; Rijkswaterstaat 2019), and two scenarios with additional mortality according to the stochastic CRM: one without stochasticity in flight height distribution, and one with stochastic flight height distribution. PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found. The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.*

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	0.9886	0.9036	1.0651	56%		
deterministic CRM (KEC)	0.9880	0.9020	1.0637	57%	0.981	50%
stoch. CRM; av. flight height	0.9880	0.9027	1.0640	57%	0.982	50%
stoch. CRM; ind. flight height	0.9880	0.9032	1.0642	57%	0.980	49%

3.3.5   Great skua

Table 3.3.5 summarizes the outcomes of the population models for the deterministic KEC-scenario and two sCRM scenarios (with and without stochasticity in flight height distribution) for great skua. The median projected population growth rate for the deterministic KEC-scenario is 1.0155, i.e. an annual increase by 1.55%. This is very similar to the median projected growth rate for each of the sCRM scenarios.

Hence, incorporating stochasticity in body length, wing span and flight speed does not strongly affect the median population growth rate. The 5% and 95% quantiles also show similarity in the width of the distribution of projected population growth rates for these two scenarios (see also figure 3.3.5).

Due to a higher estimated number of victims in the sCRM with stochastic individual-based flight height distribution (in addition to stochasticity in body length, wing span and flight speed), this scenario gives a slightly lower relative final population size (0.981) compared to the scenario based on the deterministic CRM (0.993) and the scenario based on the sCRM with only stochasticity in body length, wing span and flight speed (0.989).

Please note that Gyimesi *et al.* (2017) mention that flight altitude measurements of specifically great skua with GPS loggers come with very large error margins. The results presented here should thus be regarded as indicative only.

The percentage unaffected lower than median affected in Figure 3.3.1 shows the overlap in output of the population model for different CRM scenarios. Note that the distributions of the scenario based on two stochastic CRMs and the deterministic CRM strongly overlap with each other, and with the null scenario.
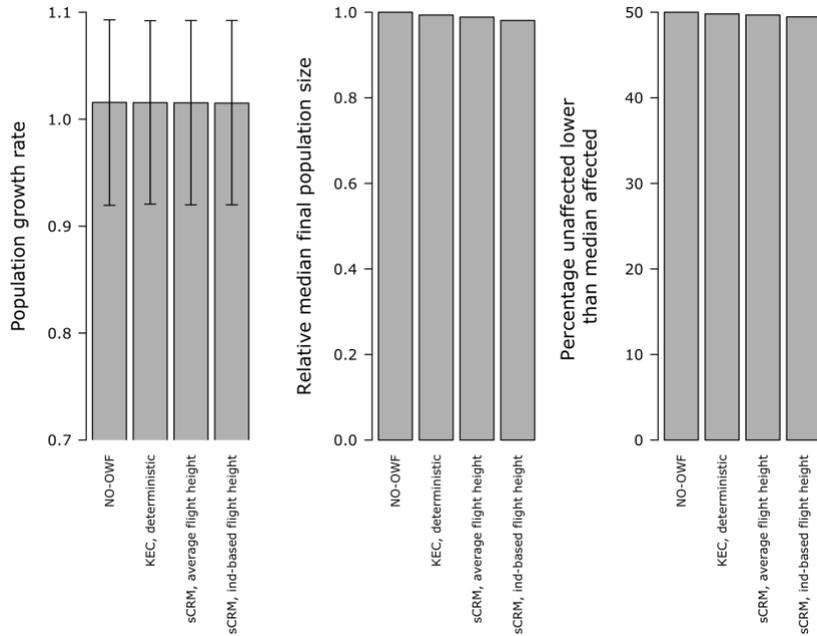
Fig. 3.3.5   Distribution of population growth rates, relative population sizes (final median population size divided by final median population size without additional mortality) and the probability of the null scenario to be below the median of the affected scenario. In each of the three subfigures, the first bar shows the null scenario without additional mortality through collisions with turbines. The second bar shows the scenario based on the deterministic collision rate calculated in the KEC-study (Rijkswaterstaat 2019). The third bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in body length, wingspan and flight speed (no stochasticity in flight height distribution). The last bar shows the scenario based on the stochastic collision rate calculated in this study with stochasticity in flight height distribution (individual-based, see paragraph 2.1), body length, wingspan and flight speed.

Table 3.3.5   Summary of model output for the null scenario, the scenario with additional mortality according to the deterministic collision rate model (KEC-scenario; Rijkswaterstaat 2019), and two scenarios with additional mortality according to the stochastic CRM: one without stochasticity in flight height distribution, and one with stochastic flight height distribution. PGR is the population growth rate (lambda). 5% and 95% quantiles represent the value below which 5% and 95% of the values are found. The following column shows the proportion of runs with a population decline of 10% or more after 30 years compared to the initial population size. The relative population size after 30 years is calculated as the final population size of the impacted scenario divided by the final population size of the unimpacted scenario. The last column shows the percentage of the results of the null scenario, which have a final population size below the median of the affected scenario.

	PGR	5% quantile	95% quantile	Prop. runs more than 10% decline after 30 years	Relative population size after 30 years	Perc. results unaffected below median affected
null scenario	1.0157	0.9195	1.0929	36%		
deterministic CRM (KEC)	1.0155	0.9206	1.0921	36%	0.993	50%
stoch. CRM; av. flight height	1.0153	0.9199	1.0923	37%	0.989	50%
stoch. CRM; ind. flight height	1.0150	0.9200	1.0923	36%	0.981	49%

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5. Literature

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IX Other anthropogenic sources of mortality

Besides collisions with wind turbines, other anthropogenic sources of mortality can impact bird populations. Based on the available literature, we assess how our study species may be impacted by different anthropogenic sources of mortality. Human activity may directly or indirectly result in bird mortality. Here we focus on direct mortality only; indirect mortality resulting through processes such as habitat loss are not taken into account.

5.2.1 Main anthropogenic sources of mortality

Loss *et al.* (2015) reviewed sources of direct anthropogenic mortality using the following categories: collisions with vehicles and manmade structures; poisoning with toxins; and predation by free-ranging pets. The authors did not study species-specific mortality, but looked at total numbers of birds killed. Erickson *et al.* (2005) made a similar categorization of anthropogenic sources of direct mortality, expanding the categorization of Loss *et al.* (2015) with electrocutions and commercial fishing by-catch.

In general, it is difficult to obtain data on how much additional mortality each anthropogenic source causes. This is mainly because the majority of victims remain unreported. Moreover, studies often focus on the impact of one specific anthropogenic source, making it hard to compare between sources of mortality. Although it is possible to get an idea of the relative impact of different sources of anthropogenic mortality by comparing ring recoveries, reporting rates can vary greatly between different sources of mortality. For example, the probabilities of finding a victim, knowing the cause of death and reporting this correctly differs greatly between sources of anthropogenic mortality. Certain causes of mortality may be more obvious whereas some causes may not be apparent to the reporter. For example, a bird found along a road will frequently be reported as a traffic victim, whereas for a dead bird found elsewhere the cause of death may not be initially apparent.

Collisions with vehicles
Birds may collide with cars, trains or airplanes, which may impact populations (Reijnen & Foppen 2006). Some species are more often found as victims from collisions with vehicles than others. Based on a review by Erritzoe *et al.* (2003), in western Europe the species dying most frequently on the roads are sparrows and blackbirds, whereas in Central and Eastern Europe not only sparrows but also corvids and barn swallows make up a high proportion of the victims. Based on the website waarneming.nl, an online platform of wildlife observations, tawny owl, common buzzard and mallard were the most reported road traffic victims in the Netherlands (Table 5.2.1). Note that this source is likely to be biased towards the more obvious and/or more spectacular species.

Of the study species in our study, herring gull is the only species in the top 10 of road traffic victims reported in the Netherlands (Table 5.2.1). Loss *et al.* (2014) reviewed species-specific data available from the USA, and conclude that barn owls are especially at risk. In Norway, corvids and gulls were the most frequently reported victims from collisions with vehicles (Husby 2016; Table 5.2.2). Note that Husby (2016) reported relatively higher proportions of gulls compared to waarneming.nl; and although gulls are quite likely to represent a higher proportion of victims in Norway, this may be in part due

to the inclusion of unidentified individuals as gull sp., while the list compiled from waarneming.nl only includes individuals identified to the species level.

Jacobson (2005) reviewed the species groups at risk of collisions with road traffic, and concluded that the main species groups are gallinaceous birds, waterbirds (such as terns), owls, ground nesters and scavengers. Note that this is a general classification, with some overlap between species groups.

Dolbeer (2006) assessed species composition of airplane strikes below and above 500 feet (ca. 150 m). Below 500 feet, the main species groups among collision victims were passerines, gulls and terns, pigeons and doves, and raptors. Above 500 feet, species-groups most frequently struck were wildfowl, gulls and terns, passerines, and vultures.

Table 5.2.1 Reported numbers of road traffic victims in the Netherlands per species between 2009 and 2018. Source: waarneming.nl

Species	Number of roadkills
Tawny owl	3001
Common buzzard	1511
Mallard	1486
Coot	1100
Blackbird	804
Black-headed gull	602
Wood pigeon	515
Long-eared owl	510
Herring gull	418
Grey heron	357

Table 5.2.2 Percentage of species groups among roadkills, as found by Husby (2016) in Norway.

Species-group	Percentage of roadkills
Gulls	28%
Common gull	*5.8%*
Herring gull	*0.8%*
Great black-backed gull	*0.8%*
Gull sp.	*20.6%*
Corvids	33%
Small passerines	22%
Thrushes	12%
Others (e.g. oystercatcher and tawny owl)	5%

Collisions with buildings and man-made structures

In addition to collisions with wind turbines, birds are known to collide with structures such as buildings, power lines and communication towers (Erickson 2005; Loss *et al.* 2015). In their review, Loss *et al.* (2015) conclude that collisions with buildings is the second largest cause of anthropogenic mortality in the USA, and the third largest cause of anthropogenic mortality in Canada (Fig. 5.2.1). Both Loss *et al.* (2014) and Arnold & Zink (2011) studied the species composition of victims of collisions with buildings in

North America, and both found mainly species on migration. To the best of our knowledge, the species composition of victims from collisions with buildings in Europe has not yet been studied. Although victims can be entered in the database of waarneming.nl, this is not done systematically. The German organisation NABU estimates that collisions with windows represent 5-10% of all bird mortality in Germany, with species using the urban habitat intensively are expected to be at most risk (NABU 2019).

Poisoning

Another form of anthropogenic mortality is poisoning. This is mainly poisoning from pesticides or illegal bait poisoning. The use of pesticides has been linked to bird population declines worldwide (Mineau 2005; Mineau & Whiteside 2013; Goulson 2013, 2014; Walker 2003). The use of neonicotinoids as insecticide has also been associated with population declines in the Netherlands (Hallmann *et al.* 2014). Oil, particularly oil spills, seems to mainly play an important factor of anthropogenic mortality for species found at sea or along the coast (Buij *et al.* 2018). Although, gulls and terns are less affected by oil spills (Buij *et al.* 2018).

Predation by free-ranging cats

The total extent of bird mortality caused by domestic cats is speculative, but widely considered to be very large (Loss *et al.* 2013). Loss *et al.* (2013) reviewed literature on bird mortality due to cat predation in the United States. Table 5.2.3 shows the top 10 most found species, as reported in Loss *et al.* (2013), based on several studies. Although this table shows the majority of species to be passerines, and also those species that occur close to human habitation, these figures are likely to be influenced by reporting rates. Nevertheless, in general it could be expected that most cat-related mortality to occur close to human habitation and affect the species that occur there. Similar species could be expected to be victim from cat predation in Europe and species of interest for our study are therefore unlikely to be strongly impacted by cat predation. Exceptions to this are feral cat populations that can result in high levels of mortality away from populated area, such as in seabird colonies (Towns *et al.* 2011).

Table 5.2.3 Average proportion of total bird mortality caused by cat predation for individual species. Source: Loss et al. (2013), Supplementary table S3. See Loss et al. (2013) for details on methodology.

Species	Average proportion[1]	Number of Studies[2]
Ring-necked Pheasant (*Phasianus colchicus*)	0.160	1
House Sparrow (*Passer domesticus*)	0.107	2
American Robin (*Turdus migratorius*)	0.085	5
Red-winged Blackbird (*Agelaius phoeniceus*)	0.056	1
Northern Bobwhite (*Colinus virginianus*)	0.050	1
Gray Catbird (*Dumetella carolinensis*)	0.048	1
American Goldfinch (*Spinus tristis*)	0.036	5
Northern Cardinal (*Cardinalis cardinalis*)	0.035	2
Rock Pigeon (*Columba livia*)	0.034	1
House Wren (*Troglodytes aedon*)	0.030	1

[1] Proportions are based on 10 U.S. studies that report species-by-species mortality counts.

[2] Number of studies documenting predation on each bird species

5.2.2 Comparison of the levels of anthropogenic mortality

As described previously, differences in the probability of detection and in reporting rates make it difficult to compare the levels of mortality resulting from different anthropogenic sources directly. Loss *et al.* (2015) reviewed available estimates of anthropogenic mortality for Canada and the United States. The authors concluded that predation by cats forms the largest source of anthropogenic mortality, followed by collisions with buildings, cars, and then power lines (Figure 5.2.1).

However, note that these are total numbers of individuals, summed over all species. Species likely differ in their vulnerability to different sources of mortality. For example, songbirds are relatively vulnerable to predation by cats, whereas large raptors are not vulnerable to this source of mortality.

Buij *et al.* (2018) reviewed sources of anthropogenic mortality based on ring recoveries of individuals ringed in the Netherlands since 1911. They analysed different sources of mortality due to energy infrastructure (oil, electrocution, high-tension cables, wind turbine and other structures). They conclude that especially in the case of species occurring at sea or close to the coast, waders and gulls and terns, mortality caused by wind energy represents a large part (>30%) of the mortality related to energy infrastructure based on ring recoveries. Again, it should be noted that this proportion of mortality caused by wind energy is likely an underestimate, as both the probability of finding and determining the cause of death of victims from collisions with offshore wind turbines is very low.
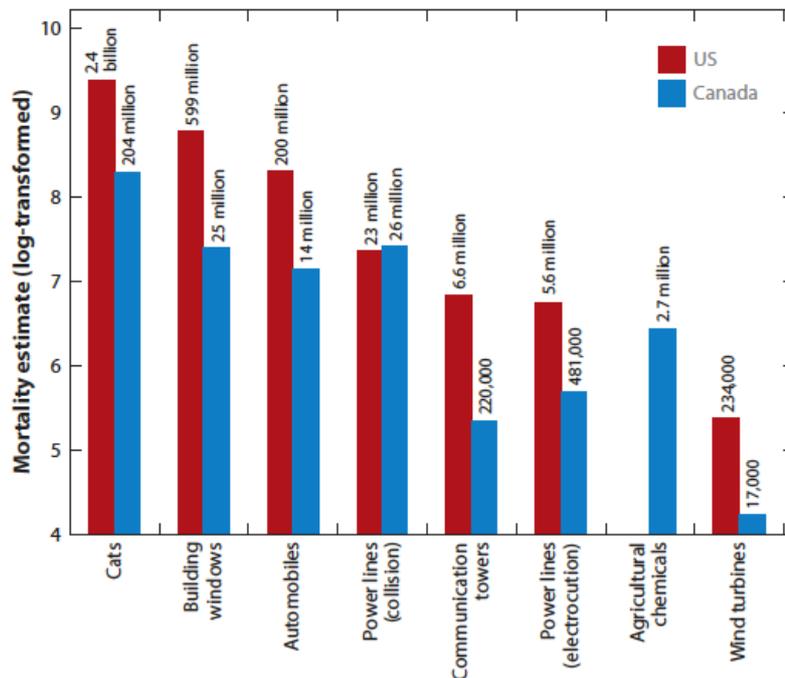
Figure 5.2.1 Comparison of major sources of direct anthropogenic bird mortality for the United States and Canada. Note the logarithmic scale. Source: Loss et al. *(2015), Figure 2a. Based on Longcore* et al. *(2012), Calvert* et al. *(2013), Loss* et al. *(2013, 2014).*

Species of interest for this study

For the species groups that are of interest for this study, we give an overview of the vulnerability for different anthropogenic sources of mortality.

As species-specific data are often lacking, we grouped our study species into gulls (lesser black-backed gull, great black-backed gull, herring gull, black-legged kittiwake), skuas (great skua, arctic skua), wildfowl (brent goose, common shelduck, Bewick's swan), waders (Eurasian curlew) and terns (black tern, common tern).

Gulls

Using data from ring recoveries of individuals ringed in the Netherlands, Buij *et al.* (2018) showed that from all sources of anthropogenic mortality (hunting, traffic, windows, others), traffic seems to be the most important for gulls. Most of these are expected to be victims from car collisions. Based on an analysis of roadkills in Norway, gulls account for 28% of the roadkills (Husby 2016; Table 5.2.2). In addition, gulls represent a large proportion of victims from collisions with airplanes (Burger 1985; Dolbeer 2006). The top 10 most found species among roadkills (waarneming.nl, Table 5.2.1) includes herring gull (9th position) and black-headed gull (6th position); although the physical characteristics of gulls may mean that they are more likely to be found and reported than some other species groups.

Based on ring recoveries from the UK, Wernham *et al.* (2002) show the main cause of death of recovered individuals in the UK to be 'deliberately taken by man' (Table 5.2.4 - 5.2.7). For lesser black-backed gull, herring gull and great black-backed gull, at least another 10% of individuals with known cause died because of a human-related cause,

for example collisions with man-made structures. Wernham *et al.* (2002) found that for black-legged kittiwake the second-largest source of mortality is pollution (Table 5.2.7).

In a review about international impacts of electrocution and collision due to power lines, Prinsen *et al.* (2011) conclude that regionally high numbers of collisions with power lines are reported. However, no significant impact on the overall species populations is expected (Prinsen *et al.* 2011).

Table 5.2.4 Finding circumstances of lesser black-backed gull based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	63%
Natural (illness)	15%
Human-related	10%
Other	12%

Table 5.2.5 Finding circumstances of herring gull based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	60%
Human-related	12%
Natural (illness)	12%
Other	16%

Table 5.2.6 Finding circumstances of great black-backed gull based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	38%
Human-related	19%
Natural (illness)	14%
Other (e.g. oil)	29%

Table 5.2.7 Finding circumstances of black-legged kittiwake based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	50%
Pollution	19%
Natural (environmental)	9%
Other	22%

Wernham *et al.* (2002) state that for great skuas for which the cause of death is known, 45% are deliberately taken by man (Table 5.2.8). In addition, 17% are accidentally captured, mainly caused by drowning by fishing nets or baited lines (Wernham *et al.* 2002). For arctic skua, Wernham *et al.* (2002) identified other (unspecified) sources of mortality and mortality from being taken by man as accounting for around a third of the causes of death each. The remaining third being from natural sources (Table 5.2.9). Regionally high numbers of skua collisions with power lines are reported internationally (Prinsen *et al.* 2011). However, no significant impact on the overall species populations is expected (Prinsen *et al.* 2011).

Table 5.2.8 Finding circumstances of great skua based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	45%
Accidental capture	17%
Pollution	13%
Other	25%

Table 5.2.9 Finding circumstances of arctic skua based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Other	33%
Deliberately taken by man	30%
Predation	22%
Natural (illness)	15%

Wildfowl

Ring recoveries of brent goose, common shelduck and Bewick's swan with known source of mortality mostly died as a result of hunting (respectively 70%, 53% and 32%) (Wernham *et al.* 2002; Table 5.2.10 - 5.2.12). According to Rees (2006), the main cause of death in Bewick's swans is flying accidents, of which most are collisions with power lines. Wernham *et al.* (2002) also report relatively high mortality due to collisions with man-made structures (mostly power lines), which make up most of the finding classified as 'other' in table 5.2.12. Prinsen *et al.* (2011) conclude based on a review that Anatidae suffer regionally from many fatalities due to collisions with power lines, but no significant impact on the overall species populations is expected.

In addition, wildfowl seems to be relatively vulnerable to collisions with civil aircraft. Dolbeer (2006) also found that Anatidae are one of the main species groups among victims of collisions with civil aircraft.

Table 5.2.10   Finding circumstances of brent goose based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	70%
Natural (environmental)	15%
Natural (illness)	9%
Human-related	6%

Table 5.2.11   Finding circumstances of common shelduck based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	53%
Natural (environmental)	15%
Natural (illness)	8%
Other	24%

Table 5.2.12   Finding circumstances of Bewick's swan based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	32%
Human-related	30%
Natural (environmental)	15%
Other	23%

Waders

In the UK, curlews are often victims from hunting. Douglas *et al.* (2014) found that curlew population growth rate is linked to gamekeeper density. Wernham *et al.* (2002) report that 70% of ring recoveries with known cause of death are victims of hunting (Table 5.2.13), with figures varying between countries (e.g. Ireland 90%, Denmark 94% and France 100%). A total of 12% of the recoveries were attributed to other human-related causes of mortality.

In contrast, Buij *et al.* (2018) reviewed anthropogenic causes of mortality based on ring recoveries from the Netherlands, and showed that most of the reported dead waders were killed by traffic (in comparison to hunting, traffic, windows and others). As pointed out previously, although this high number of reported road kills could suggest that mortality due to traffic plays a large role, this may be partly caused by higher reporting rates of road kills.

Moreover, note that this relatively high number of reported road kills may be caused by a few wader species (for example lapwing). No clear data suggests that curlews strongly suffer from traffic mortality. The database of waarneming.nl only contains 11 reported curlews in the period 2009-2018. Within the same period, 80 victims of lapwing have been reported in this database as road kill.

Collisions with power lines regionally cause many fatalities, which may threaten the overall species populations (Prinsen *et al.* 2011).

Table 5.2.13 Finding circumstances of Eurasian curlews based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	70%
Human-related	12%
Natural predator	7%
Other	11%

Terns

Based on ring recoveries, the majority of the common terns recovered dead from known causes were deliberately taken by man (55%), mainly in Africa (Wernham *et al.* 2002; Table 5.2.14). No recovery data of black terns is available from birds ringed in the UK (Wernham *et al.* 2002). Based on ring recoveries of individuals ringed in the Netherlands, Buij *et al.* (2018) showed that from all sources of anthropogenic mortality (hunting, traffic, windows, others), the most reported cause of mortality of terns is traffic. Terns regularly collide with vehicles. According to a review on victims from car collisions by Jacobson (2005), terns are among the species groups with expected higher risk of car collisions. In addition, terns are one of the main species groups among victims of collisions with civil aircraft (Dolbeer 2006).

Collisions with power lines may impact terns regionally (Prinsen *et al.* 2011). However, no significant impact on the overall species populations is expected.

Table 5.2.14 Finding circumstances of common terns based on ring recoveries. Source: Wernham et al. *(2002).*

Finding circumstances	Percentage of findings
Deliberately taken by man	55%
Accidental capture	8%
Found on ship	8%
Other	29%

Conclusion

Overall, the differences in probability of detection and reporting rates make it difficult to compare the levels of mortality resulting from different anthropogenic sources directly. Loss *et al.* (2015) gave ranges of estimated mortality per anthropogenic source for all species. However, species likely differ in relative mortality per anthropogenic source.

Within this chapter, we presented an overview of the literature available on relative vulnerability of species (groups) to certain sources of anthropogenic mortality. This gives an impression of the expected impact of this source of anthropogenic mortality for each species (group). However, it is not possible to assess per species (group) how many victims are expected from each source of anthropogenic mortality.

This means that, even though for all bird species, the numbers of victims due to turbine collisions seems lower than due to other anthropogenic sources (Loss *et al.* 2015), the relative impact likely strongly differs between species (groups). Predation by cats for

example, which is an important source of anthropogenic mortality as pointed out by Loss *et al.* (2015), is unlikely to play a large role for the species selected in this study.

The species selected in this study may be impacted by collisions with vehicles, buildings and man-made structures or poisoning. We reviewed available data on mortality due to these factors for the selected species, mainly based on ring recoveries. Due to differences in probability of finding and reporting victims, data on impacts of these anthropogenic factors are not comparable. For that reason, it is not possible to directly compare mortality rates due to these different anthropogenic sources. Therefore it is currently not feasible to incorporate other sources of mortality into our population models.

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