



# Arklow Bank Wind Park 2

## Environmental Impact Assessment Report

Volume III, Appendix 12.10: Offshore Ornithology Technical Report -  
Kittiwake Population Viability Analysis



MacArthur  
Green

## Arklow Bank Wind Park 2

### Technical Appendix 12.10 Offshore Ornithology

### Kittiwake Population Viability Analysis

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**CONTENTS**

Glossary ..... III

Acronyms.....IV

1 OFFSHORE ORNITHOLOGY TECHNICAL REPORT – KITTIWAKE POPULATION VIABILITY ANALYSIS.....1

1.1 Introduction .....1

1.2 Methods .....1

1.2.1 Population counts .....1

1.2.2 Population modelling..... 2

1.3 Results ..... 5

1.4 Discussion.....9

1.5 References..... 11

ANNEX A. MODEL INPUT PARAMETERS AND SETTINGS.....12

**LIST OF TABLES**

Table 12.10.1: Demographic rates used in the population models..... 3

Table 12.10.2: Kittiwake, Density independent (DI) CPGR and CPS. .... 7

Table 12.10.3: Kittiwake, Density dependent (DD) CPGR and CPS.....8

**LIST OF FIGURES**

Figure 12.10.1: Wicklow Head kittiwake population data, 1999-2023. .... 2

Figure 12.10.2: Density independent (left) and density dependent (right) population simulations for baseline (no impact) and adult mortality up to 20. Solid lines are the median from 1,000 simulations and the dashed lines are the 95% confidence intervals. .... 6

Figure 12.10.3: Median and 95% confidence intervals for the counterfactuals of population growth rate (CPGR; left) and population size (CPS; right) for the density independent (DI, red lines) and density dependent(DD, blue lines) simulations. Note the different scales on the y-axes, with a much wider range required for CPS..... 7

## Glossary

| Term  | Meaning  |
|---|--|
| a   | Scale parameter for the Weibull function   |
| Arklow Bank Wind Park 1 (ABWP1)                   | Arklow Bank Wind Park 1 consists of seven wind turbines, offshore export cable and inter-array cables. Arklow Bank Wind Park 1 has a capacity of 25.2 MW. Arklow Bank Wind Park 1 was constructed in 2003/04 and is owned and operated by Arklow Energy Limited. It remains the first and only operational offshore windfarm in Ireland.   |
| Arklow Bank Wind Park 2 – Offshore Infrastructure | “The Proposed Development”, Arklow Bank Wind Park 2 Offshore Infrastructure: This includes all elements under the existing Maritime Area Consent.  |
| Arklow Bank Wind Park 2 (ABWP2) (The Project)     | <p>Arklow Bank Wind Park 2 (ABWP2) (The Project) is the onshore and offshore infrastructure. This EIAR is being prepared for the Offshore Infrastructure. Consents for the Onshore Grid Infrastructure (Planning Reference 310090) and Operational and Maintenance Facility (Planning Reference 211316) has been granted on 26th May 2022 and 20th July 2022, respectively.</p> <ul style="list-style-type: none"> <li>• Arklow Bank Wind Park 2 Offshore Infrastructure: This includes all elements to be consented in accordance with the Maritime Area Consent. This is the subject of this EIAR and will be referred to as ‘the Proposed Development’ in the EIAR.</li> <li>• Arklow Bank Wind Park 2 Onshore Grid Infrastructure: This relates to the onshore grid infrastructure for which planning permission has been granted.</li> <li>• Arklow Bank Wind Park 2 Operational and Maintenance Facility (OMF): This includes the onshore and nearshore infrastructure at the OMF, for which planning permission has been granted.</li> <li>• Arklow Bank Wind Park 2 EirGrid Upgrade Works: any non-contestable grid upgrade works, consent to be sought and works to be completed by EirGrid.</li> </ul> |
| b   | Shape parameters for the Weibull function  |
| EirGrid   | State-owned electric power Transmission System Operator (TSO) in Ireland and Transmission Asset Owner (TAO) for the Project’s transmission assets.   |
| Counterfactual                                    | The ratio of the impacted to unimpacted population sizes or the impacted population growth rate to unimpacted population growth rate.  |
| F   | Reproduction for the Weibull function  |
| maxF  | The estimated biological maximum reproductive rate for the species being modelled using the Weibull function   |
| N   | Population size for the Weibull function   |
| PVA   | Population Viability Analysis is a species-specific method of risk assessment frequently used in conservation biology. It is traditionally defined as the process that determines the probability that a population will become extinct within a given number of years.  |
| Weibull function                                  | An equation used for density dependent modelling   |

## Acronyms

| Term  | Meaning                                  |
|-------|--|
| AON   | Apparently Occupied Nests                |
| BTO   | British Trust for Ornithology            |
| c.i.  | Confidence Intervals                     |
| CPGR  | Counterfactual of Population Growth Rate |
| CPS   | Counterfactual of Population Size        |
| DD    | Density Dependent                        |
| DI    | Density Independent                      |
| JNCC  | Joint Nature Conservation Committee      |
| NEPVA | Natural England PVA                      |
| NPWS  | National Parks and Wildlife Service      |
| OMF   | Operational and Maintenance Facility     |
| PVA   | Population Viability Analysis            |
| SPA   | Special Protected Area                   |

# 1 OFFSHORE ORNITHOLOGY TECHNICAL REPORT – KITTIWAKE POPULATION VIABILITY ANALYSIS

## 1.1 Introduction

1. This Technical Report provides details of Population Viability Analysis (PVA) for the kittiwake breeding population at the Wicklow Head Special Protection Area (SPA).
2. The Natural England commissioned PVA tool (hereafter NEPVA) developed by the Centre for Ecology and Hydrology (CEH; Searle *et al.*, 2019) was used for this analysis. The NEPVA, which is the recommended tool for undertaking seabird PVA in the UK, is written using the R programming language and the scripts are available to download from the project's webpage<sup>1</sup> although for most users it is expected that the online version of the model<sup>2</sup> will be used since this requires less technical experience. However, the online version of the model has a reduced range of options, most notably in how density dependent population regulation can be specified. For this reason the current kittiwake modelling was conducted using the downloaded NEPVA R scripts.
3. The parameters for the model were derived from counts of breeding adults and productivity data collected at the SPA<sup>3</sup> combined with survival and age at first breeding rates from a review of seabird literature conducted by the British Trust for Ornithology (BTO; Horswill and Robinson, 2015). The latter were used as no studies of survival rates have been conducted at Wicklow Head.
4. Outputs are provided as the ratios of the impacted to unimpacted population sizes and impacted to unimpacted population growth rate, respectively referred to as the Counterfactual of Population Size (CPS) and the Counterfactual of Population Growth Rate (CPGR). These measures of relative, rather than absolute, effects of additional mortality are preferred because they have been found to be comparatively less sensitive to assumptions about demographic rate values and therefore are considered more robust and reliable.
5. These have been obtained across a range of mortality levels. The discussion reviews the results and provides a guide for their interpretation.

## 1.2 Methods

### 1.2.1 Population counts

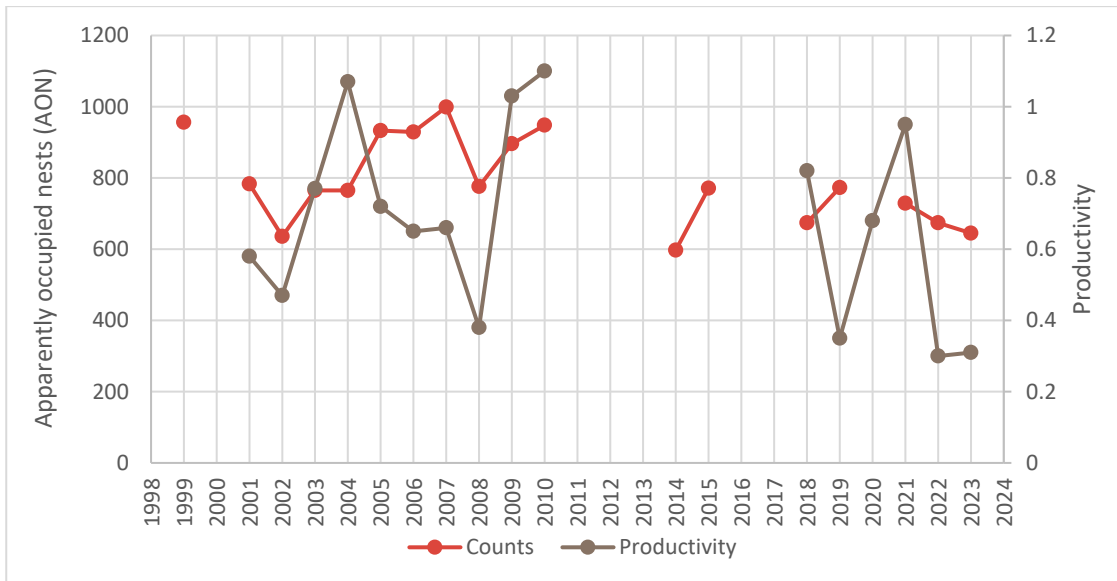
6. Population data for this colony were obtained from the monitoring reports prepared for Arklow Bank Wind Park 1 (ABWP1), counts supplied by Birdwatch Ireland (S. Newton, pers. comm.), the National Parks and Wildlife Service (NPWS, 2022) and Cork Ecology (2023). The counts and productivity data are presented in Figure 12.10.1.

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<sup>1</sup> [https://github.com/naturalengland/Seabird\\_PVA\\_Tool](https://github.com/naturalengland/Seabird_PVA_Tool)

<sup>2</sup> [http://ec2-34-243-66-127.eu-west-1.compute.amazonaws.com/shiny/seabirds/PVATool\\_Nov2022/R/](http://ec2-34-243-66-127.eu-west-1.compute.amazonaws.com/shiny/seabirds/PVATool_Nov2022/R/)

<sup>3</sup> Data provided by National Parks and Wildlife Service (via email 17/04/2020) and Birdwatch Ireland (at a meeting on 10/12/2018)



**Figure 12.10.1: Wicklow Head kittiwake population data, 1999-2023.**

7. During the period of available data (1999 to 2023), the kittiwake breeding population has varied between 600 and 1,000 pairs (also referred to as apparently occupied nests) with an average of approximately 800, and the annual population growth rate, estimated between consecutive counts has varied between 0.78 and 1.29, with an average of 0.992 (i.e. 0.8% annual decline). Productivity, measured as fledged young per pair, has varied between 0.30 and 1.1 with an average of 0.68. An average of 0.68 chicks per nest is relatively high productivity for kittiwakes at colonies in Britain and Ireland in recent years (Coulson, 2017) and is close to the level where population stability would be expected (Coulson, 2017). This is, therefore, consistent with the observed approximate stability of breeding numbers at this colony, with an indication of a gradual long-term decline.

#### 1.2.2 Population modelling

8. The population model included the following aspects:
- Environment stochasticity;
  - Demographic stochasticity;
  - Density independent and density dependent formulations; and
  - Matched runs, with baseline and impact simulations conducted with identical sequences of demographic rates.
9. The models use a matrix formulation and generate population estimates corresponding to an annual post-breeding census for a period of 37 annual time steps. The population is made up of single year age classes (e.g. 0-1, 1-2, etc.) up to the adult class, which is a multi-age class for all individuals from age of first breeding (4) and older. The initial population size was 1,600 breeding adults (i.e. 800 Apparently Occupied Nests (AON)), derived from the average number of pairs recorded at the colony.



10. Rates of survival have not been estimated for the Wicklow Head colony, therefore the survival rates and age of first breeding used in the model were obtained from a review conducted by the BTO for the Joint Nature Conservation Committee (JNCC) (Horswill and Robinson, 2015), while productivity data for Wicklow Head kittiwakes were used in the models (Table 12.10.1).

**Table 12.10.1: Demographic rates used in the population models.**

|                    | Survival |       |       |       |       | Productivity                        |                    |
|--------------------|----------|-------|-------|-------|-------|-------------------------------------|--------------------|
|                    | 0-1      | 1-2   | 2-3   | 3-4   | Adult | Fledged young per pair <sup>a</sup> | Age first breeding |
| Mean               | 0.79     | 0.854 | 0.854 | 0.854 | 0.854 | 0.677                               | 4                  |
| Standard Deviation | 0.051    | 0.051 | 0.051 | 0.051 | 0.051 | 0.270                               |                    |

<sup>a</sup> Productivity data taken from the Wicklow Head colony productivity data for the period 2001 to 2010 (see Volume III, Appendix 13.9: Offshore Ornithology Technical Report – Review of Seabird Monitoring Data: 2000 to 2010) and 2018 to 2022 from NPWS (2022).

11. The online version of the NEPVA provides an option to run the model projections as either density independent, with no feedback between demographic rates and the population size, or density dependent using a function that relates change in selected demographic rates to change in population size. However, only one relationship between population size and demographic rate is available, based on a log<sup>10</sup> change in population size. In practice this is an extremely weak form of density dependent regulation, since a ten-fold change in population size is required to observe a meaningful change in demographic rates. Such a large degree of change in long-lived slow breeding species such as seabirds is highly unlikely, with the consequence that the density dependent results obtained using this option are barely distinguishable from density independent ones.
12. Limiting the options available in the NEPVA in this manner is surprising since there is strong evidence that density dependence influences the size and growth rate of seabird colonies. For example, with respect to kittiwake, Coulson (1983, 2011) and Suryan and Irons (2001) reported density dependent effects, probably mediated through competition for food in the sea surrounding each colony (Furness and Birkhead, 1984; Wakefield *et al.*, 2017). Horswill and Robinson (2015) reviewed the evidence for density dependent regulation in seabird populations and found that regulation can operate via a range of mechanisms. At the scale of the population being modelled for this report it is therefore likely that regulation may be operating on different components of the population by different means. Ecological theory suggests that long lived slow breeding species, such as seabirds, buffer themselves against variations in their environment through varying reproductive success rather than survival. Thus, the demographic rate most likely to reflect density dependent effects will be reproduction, with breeding success declining as population approaches the ceiling set by food resources. Thus, it was considered more appropriate to model regulation through reproduction rather than across multiple rates. This is also more precautionary for assessing mortality impacts, since seabird population growth is more sensitive to variation in survival (particularly of adults). Thus, the modelled population’s ability to recover is lower when density dependence operates through reproduction rather than through survival since production of additional chicks represents a slower means of population recovery than an increase in adult survival.

13. Relating the reproductive rate to the population size also corresponds with studies which indicate that foraging ranges are negatively related to food availability which in turn affects variations in reproductive success between colonies. A wide range of values of density dependence could be explored, but the aim of the work was to indicate the possible difference in output between a biologically unrealistic density independent (worst case) scenario, and a plausible density dependent model (more realistic but with a precautionary density dependent formulation in the absence of empirical evidence on density dependent mechanisms in this population).
14. The R scripts which underpin the NEPVA include several additional options for applying density dependence, as well as the facility to define extra ones, but this can only be done when the model is run using R offline on a local machine. Thus, the R scripts were downloaded<sup>1</sup> to permit running of more realistic population models that incorporate population regulation. One of the density dependent models pre-coded in the NEPVA is named as a Weibull function, however this model can be defined in several ways and the one built into the NEPVA does not prevent nonsensical values. For example, as population size grows productivity or survival values less than zero can be obtained, or at low population sizes survival rates greater than 1 can be obtained. Therefore, an alternative version of the Weibull function was added which avoids these problems (i.e. the equation generates an asymptotic curve which never breaches biological realistic thresholds).
15. This function relates productivity ( $F$ ) to population size ( $N$ ) using the following equation:

$$F = \text{max}F * \exp(-a * (N^b))$$

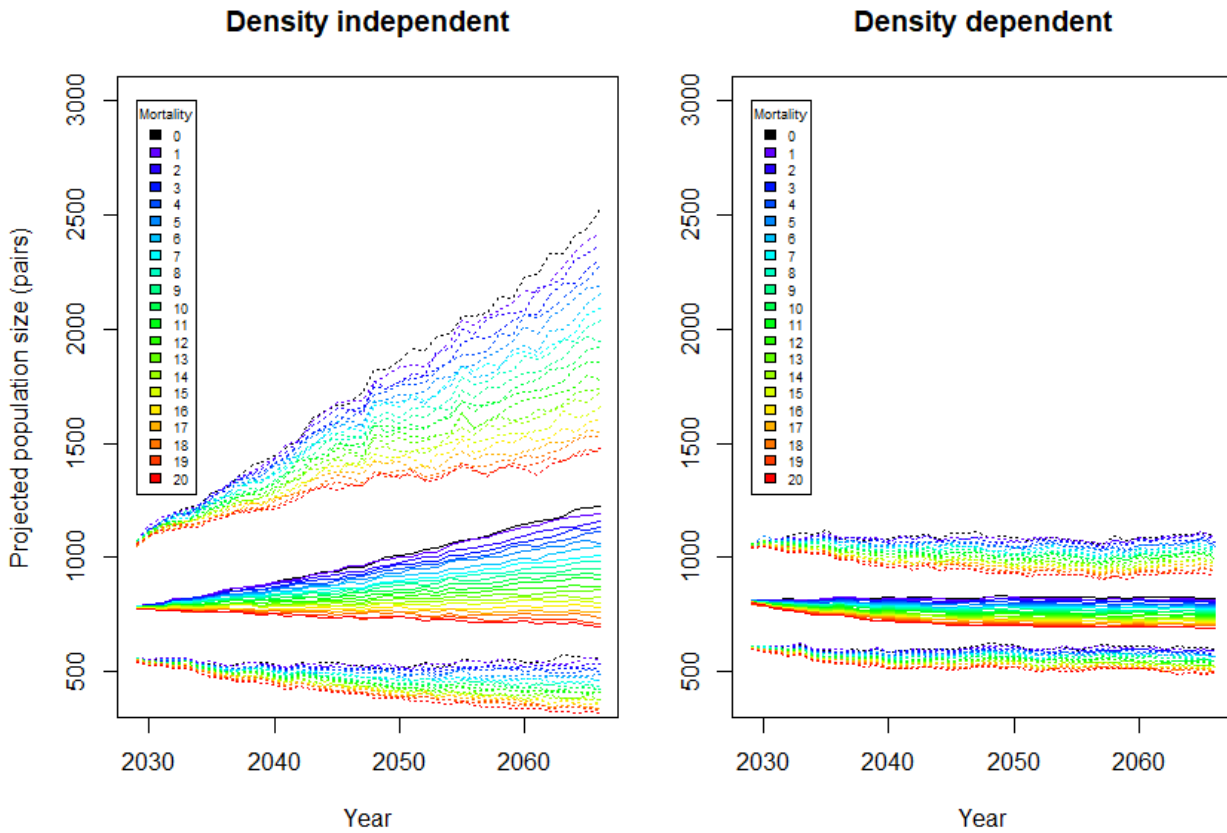
16. Where  $\text{max}F$  is the estimated biological maximum reproductive rate for the species being modelled and  $a$  and  $b$  are scale and shape parameters (respectively) for the Weibull function.
17. Previous population modelling of this species (MacArthur Green, 2014) included a review of available evidence for density dependent regulation and determined that a precautionary, but realistic value for  $b$  was 1.2 as this generates population trends similar to those observed for a range of seabird species and populations (Cury *et al.*, 2011). Following this, the value for  $a$  was calculated using the equation above with  $b$  equal to 1.2,  $F$  equal to the mean reproductive rate (0.702),  $\text{max}F$  equal to the species' estimated biological maximum (1.25) and  $N$  equal to the initial population estimate (1,600).
18. Estimating the value for  $a$  in this manner makes the assumption that the population is currently at its carrying capacity, and ensured that baseline simulations (i.e. with no additional mortality) were tuned to remain around this size (although with variations due to stochastic variation in the parameters).
19. The complete list of input parameters used in the model, both density independent and density dependent is provided in Annex 1. A 'burn-in' period of 10 years was included in the simulations, which allows the population age ratios to stabilise and also reduces the risk of initial parameter values influencing the outputs. During preliminary model runs it was found that the initial population size needed to be defined slightly differently for density independent and density dependent simulations in order for them both to have an initial number of pairs of 800 following the burn-in. Thus, density independent runs had an initial (pre- burn-in) size of 750 AON while for density dependent runs this value was 650 AON.

20. It was necessary to assume the population was closed (i.e. no exchange with other colonies) because rates of exchange between colonies are unknown. While this is unrealistic, it was considered to be a pragmatic and precautionary approach, since immigration from other colonies will tend to buffer any additional mortality impact on the focal population.
21. It is also important to note that since all the predictions use past data on demographic parameters, they take no account of impacts of continuing climate change on seabird demography or the potential risks from disease outbreaks such as the current occurrence of Avian Influenza in seabird colonies.
22. A range of additional annual mortality values was modelled, from 0 to 20, at increments of 1. However, to ensure that this mortality remained in proportion to the population size, this was applied as an adjustment to the survival rate by dividing the modelled mortality for that simulation (1-20) by the initial population size (800). The additional mortality was applied to all age classes in proportion to their presence (i.e. windfarm mortality was not considered to target specific age classes). This means that when comparing the outputs from the PVA obtained for a particular mortality level to collision mortality predictions, the latter need to be defined as the number of breeding adults (i.e. the overall collisions need to be apportioned both to the Wicklow colony and also to adults only).
23. Although additional mortality was applied to all age classes, the outputs are presented as the number of breeding pairs, thereby ensuring the outputs are consistent with the count unit used at the breeding colony.
24. At each level of mortality, 1,000 simulations were conducted and summary outputs calculated, with simulations run for a projection period of 37 years from 2029 to 2066.
25. The following outputs are provided as graphs and tables:
  - Ratio of impacted to baseline population growth rate (Counterfactual of Population Growth Rate, CPGR) at each mortality level; and
  - Ratio of impacted population size to baseline population size (Counterfactual of Population Size, CPS), for each mortality level.

### 1.3 Results

26. The median population projections (and 95% confidence intervals) for the baseline and impact simulations from the density independent and density dependent model are provided in Figure 12.10. These figures illustrate a key difference between these two model options. In the case of density independent simulations, if growth is positive the populations experience exponential growth (ultimately reaching infinity). As a consequence, baseline and impact projections diverge by an ever increasing amount with the CPS for any given impact level thus reflecting in large part the duration of simulation, while the growth rates are effectively constant meaning the CPGR is unaffected by the simulation duration. In contrast, when density dependent regulation is included, the model generates level population projections which settle at equilibrium levels irrespective of duration. Thus the CPS is insensitive to simulation duration, while the growth rate will average around 1 for both baseline and impacted runs, meaning that there is little variation in growth rates and the CPGR is somewhat uninformative.

27. The outcome of this is that for density independent simulations the CPGR is the more useful metric and for density dependent ones the CPS is the more useful one.



**Figure 12.10.2: Density independent (left) and density dependent (right) population simulations for baseline (no impact) and adult mortality up to 20. Solid lines are the median from 1,000 simulations and the dashed lines are the 95% confidence intervals.**

28. The CPGR and CPS are provided for both density independent and density dependent models in Figure 12.10.. The values are also provided in Tables 12.10.2 and 12.10.3, respectively.
29. As discussed above, the CPGR is more appropriate to consider for a density independent model and the CPS is more appropriate for a density dependent one. Thus, at the maximum modelled mortality of 20 individuals, the greatest reduction in growth rate was 1.5% (a CPGR of 0.985 for the density independent model), while the greatest reduction in population size was 16% (a CPS of 0.84 for the density dependent model).

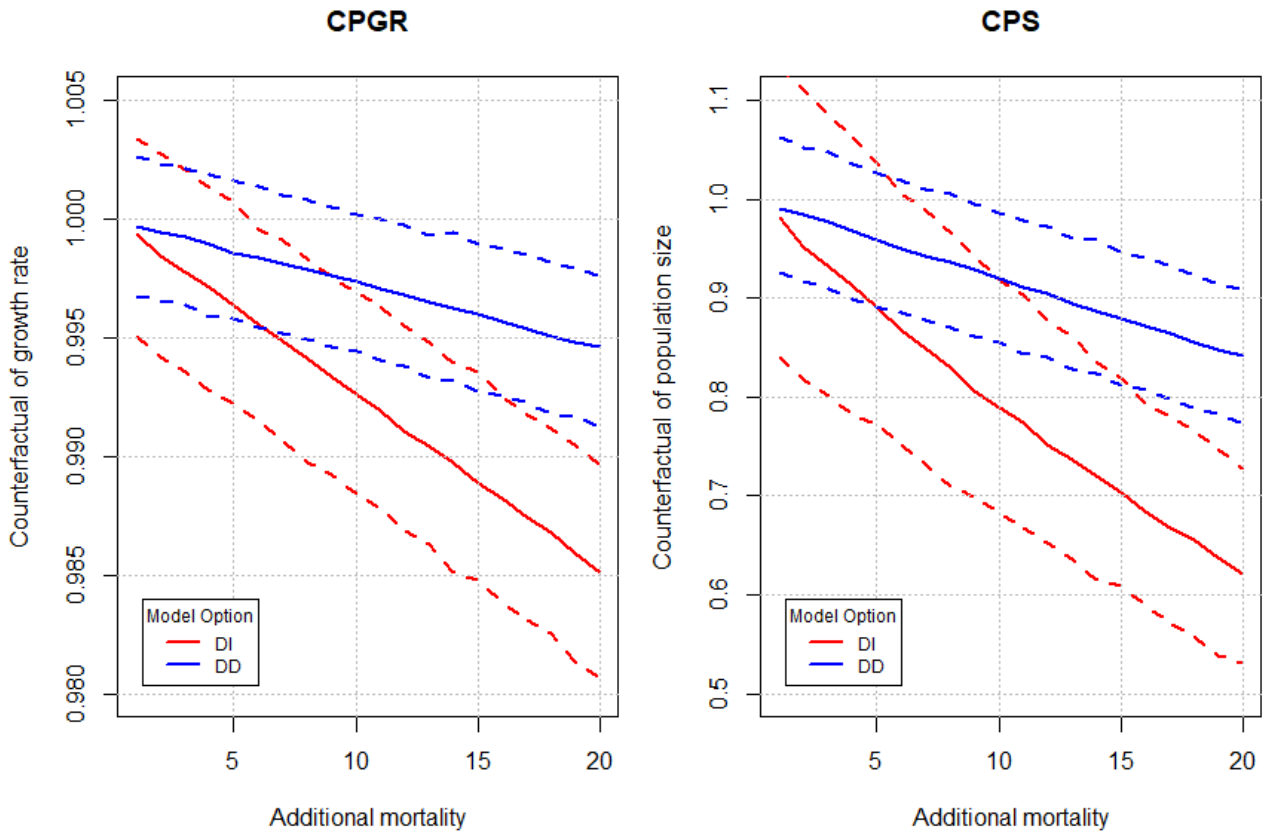


Figure 12.10.3: Median and 95% confidence intervals for the counterfactuals of population growth rate (CPGR; left) and population size (CPS; right) for the density independent (DI, red lines) and density dependent (DD, blue lines) simulations. Note the different scales on the y-axes, with a much wider range required for CPS.

Table 12.10.2: Kittiwake, Density independent (DI) CPGR and CPS.

| Additional adult mortality | CPGR   |                 | CPS    |                 |
|----------------------------|--------|-----------------|--------|-----------------|
|                            | Median | 95% c.i.        | Median | 95% c.i.        |
| 1                          | 0.9994 | 0.9951 - 1.0034 | 0.9824 | 0.8402 - 1.1335 |
| 2                          | 0.9984 | 0.9942 - 1.0027 | 0.9522 | 0.8179 - 1.1112 |
| 3                          | 0.9977 | 0.9936 - 1.0021 | 0.9318 | 0.8013 - 1.0851 |
| 4                          | 0.9971 | 0.9928 - 1.0013 | 0.9130 | 0.7820 - 1.0630 |
| 5                          | 0.9964 | 0.9922 - 1.0007 | 0.8920 | 0.7721 - 1.0368 |
| 6                          | 0.9956 | 0.9915 - 0.9996 | 0.8679 | 0.7523 - 1.0050 |
| 7                          | 0.9949 | 0.9907 - 0.9992 | 0.8491 | 0.7326 - 0.9889 |
| 8                          | 0.9941 | 0.9897 - 0.9983 | 0.8306 | 0.7096 - 0.9669 |
| 9                          | 0.9934 | 0.9893 - 0.9976 | 0.8065 | 0.6986 - 0.9429 |
| 10                         | 0.9926 | 0.9884 - 0.9969 | 0.7901 | 0.6831 - 0.9180 |
| 11                         | 0.9919 | 0.9878 - 0.9963 | 0.7750 | 0.6678 - 0.9036 |
| 12                         | 0.9911 | 0.9870 - 0.9955 | 0.7514 | 0.6529 - 0.8784 |
| 13                         | 0.9904 | 0.9863 - 0.9948 | 0.7368 | 0.6362 - 0.8613 |
| 14                         | 0.9897 | 0.9851 - 0.9940 | 0.7198 | 0.6157 - 0.8364 |

| Additional adult mortality | CPGR   |                 | CPS    |                 |
|----------------------------|--------|-----------------|--------|-----------------|
|                            | Median | 95% c.i.        | Median | 95% c.i.        |
| 15                         | 0.9889 | 0.9848 - 0.9935 | 0.7030 | 0.6090 - 0.8196 |
| 16                         | 0.9882 | 0.9839 - 0.9925 | 0.6848 | 0.5887 - 0.7940 |
| 17                         | 0.9874 | 0.9831 - 0.9917 | 0.6673 | 0.5721 - 0.7811 |
| 18                         | 0.9868 | 0.9825 - 0.9912 | 0.6559 | 0.5585 - 0.7655 |
| 19                         | 0.9859 | 0.9813 - 0.9905 | 0.6370 | 0.5383 - 0.7472 |
| 20                         | 0.9851 | 0.9807 - 0.9896 | 0.6217 | 0.5316 - 0.7270 |

**Table 12.10.3: Kittiwake, Density dependent (DD) CPGR and CPS.**

| Additional adult mortality | CPGR   |                 | CPS    |                 |
|----------------------------|--------|-----------------|--------|-----------------|
|                            | Median | 95% c.i.        | Median | 95% c.i.        |
| 1                          | 0.9997 | 0.9967 - 1.0026 | 0.9914 | 0.9258 - 1.0626 |
| 2                          | 0.9994 | 0.9965 - 1.0023 | 0.9846 | 0.9171 - 1.0529 |
| 3                          | 0.9993 | 0.9963 - 1.0021 | 0.9769 | 0.9099 - 1.0481 |
| 4                          | 0.9990 | 0.9959 - 1.0019 | 0.9675 | 0.8994 - 1.0356 |
| 5                          | 0.9986 | 0.9958 - 1.0017 | 0.9590 | 0.8910 - 1.0272 |
| 6                          | 0.9984 | 0.9954 - 1.0014 | 0.9503 | 0.8855 - 1.0191 |
| 7                          | 0.9981 | 0.9952 - 1.0010 | 0.9423 | 0.8775 - 1.0097 |
| 8                          | 0.9979 | 0.9949 - 1.0008 | 0.9365 | 0.8706 - 1.0051 |
| 9                          | 0.9976 | 0.9946 - 1.0005 | 0.9285 | 0.8613 - 0.9954 |
| 10                         | 0.9974 | 0.9944 - 1.0002 | 0.9196 | 0.8554 - 0.9868 |
| 11                         | 0.9970 | 0.9940 - 1.0000 | 0.9117 | 0.8440 - 0.9781 |
| 12                         | 0.9968 | 0.9938 - 0.9998 | 0.9053 | 0.8404 - 0.9723 |
| 13                         | 0.9965 | 0.9933 - 0.9993 | 0.8952 | 0.8278 - 0.9608 |
| 14                         | 0.9962 | 0.9932 - 0.9994 | 0.8875 | 0.8235 - 0.9594 |
| 15                         | 0.9960 | 0.9927 - 0.9989 | 0.8799 | 0.8125 - 0.9469 |
| 16                         | 0.9957 | 0.9925 - 0.9987 | 0.8723 | 0.8067 - 0.9405 |
| 17                         | 0.9954 | 0.9923 - 0.9985 | 0.8648 | 0.7982 - 0.9324 |
| 18                         | 0.9951 | 0.9918 - 0.9982 | 0.8559 | 0.7890 - 0.9232 |
| 19                         | 0.9948 | 0.9917 - 0.9979 | 0.8484 | 0.7835 - 0.9148 |
| 20                         | 0.9946 | 0.9912 - 0.9976 | 0.8419 | 0.7726 - 0.9085 |

## 1.4 Discussion

30. The potential maximum annual rate at which kittiwake populations can grow, calculated using the method described by Niel and Lebreton (2005), is estimated to be 13.3%. This is considerably higher than the maximum reduction predicted of 1.5% by the more precautionary density independent model at an adult mortality of 20. While this suggests that kittiwake populations may possess a high degree of resilience to additional sources of mortality, it is important to note that the theoretical maximum growth rate is only likely to be achieved under optimal conditions (e.g. plentiful prey and favourable weather conditions). The maximum reduction in population size predicted by the density dependent model was 16% which equates to a decline in the number of pairs from 800 to 670.
31. The variation in counts of apparently occupied nests at Wicklow Head in different years is likely to be a combination of counting errors, chance variation caused by severe weather such as storms washing nests away, or variation in the extent of nonbreeding by adults in some years when environmental conditions adversely affect their body condition (Coulson, 2011). These variations notwithstanding, the population has remained relatively stable over the period of available counts, averaging around 800 pairs. This suggests that density dependent competition for resources has been operating to maintain the population around this size, although the mechanism for this is unknown. This being the case, the density dependent model would be expected to be the more reliable model on which to base predictions.
32. Therefore, reductions in population size which might be predicted due to additional mortality would be expected to reduce competition for resources and allow increases in other demographic rates, which will maintain the population. In other words, populations subject to density dependent regulation are buffered against potentially negative effects which do not impinge on the limiting resource.
33. When reviewing population model outputs it is important not to place undue weight on absolute predictions (e.g. of future population sizes), since these will only be reliable if the conditions experienced during the period of data collection are maintained. Models aiming to make such predictions need to include estimates of how demographic parameters will change in future. The present models cannot do that, and instead are based on historic demographic parameter values. Predicting future numbers would require estimates of how survival and productivity will be influenced by anticipated climate change and by the dynamics of fish populations, possibly with stocks of large predatory fish recovering (as a consequence of conservation measures) but with potential harvest of forage fish such as sandeels and sprats. Such absolute predictive modelling of population sizes is extremely challenging and is not within the scope of the work presented here. For this reason, while the population projections have been provided, these are intended to illustrate the differences in density independent and density dependent projections rather than as absolute predictions of future population size. The counterfactual outputs are considered to be more reliable metrics since these have been found to be relatively insensitive to input parameter values, which in the case of demographic rate estimates are necessarily subject to a degree of uncertainty.
34. Assessing the counterfactual leads to a more important requirement that the models provide robust simulations of the population dynamics (i.e. how they change) but not of the actual

population sizes themselves. In this context, the key consideration is that underlying factors which regulate populations are incorporated. These are included in the density dependent models, where reproduction in any given year of a simulation reflects the population size on the basis that breeding individuals compete (e.g. for breeding space, food, mates, etc.). The density independent models lack any form of inherent regulation, something which will nearly always be present in natural populations, so they are considered to be less suitable tools for the current exercise.

35. It must be stated that the density dependent models also have limitations. The density dependent mechanism used has been assumed to apply to reproduction rather than survival and, as the environment's carrying capacity is unknown, it has been assumed that the carrying capacity is the current population size. Neither of these assumptions is expected to be accurate, however they are both more reasonable than assuming there is no limit to the population size as the density independent models do. A density independent model will either predict continuous increase in a population or continuous decline to extinction. Neither is realistic. It should also be noted that modelling regulation of seabird populations through reproduction rather than survival represents a precautionary mechanism (i.e. the results presented here are very unlikely to over-state the buffering effect). This is because modelled increases in mortality can only be replaced through increased numbers of fledglings, and these individuals can only contribute to elevated population if they survive to breeding age (of which typically 50% do in seabird populations). Thus, there is both a time lag (to reach maturity) and also natural losses (younger age classes suffer higher natural mortality) which reduce the degree of compensation. If density dependence in this population in fact operates through elevated survival rates, mortality losses will be much more rapidly offset by enhanced survival of the remaining population, and recovery is therefore much more closely linked to losses.
36. Therefore, from the perspective of generating reasonable comparative predictions of novel impacts the density dependent models presented here are considered to be more reliable, whilst also retaining precaution.
37. Additional discussion on aspects of population modelling is provided in MacArthur Green (2014) along with modelling outputs which explore variations in the strength of density dependence.



## 1.5 References

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**ANNEX A. MODEL INPUT PARAMETERS AND SETTINGS**

These are the input parameters as entered in the NEPVA.

| Parameter / setting   | Density Independent                        | Density dependent  |
|---|--|--------------------|
| Environmental stochasticity   | Beta/gamma                                 | Beta/gamma         |
| Demographic stochasticity   | Included                                   |                    |
| Density dependent model   | N/A  | Weibull            |
| No. simulations   | 1,000                                      |                    |
| Random seed value   | 19   |                    |
| Years for burn-in   | 10   |                    |
| Productivity upper limit  | 3  |                    |
| Population units (input and output)   | Breeding pairs                             |                    |
| Initial population at start of burn in period (year)  | 750 (2020)                                 | 650 (2020)         |
| Initial population at end of burn-in/start of simulation (year)   | 800 (2029)                                 |                    |
| Productivity rate: mean (SD)  | 0.677 (0.27)                               |                    |
| Weibull a, b, c (density dependence on productivity)  | NA   | 1.25, -1.0e-4, 1.2 |
| Adult survival rate: mean (SD)  | 0.854 (0.051)                              |                    |
| Age class 0-1 survival rate: mean (SD)  | 0.79 (0.051)                               |                    |
| Age class 1-2 survival rate: mean (SD)  | 0.854 (0.051)                              |                    |
| Age class 2-3 survival rate: mean (SD)  | 0.854 (0.051)                              |                    |
| Age class 3-4 survival rate: mean (SD)  | 0.854 (0.051)                              |                    |
| Are impacts of scenarios specified separately for immatures?  | No   |                    |
| Are standard errors of impacts available?   | No   |                    |
| Should random seeds be matched for impact scenarios?  | No   |                    |
| Are impacts specified as a relative value or absolute harvest?  | Relative                                   |                    |
| Years in which impacts are assumed to begin and end:  | 2029 to 2066                               |                    |
| Impact levels   | 1 : 20 @ increments of 1                   |                    |
| Additional mortality (impact converted to increase in mortality as: impact/adult population, e.g. 1/1600, 2/1600, etc.) | 0.000625 : 0.0125 @ increments of 0.000625 |                    |
| First / last year of outputs  | 2029 / 2066                                |                    |