Client: The Crown Estate SOSS

Project: SOSS-04 Gannet Population Viability Analysis

Date: February 2012

Report: Demographic data, population model and outputs

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The SOSS steering group includes representatives of regulators, advisory bodies, NGOs and offshore wind developers (or their consultants). All SOSS reports have had contributions from various members of the steering group. However the report is not officially endorsed by any of these organisations and does not constitute guidance from statutory bodies. The following organisations are represented in the SOSS steering group:

SOSS Secretariat Partners:	The Crown Estate
	British Trust for Ornithology
	Bureau Waardenburg
	Centre for Research into Ecological and Environmental Modelling, University of St. Andrews
Regulators:	Marine Management Organisation
	Marine Scotland
Statutory advisory bodies:	Joint Nature Conservation Committee
	Countryside Council for Wales
	Natural England
	Northern Ireland Environment Agency
	Scottish Natural Heritage
Other advisors:	Royal Society for the Protection of Birds
Offshore wind developers:	Centrica (nominated consultant RES)
	Dong Energy
	Eon (nominated consultant Natural Power)
	EdF Energy Renewables
	Eneco (nominated consultant PMSS)
	Forewind
	Mainstream Renewable Power (nominated consultant Pelagica)
	RWE npower renewables (nominated consultant GoBe)
	Scottish Power Renewables
	SeaEnergy/MORL/Repsol (nominated consultant Natural Power)
	SSE Renewables (nominated consultant AMEC or ECON)
	Vattenfall
	Warwick Energy

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year in offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality.

Tables

1. NON-TECHNICAL SUMMARY

1.1 In order to assess the cumulative impacts of existing and consented wind farms on UK Northern Gannets *Morus bassanus* (gannets hereafter) populations and to determine a threshold 'harvesting rate' of gannets from these and future offshore developments population viability analysis (PVA) was undertaken in combination with Collision Risk Modelling.

1.2 Demographic parameters were obtained from an extensive literature review, providing data on colony sizes, age structure, adult and immature survival rates, breeding success, age at first breeding and non-breeding rates, immigration and emigration rates and migration routes.

1.3 The population model structure formulated was based on a matrix representing five gannet age classes; years one to four and from age five onwards, the assumed age of first breeding. The demographic rates were applied to this and environmental and demographic stochasticity incorporated. All modelling was conducted in the 'R' environment. Sensitivity analysis was conducted to assess the influence of changes to each demographic rate on the model prediction.

1.4 Because the literature review showed that there was no evidence for density dependence operating in the gannet population, the initial model developed was density independent. Following a meeting of the Strategic Ornithological Support Services group (SOSS) steering group it was decided that a density-dependent model should also be developed. Both models and their outputs are presented here.

1.5 The recently developed offshore collision risk model (Band 2011) was used to generate estimates of monthly collision mortality for all existing and consented offshore wind farms (Rounds 1 & 2). Gannet density estimates were obtained from the Environmental Statements from these wind farm developers, or where unavailable, calculated from other survey data sources. Mortality was assigned to age groups using regional proportions of adults to immatures from WWT Consulting's aerial survey database. An avoidance rate of 98% was used in the assessment of impacts, although outputs were also calculated for avoidance rates of 99%, 99.5% and 99.9%.

1.6 Having apportioned mortality to age groups, work was then done to apportion to the British Isles population and then to individual colonies, based on migration and foraging range data, particularly from recent geolocation logger studies. Proportions of gannets from each country were derived for each sea area through the winter. In the summer it was assumed that all adult gannets were in their national waters (with respect this study area).

1.7 As expected for a large 'k' strategist, the predicted population growth rate produced by the models was most sensitive to change in adult gannet survival rate with immature survival and breeding success having comparatively little impact.

1.8 In the absence of additional wind farm mortality, the baseline density independent model predicted a stochastic population growth rate of 1.28% per annum. This closely fitted the observed population growth rate between 1995 and 2004 of 1.33%. That obtained from the density dependent model was slightly lower at 0.87% per annum.

1.9 The risk of the British Isles gannet population declining by more than 5% within 25 years was 0.5% from the density independent model and 0.9% for the density dependent model.

1.10 The Collision Risk Modelling provided an estimate of annual mortality of 2,603 adult and 1,056 immature gannets in existing and consented wind farms around the UK coast. These estimates were critically sensitive to the avoidance rate applied and less so to estimated gannet densities and proportions. Changing the avoidance rate to 99% reduced the numbers killed to approximately 1,400 and 620 adults and immature respectively.

1.11 Using demographics, this mortality was allocated in the study to two regions; the Irish Sea and Solway (including the wind farms Barrow, Rhyl Flat, North Hoyle, Burbo Bank, Ormonde, Walney, W Duddon, Gwynt Y Mor, Robin Rigg E and Robin Rigg W and the gannet colonies Ailsa Craig (27,130 AONs), Scar Rocks (2,394 AONs), Ireland's Eye (285 AONs), Great Saltee (2,446 AONs), Grassholm (32,094 AONs) and St Margarets (1 AON)) and the south western North Sea (including the wind farms Teeside, Westernmost Rough, Humber, Lynn, Inner Dowsing, Scroby Sands, Docking S, Dudgeon, Race Bank, Sheringham Shoals, Lincs, Kentish Flats, Greater Gabbard, Gunfleet Sands I and II, London Array and Thanet and the colonies Bass Rock (48,065 AONs), Bempton (3,940 AONs) and Helgoland, Germany (190 AONs)). Approximately 1,310 adult gannets deaths could be allocated to gannetries in the former area and 1,520 to those in the latter. Due to mixing of populations at sea in the winter these estimates may include individuals from gannetries in other areas, so useful minimum estimates are 1,220 gannets killed in the former area and 765 in the latter area in the summer, when the adults are attending local colonies.

1.12 Impacts on colonies in other areas, such as those in Shetland or St. Kilda will be very much less from the existing wind farms as they may only encounter them in the winter during migration and wintering, most of which is spent in other areas, such as southern Europe and Africa. Allocation of mortality to immature birds was much harder since there are far fewer data on their movement from individual colonies. A cautious approach would be to allocate mortality in the same proportion as for adults, though this would certainly lead to an overestimate.

1.13 Calculations of the probability of population decline for the gannet population were made for the whole of the British Isles and for individual colonies. These were based on 5,000 runs of 25 years from 2004, the last census year, onwards, with the final population growth rate being the mean of all runs estimated over the final five years. The range of additional mortality, due in this case to collision with offshore wind farms, was set at 0-15,000 fatalities, in intervals of 500. The probability of decline was estimated as the proportion of each suite of simulations which declined at any point in the 25 year run below each stipulated threshold. These thresholds were defined at 5% intervals across the range 5-20%.

1.14 The density-independent model predicted that, on average, gannet numbers will continue to increase with additional gannet mortality due to collisions with offshore wind farms, up to a threshold of approximately 10,000 additional birds killed per year. At this level of additional mortality 50% of simulations would have negative population growth. A much lower level of risk, when 95% of simulations maintain positive population growth (i.e. >1), was achieved with a threshold additional mortality of approximately 3,500 birds per year.

1.15 Very small differences between the density dependent and density independent models' predictions were found over the range of conditions likely to pertain in reality (fewer than 10,000 gannet deaths per year at offshore wind farms in UK waters). Given this, the choice of which model to use is comparatively unimportant, particularly as density dependence in these gannet populations is not currently supported by empirical data.

1.16 The density-independent model was used to derive the threshold mortalities that could be sustained for each UK Special Protection Area (SPA) colony in the study area before the colony size was predicted to decline. For Bass Rock this was calculated as 2,000 birds, for Bempton Cliffs and Flamborough this was 150 birds, for Ailsa Craig and Scar Rocks (pooled) it was 1,100 birds, for Grassholm and Great Saltee, 1,400 birds and for St. Kilda and Flannans (pooled) 2,200 birds per year.

1.17 These results suggest that the presently expanding colonies can support higher levels of mortality than are currently experienced due to existing and consented offshore wind farms before the populations will decline. This study has provided a framework in which outputs from Collision Risk Modelling for individual wind farm sites and in assessing cumulative effects can be put into the context of national and colony specific gannet populations.

2. INTRODUCTION

Background

2.1 The Strategic Ornithological Support Services group (SOSS) has commissioned Wildfowl & Wetlands Trust (Consulting) Ltd (WWT Consulting), in collaboration with Bob Furness (MacArthur Green) and Mark Trinder (RPS) to undertake a population viability analysis (PVA) of Northern Gannets *Morus bassanus* (gannets hereafter) to inform on the cumulative effect on the species' population from collisions with existing and planned offshore wind farms in the UK.

Aims and objectives

2.2 The objectives of this project are to:

- Assess the cumulative impact of all existing and consented wind farms on UK gannet populations; and
- Determine, through PVA, a threshold of the 'harvesting rate' of gannets that can be sustained from the cumulative effects of collisions with existing and future wind farm developments without having a significant negative impact on UK gannet populations.

Scope of this document

2.3 The first stage of the project involved collating demographic data for the gannet at colonies in Britain and Ireland. These data were circulated to SOSS as an Excel spreadsheet, together with a narrative, on 10 August 2011, and comments were uploaded to the SOSS website shortly thereafter.

2.4 Following this, a brief report providing guidelines on the development and use of PVA was produced and circulated for comments in September 2011. This will be updated with any additional considerations that have arisen during the production of the gannet PVA presented here.

2.5 For the gannet PVA, a population model was developed in order to generate predictions of the potential impact on the UK gannet population of additional mortality, in this case caused through collisions with offshore wind turbines, although other sources of mortality would be expected to generate the same effects. This model was presented at a SOSS-04 project steering group meeting on 31st October 2011. Following discussions at the meeting this report presents the final versions of the population model and their outputs, incorporating requests from the different stakeholders. The appraisal of demographic data is included for ease of reference.

3. GANNET DEMOGRAPHIC DATA: APPRAISAL

The sources

3.1 Official estimates of gannet colony size are available from published papers on the status of gannets (Wanless *et al.* 2005, Murray *et al.* 2006) and from the books describing results of the national seabird surveys (Cramp *et al.* 1974, Lloyd *et al.* 1991, Mitchell *et al.* 2004, Forrester *et al.* 2007). Early counts were based on estimates of the numbers of 'breeding pairs' at colonies. Since the 1990s counts have followed the standardized protocol described for this species by Walsh *et al.* (1995) and are of 'apparently occupied nests' (AONs) or for photographs where nests are obscured by adults, 'apparently occupied sites' (AOSs). These units are essentially equivalent to 'breeding pairs'. There are a few estimates of breeding numbers that are not included in this study as they relate to years outwith the national census periods. The decadal census periods provide near complete coverage of the colonies, however data between are sparse for most colonies.

3.2 Breeding success data from British gannet colonies are summarised by Nelson (2002) for the period 1961 to 2000, and by the Joint Nature Conservation Committee (JNCC) annual reports 'Seabird

Numbers and Breeding Success in Britain and Ireland' for the years 1986 to 2006 (e.g. Mavor *et al.* 2008). JNCC have not yet published detailed data for seabird breeding success in 2007 and subsequently. Additional data on breeding success are provided in various annual bird reports, including Shetland Bird Report (e.g. Pennington 2010), and Fair Isle Bird Observatory Reports (e.g. Shaw 2008), both of which cover a longer run of years than provided by the JNCC reports. There are, however, some minor discrepancies in data reported by JNCC and by Shetland Bird Report, and Fair Isle Bird Observatory Reports, so we have tabulated data from both sources so that these errors are visible (the data clearly come from the same original source but some transcription errors appear to have been incorporated in some cases). Altogether, we have found 138 independent estimates of breeding success from nine colonies across the period 1961-2009. There may be a few more that could be added to the study but this sample size provides plenty of detail to look for variation among years or among colonies.

3.3 Nelson (1978) summarised what was then known of gannet age of first breeding, frequency of nonbreeding and age-specific survival rates. Much of this was reiterated, with updates when available, in Nelson (2002). Survival rates presented by Nelson are based on study of colour ringed breeding adult gannets, with immature survival inferred from a population model adjusted to output the observed rate of population growth. A detailed and comprehensive analysis of British and Irish ring recovery data was carried out by Wanless *et al.* (2006) to estimate age-specific survival rates of gannets. Gannet ringing at colonies in Britain and Ireland was mostly at the Bass Rock, Ailsa Craig, Hermaness and Great Saltee, with highest numbers ringed at the Bass Rock, and far more chicks ringed than adults. Data limitations constrain the analysis, and Wanless *et al.* (2006) concluded that there was insufficient data to assess with confidence whether survival rates differ between colonies or regions, or across years. Given that there have been rather few additional recoveries since their analysis it was considered not to be cost-effective to try to update this analysis at the present time, as it is unlikely that conclusions would change.

The data

Colony sizes

3.4 The gannet is probably the most accurately and extensively censused seabird in the world. There are few colonies (Figure 1, Appendix I for locations of gannetries in the UK and Ireland), the adults are large and nest in the open and are relatively easy to count by direct observation or by aerial photography. In addition to frequent counts at many colonies, there have been comprehensive surveys of gannet colonies in Britain and Ireland in 1902-05, 1929, 1939, 1949, 1959, 1969, 1984, 1994, and 2004. The count data permit measurement of the rate of growth of individual colonies, of regional, national and global breeding population size (though count data from some countries are less complete than for the UK). Only a few colonies pose significant counting problems, and with the use of high definition aerial photography the main problem is of discriminating between breeding nest sites and sites occupied by nonbreeders (the latter are excluded from the definition of 'apparently occupied nests' (and therefore from 'apparently occupied sites'), but at a few colonies the areas occupied by nonbreeders can be difficult to identify. Data in Table 1, Appendix II show that numbers have increased at all colonies, and several new colonies have been founded during the 20th century, a trend that has continued during the early 21st century. The rate of growth of the gannet breeding population in Britain and Ireland increased by an average of 2% per annum from 1969-1985, 2.2% per annum from 1985-1995, and 1.33% per annum from 1995-2005. The slower rate of increase since 1995 is consistent with a view that gannets will reach a population ceiling set by food supply, and that the population may be starting to get near to that limit. We should, therefore, (and did) investigate whether there is evidence of a density-dependent influence on breeding success or some other demographic parameters. The data presented represent more than adequate detail on colony sizes for the purposes of this study.

Breeding success

3.5 Estimates of breeding success (chicks reared per pair) were available for the period 1961 to 2009 from nine colonies in the UK; Hermaness (21 years), Noss (24 years), Fair Isle (21 years), Noup Head (2 years), Troup Head (11 years), Bass Rock (12 years), Bempton (25 years), Ailsa Craig (20 years), and Grassholm (2 years) (Table 2, Appendix II). All but the last two of these are colonies in the North Sea; west coast colonies are relatively poorly represented. Overall, mean breeding success was 0.698 chicks raised per apparently occupied nest (SD 0.089, se 0.008, 95% confidence interval 0.683-0.712). Excluding colonies with only one or two estimates of breeding success, the lowest breeding success was at Troup Head (0.601) and the highest at the Bass Rock (0.769). Troup Head is a small young colony where there will be a high proportion of inexperienced breeders. Bass Rock breeding data came from the 1970s and there may be decadal patterns to consider. Any further data on breeding success to add to the existing compilation would be welcome, together with the full citation of its source.

Survival rates

3.6 Nelson (2002) reported return rates of 0.889 for adult males and 0.913 for adult females based on colour ringed birds at the Bass study area from 1960-75. These will tend to be underestimates of survival rates since birds returning but moving to a nest site outwith the study area will be overlooked, as may be birds that lose colour rings, or avoid detection. From these rates he suggested that annual adult survival rate was about 0.94. Coupling this adult survival rate with his observed breeding success for Bass gannets and the observed rate of population growth he estimated survival of first year birds at 0.35, 2nd years at 0.9, 3rd and 4th years at 0.94. Wanless *et al.* (2006) calculated age-specific survival rates for British and Irish gannets, and separately for Bass gannets and for birds from other British and Irish colonies. These estimates were derived from analysis of ring recoveries of birds ringed as adults and chicks at four (Bass Rock, Hermaness, Ailsa Craig and Great Saltee) and ten (Bass Rock, Hermaness, Ailsa Craig and Great Saltee, Fair Isle, Noss, Grassholm, Little Skellig, Bull Rock and Scar Rocks) UK colonies, respectively. Their estimates are not too far from those of Nelson (2002) (see Table 3, Appendix II for actual figures compared with Nelson (2002)), but because they are based on the best available analysis of ring recovery data we plan to use the Wanless *et al.* (2006) data. Their analysis found little or no statistical support for differences in survival rates between colonies or between years, although the best model estimating adult survival rate across years suggested a slight decline in survival rate from 0.93 in 1960 to 0.89 in 2000. This decline is also consistent with the observed reduction in rate of population growth since 1995, hinting at a possible density-dependent effect. It should be emphasised that the estimates of survival rates are for a time period before construction of Round 1 and Round 2 offshore wind farms, and therefore these estimates do not require any correction to account for mortality at offshore wind farms. Monitoring of adult survival rates in future would include a mortality component due to offshore wind farms and this would need to be taken into account before inputting into population models.

Age at first breeding and nonbreeding rates

3.7 The only data on these parameters come from Nelson (1978, 2002) and are based on his detailed study of a selected part of the Bass Rock colony where many adults were individually colour ringed and observed over the 1960s and 1970s. It is widely accepted that gannets normally start to breed when 5 years old, but that a few 4 year olds also breed. Undoubtedly some older birds do not breed, and so the mean age of first breeding is likely to be similar to the mode of 5 years. However, there are no detailed data on the range of ages of first breeding, and we must assume that age of first breeding is similar at colonies away from the Bass. Similarly, there is very little quantitative information on rates of nonbreeding by adult gannets, or whether this varies among years, or among colonies. Nelson (2002) suggests that nonbreeding is very exceptional, based on his observations of colour ringed adults in the Bass study area in the 1960s and 1970s. Kubetzki *et al.* (2009) anecdotally reported one instance of nonbreeding by a gannet equipped with a data logger (out of a sample of about 30 equipped birds), all the others having a nest and egg in the year after the loggers were deployed on them. In the absence of data to the contrary, we plan to assume that nonbreeding rates are negligible in this species. This accords with the fact that gannets normally return year after year to exactly the same nest site. Nonbreeding in one year would result in birds losing their nest site and would lead to conflict over site ownership when the bird attempts to breed in a subsequent year and finds the site occupied by another. Nonbreeding has not been evident in gannet colonies up to now, but it is likely to be a consequence of density dependent competition if numbers reach a carrying capacity in future.

Migration routes and winter distributions of gannets

Historical information

3.8 Until recently, our knowledge of gannet migrations and wintering areas came from three sources: 'seawatching', ringing, and surveys at sea. Observations of gannets passing seawatching sites (mostly headlands along the European coast where birdwatchers record numbers of seabirds flying past during autumn and spring migrations) have been made since at least the 1950s, especially in the UK but more recently also in considerable detail in The Netherlands and countries down to southern Europe (Meraz Hernando 2011). Such observations indicate timings of migration, but numbers counted are strongly affected by whether winds are onshore or offshore.

3.9 Recoveries of dead gannets that had been ringed at colonies (mostly as chicks at the few colonies that are easily accessible and where catching gannets does not involve climbing cliffs) have given considerable information on the distributions of different age-classes of gannets from colonies in Britain and Ireland (Wernham *et al.* 2002). There is no clear evidence from ringing data that gannets from colonies in Britain and Ireland show differences among colonies in their wintering areas (Thomson 1974, Veron 1988, Wernham *et al.* 2002, Veron and Lawlor 2009). However, ringing effort has been high at the Bass Rock, moderate at Ailsa Craig, Hermaness, Grassholm and Great Saltee, and low or non-existent at other colonies. In particular, very little gannet ringing has been done at St Kilda, Sule Stack or Sula Sgeir, long-established and large colonies that represent a high proportion of the population and that are all located in the NW of the British Isles. Fewer ring recovery data are available from gannet colonies in other European countries, but these also provide some evidence of the migrations of birds from Iceland, Faeroes, Norway and Canada (Barrett 1988, Wernham *et al.* 2002), which do differ from the migrations of birds from colonies in Britain and Ireland.

3.10 At sea surveys include the European Seabirds at Sea (ESAS) database which holds information on numbers of gannets at sea counted using standardized methodology (Tasker *et al.* 1987). These data can be used to estimate gannet densities at sea in different months and locations and hence can provide evidence of seasonal changes in distribution (Tasker *et al.* 1985). ESAS data primarily relate to the North Sea and data are predominantly from the 1970s and 1980s with fewer surveys in more recent decades, but ESAS methods have also been used in marine areas west and south of the UK and to some extent in years from the 1990s to the present. In the North Sea, gannets in summer show distributions that relate to the locations of breeding colonies, with birds travelling out from the colony to forage up to 540km from the colony in the case of the largest colonies such as the Bass Rock (Hamer *et al.* 2001), predominantly on pelagic fish such as sandeels Ammodytidae, herring *Clupea harengus* and mackerel *Scomber scombrus*. Foraging ranges from smaller colonies are much shorter, and there is a strong tendency for gannets from neighbouring breeding colonies to avoid waters close to colonies other than their own colony. Many immature gannets, particularly of the 3 and 4 year old cohorts, attend colonies during the summer (mostly from May to August so for a shorter period than breeding adults are present), and those birds tend also to show 'Central Place foraging' with their feeding flights radiating out from the colony, but over larger areas of sea than used by breeding adults (Votier *et al.* 2011). Tasker *et al.* (1985) found that about 60% of gannets in the southern North Sea in summer were immatures, whereas in the northern North Sea this was only 20%. In winter, there are few immature gannets in the North Sea (fewer than 7% of all records), and densities of adults are lower than in summer (Tasker *et al.* 1985). Tasker *et al.* (1985) reported an average across the North Sea of 0.4 gannets per km^2 in October but only 0.04 per km^2 in December to February. gannet distribution in the North Sea show a stronger correlation in winter with the distribution of fishing vessels, as they scavenge extensively on trawl fishery discards in winter when pelagic fish are less available (Garthe *et al.* 1996). Off the west of Britain, gannets were found to be present in relatively much lower numbers in winter than in summer, with gannets in winter mostly associated with fishing vessels (Webb *et al.* 1990). Surprisingly few occurred within the Irish Sea at any time of year with peak abundance there (in September) still below 0.5 birds per km^2 (Webb *et al.* 1990).

3.11 Gannets from European colonies migrate southwards in autumn, as do Canadian gannets, but the latter predominantly remain in the west Atlantic and very few cross the Atlantic to winter in Europe (Wernham *et al.* 2002), so we do not need to consider Canadian gannets any further in this study. Younger birds (especially juveniles) tend to travel further south than adults. Ring recoveries of gannets are often caused by accidental capture of birds on baited lines or drowning in nets (34% of reported recoveries), or birds being deliberately killed (24% of reported recoveries) (Wernham *et al.* 2002). The distribution of these recoveries therefore is biased by locations where fisheries may take a bycatch of gannets and where it is legal to hunt gannets, and complicates understanding gannet migration patterns. Nevertheless, taking these biases into consideration, the broad pattern is clearly evident. Ring recoveries from 1939 to the end of the 1990s, indicate that juveniles from colonies in Britain and Ireland mostly wintered in areas from the Bay of Biscay to west Africa. Immatures (1 to 4 years old) had a similar winter distribution, but generally not as far south as juveniles. Adults (5 years old and older) mostly wintered fairly close to breeding colonies, in the North Sea or west of the British Isles, with some travelling to the Bay of Biscay but very few going further south. Only three out of 639 recoveries up to 1997 of adult gannets were from as far south as west Africa, with a further four from the Atlantic coast of north Africa (Wernham *et al.* 2002). These ringing data match well with observations from coastal seawatching and from at sea surveys (e.g. Woolridge 1974, Meraz Hernando 2011). Gannet southward migration is clearly evident in October, with spring northwards migration occurring as early as February. At sea surveys indicate that during the 1980s the North Sea held similar numbers of adult gannets to the numbers nesting on the North Sea coasts. On the basis of ring recovery data and observations of gannets on migration and in winter, Nelson (1978, 2002) suggests that most gannets breeding at the Bass Rock probably spend the winter in the North Sea or no further south than the English Channel. Ring recoveries of gannets from Iceland, Faeroes and Norway indicate that those birds occur in UK waters during winter, especially in the North Sea and the English Channel (Wernham *et al.* 2002). The ring recovery data suggest that Icelandic gannets were mainly recovered from the west coasts of Britain and Ireland whereas Norwegian gannets were only recovered from North Sea coasts (Barrett 1988, Wernham *et al.* 2002).

Recent studies deploying data loggers

3.12 In recent years, new technologies have become available that can be used to study gannet migrations and wintering areas. In particular, several studies have deployed geolocation data loggers on breeding gannets at various colonies. Geolocation data loggers are very small devices that can be attached to a leg ring on a breeding gannet. About a year later, if the gannet can be recaptured, the logger can be removed and data downloaded for analysis. These loggers record light intensity, and usually also temperature, on a time base. In principle, analysis of location from these data is simple. In the northern hemisphere, day length is longer further north in summer, shorter further north in winter. At any given latitude, sunrise occurs earlier further east. Using light intensity data recorded in the logger, the location of a bird can be estimated twice each day from the light data (Phillips *et al.* 2004). In some situations, temperature data can help with estimating location (Teo *et al.* 2004) since the temperature recorded will be sea surface temperature when the bird is sitting on the water (which gannets always do at night when away from the colony). Location estimates are imprecise. The average error is around 180km (Phillips *et al.* 2004, Teo *et al.* 2004). But this is adequate to establish the general area in which the bird is present. Geolocation does not work at the equinoxes, but this results in the loss of only a few weeks of data at those times of year. Logger data can also be used to infer behaviour of birds, especially amounts of time spent flying, and spent sitting on the water (Mackley *et al.* 2010). However, geolocation data loggers do not provide precise enough information on location to allow us to infer whether individuals passed through particular offshore wind farms or detoured around them (Masden *et al.* 2009), for which GPS accuracy data loggers would be more appropriate technology.

3.13 Kubetzki *et al.* (2009) deployed geolocation data loggers on breeding gannets at the Bass Rock in August 2002 and August 2003. They recovered ten loggers in 2003 and twelve in 2004. The loggers showed that, on average, these breeding birds left the colony on autumn migration on 5 October. Several birds initially moved north and spent some time feeding in waters off Norway before starting their southward migration. Movement south took about four weeks, and then each individual spent the winter in a relatively small area of sea before migrating back to the breeding colony in February. Four examples from their study, of movements of adult gannets nesting on the Bass Rock are shown in Figure 2, Appendix I.

3.14 Although, according to Garthe *et al.* (2007), gannets fly at an average speed of about 58km per hour (about 16 m/sec), migration took up to four weeks to complete, as birds spent considerable amounts of time sitting on the water or foraging locally rather than travelling consistently towards their goal, so net movement was often only 200 to 400km per day. Four adults wintered within the North Sea/English Channel. Six wintered in the Bay of Biscay. Two wintered in the western Mediterranean. Ten wintered off West Africa. These results are in strong contrast to the previously established view that adult gannets from the Bass Rock predominantly winter in the North Sea and only extremely exceptionally travel as far as Africa. Kubetzki *et al.* (2009) suggest that gannet migration behaviour may have changed in recent years, in response to changes in fish stocks and fisheries. In particular, amounts of fish discarded in the North Sea have been drastically reduced in recent years, whereas large fisheries have developed on the West African continental shelf and large quantities of discards are generated in that region (Meraz Hernando 2011). Almost all gannets (over 88%) seen on the West African shelf occur behind fishing vessels (Camphuysen and van der Meer 2005). In support of this suggested change in gannet winter distribution, Garthe (unpublished) analysed the ESAS database and found that the numbers of adult-plumaged gannets present in the North Sea in winter have declined since the 1980s despite very large increases in the gannet population. None of the birds carrying loggers wintered over deep water; all were on the continental shelf sea, wintering in areas where there are large fisheries as well as large stocks of pelagic fish (Meraz Hernando 2011).

3.15 For birds where the logger data indicated migration routes used by breeding adults from the Bass Rock, twelve individuals migrated southwards through the English Channel, and eight left the North Sea around the north coast of Scotland and flew southwards west of the British Isles. On spring northward migration, only three birds moved back into the North Sea through the English Channel, while six moved into the North Sea around the north of Scotland (some loggers failed to record spring migration route because battery power was depleted). Birds that left in autumn through the English Channel did not consistently return by the same route but in several cases moved north by a westerly route. A further deployment of loggers on Bass breeding adults in summer 2008 showed similar results (Garthe *et al.* 2010). Out of 21 loggers recovered, none of the birds wintered in the North Sea, eight wintered in the Celtic Sea/Bay of Biscay/Iberia, two wintered in the western Mediterranean, and eleven wintered off West Africa (Meraz Hernando 2011). On southward migration, 14 left the North Sea through the English Channel, and seven around the north of Scotland (apparently none of these birds flew overland from the North Sea to the Irish Sea or Atlantic). On northward migration in early spring, five entered the North Sea through the English Channel, and 16 flew up the west coast of Ireland and into the North Sea around the north of Scotland.

3.16 Geolocation data loggers have been deployed on breeding adult gannets at several other colonies since 2004, including in Norway, Iceland, Wales and France (Fort *et al.* 2012). In summary, Norwegian breeding gannets mainly wintered in the North Sea to Bay of Biscay from late October to late January then returned to Norwegian waters in February. Welsh gannets (from Grassholm) dispersed in autumn to areas from the west of Scotland to West Africa but wintered predominantly in the Bay of Biscay area. French gannets mainly wintered off West Africa. In all cases, birds wintered on average a maximum of about 3000-4000km south of their breeding area, as also did birds from the Bass Rock, suggesting that migration followed a 'chain-migration' pattern, with birds shifting consistent distances south and so preserving in winter the same relative distribution to birds from other colonies as they have in the breeding season. More recently, 12 loggers deployed on gannets at a colony in Iceland in summer 2010 were recovered in summer 2011 and preliminary analysis of these loggers indicates that the Icelandic gannets wintered from West Africa to west of Scotland (Garthe, Furness, Montevecchi and Halgrimsson unpublished data). During autumn migration, some of these birds passed through the North Sea and English Channel (5 out of 12) whereas in spring all returned northwards past the west of Ireland. Similar results have emerged for adults satellite tracked from Bempton in 2011 (Langston unpubl.). Most breeding adults left Bempton in late September, with one bird heading north and travelling around the north of Scotland and down the west coast of Ireland before continuing southward. Other individuals spent time in the southern North Sea in relatively limited areas before at least one of these headed south. The last, most southerly records, were obtained off northwest Africa and in the Bay of Biscay in the second half of October when unfortunately tags ceased operation (Langston in litt.).

4. GANNET POPULATION MODEL: METHODOLOGY

Demographic rates

4.1 Age specific survival rates, estimated by Wanless *et al.* (2006) (summarised in Table 4, Appendix II) were used to develop the population model, and since no support for colony differences in survival was found the combined survival estimates were used. The model comprised five age classes, the first four of which covered the period from fledging to breeding at age five (i.e. transitions for 0-1, 1-2, 2-3, 3-4) The final survival rate was a composite adult age class from the fifth year onwards.

4.2 Wanless *et al.* (*op. cit.,* Appendix II) also provided estimates of process variance on the survival rates and these were used in the model (Wanless *et al.* (2006) provide details of the methods used). The benefit of removing sampling variance from the overall variance estimates is through improved precision of the subsequent population predictions. The difference in model predictions obtained when using total variance (i.e. process and sampling) versus just process variance can be seen in Figures 3 and 4 in Appendix I.

4.3 The assumed age at first breeding used in the model was five. The number of chicks produced per pair was derived from all the available estimates of productivity (see Table 2, Appendix II and Paragraph 3.5 above).

4.4 To generate an overall productivity value for use in the population model, an annual average was calculated for each year between 1986 and 2009 (the more recent run of continuous values). The average and standard deviation of each of these was then calculated, giving an average number of chicks raised per pair of 0.69 (SD 0.035).

Population model structure

4.5 The age based survival rates lent themselves to the development of a five-age class matrix model. This had annual survival transitions between the ages of 0-1, 1-2, 2-3, 3-4 and an adult age class for all remaining individuals. Reproduction was confined to just this adult age class, hence the population matrix of average rates (*a*) was:

$$a = \begin{bmatrix} 0 & 0 & 0 & 0 & 0.32 \\ 0.42 & 0 & 0 & 0 & 0 \\ 0 & 0.83 & 0 & 0 & 0 \\ 0 & 0 & 0.89 & 0 & 0 \\ 0 & 0 & 0 & 0.89 & 0.92 \end{bmatrix}$$

4.6 The model represents a post-breeding census. Thus, the annual cycle of the model can be considered as comprising a census of the population, carried out immediately after fledging, with the first age class defined as birds hatched that year, followed by a 12 month period of survival, after which surviving adults breed, followed by the next census.

4.7 The values on the sub-diagonal (from row 2, column 1 to row 5 column 4) represent each of the annual sub-adult survival rates and the value in row 5 column 5 is the adult survival rate. The value in the top row represents fecundity and was derived as half the annual productivity rate (to turn the per pair value calculated above into an individual rate) multiplied by adult survival.

4.8 Simulation models can include two types of stochasticity: environmental and demographic. Demographic stochasticity is defined as temporal variation driven by chance variation in the fate of individuals. As such its magnitude is strongly dependent on population size. As a rule of thumb, it can be ignored (from a modelling perspective) in populations of more than 100 individuals or in which the population of the most important life stage (e.g. adults for long-lived, slow breeding species) is more than 20 individuals. Environmental stochasticity is temporal variation driven by changes in the environment (e.g. weather), and is not a function of population size.

4.9 Environmental stochasticity was incorporated using the standard deviation (SD) of the mean demographic rates. The SD provides a measure of the temporal variation present around the mean value for each rate. The mean and SD can therefore be used to generate random values which share the same pattern of variation as observed in the study population. It is important to select a suitable probability distribution for the demographic rate in question, to ensure the randomly generated values are appropriate to that rate. Thus for survival, a probability distribution bounded at zero and one is essential, while for productivity a distribution bounded at zero is required. For the modelling presented here, random survival rates were generated using a beta distribution and productivity rates were generated using a log-normal distribution. These two distributions were selected as they have been found to provide close correspondence between observed and simulated values (Morris and Doak 2002).

4.10 Demographic stochasticity was included in survival by drawing the estimated number of surviving individuals at each time step from a binomial distribution. While its inclusion may appear unnecessary, given the above guidance that its effects are undetectable in populations larger than 100 individuals, this is considered to be a precautionary step, should any particular modelling scenario trigger a sufficiently large population decline that its effects could become important. A comparison of model outputs was conducted to ensure that including both forms of stochasticity did not inflate variance in simulated population trajectories and no such effect could be detected.

4.11 All population modelling was conducted with the R environment (http://www.r-project.org/). The model was originated and developed specifically for the British Isles gannet populations and does not make use of any published PVA software.

Exploration of density dependent relationships

4.12 There is currently no evidence to indicate that density dependent population regulation is operating in the British Isles gannet population. Thus for the gannet population a density-independent model appears to be a more appropriate model to employ than one including density-dependence. Nevertheless, we chose to examine both density independent and density dependent models, since it is inevitable that density dependence will affect the gannet population at some population size, if the numbers continue to increase. The density dependent and density independent models we developed are closely similar, especially over small variations in population size. Differences between these two models become more apparent over large ranges of population size, and so the models may diverge where projections are being made over long time scales, but we would caution against extensive extrapolations of models. There is a case that using the density independent model is appropriate because empirical data show no evidence of density dependent variation in demographic parameters, but reasons for considering density dependence are explained below, and our exploration of both types of model was strongly mandated by SOSS during planning.

4.13 There are suggestions that breeding adults may be modifying their behaviour as breeding colonies increase in size (Hamer *et al.* 2000, Lewis *et al.* 2001). This has been seen in increases in foraging trip duration and other aspects of breeding ecology. It is notable, however that none of these changes have yet led to detectable impacts on demographic rates. Nonetheless, it seems likely that if adults are expending more energy to achieve successful reproduction then they may be compromising their future survival. Indeed, there is evidence from changes in breeding numbers, that the rate of growth of the gannet population has declined as population size has increased (Mitchell *et al.* 2004, Wanless *et al.* 2005, Forrester *et al.* 2007), and growth rates of individual colonies suggest density-dependent reduction in population growth rates in the larger colonies (Lewis *et al.* 2001). Analysis of the available data suggest that if density dependence is operating within British Isles gannet population it does *not* appear to be operating on:

- reproduction: there is no evidence from any of the colony data for any trends in reproductive rates. This is based on a very large data set on breeding success, lending considerable support to the assertion that this parameter has not shown density dependent variation over the current range of densities experienced at colonies in the British Isles;
- age at first breeding. There is no evidence to suggest the presence of large numbers of non-breeding adults at any monitored colonies, and it is felt that such non-breeding individuals would be obvious if they were present; and
- juvenile and immature survival. There is no support for this in the data, and moreover the mechanism by which immature survival would respond to an increase in the population is not clear, since young birds can, and apparently do, disperse away from colonies to regions where gannet densities at sea are lower.

4.14 On this basis, it seems reasonable that if density dependence is operating in the population its effects will only impact on adult survival.

4.15 The analysis of survival rates reported by Wanless *et al.* (2006) included models with a time-trend (across the period 1960 to 2000). These models received weak support, but the ring recovery data were too limited to provide a robust estimate of the rate of decrease in adult survival. Nevertheless, the small negative trend reported in adult survival formed the basis of investigations into the presence of a density dependent effect.

4.16 A negative relationship was identified between the adult survival rates generated from the time-trend model (of Wanless *et al.* 2006) when plotted against the total number of AON ($p < 0.001$, Figure 5, Appendix I). With so few data points a linear relationship was considered the most robust model to explain this apparent relationship:

$$\textbf{Adult survival} = -2.04\text{e-}7 \times \textbf{AON} + 0.951 \qquad \text{[Equation 1]}$$

4.17 This relationship was incorporated into the population model. In order to include both density dependence and stochasticity in the adult survival rate, the following approach was taken:

1. The adult population in the previous year of each simulated population was divided by two (to turn this value into breeding pairs, equivalent of AON counts);
2. the estimated AON was entered into Equation 1 to generate a predicted mean estimate of density dependent adult survival; and
3. this mean value, together with the process variance on survival (from Wanless *et al.* 2006) was used to generate random survival rates for each simulation (from a beta probability distribution) which were centred on the mean value.

4.18 The ability of the density dependent relationship described by Equation 1 to predict the observed trend in the British Isles population was tested using the model. The initial population size was set at the 1962 population count, with the model run for 43 time steps (years) to match the period of censuses (the most recent was in 2004). In this format the model provided a poor match to the observed trend (Figure 6, Appendix I).

4.19 It was assumed that the empirical data on breeding numbers of gannets in the population are reliable. Therefore the empirical changes in breeding numbers were prioritised over the weakly supported trend in adult survival rate. Thus the slope of the line in Equation 1 was adjusted sequentially and the fit of the median population projection to the observed population counts determined using a least squares approach.

4.20 The best-fit slope identified by this process was -1.29e-07. The population model output obtained using this rate, and the fit of the line to the survival and AON data are provided in Figure 7, Appendix I.

4.21 Thus, a density dependent decline in adult survival derived from the trend reported in Wanless *et al.* (2006) generated a poor fit of the model to the observed counts. In order to obtain a satisfactory fit, the slope of the line needed to be reduced by approximately one third. We consider this latter model to be the most appropriate density-dependent model, given the greater level of confidence in the reliability of the population size data and the lower level of confidence in the trend in adult survival rate.

4.22 For the outputs presented 5,000 simulations of 25 years were performed. The initial population size was set at 890,000, distributed as (170,000, 77,000, 64000, 57,000, 522,000) for each age group. This was obtained from the stable age distribution generated by the model, using the most recent estimate of breeding pairs (261,000; Table 1, Appendix II).

Changes to probability of population decline – British Isles populations

4.23 Calculations of the probability of population decline for the gannet population of the British Isles are based on the following:

- population model runs from 2004 onwards (2004 being the last national census year);
- population size is based on the 2004 gannet census that found 261,000 AONs (=pairs);
- age of first breeding is taken to be 5 years old;
- all adults 5 years and older breed each year;
- for baseline model runs (either density dependent or density independent) there is no additional mortality due to offshore wind farms;
- for density independent simulations mean adult survival rate is 0.919 (SD 0.0118) (Wanless *et al.* 2006 value for all colonies; Table 4, Appendix II);
- for density-dependent simulations adult survival was modelled as a straight line relationship with AON in the previous time step (Equation 1). The intercept on the y-axis (at population size 0) was 0.9506 and the negative slope with AON was -1.29e-07 (NB- this is the modified slope determined by fitting the line to the observed trend, not from the original survival and AON data). This relationship was used to define a

nominal 'mean' adult survival rate for each iteration of each simulation independently. The density-independent standard deviation on adult survival was used in combination with these mean values to generate stochastic density-dependent survival rates for each simulation at each time step;

- survival rates of immatures are: 1st year 0.424 (SD 0.079), 2nd year 0.829 (SD 0.032), 3rd year 0.891 (SD 0.032), 4th year 0.895 (SD 0.032) (values from Wanless *et al.* 2006 for all colonies);
- breeding success averages 0.69 chicks per pair (mean of 137 values collated in this study, SD 0.035) in all models;
- stochasticity is incorporated as measured process error for survival rate reported in Wanless *et al.* (2006), and as measured variance in breeding success from the 137 values collated in this study);
- for all models, 5,000 runs were used to produce confidence intervals and mean output values;
- additional mortality at offshore wind farms is allocated among age classes in proportion to their abundances in the population; and
- survival rates were modelled using random draws from a beta distribution and productivity was modelled using random draws from a log-normal distribution.

4.24 In analysis of the models with additional mortality (attributed here to offshore wind farms), the range of collision rates was set at initial values between 0-15,000 at intervals of 500 (note this value was converted into a proportion of the initial population size in order that the mortality was proportional to the population size throughout the simulation, hence use of the word 'initial'. The actual mortality levels experienced in each year can be calculated as the initial value multiplied by the population growth rate). The population growth rate is the mean across all 5,000 simulations, estimated after removing the first 5 years of each simulation to remove any starting effect influences. The probability of decline is estimated as the proportion of each suite of simulations which decline at any point in the 25 year run below each threshold. These thresholds were defined at 5% intervals across the range 5 – 20% (higher values were also used but they all returned very low / zero probabilities).

Collision Risk Modelling

4.25 The recently updated offshore wind farm collision risk model (Band 2011) was used to generate estimates of monthly collision mortality for all existing and consented offshore wind farms within UK waters. This model requires estimates of the monthly density of birds in flight as input to the process.

4.26 Gannet survey data were obtained in a range of formats for different Round 1 and 2 offshore wind farm sites and was collected using different techniques. For some sites data were extracted from Environmental Statements (ES) provided on request from developers, but for other sites an ES could not be obtained. For a few of these cases the wind farm companies provided raw survey data, from which densities of flying gannets within the wind farm footprint were calculated. For sites where neither an ES nor boat survey data were provided, gannet densities were estimated using nearby sites (on the assumption that densities would be similar). If there were no suitable nearby sites, WWT Consulting aerial survey data were used. These data cover many sites in most months, however using these data generates density estimates across a much larger scale (e.g. 1,000km^2) than the wind farm project sites. Hence using these data makes an assumption of even distributions of gannet across the entire aerial survey areas. The data source used for each month for each wind farm is indicated in Table 9 (Appendix III).

4.27 The CRM uses the density of birds in flight. However, despite scrutiny of the data provided, for some wind farm density estimates it was not possible to determine if the values provided referred to the density of birds on the sea surface, in flight or a combination of both. Thus the density estimates used may include data for birds sat on the sea surface, which will increase the estimated collision mortality. For all those sites where the source of the data was known, only birds in flight were used in the calculations.

4.28 Comparison of in-flight density estimates obtained from aerial and boat data, for sites and months where both were available, indicated that aerial derived densities were typically lower. This comparison used data for eight wind farms, with a total of 31 possible comparisons. Boat data gave higher densities in 24 cases (77%). The average ratio of boat density to aerial density was 20.3, but this was heavily influenced by a few large values, thus the median ratio of 7.3 was considered to be a more reliable indicator of the differences between the datasets. This value was used as a correction factor to multiply the aerial density estimates for those sites where only an aerial density estimate was available. In this manner, aerial data were 'converted' into boat density estimates. Converting the aerial data to match the boat data ensures that the collision estimates obtained are equivalent to those typically submitted with wind farm EIAs. It is worth noting that the reason higher gannet densities are obtained from boat surveys than aerial is not known, but may include the attraction of seabirds such as gannets to vessels or possible avoidance of the flight line of aircraft.

4.29 Collision mortality estimates were generated for each wind farm site in each month using the gannet density estimates (Table 9), wind farm specifications (e.g. site dimensions, turbine specifications, etc.) and gannet biometric data taken from the literature (Table 10; Snow and Perrins 1994, Pennycuick 1997). The proportion of gannets at potential collision height (PCH; 0.14) was taken from the draft SOSS-02 report (Cook *et al.* 2011; note this value was updated to 0.158 in a later draft after the current analysis was conducted. If required the updated PCH can be applied to the final results by multiplying mortality by 0.158/0.14).

4.30 The ratio of immature birds to adults in each region and month (Bradbury 2011) was used to apportion the total collision mortality to age classes. In all collision estimates presented in this report an avoidance rate of 98% was adopted.

Apportioning mortality to populations

4.31 Caution needs to be exercised in describing gannet winter distributions and migration routes since these appear to be changing in response to changes in fishing patterns in Europe and off West Africa. Data from 120 gannets equipped with geolocation data loggers have provided some information on the movements of adult birds from different populations. Many of the data are reported in a paper by Fort et al. (2012). That study reported on 86 loggers in total, from two colonies in Norway, from Bass Rock, Grassholm and one colony in France (Rouzic). In addition we have data from another 20 gannets from Bass Rock a few years after the birds reported in the Fort *et al.* paper, and from 18 gannets from a colony in Iceland (both these latter data sets shared by Stefan Garthe, Bob Furness and others and not yet published). From the above data we can make the following generalisations :

- Very few gannets from Canadian colonies cross the Atlantic to winter in Europe, so we can effectively exclude Canadian gannets from further consideration. No European gannets are known to cross the Atlantic to winter in Canadian or American waters, so we can assume that all European gannets are in European or African waters during winter;
- some Icelandic, Faeroese and Norwegian adult gannets appear to winter in waters around the British Isles, and so estimates of collisions at offshore wind farms in UK waters in winter will include some gannets from those countries. The Norwegian gannets are more likely to be in the North Sea and the Icelandic gannets more likely to be west of the British Isles, but there are overlaps in their winter ranges, especially in the Celtic Sea. Some immature gannets from those countries may also be in UK waters in winter, but some are likely to be further south;

- some UK and Irish gannets winter in UK waters, but it seems that most now migrate further south to winter from the Bay of Biscay to west Africa;
- French gannets mostly winter off west Africa and will spend very little time in UK waters;
- autumn migration of adult gannets occurs mainly during October. Some (but a minority) from the Bass Rock, and presumably also from other North Sea colonies, leave the North Sea around the north of Scotland rather than through the English Channel. During return migration of adults in spring (mostly in February), gannets heading for North Sea colonies appear more likely to use a route around the north of Scotland rather than through the English Channel. According to the data logger data, rather few gannets spend time in the Irish Sea during migrations or in winter; and
- logger data and ring recovery data suggest that 'foreign' adult gannets will only be in UK waters from mid-October to mid-February. During March to September those foreign adult gannets will be in their own national waters associated with colonies that are too far distant for breeding adults to travel to UK waters while foraging. So we can exclude foreign adult gannets from consideration in relation to UK offshore wind farms from March to September. However, some immature gannets (especially the youngest age classes) from more northern countries may remain in UK waters throughout the summer, as may some local immatures.

4.32 Having established these broad patterns, we need to decide on how to apportion any estimated collision mortality at particular sites in particular months between gannets from the British and Irish population and gannets from the Icelandic, Faeroese and Norwegian populations, based on the information available on these movement patterns and the sizes of the relevant populations. Our best estimate of the numbers of adult gannets wintering in each area from each breeding population is based primarily on the geolocation logger data, and is shown in Table 6, Appendix II.

4.33 A similar apportioning of collision mortality to immature birds will be required, but that is more difficult since the distributions of immatures are less well known. For the purposes of that apportionment it may be best to simply assume that all immatures in British and Irish waters in summer are from British and Irish colonies, and that in winter immatures from all countries show the same general distribution.

5. RESULTS

Demographic rate sensitivity analysis.

5.1 Figure 8, Appendix I shows the mean population growth rate obtained from proportional change in each demographic rate. This shows rate changes using the density independent model, but note, as discussed above, results from the density dependent model would be very similar. The outputs have been combined into a single plot, but the simulations were run separately for each rate. The outputs for the first four survival rates gave nearly identical results, hence the lines for 0-1, 1-2 and 2-3 are hidden under that for 3-4.

5.2 The rates were generated using 5,000 simulations of 25 years at each level of demographic rate change. The population growth rate was calculated in each case after the first 5 years of a simulation were removed to avoid any effects due to starting conditions. Prior to each sequential simulation, the chosen demographic rate was perturbed by a value in the range 0.95-1.05 (at intervals of 0.005), and the mean population growth rate calculated from the resulting 5,000 simulations. Thus, each rate was adjusted between +/-5% of its mean value. Adult survival also impacts on reproduction, since anything that reduced the number of adults in the population also reduces the production of young.

5.3 This sensitivity analysis clearly shows the population growth rate is most sensitive to change in the adult survival rate, while changes in immature survival and reproduction have comparatively little impact on the population growth rate. It should be noted that, since adult survival also impacts on reproduction, anything that reduces the number of adults in the population also reduces the production of young. Thus changes in adult survival have a double effect.

Baseline model outputs

5.4 A comparison of the density dependent and density independent model outputs in the absence of additional mortality, running them from the 2004 population size can be seen in Figures 9 and 10, Appendix I. The stochastic population growth rate predicted by the density dependent model was 1.008 approximating to a rate of 0.87% pa. The equivalent for the density independent model was 1.013 approximating to 1.28% pa. The observed population growth rate between 1995 and 2004 for the British Isles population was 1.33%. Thus the density independent model predicts this rate of growth into the future, while the density dependent one predicts a slower rate of growth.

5.5 Approximately 22% of density independent simulations declined below the initial population size at some point during the 25 year projection, however only 0.16% were smaller than the initial population size in the 25th year. The risk of the population declining by more than 5% within 25 years was 0.5%.

5.6 Approximately 26% of density dependent simulations declined below the initial population size at some point during the 25 year projection, however only 0.02% were smaller than the initial population size in the 25th year. The risk of the population declining by more than 5% within 25 years was 0.9%.

Existing wind farm mortality

5.7 The density estimates used to generate collision mortality for Round 1 and 2 offshore wind farms are presented in Table 9 (Appendix III). This table also indicates the data source used in each case. The other parameters used to generate collision mortality are presented in Table 10 (Appendix III).

5.8 Figure 11, Appendix I shows the estimated numbers of adult gannet collisions at Round 1 and 2 wind farms in the UK sector of the North Sea. The peak predicted in October coincides with geolocation data logger evidence that gannet autumn migration occurs in October and many birds move southwards through the North Sea during this period. The higher numbers in summer than in winter fits with study data with relatively few gannets remaining in the North Sea in winter (but with some from Iceland, Faeroes and Norway). Summer (April to September) adult mortality will effectively be entirely from colonies in the North Sea area.

5.9 Figure 12, Appendix I shows the estimated mortality (individuals) for gannets at wind farms in the Irish Sea and Solway Firth. The peaks in May and September are not expected and are difficult to explain. Spring migration occurs in February/March and autumn in October, so April to September mortality will presumably be of adults foraging from colonies to the south-west (e.g. Grassholm) and north-west (Ailsa Craig). Possibly breeding adults travel further from these main colonies early and late in the breeding season and so interact more with these wind farms at these times. The lack of gannets during winter months in the Irish Sea is evident from these data.

5.10 Table 7, Appendix II shows estimated total numbers of adults killed each month by all Round 1 and Round 2 offshore wind farms in UK waters based on the new Band model with 98% avoidance. Apportioning these among populations of the British Isles, Iceland, Norway, and France, assumes that all adults are in national waters in summer (March to September) and distribution in winter is estimated from geolocation data logger data. This suggests that during winter in waters of the North Sea, English Channel, Celtic Sea, Irish Sea and west of Scotland, there are 131,500 gannets, of which 90,000 are from the British Isles, 32,000 from Iceland, 4,400 from Norway and 5,100 from France. Note that only 223 of the adults killed each year appear to be from countries other than the British

Isles so this adjustment for 'foreign' adults makes only a small difference to the estimated total killed each year (2,603 rather than 2,826). This is because very little of the annual mortality occurs in winter, and the population sizes from Iceland and Norway are very much smaller than from the British Isles, and relatively few French gannets winter in UK waters. Minor adjustments to estimated distributions of adults in winter will alter the estimated total mortality of the British Isles adult gannets only very marginally. In contrast, altering avoidance rate from 98% to 99% would dramatically alter estimated numbers killed (from ca.2,800 to ca.1,400). We can conclude that the avoidance rate estimate is critically important in this calculation while slight adjustments to details of at sea distributions of adults have a trivial effect on totals.

5.11 Band model estimates that 1,241 immature gannets are killed at offshore wind farms each year (Table 8, Appendix II). Of these, 1,056 are likely to be from the British Isles colonies if all populations are mixed at sea. If immatures are predominantly in their own national waters in summer, the total for the British Isles may be closer to 1,241 than to 1,056. Since very few of the casualties (less than 6%) occur during winter, the relative distribution of these immatures in winter is not going to have any significant influence on estimated totals. As with adults, the avoidance rate is far more important in influencing the estimated numbers killed. Changing avoidance rate in the CRM from 98% to 99% alters the total of the British Isles immatures estimated to be killed from 1,241 to 620.

5.12 Breakdowns of monthly adult and immature collision estimates for each wind farm included in this analysis are provided in Tables 11 and 12 (Appendix III). Owing to the variety of data sources it was necessary to use to estimate gannet densities it is likely that some of the collision mortality values presented do not exactly match those presented within individual ESs. Without complete access to original survey data (which was not available for this study) it is inevitable that such differences will occur. Nevertheless, the collision estimates still provide a useful guide to the total gannet collision mortality which CRM attributes to Round 1 and 2 wind farms.

5.13 It should also be noted that only 33% and 28% of the total collision mortality calculated (for adults and immatures respectively) is attributable to existing (i.e. operational) wind farms. A complete breakdown of estimated mortality by wind farm status (approved, operational, submitted and under construction) is provided in Table 13.

Allocation of mortality to colonies within the British Isles

5.14 Taking the estimated annual mortality of 2,830 adults (including those from populations beyond the British Isles), of which about 2,603 adults and 1,056 immatures are from the gannet population of The British Isles, can this be allocated to birds from particular gannet colonies? It can, as follows, for adults. Of this total of 2830 adults, 1,310 adults are estimated to be killed at offshore wind farms in the Irish Sea and Solway (Barrow, Rhyl Flat, North Hoyle, Burbo Bank, Ormonde, Walney, W Duddon, Gwynt Y Mor, Robin Rigg E and Robin Rigg W), of which 1,220 are killed in 'summer' (defined as April to September). During winter (November to February) and migration (March, October), birds from many colonies may be present in this area. However, during the breeding season (April to September) offshore wind farms in the Irish Sea and Solway are only likely to affect adult gannets from nearby colonies. These are Ailsa Craig (27,130 AONs), Scar Rocks (2,394 AONs), Ireland's Eye (285 AONs), Great Saltee (2,446 AONs), Grassholm (32,094 AONs) and St Margarets (1 AON). Adults from colonies further away than these are assumed to be unlikely to forage within the Irish Sea, both because the commuting distance is apparently excessive, but also because telemetry studies suggest that adult gannets rarely forage within the main foraging grounds of birds from other colonies (Grecian *el al.* in press), so birds from further afield are unlikely to forage within waters close to Ailsa Craig, Grassholm etc. This comes to a total breeding population of 64,350 AONs (128,700 adult birds) affected by mortality of an estimated 1,310 adults over the whole year (a few of which are from populations beyond the British Isles), or 1,220 adults during the 'summer'. We can take 1,310 adults from this population of 128,700 adults as the 'worst case scenario' and 1,220 adults from this population of 128,700 adults as a more conservative estimate (i.e. assuming that none of the local birds are killed during migration and winter and that all of that mortality can be allocated to birds from other colonies). The true situation is likely to lie between these two scenarios.

5.15 In the southwestern North Sea, annual mortality of adult gannets at offshore wind farms in UK waters (Teeside, Wm Rough, Humber, Lynn, Inner Dowsing, Scroby Sands, Docking S, Dudgeon, Race Bank, Sheringham Shoals, Lincs, Kentish Flats, Greater Gabbard, Gunfleet Sands I and II, London Array, Thanet) is estimated at 1,520 adults, with 765 of these killed in 'summer' (April to September). For the reasons outlined above, in summer, these deaths will be drawn from the local breeding colonies: Bass Rock (48,065 AONs), Bempton (3,940 AONs) and Helgoland, Germany (190 AONs). This comes to a total breeding population of 52,195 AONs (104,390 adults). However, during migration and in winter the mortality may be spread over a larger population of gannets from further afield, migrating through the area and wintering in the area. We can take 1,520 adults from this population of 104,390 adults as the 'worst case scenario' and 765 adults from this population of 104,390 adults as a more conservative estimate (i.e. assuming that none of the local birds are killed during migration and winter and that all of that mortality can be allocated to birds from other colonies). The true situation is likely to lie between these two scenarios.

5.16 Impacts of mortality on other colonies of gannets (for example those in Shetland, or on St Kilda) will be very considerably less than for colonies within foraging range of offshore wind farms during summer, since birds at Shetland or St Kilda will only come into range of offshore wind farms during migration (October, February/March) or during their relatively brief winter period (November to February), most of which is spent in areas where there are no offshore wind farms (e.g. over the continental shelf of west Africa).

5.17 It is much more difficult to allocate mortality of immature gannets among colonies, since we know little about the movements of immature gannets. The youngest age classes may not associate with their natal colony but may disperse widely at sea (Nelson 1978, 2002). However, as the birds get older, they tend to spend increasing amounts of time in summer at a colony, and this is usually their natal colony, although some individuals move to recruit elsewhere (Nelson 1978, 2002). The worst case scenario would be to allocate mortality to colonies in the same proportion as done above for adults. However, this would certainly overestimate immature mortality from these specific colonies, since all the evidence points towards immature gannets ranging more widely than adults and so mortality will affect immature birds from a wider range of colonies than it will for adults.

Density-independent model predictions

5.18 The density-independent model predicted that, on average, gannet numbers will continue to increase with additional gannet mortality due to collisions with offshore wind farms, up to a threshold of approximately 10,000 additional birds killed per year. Above this value the average population growth rate fell below 1. Ninety five percent of simulations had positive population growth (i.e. >1) up to a threshold additional mortality of approximately 3,500 birds per year (Figure 13, Appendix I).

Density-dependent predictions

5.19 The predictions from the density-dependent model (Figure 14, Appendix I) look very similar to those from the density-independent model, and comparison can be made more easily by plotting the two sets of predictions in the same graph (Figure 15, Appendix I).). The values used to generate this plot are provided in Table 5.

5.20 The two models differ only slightly, with the density-dependent model predicting lower growth rates for low levels of additional mortality, but lower rates of population decline for the highest levels of additional mortality. This is to be expected, since the density-dependent model is buffered against the effects of additional mortality through increases in adult survival. The two models predict closely similar outcomes where collision mortality is in the range 8,000 to 12,000 gannets per year.

5.21 The differences in probability of population decline for the density independent model and the model with density dependent adult survival are shown in Figure 16, Appendix I. Differences between these two models are trivial when small declines are considered (red and green lines in Figure 16, Appendix I) but become more substantial where large declines are predicted (and additional mortality rates are very high). As would be expected from a density dependent model, very high rates of additional mortality (over 12,000 birds per year) are somewhat buffered in the density dependent model so cause a lower probability of population declines exceeding 10% than seen in the density independent model.

5.22 Given the very small differences between the two models' predictions over a range of conditions likely to pertain in reality (fewer than 10,000 gannet deaths per year at offshore wind farms in UK waters), the choice of which model to use becomes comparatively unimportant.

5.23 Although there are theoretical attractions to incorporating density dependence in population models, since the density dependent model is not statistically supported by empirical data (Wanless *et al.* 2006 showed that the suggested trend in adult survival rate was not statistically significant), and is only weakly supported by behavioural data (gannets from larger colonies spend longer on foraging trips than gannets from smaller colonies, but foraging trip duration also varies as much or more among years according to food abundance, and apparently has no consequence for breeding success), we consider that the density independent model should be used. The basis for this is that, in the absence of compelling support for density dependence, the simpler modelling approach should be favoured. This model is also slightly more precautionary, due to the absence of the buffering effect of increased adult survival at lower population sizes.

Model results - Changes to probability of population decline – SPA outputs

5.24 Since density independent and density dependent variants of the model produced closely similar outputs in the analysis of the UK and Ireland population, we have carried out all modelling of individual colony data using the density independent variant of the model. In addition to convenience and the benefit of keeping modelling relatively simple, there are several other powerful reasons for doing this.

5.25 Firstly, we scaled density dependent adult survival against empirical data on the growth rates of the UK and Ireland population, and not on the basis of growth rates of individual colonies, which have not been uniform. Secondly, density dependence is unlikely to act in the same way at all colonies. Some colonies are close to others, and so the density of foraging birds at sea near small colonies will potentially be influenced as much, or more, by numbers of gannets breeding at nearby much larger colonies. It would be nonsense, for example, to consider a density dependent model for gannets at Scar Rocks (2,394 AONs) without reference to the order of magnitude larger population at nearby Ailsa Craig (27,130 AONs). It would be equally unreasonable to assume that prey abundance is the same in all waters around the British Isles, and so any relationship between gannet density and demography is unlikely to be consistent among areas.

5.26 Densities of gannets foraging at sea will also be affected by colony position in relation to coastline. For example, gannets foraging from St Kilda can use 360^o of open sea around their colony. Gannets foraging from Bempton can only use 180^o of open sea since their colony lies on a mainland coastline.

5.27 Finally, while demographic data could be dis-aggregated specifically for the Bass Rock, this would not be true for any other gannet colony, since survival rates cannot be estimated reliably for any individual colony other than the Bass Rock. Even for the Bass Rock, it is not self-evident that the estimates of survival rate and breeding success derived from data pertaining only to the Bass Rock are more appropriate than data based on all gannet colonies. The latter may not necessarily be applicable to the individual circumstances at the Bass Rock if parameters vary among colonies, but data for all colonies represent a much larger data set with correspondingly smaller standard errors. So it is quite plausible that the data for all colonies may be more suitable for application to the Bass

Rock model than estimates derived specifically from Bass Rock data with correspondingly much larger standard errors.

5.28 For all these reasons, we have limited the colony-specific modelling to use of density independent models, with the same parameter values as used in the British Isles density-independent model, apart from the starting size of the population (which is based on individual colony census data from the national census in 2004).

5.29 The colony-specific models follow the structure of the density independent model described above, used to investigate impacts on the UK and Ireland gannet population. However, when applied to an individual colony, the model makes some further assumptions. Most importantly, it assumes that the focal colony is a closed population, or that, if open to immigration and emigration, there is no net flow of birds to/from other colonies. We know that adult gannets normally return to nest in exactly the same nest site each year, and it is thought that young birds mostly recruit into their natal colony when they start to breed (Nelson 1978, 2002). However, there are some anecdotal observations of individuals born at one colony recruiting to breed in another. It is uncertain how extensive such emigration may be, though it is likely to be a small influence except at the smallest recently-founded colonies. It is unknown whether emigration rates may respond to factors such as the presence of offshore wind farms within flightlines or foraging areas used by gannets. It is assumed that birds from colonies such as St Kilda are too far away from the offshore wind farms considered in this report to be affected during the breeding season. Thus they migrate and overwinter and their distributions may then bring them in to contact; apportionment is according to what is known of the migration routes and wintering areas, so for adults based on the logger data and for immatures based on the assumption that immatures from all populations winter in similar areas.

Bass Rock

5.30 For the Bass Rock colony (48,065 AONs in 2004) the model indicates that additional mortality in excess of 2000 birds per year would be expected to lead to decline in colony size (Figure 17, Appendix I). Probability of decline by 5, 10, 15 or 20% at any point within the 25-year simulation (Figure 18, Appendix I) is low for additional mortality below 500 birds per year, but increases rapidly with additional mortality in excess of 1000 birds per year.

Bempton

5.31 For the Bempton colony (3,940 AONs in 2004) the model indicates that additional mortality in excess of 150 birds per year would be expected to lead to decline in colony size (Figure 19, Appendix I). Probability of decline by 5, 10, 15 or 20% at any point within the 25-year simulation (Figure 20, Appendix I) is low for additional mortality below 50 birds per year, but increases rapidly with additional mortality in excess of 100 birds per year.

Ailsa Craig and Scar Rocks

5.32 Because Ailsa Craig and Scar Rocks are so close together, we have pooled data for these two colonies and treated them as a single population. For the Ailsa Craig and Scar Rock population (27,130 and 2,394 AONs in 2004 respectively) the model indicates that additional mortality in excess of 1100 birds per year would be expected to lead to decline in colony size (Figure 21, Appendix I). Probability of decline by 5, 10, 15 or 20% at any point within the 25-year simulation (Figure 22, Appendix I) is low for additional mortality below 500 birds per year, but increases rapidly with additional mortality in excess of this number of birds per year.

Grassholm and Great Saltee

5.33 Because Grassholm and Great Saltee are so close together, we have pooled data for these two colonies and treated them as a single population. For the Grassholm and Great Saltee population (32,094 and 2,446 AONs in 2004 respectively) the model indicates that additional mortality in excess of 1,400 birds per year would be expected to lead to decline in colony size (Figure 23, Appendix I). Probability of decline by 5, 10, 15 or 20% at any point within the 25-year simulation (Figure 24,

Appendix I) is low for additional mortality below 500 birds per year, but increases rapidly with additional mortality in excess of 1,000 birds per year.

St Kilda and Flannans

5.34 Because St Kilda and the Flannans are so close together, we have pooled data for these two colonies and treated them as a single population. For St Kilda and the Flannans population (59,622 and 2,760 AONs in 2004 respectively) the model indicates that additional mortality in excess of 2,200 birds per year would be expected to lead to decline in colony size (Figure 25, Appendix I). Probability of decline by 5, 10, 15 or 20% at any point within the 25-year simulation (Figure 26, Appendix I) is low for additional mortality below 1000 birds per year, but increases rapidly with additional mortality in excess of 1,500 birds per year.

6. DISCUSSION

6.1 Using published data for the UK gannet population, a stochastic simulation model was developed. This model produces average predictions which closely match the recent observed trend in the UK breeding colony estimates. There is no evidence from the colony counts, survival rates or breeding success for density dependent regulation, however there are indications that adults may be modifying their behaviour at larger colonies (Hamer *et al.* 2000, Lewis *et al.* 2001). This could potentially lead to reductions in adult survival. Therefore two forms of the population model were developed; density independent and density dependent.

6.2 The predictions generated by the two models were very similar over the range of projected populations and mortality estimates investigated. The relationship between adult survival and population size was estimated empirically and tested against the known trend in population counts. This model performed poorly and the slope of the relationship required considerable modification before a satisfactory fit between model output and known trend was obtained. Thus, since the density independent model was based on fewer assumptions, and the fact that the two forms of model produced similar outputs, it is considered as more suitable for investigating collision risk impacts. While the density independent model can permit very large population increases, it is also conservative in the sense that there are no mechanisms in the model to buffer declines. Thus, simulated populations which decline do not experience elevated survival or reproduction. However, a density-dependent version of the model provided very similar results over the range of population sizes likely to be present in the immediate future.

6.3 No attempt has been made here to adapt the population model to smaller ‘sub-populations’, since there is a lack of colony-specific demographic data apart from breeding success. Breeding success shows very little variation among colonies. Of the data sets currently available this will be most straightforward for the Bass Rock colony, however the general similarity of productivity rates from the sites studied suggests that the rates used in the baseline model will probably be suitable for most ‘sub-populations’, although this has still to be investigated. However, since the model represents a closed population, using it to simulate smaller sub-populations is therefore based on the assumption of no inter-colony movement. Estimates of exchange between breeding colonies are considered to be low, but this assumption will need careful examination before predictions for sub-populations are generated.

6.4 Demographic data provide detailed information on breeding success of gannets, but there is little information on survival rates and those are based only on ring recovery data and not on individually marked breeding adults. The reliability of population models for gannets could be improved considerably by establishing long-term monitoring of colour marked individuals at a range of colonies. Sensitivity analysis shows clearly that the models we have used are most strongly influenced by errors in adult survival estimates, as would be expected for a long-lived bird with deferred maturity and low fecundity; a reduction in mean adult survival from 0.92 to 0.903 was sufficient to reduce the mean population growth rate to 1.

6.5 Although estimates of breeding numbers at gannet colonies in Britain and Ireland are among the best data on any seabird population in the world, there is a lack of information on the rates of immigration and emigration. Many gannets recruit to breed in their natal colony, and once established they seem to return to the same part of the same colony every year to breed, However, colony data indicate that growth rates of small colonies must be 'subsidised' by immigration from elsewhere. So it is also likely that additional mortality at particular colonies may be somewhat buffered by net immigration of birds to take up any vacated nest sites or pairing opportunities created by collision mortality of birds at offshore wind farms. Better understanding of colony-specific dynamics of gannets will require an improved knowledge of the roles of emigration and immigration.

6.6 Estimates of the numbers of gannets that could be killed at offshore wind farms without a high risk of population decline are relatively large: about 10,000 gannets per year from the British and Irish population, about 2,000 gannets per year from the Bass Rock colony (if it was a closed population). The key words in the previous sentence are "high risk". While true that 10,000 gannets per year are predicted to be "harvestable" before the average population growth rate will fall to 1, at this harvest level 50% of the simulations will have generated growth rates less than 1 i.e. at a harvest of 10,000 there is a 50% chance that a population which has been growing steadily for several decades will no longer be able to do so. A much lower level of risk is reached at the harvest level at which one can be almost certain (95% confident) that the growth rate will not fall below 1 i.e. the point at which the lower 95% CI crosses the growth =1 point – in this case c. 4000 birds which is almost exactly the number predicted to be accounted for by the Round 1 & 2 sites (Fig 13, Appendix I, Tables 6 & 7, Appendix II). So interpretation of risk depends crucially on what level of risk is acceptable to regulators. The thresholds provided in the report need to be caveated to explain that they are the values at which population growth is expected to become zero, even if colonies were exhibiting high growth rates prior to harvesting. This threshold is not one presently supported by SNCBs though further clarification is required on what is an appropriate threshold for conservation aims given that no animal population can sustain growth indefinitely. Historical studies of seabird populations suggest that populations showing highest rates of increase often cease their growth and sometimes decline to more sustainable levels in line with resources available (Mitchell *et al.* 2004). Most global conservation objectives involve sustainability, and populations tend to be sustainable if they remain around a level that is in balance with the ecosystem in which they live albeit with some natural fluctuations. So there are sound theoretical reasons for suggesting that maintenance of numbers is a more appropriate threshold against which to assess impacts than sustained growth if the natural carrying capacity is not known. In the context of the target numbers for gannet populations, this requires some further discussion, particularly between the SNCBs and regulators.

6.7 Estimates of numbers of gannets that may be killed each year at Round 1 and 2 offshore wind farms depend on data on gannet densities at these sites and application of the Band (2011) model to estimate likely casualties (Fox *et al.* 2006). Density estimates each month for each wind farm are one of the constraints on the accuracy of such calculations, and there are difficulties in arriving at satisfactory density estimates because surveys by boat and by air produce discrepant figures, and because the relatively small amount of survey effort at any particular site in a particular month is subject to stochastic variation with highly aggregated data such as gannet distribution in space and time (Pettex *et al.* 2010). Furthermore, the Band model estimates are currently considered to use a 98% default avoidance rate as the standard for bird species that have not been studied in sufficient detail to provide an accurate species-specific value (Desholm 2009, Desholm and Kahlert 2005, 2006, Desholm *et al.* 2006). Given that many birds that have been studied in detail show a much higher avoidance rate (for example geese achieve at least 99% avoidance) it is likely that the use of a default 98% avoidance rate for gannets is unduly conservative (so will report higher death rates than are likely to occur in the real world; Chamberlain *et al.* 2006, Baisner *et al.* 2010). Combining the available gannet density data and applying the Band model estimated 2,826 adults killed per year at Round 1 and 2 offshore wind farms of which 2,603 would be from the British and Irish population.

A further 1,242 immatures (1,056 from the British and Irish population) would be killed. These totals are well below the threshold levels that would be expected to stop the British and Irish gannet population from increasing. It should be noted, however, that these calculations take no account of Round 3 and Scottish Territorial Waters (STW) developments.

For future modelling of gannet populations in relation to additional mortality caused by offshore wind farms, the greatest improvements to model reliability would be:

- Measurement of gannet avoidance rate to replace the default 98% estimate;
- measurement of adult gannet annual survival rates at key colonies;
- information on net migration rates between gannet colonies and populations;
- improved estimation of gannet densities at offshore wind farms by investigation of the intercalibration between estimation from ship-based surveys and aerial surveys;
- collection of further data on gannet densities at offshore sites through the year; and
- improved understanding of the foraging ranges and migrations of gannets through deployment of data loggers (Burger and Schaffer 2008).

6.8 The results of this modelling work should assist regulators and developers with regard to the assessment of cumulative impacts of forthcoming offshore wind farm developments on gannets.

6.9 Our models indicate that the gannet population of the British Isles is robust to collision mortality because the population is large and increasing. Some 10,000 gannets per year (across all age classes in proportion to their abundance) could be taken from the population before there would be a high likelihood of a decline in overall breeding numbers, providing other aspects of gannet ecology do not change drastically (for example changes in fisheries may alter conditions for gannets; Votier *et al.* 2004, 2010).

6.10 Estimated mortality at Round 1 and 2 offshore wind farms accounts for around a third of this amount if a 98% avoidance rate is taken as appropriate, and an even smaller proportion if the avoidance rate is higher than 98%, as seems likely based on data for comparable large birds such as geese. At the level of the British Isles gannet population we can be confident that collision mortality rates predicted by observed gannet densities and the assumption of a 98% avoidance rate are below the threshold at which decline of the population is likely. At the level of local populations, such as the Bass Rock, Bempton, Ailsa Craig, or Grassholm, the modelling indicates that threshold numbers that may risk decline are of course much smaller than for the whole British Isles population.

6.11 The existing offshore wind farm sites, predominantly in the southern North Sea and Irish Sea, are mostly fairly distant from major gannet colonies, but Round 3 and STW sites include areas much closer to major colonies. Overall, all offshore wind farm sites tend to be far from some gannet colonies (e.g. St Kilda, and all Shetland colonies) but relatively close to certain colonies (particularly Bempton, Bass Rock, Grassholm, St Margarets, Great Saltee and Ireland's Eye). Mortality to gannets from the distant colonies in St Kilda and Shetland may occur during spring and autumn passage, when these birds pass southwards to wintering areas, but is likely to be very much less than for gannets from Bempton, Bass Rock, Grassholm, St Margarets, Great Saltee and Ireland's Eye.

6.12 There is a clear prediction here, that if mortality of gannets at offshore wind farms is high, the colonies where an impact would be most evident would be Bempton, Bass Rock, Grassholm, St Margarets, Great Saltee and Ireland's Eye. Of these, Bempton, St Margarets and Ireland's Eye may be the colonies most at risk given their relatively small size, although small colonies may benefit from higher levels of immigration sustaining higher than expected rates of growth (Moss *et al.* 2002).

6.13 Monitoring of breeding numbers of gannets will almost certainly continue, with a new national census due in the next couple of years, which may coincide with a national census of all seabird populations. Particular attention should be placed on the future growth rates of the colonies at Bempton, Bass Rock, Grassholm, St Margarets, Great Saltee and Ireland's Eye, since increases at these colonies comparable to the increases at others would provide evidence suggesting that mortality at offshore wind farms is at a level too low to be a conservation concern in terms of gannet numbers. Conversely, since net migration rates between colonies appear to be low in the gannet (Nelson 2002), declines in breeding numbers at these colonies but not at others, would tend to implicate offshore wind farm mortality as the cause. These predictions provide a mechanism for regulators to monitor the responses of gannet numbers to offshore wind, so can provide an independent check on the cumulative impact assessment based on the modelling approach outlined in this report. It seems likely that RSPB and others will be looking at gannet numbers at Bempton (for example) to see if the rate of increase there slows as offshore wind farms are developed, and it is easy to jump to a conclusion that a reduction in the rate of increase there might be attributed to offshore wind farm collisions. Of course other factors may be important, and the challenge is to assess whether these can be separated. Since some gannet colonies lie close to offshore wind farms and others do not, there is the possibility that effects may be seen at some colonies and not at others. However, changes in fisheries discarding policy, and distributions and abundances of fish stocks, especially pelagic fish species, are likely to strongly influence food availability to gannets, and such changes may vary around the British Isles.

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APPENDIX I. Figures

Figure 1 – Locations of gannetries in the UK and Ireland. See Table 1, Appendix II for names

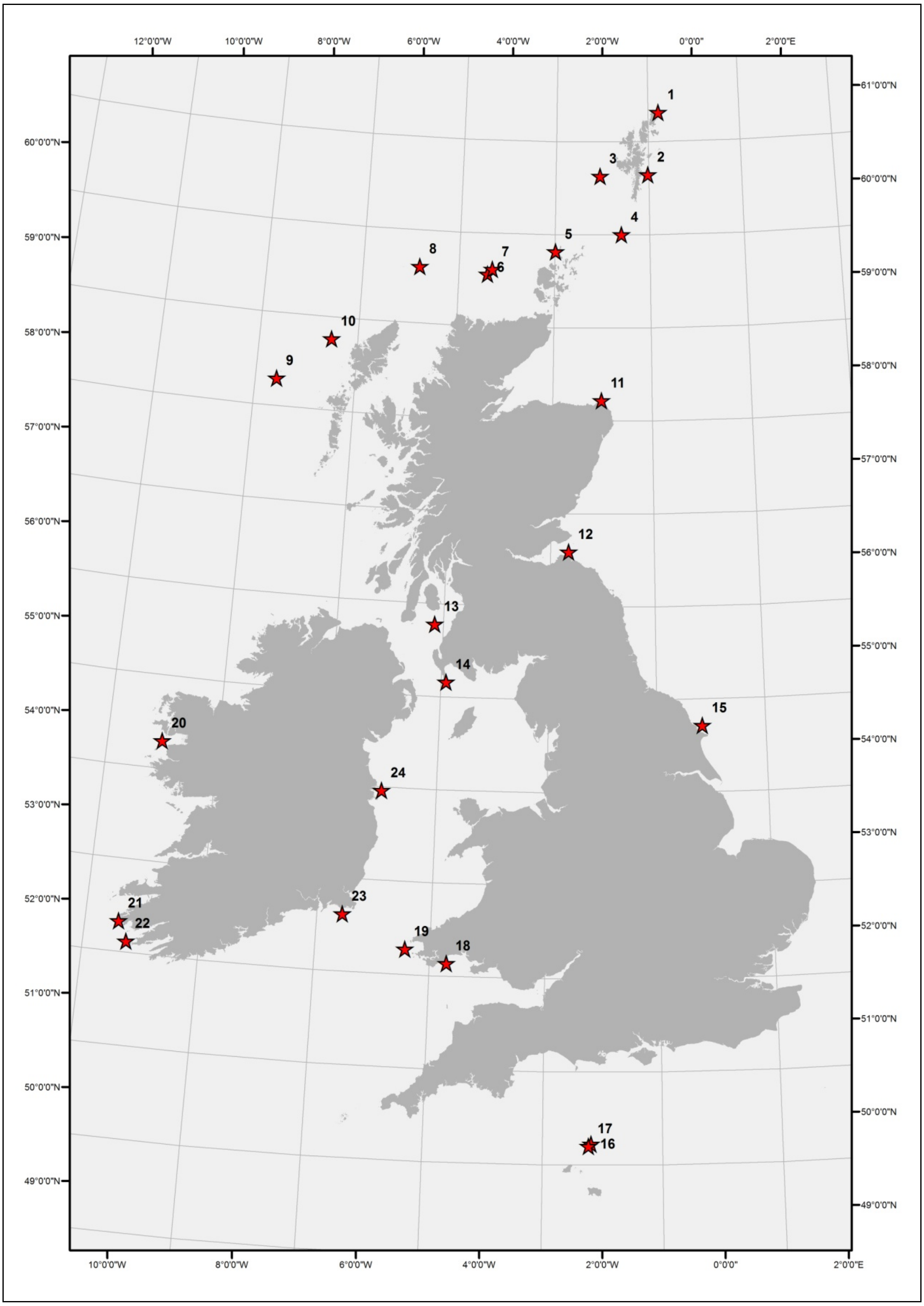


Figure 2 - Four examples from geolocator deployments of the autumn migrations and wintering areas of gannets that bred on the Bass Rock (birds 26, 27, 50 and 51). From Kubetzki *et al.* (2009). Bird 26 wintered in the North Sea and English Channel. The arrow points to the location of the Bass Rock. Bird 51 spent time in the southern North Sea before moving through the English Channel to winter in the southern part of the Bay of Biscay, and then returning to the North Sea via the Irish Sea. Bird 27 moved north of Shetland before migrating through the North Sea and English Channel, through the Straits of Gibraltar and wintering in the western Mediterranean. Bird 50 left the southern North Sea rapidly to Iberia and wintered off west Africa, then returned to the Bass Rock via the west of Ireland and waters north of Shetland

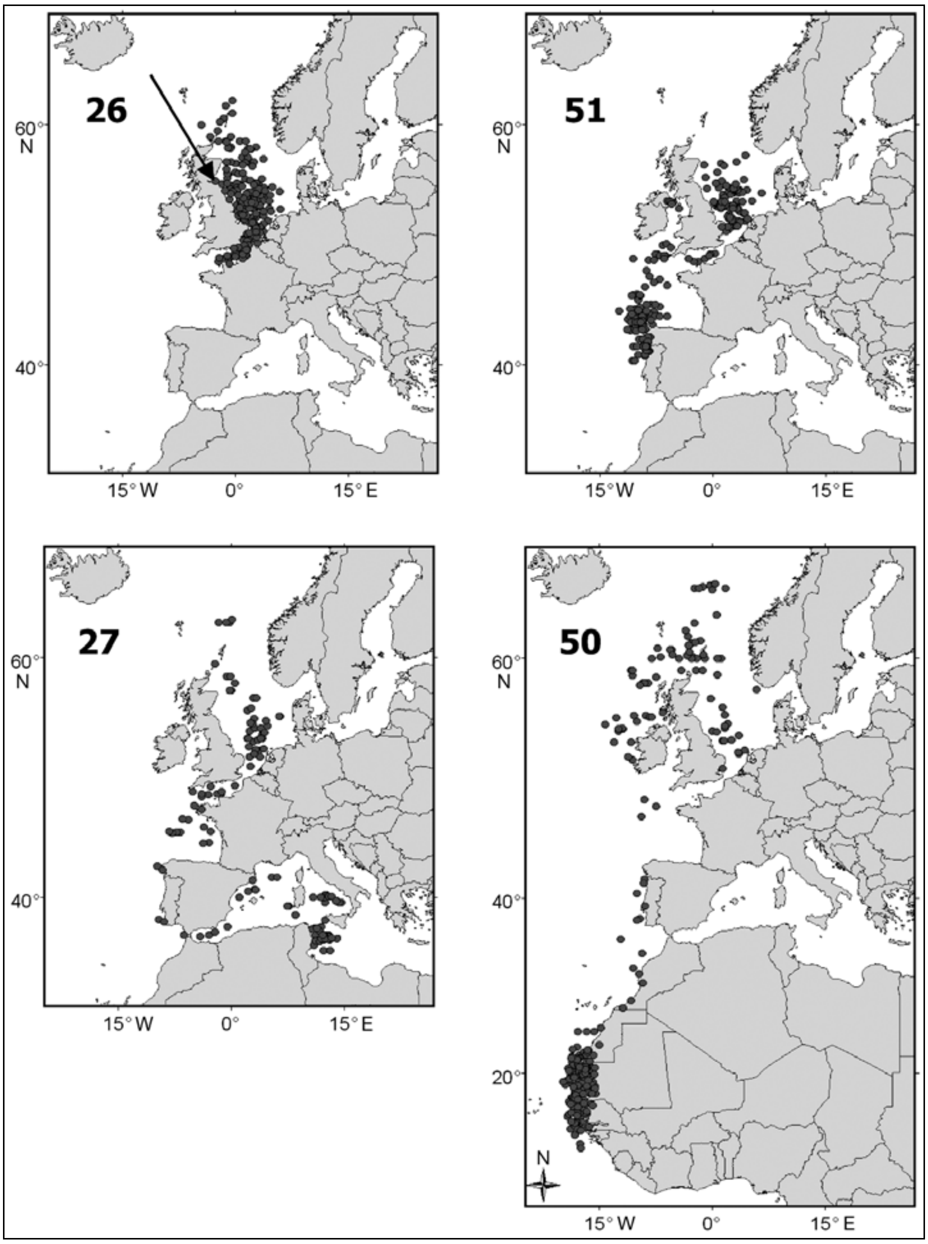


Figure 3 -. Model predictions using the density independent model. The variance on survival included process and sampling variance

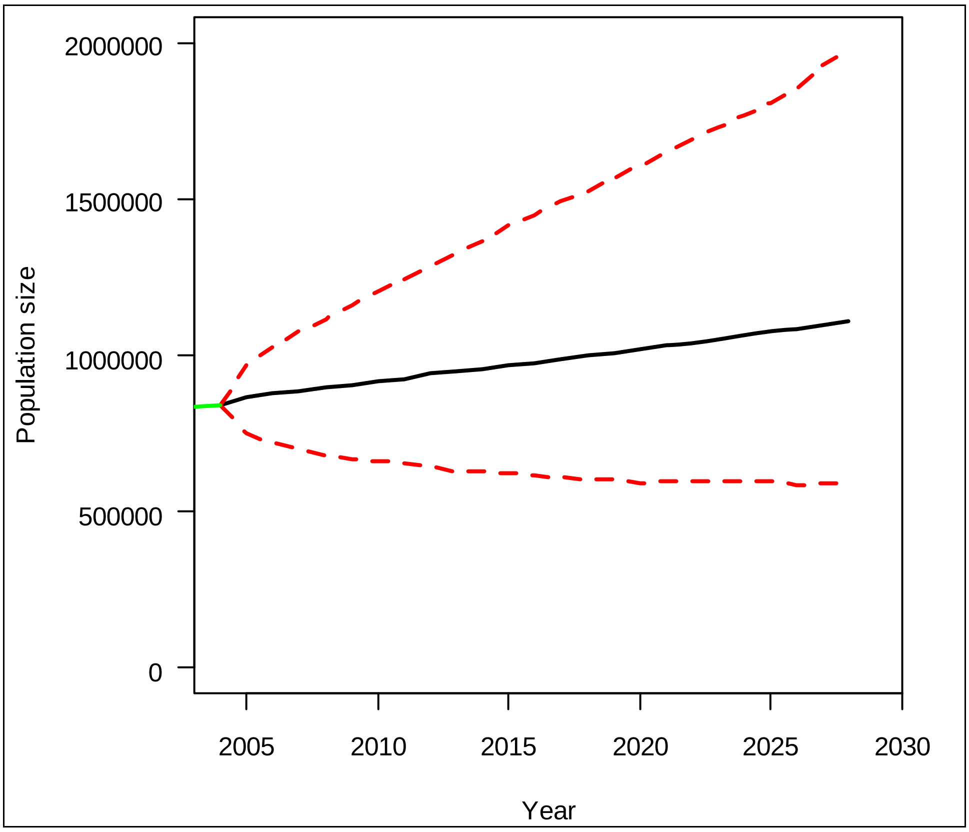


Figure 4 - Model predictions obtained using revised survival rate estimation in the density independent model. Survival rate is modelled using just process variance

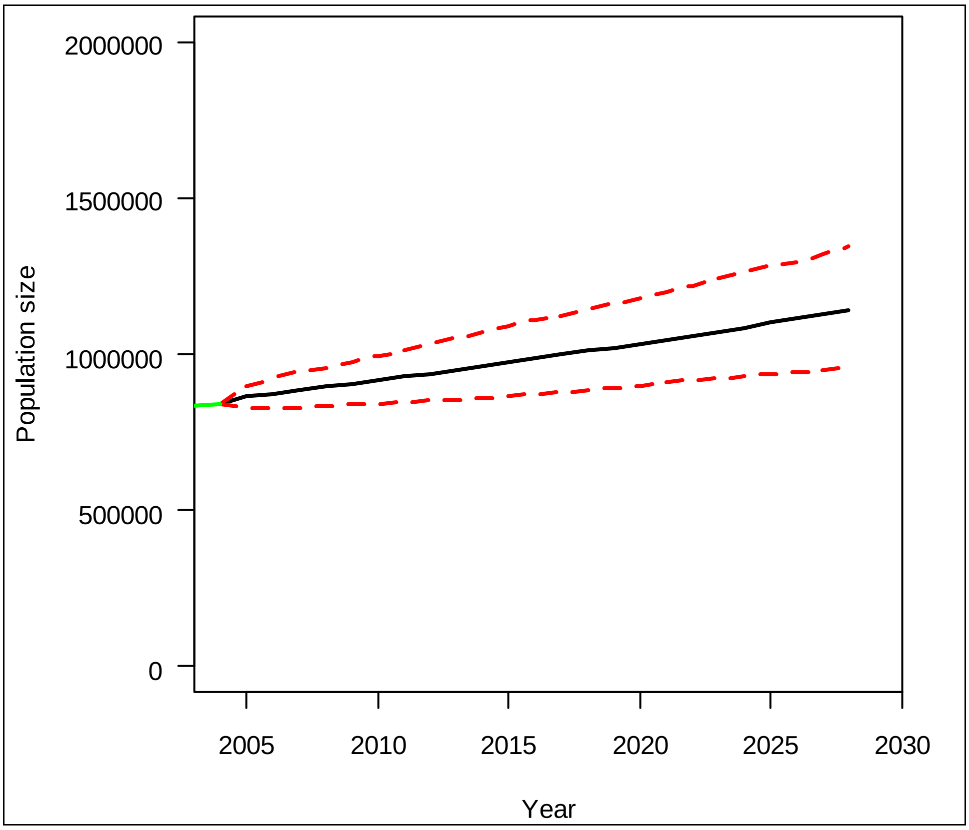


Figure 5 - Relationship between The British Isles AONs and adult survival trend derived from Wanless *et al.* (2006). Dots represent the data, the dashed red line is the best fit linear model

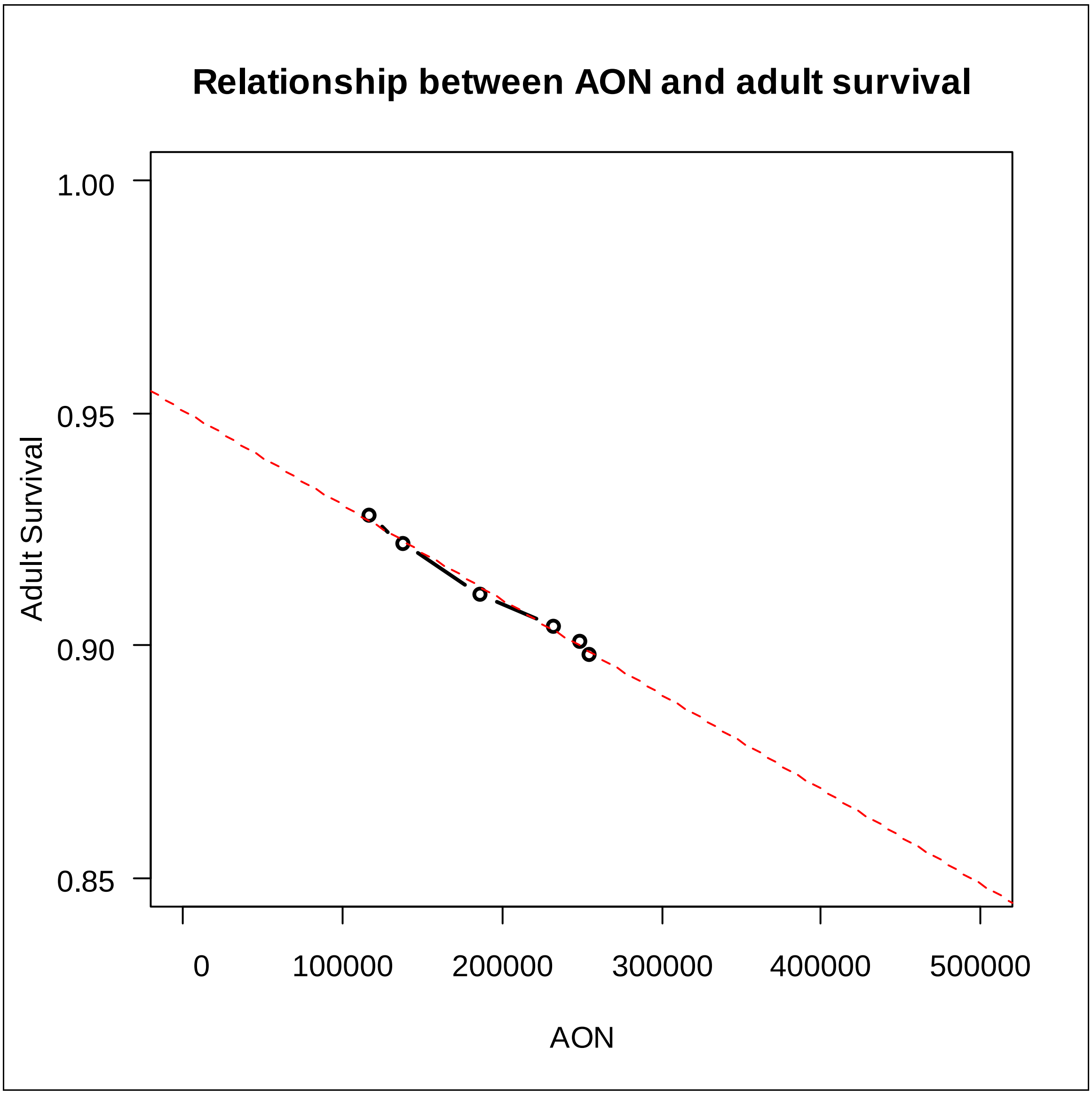


Figure 6 - Model prediction based on the density dependent relationship in equation 1. The upper panel provides the median (black line) and 95% confidence range (dashed red lines) obtained from the model plotted with the observed trend in the population (green line) based on the counts of AONs. The lower panel provides the relationship between adult survival and population size used to generate these predictions. The black line represents the linear model (equation 1)

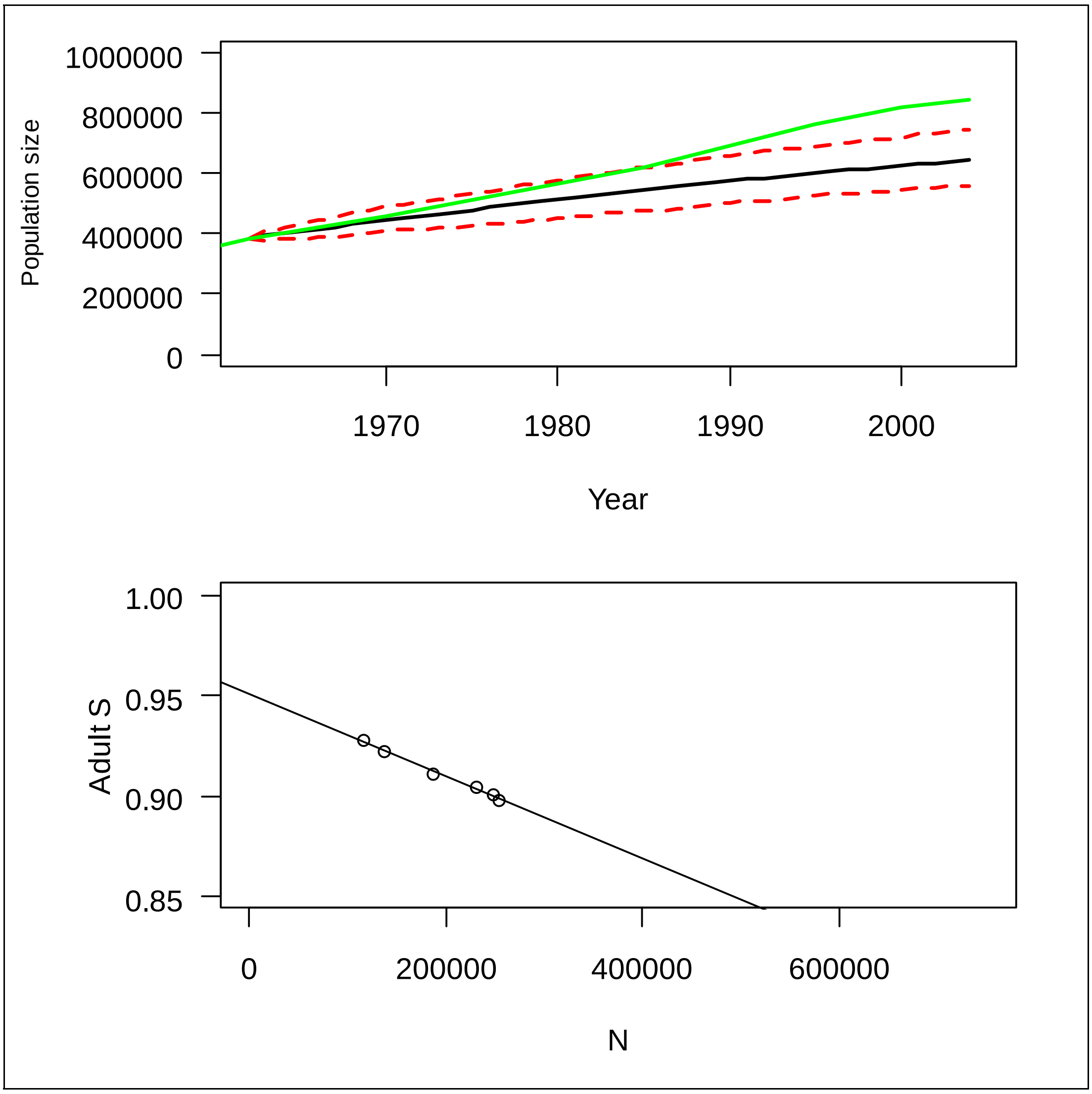


Figure 7 - Model prediction based on the best-fit adjustment to the slope of the density dependent relationship in equation 1. The upper panel provides the median (black line) and 95% confidence range (dashed red lines) obtained and the observed trend in the population (green line) based on the counts of AONs. The lower panel provides the modified relationship used to generate these predictions (the black line) and the dots are the data used to generate the original linear model (equation 1)

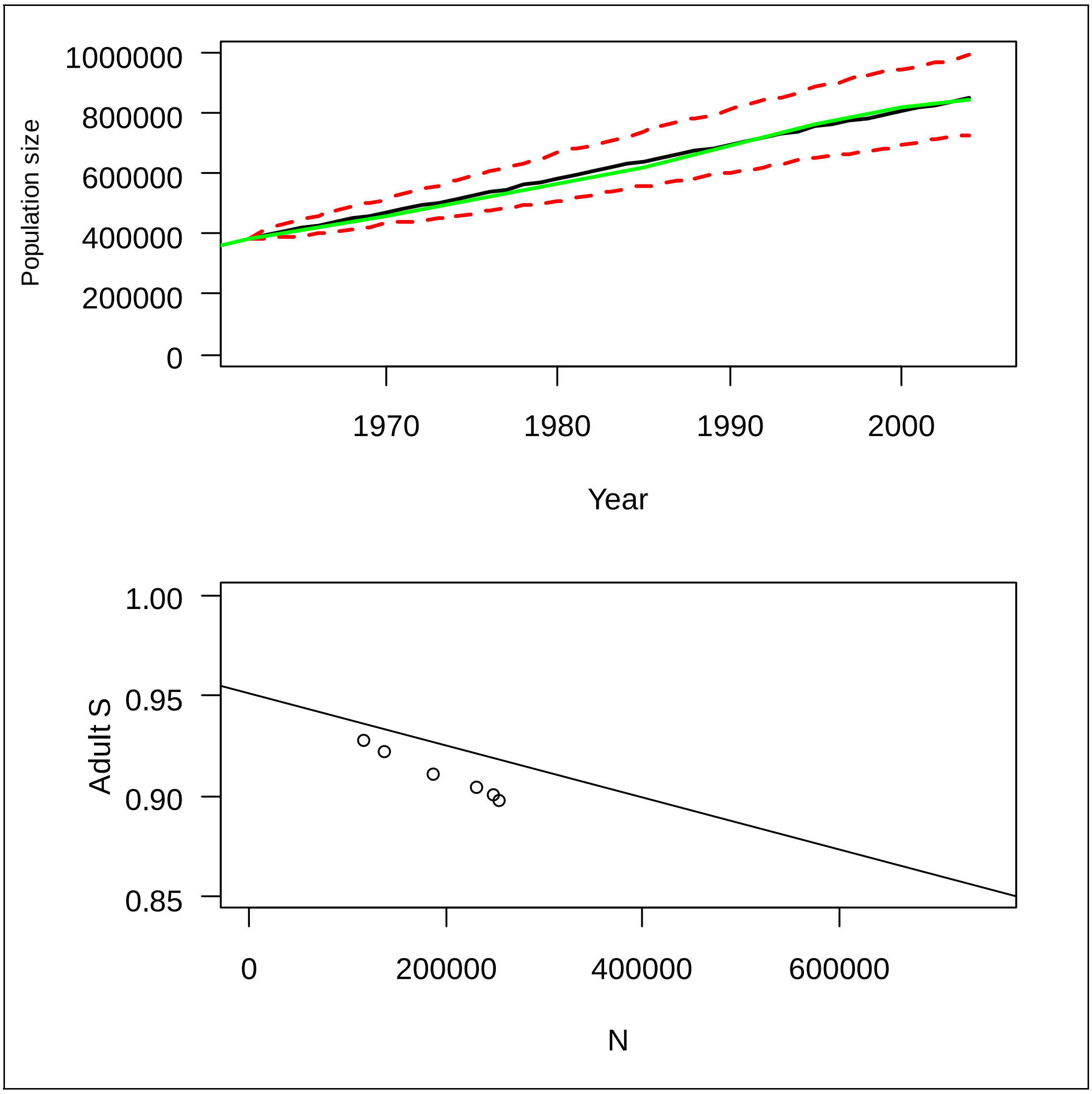


Figure 8 – Demographic sensitivity analysis – impacts of proportional rate change on population growth rate. The outputs for the first four survival rates gave nearly identical results, hence the lines for 0-1, 1-2 and 2-3 are hidden under that for 3-4

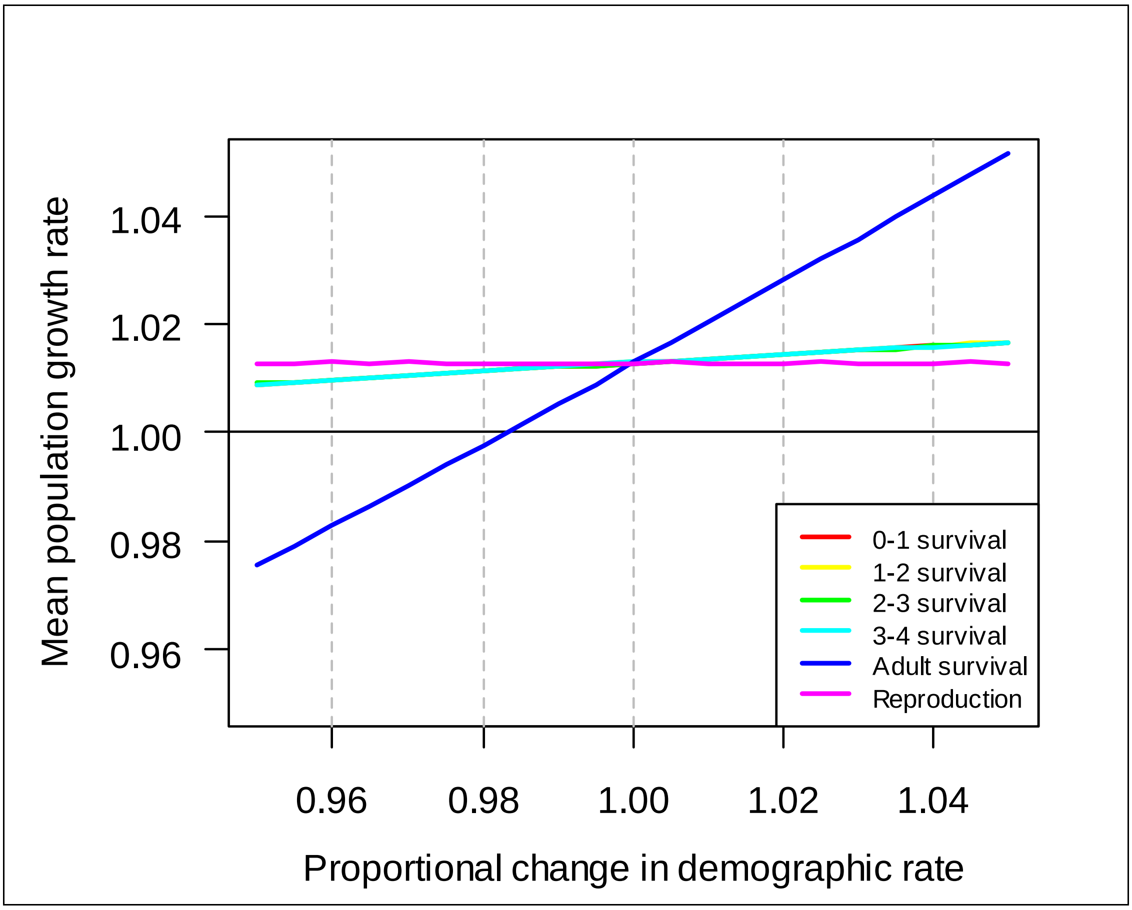


Figure 9 - Density dependent predictions based on the adjusted density dependent relationship illustrated in Figure 7

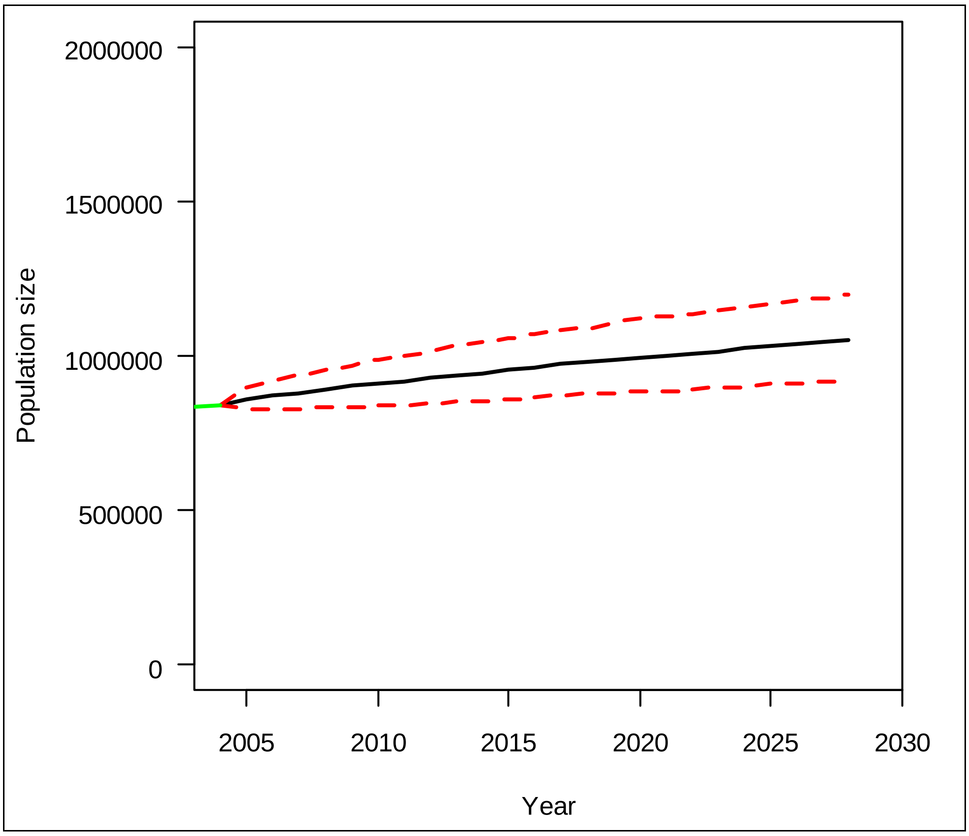


Figure 10 - Density independent predictions

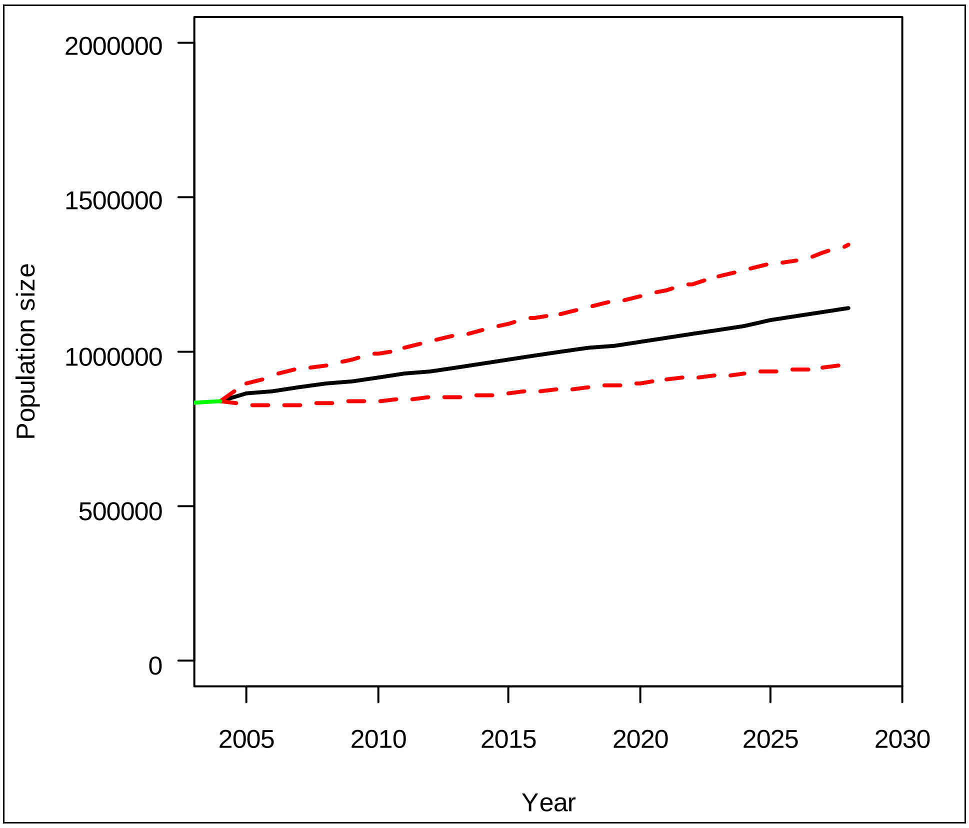


Figure 11 - Estimated numbers of adult gannet collisions at Round 1 and Round 2 offshore wind farms in the UK sector of the North Sea each month

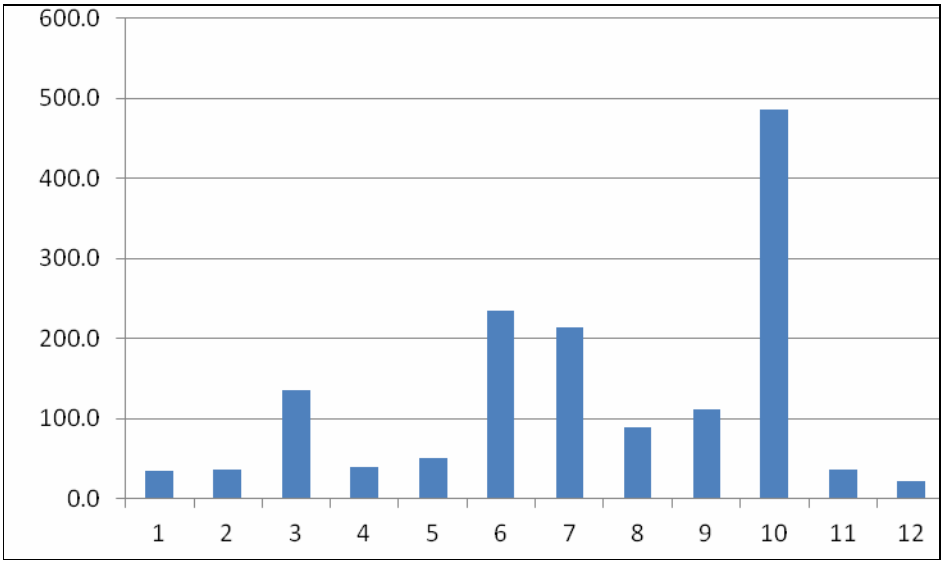


Figure 12 - Estimated numbers of adult gannet collisions at Round 1 and Round 2 offshore wind farms in the Irish Sea/Solway each month

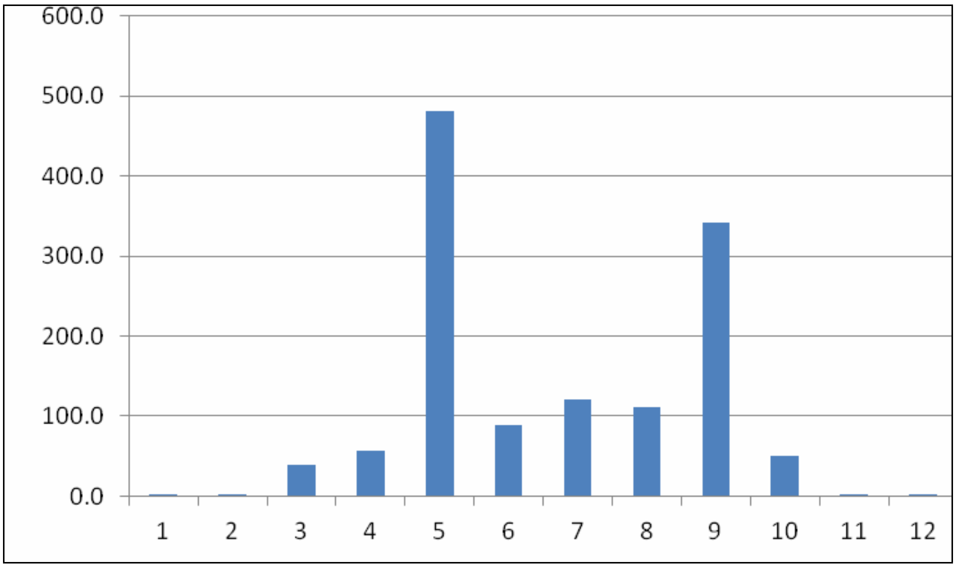


Figure 13 - Predicted population growth rate (mean and 95% confidence intervals) in relation to initial number of gannets killed per year at offshore wind farms, for the UK and Ireland gannet population based on the density-independent model (note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality)

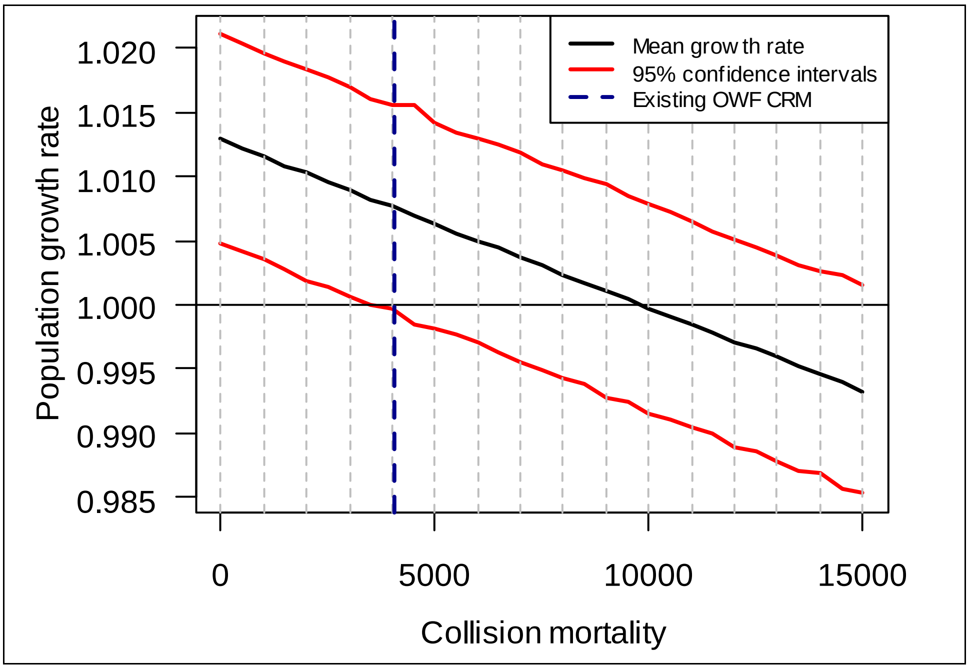


Figure 14 - Predicted population growth rate (mean and 95% confidence intervals) in relation to initial numbers of gannets killed per year at offshore wind farms, for the UK and Ireland gannet population based on the model incorporating density-dependence in adult survival rate (note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality)

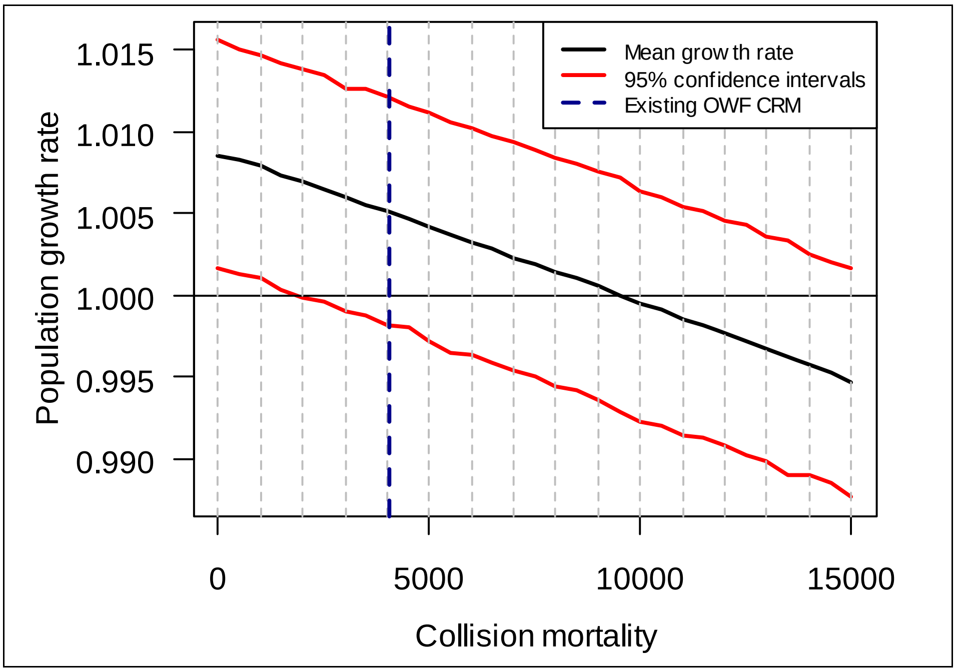


Figure 15 - Comparison of predicted population growth rates generated using the density-independent and density-dependent models. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

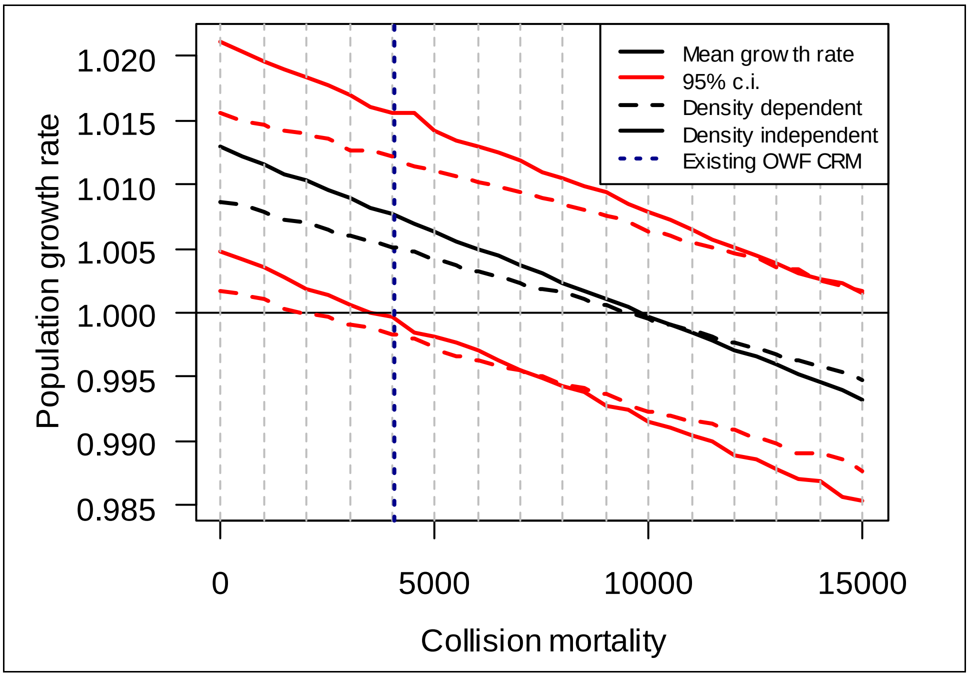


Figure 16 - Comparison of the estimated probability of population decline for the breeding population of the UK and Ireland, below a set threshold at any point in the 25 year simulation, in relation to initial numbers of gannets killed at offshore wind farms each year, using both the density-independent and density-dependent models (see Table 5 for the values used to generate this figure). Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

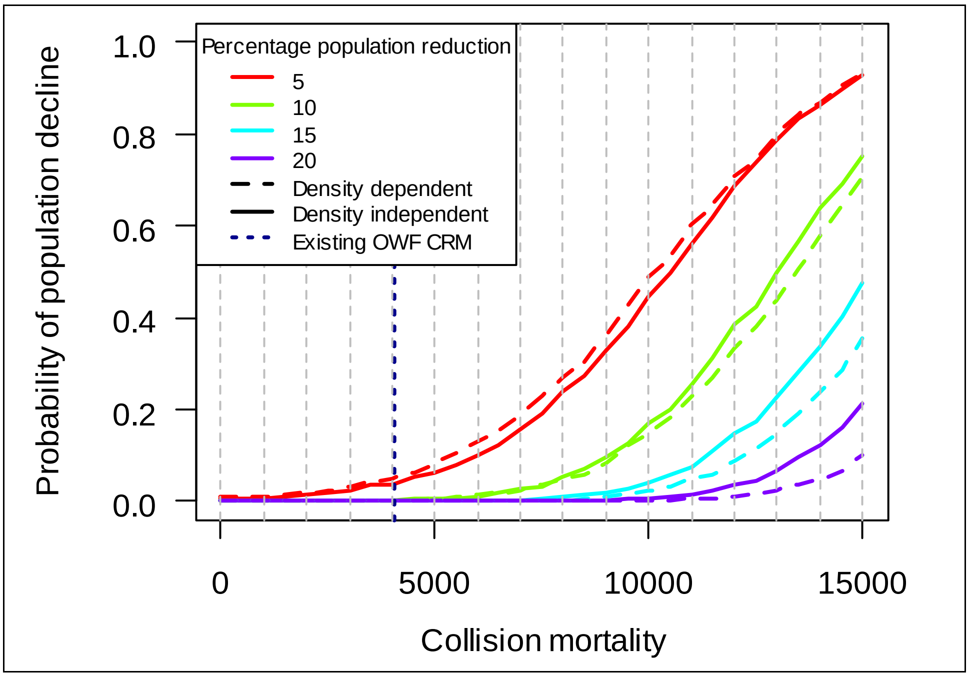


Figure 17 - Estimated growth rate of the Bass Rock gannet population with different levels of additional mortality due to offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

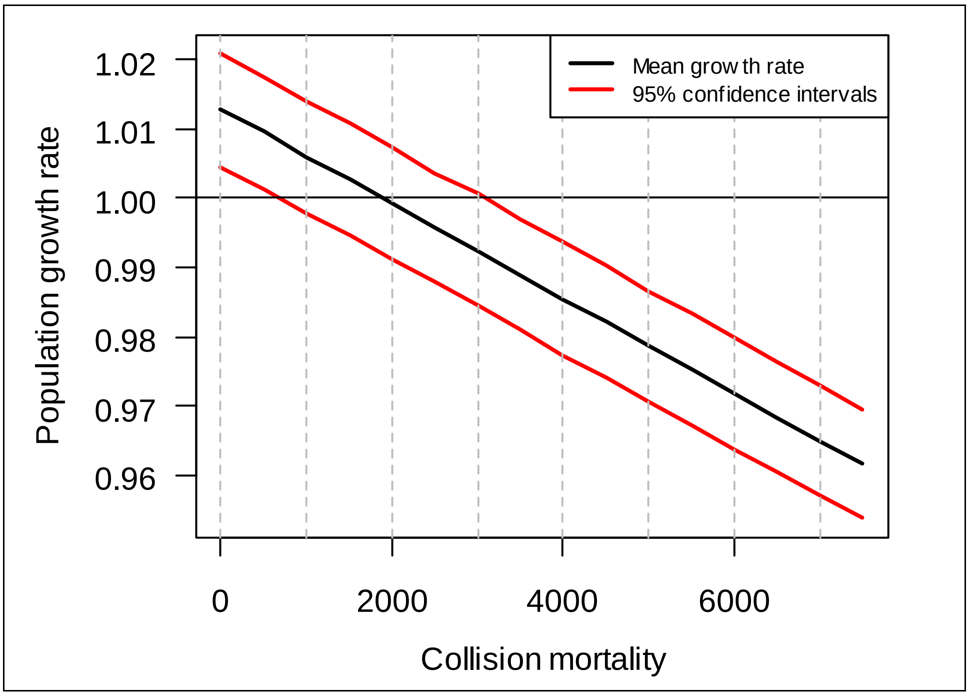


Figure 18 - Estimated probability of decline of Bass Rock gannet numbers by 5%, 10%, 15% and 20% (at any time within the 25 year simulation) in relation to numbers killed per year in offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

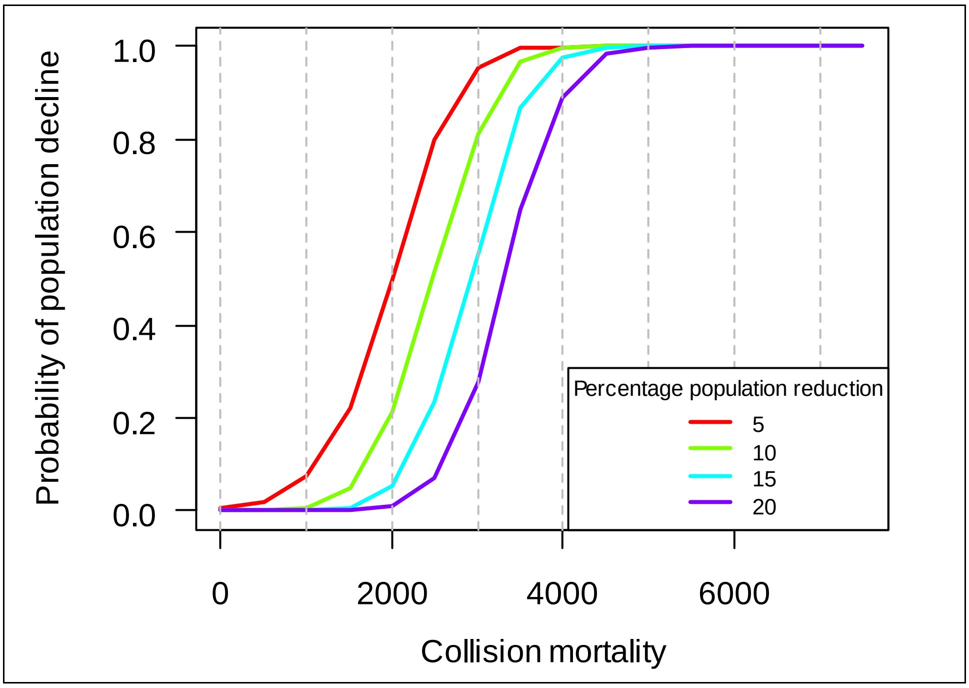


Figure 19 - Estimated growth rate of the Bempton gannet population with different levels of additional mortality due to offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

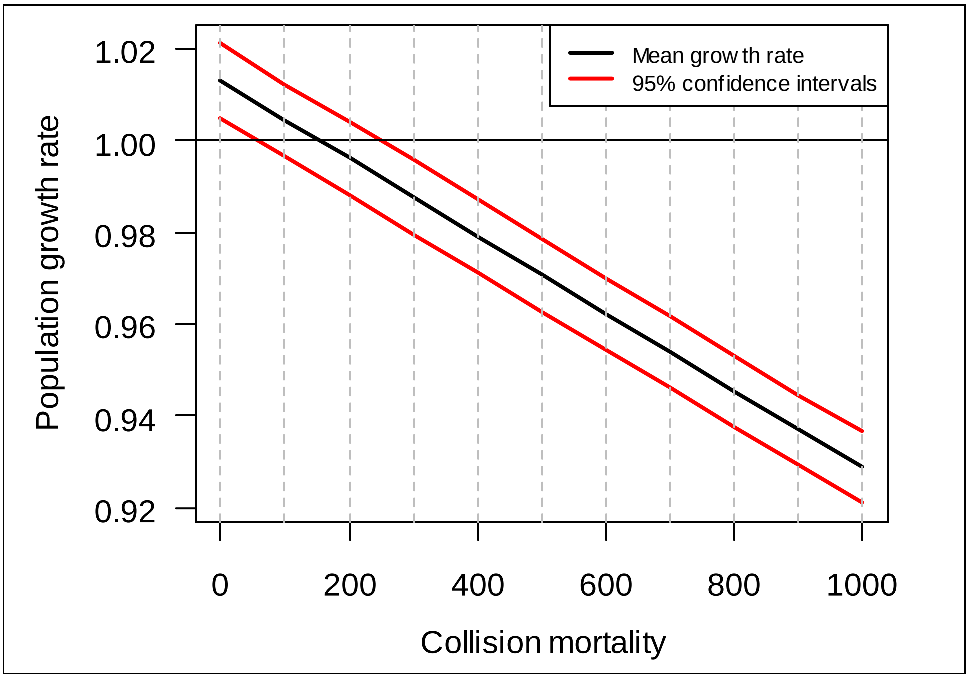


Figure 20 - Estimated probability of decline of Bempton gannet numbers by 5%, 10%, 15% and 20% (at any time within the 25 year simulation) in relation to numbers killed per year in offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

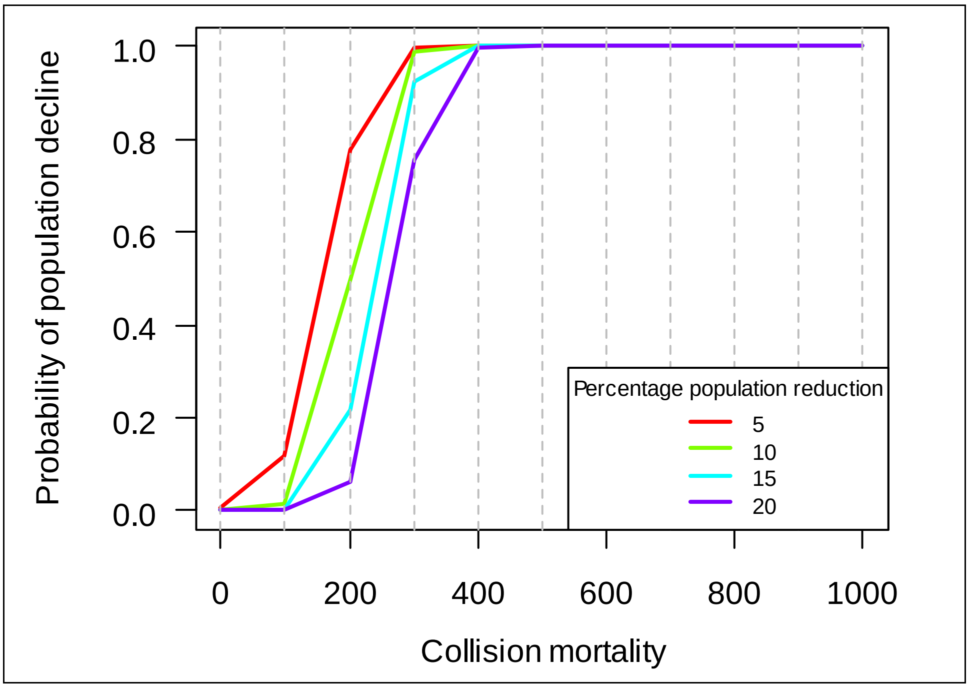


Figure 21 - Estimated growth rate of the Ailsa Craig plus Scar Rocks gannet population with different levels of additional mortality due to offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

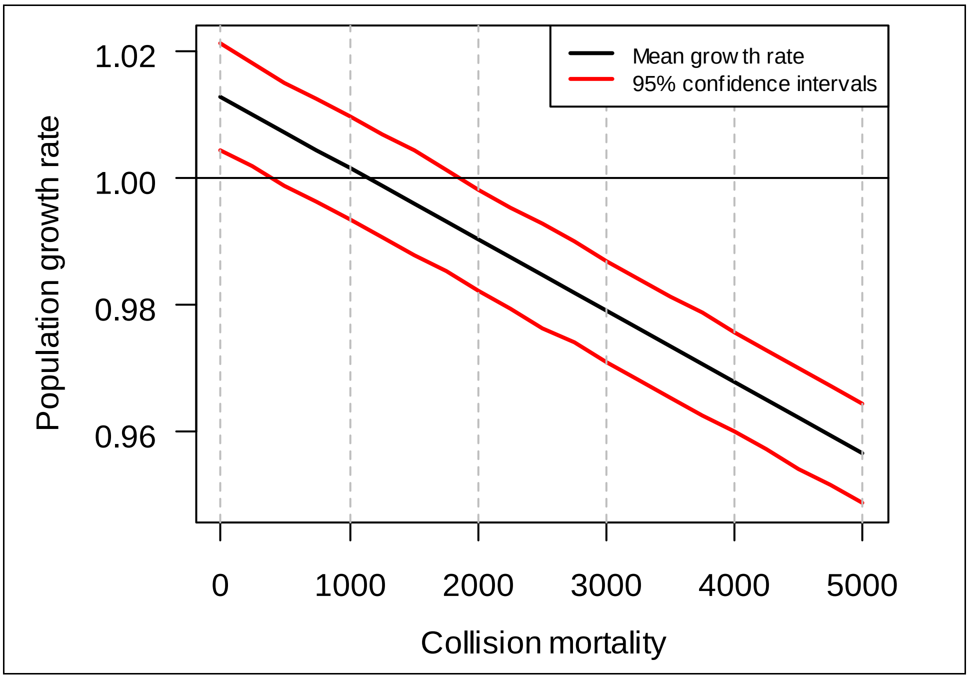


Figure 22 - Estimated probability of decline of Ailsa Craig and Scar Rocks gannet numbers by 5%, 10%, 15% and 20% (at any time within the 25 year simulation) in relation to numbers killed per year in offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

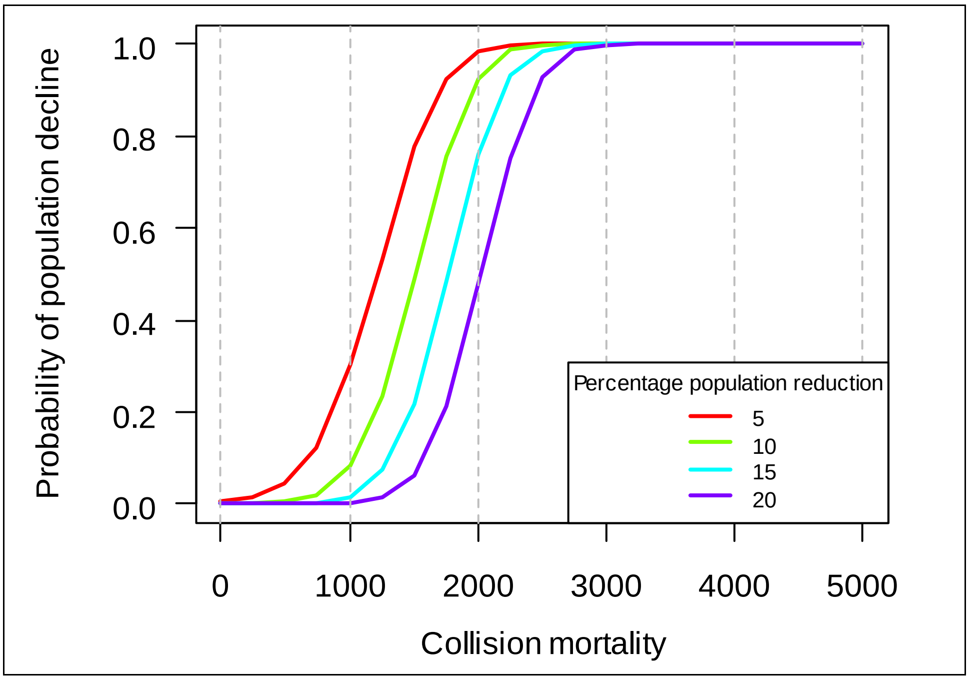


Figure 23 - Estimated growth rate of the Grassholm and Great Saltee gannet population with different levels of additional mortality due to offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

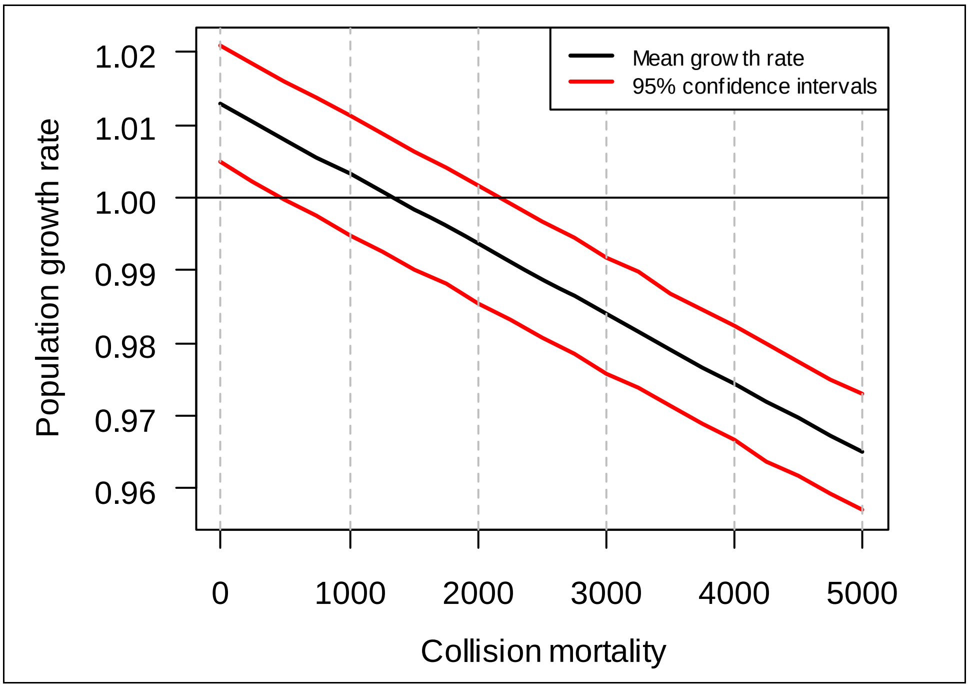


Figure 24 - Estimated probability of decline of Grassholm and Great Saltee gannet numbers by 5%, 10%, 15% and 20% (at any time within the 25 year simulation) in relation to numbers killed per year in offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

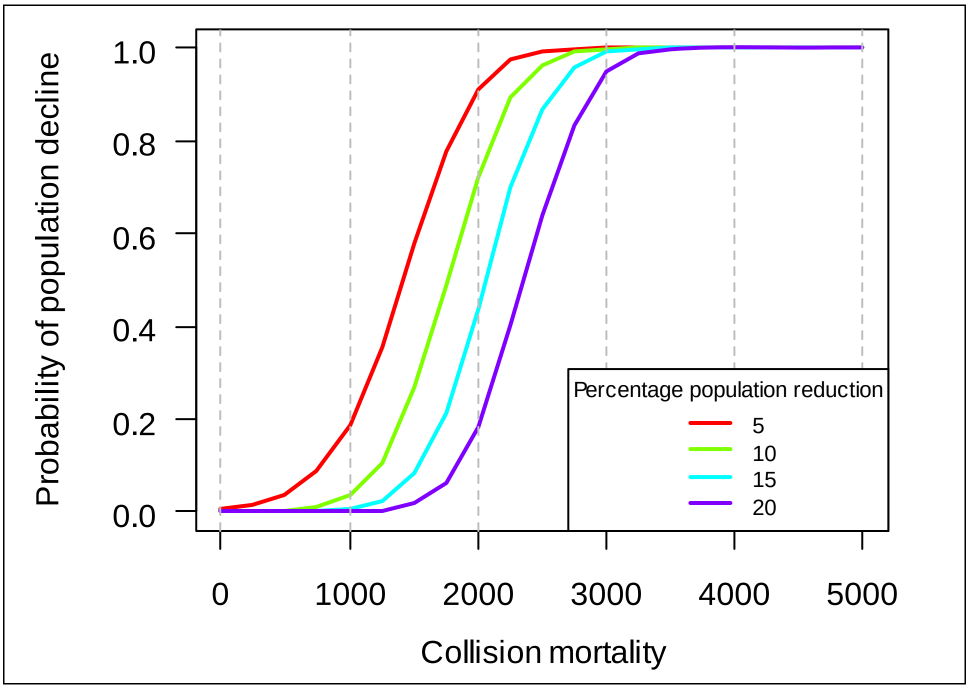


Figure 25 - Estimated growth rate of the St Kilda and Flannans gannet population with different levels of additional mortality due to offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality

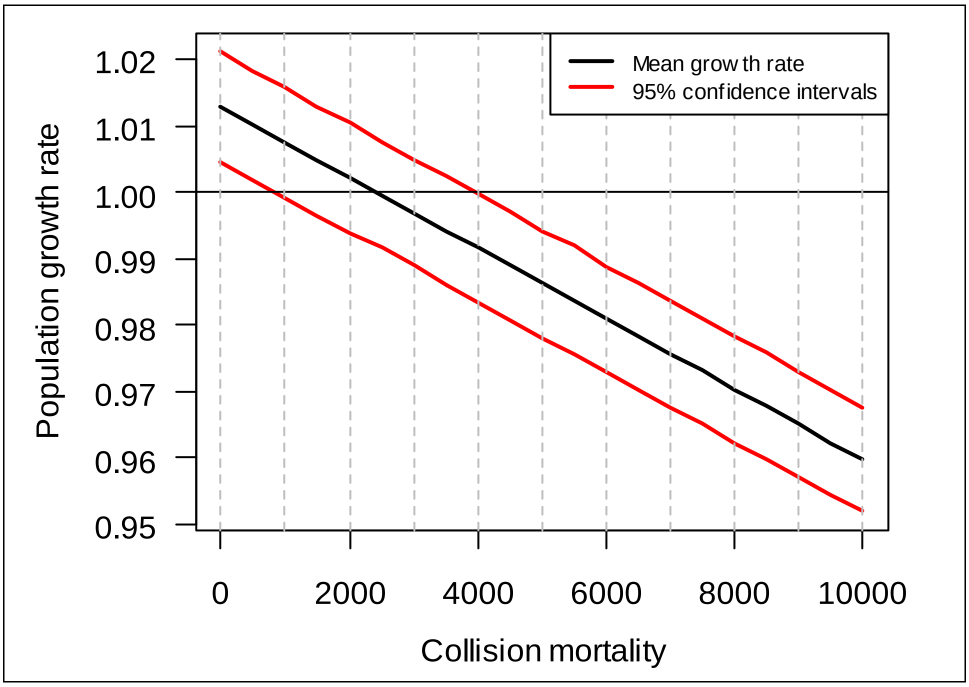
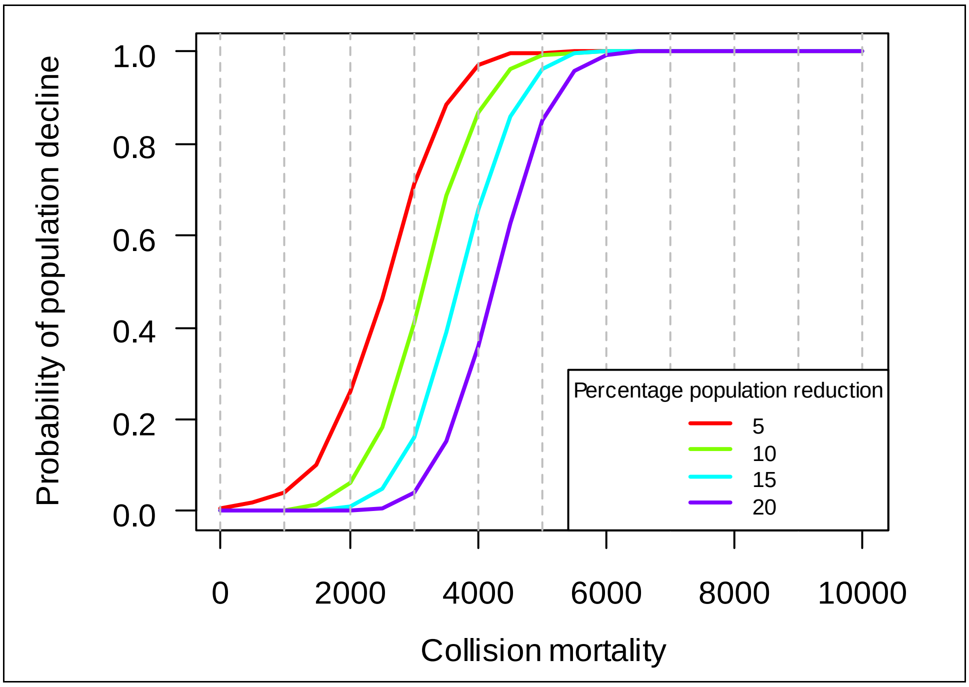


Figure 26 - Estimated probability of decline of St Kilda and Flannans gannet numbers by 5%, 10%, 15% and 20% (at any time within the 25 year simulation) in relation to numbers killed per year in offshore wind farm collisions. Note that actual mortality varies in line with the population growth rate in order to maintain proportional mortality



APPENDIX II. Tables

Table 1 – Numbers of 'apparently occupied nests' from gannet colony counts carried out between 1902 and 2004[1]. Colony numbers are given for reference to locations in Figure 1, Appendix I

	Colony	1902-05	1929	1939	1949	1959-62	1968-70	1984-85	1994-95	1998-2000	2003-04
1	Hermaness	0	1000	2611	3750	4500	5894	9904	11993	16386	15633
2	Noss	0	800	1830	3150		4300	6900	7310	8017	8652
3	Foula	0	0	0	0	0	0	124	600	723	919
4	Fair Isle	0	0	0	0	0	0	258	975	1123	1875
5	Westray	0	0	0	0	0	0	0	0	0	14
6	Sule Stack	4000		3490	2010	2900	4018	5900	4888		4618
7	Sule Skerry	0	0	0	0	0	0	0	0	0	57
8	Sula Sgeir	5000	5000	3970	6182		8964	9143	10440		9225
9	St Kilda	14750	16500	16900	17035	44526	52099	50050	60428		59622
10	Flannans	0	0	0	0	0	0	414	1438	1244	2760
11	Troup Head	0	0	0	0	0	0	2	530	1085	1547
12	Bass	3000	4147	4374	4820	6908	8077	21591	34397		48065
13	Ailsa	3250	7000	5149	4947	9390	13058	22811	32456		27130
14	Scar Rocks	0	0	2	100	167	450	770	1952	1670	2394
15	Bempton	0	0	0			18	780	1631	2552	3940
16	Ortac	0	0	0	225	925	1000	1985	2098	2500	x
17	Les Etacs	0	0	0	418	1010	2000	2536	3380	3450	x
18	St Margarets	0	0	0	0	0	0	0	0	0	1
19	Grassholm	275	4750	5875	9200	15528	16128	28600	27500	30688	32094
20	Clare Island	0	0	0	0	0	0	2	3	3	3
21	Little Skellig	17500	10000	9500	12000	17700	20000	22500	26436	28799	29683
22	Bull Rock	300	400	575	295	500	1500	1511	1815	1879	3694
23	Great Saltee	0	1	0	2	125	155	710	1250	1930	2446

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[1] Data for Scottish colonies from Forrester *et al.* 2007 except where stated otherwise
Data for 1968-70, 1984-85 1994-95, 1998-2000 from Mitchell *et al.* 2004
Data for 2003-04 from Murray *et al.* 2006 and Wanless *et al.* 2005
Data for pre-1969 from Cramp *et al.* 1974

	Colony	1902-05	1929	1939	1949	1959-62	1968-70	1984-85	1994-95	1998-2000	2003-04
24	Ireland's Eye	0	0	0	0	0	0	17	45	147	285
	France	0	0	10			2500			14900	17000
	Germany	0	0	0	0	0	0			69	190
	Faeroes			1473			1081		2340		
	Iceland			13732		20826	21000		25400	28536	
	Norway	0	0	0	10		550			3850	
	Russia	0	0	0	0	0	0			35	
	Canada			13000			31050			77700	103026
	WORLD			82763			194500			390000	418000
	BRITAIN and IRELAND	48075	53598	54548	64136	140000	138474			230000	261000

Table 2 – Average breeding success for gannets in nine UK colonies

Colony	n years	Mean breeding success	Standard deviation	Standard error
Hermaness	21	0.6514	0.0497	-
Noss	24	0.722	0.0548	-
Fair Isle	21	0.6895	0.0819	-
Ailsa	20	0.711	0.078	-
Noup Head	2	0.565	-	-
Grassholm	2	0.795	-	-
Troup Head	11	0.6009	0.0981	-
Bempton (recent)[2]	8	0.7712	0.0643	-
Bempton (old)[3]	17	0.69988	0.1102	-
Bass	12	0.7692	0.0496	-
All data	138	0.69797	0.08888	0.00757

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[2] 1990-1998, excluding 1995

[3] 1961-1976

Table 3 – Estimated survival rates of gannets at different age classes from Nelson (2002) and Wanless *et al.* (2006). See Table 4, Appendix II for summary of Wanless *et al.* (2006) data

Age class	Source	Colony	Period	Survival rate	95% CI
Adult	Nelson 1978, 2002	Bass		0.94	
males	Nelson 1978, 2002	Bass	1960-75	0.889	
females	Nelson 1978, 2002	Bass	1960-75	0.913	
	Wanless *et al.* 2006	Bass		0.916	0.910-0.922
	Wanless *et al.* 2006	Other UK and Irish colonies (non-Bass)		0.922	0.916-0.927
	Wanless *et al.* 2006	All UK and Ireland		0.919	0.915-0.922
1st Year	Nelson 1978, 2002	Bass		0.35	
	Wanless *et al.* 2006	Bass		0.542	0.516-0.567
	Wanless *et al.* 2006	Other UK and Irish colonies (non-Bass)		0.42	0.394-0.445
	Wanless *et al.* 2006	All UK and Ireland		0.424	0.410-0.439
2nd Year	Nelson 1978, 2002	Bass		0.9	
	Wanless *et al.* 2006	Bass		0.779	0.765-0.793
	Wanless *et al.* 2006	Other UK and Irish colonies (non-Bass)		0.852	0.842-0.861
	Wanless *et al.* 2006	All UK and Ireland		0.829	0.821-0.863
3rd Year	Nelson 1978, 2002	Bass		0.94	
	Wanless *et al.* 2006	Bass		0.859	0.848-0.869
	Wanless *et al.* 2006	Other UK and Irish colonies (non-Bass)		0.908	0.901-0.915
	Wanless *et al.* 2006	All UK and Ireland		0.891	0.886-0.896
4th Year	Nelson 1978, 2002	Bass		0.94	
	Wanless *et al.* 2006	Bass		0.863	0.852-0.874

Age class	Source	Colony	Period	Survival rate	95% CI
	Wanless *et al.* 2006	Other UK and Irish colonies (non-Bass)		0.91	0.903-0.917
	Wanless *et al.* 2006	All UK and Ireland		0.895	0.889-0.9
Nonbreeding adults	Nelson 1978, 2002	Bass		0.01	

Table 4 – Age specific average survival rates and confidence intervals from Wanless *et al.* (2006)

Age class	All colonies	Bass Rock	Other colonies
Adult	0.919 (0.915-0.922)	0.916 (0.910-0.922)	0.922 (0.916-0.927)
4th year	0.895 (0.889-0.900)	0.863 (0.852-0.874)	0.910 (0.903-0.917)
3rd year	0.891 (0.886-0.896)	0.859 (0.848-0.869)	0.908 (0.901-0.915)
2nd year	0.829 (0.821-0.863)	0.779 (0.765-0.793)	0.852 (0.842-0.861)
1st year	0.424 (0.410-0.439)	0.542 (0.516-0.567)	0.420 (0.394-0.445)

Table 5 - Probability of the GB and Eire breeding population declining below set thresholds at any point in the 25 year simulations in relation to additional mortality, obtained using both the density dependent and density independent models

Threshold of decline (proportion population reduction)	0.05	0.1	0.15	0.2	0.05	0.1	0.15	0.2
Collision mortality	Probability of population decline - density independent results				Probability of population decline - density dependent results			
0	0.0066	0	0	0	0.0084	0	0	0
500	0.0076	0.0002	0	0	0.0086	0.0002	0	0
1000	0.008	0.0004	0	0	0.0118	0.0002	0	0
1500	0.0104	0	0	0	0.0162	0.0002	0	0
2000	0.0162	0.001	0	0	0.0198	0.0002	0	0
2500	0.0196	0.0006	0	0	0.0244	0.0008	0	0
3000	0.025	0.0014	0.0002	0	0.0304	0.0008	0	0
3500	0.0342	0.002	0	0	0.0436	0.0016	0	0
4000	0.0356	0.0026	0	0	0.0482	0.0036	0	0
4500	0.0552	0.0046	0.0002	0	0.0604	0.002	0	0
5000	0.0624	0.0056	0.0004	0	0.082	0.005	0	0
5500	0.079	0.008	0.001	0	0.1052	0.0096	0.0004	0
6000	0.1018	0.0092	0.0012	0	0.1324	0.014	0.0004	0
6500	0.1228	0.0198	0.0022	0	0.1532	0.0184	0.0012	0
7000	0.1554	0.0256	0.0028	0.0004	0.189	0.0244	0.002	0
7500	0.1908	0.0336	0.0052	0.0004	0.2288	0.0364	0.0024	0.0002
8000	0.2376	0.0516	0.0084	0.001	0.2694	0.0498	0.0036	0.0002
8500	0.2736	0.0716	0.0146	0.0012	0.3056	0.0588	0.0072	0
9000	0.3294	0.0962	0.02	0.0028	0.3636	0.0818	0.012	0.0004
9500	0.379	0.1268	0.0288	0.004	0.4284	0.1216	0.0194	0.0016
10000	0.4458	0.1688	0.0418	0.0066	0.4882	0.1474	0.0238	0.0022
10500	0.4992	0.2002	0.0582	0.0116	0.5326	0.1818	0.0334	0.0028

Threshold of decline (proportion population reduction)	0.05	0.1	0.15	0.2	0.05	0.1	0.15	0.2
Collision mortality	Probability of population decline - density independent results				Probability of population decline - density dependent results			
11000	0.5636	0.2562	0.0744	0.015	0.6052	0.232	0.0478	0.0064
11500	0.6198	0.3112	0.1078	0.0228	0.6434	0.2688	0.0582	0.007
12000	0.688	0.3838	0.1494	0.0382	0.7086	0.3336	0.0884	0.0094
12500	0.7378	0.422	0.1762	0.047	0.744	0.3826	0.114	0.0184
13000	0.7864	0.4982	0.2252	0.068	0.8008	0.4378	0.1464	0.0212
13500	0.8326	0.5652	0.2814	0.096	0.8428	0.505	0.1928	0.037
14000	0.8634	0.6382	0.338	0.1208	0.8698	0.5776	0.2398	0.0506
14500	0.8964	0.691	0.4042	0.1612	0.9088	0.6426	0.2874	0.0664
15000	0.9286	0.7504	0.4754	0.2114	0.9308	0.7036	0.3562	0.099

Table 6 - Estimates, based on geolocation data logger deployments, of the percentages of adult gannets from each major European population that winters (from mid-October to mid-February) in each defined marine area, and the numbers of individuals those percentages represent

	Norwegian (including Russian)	Icelandic (including Faeroese)	British & Irish (including Channel Islands)	French
Winter in North Sea	25%	0%	5%	0%
Winter in English Channel	10%	5%	5%	5%
Winter in Celtic Sea	10%	25%	5%	10%
Winter in West of Scotland	10%	20%	0%	0%
Winter in Bay of Biscay	25%	10%	30%	0%
Winter in Iberia/Mediterranean	10%	10%	10%	25%
Winter in West Africa	10%	30%	45%	60%
Breeding adults in population	8,000	64,000	600,000	34,000
Winter in North Sea	2,000	0	30,000	0
Winter in English Channel	800	3,200	30,000	1,700
Winter in Celtic Sea	800	16,000	30,000	3,400
Winter in West of Scotland	800	12,800	0	0
Winter in Bay of Biscay	2,000	6,400	180,000	0
Winter in Iberia/Mediterranean	800	6,400	60,000	8,500
Winter in West Africa	800	19,200	270,000	20,400

Table 7 - Estimates of numbers of adult gannets killed each month at Round 1 and Round 2 wind farms in relation to their population of origin

TOTAL		British Isles	Iceland	Norway	France
35.4	Jan	24.22	8.61	1.17	1.381
40.6	Feb	27.77	9.87	1.34	1.584
184.6	Mar	184.6	0	0	0
83.7	Apr	83.7	0	0	0
597.8	May	597.8	0	0	0
268.6	Jun	268.6	0	0	0
338.3	Jul	338.3	0	0	0
204.8	Aug	204.8	0	0	0
441.6	Sep	441.6	0	0	0
529.0	Oct	362.2	129	17.5	20.65
72.0	Nov	49.31	17.5	2.38	2.811
29.9	Dec	20.45	7.26	0.99	1.166
2826.2	TOTAL	2603.3	171.9	23.3	27.6

Table 8 - Estimated numbers of immature (0-4 year old) gannets killed each month at Round 1 and Round 2 wind farms in relation to their population of origin. For immatures we assume that immatures from all countries are homogeneously mixed at sea in proportion to population sizes

TOTAL		British Isles	Iceland	Norway	France
0.00	Jan	0	0	0	0
0.00	Feb	0	0	0	0
9.87	Mar	8.3889	0.8882	0.099	0.493
0.00	Apr	0	0	0	0
158.95	May	135.11	14.306	1.59	7.948
196.37	Jun	166.91	17.673	1.964	9.818
357.97	Jul	304.28	32.218	3.58	17.9
222.81	Aug	189.39	20.053	2.228	11.14
186.27	Sep	158.33	16.765	1.863	9.314
71.28	Oct	60.591	6.4155	0.713	3.564
35.08	Nov	29.821	3.1575	0.351	1.754
3.11	Dec	2.6403	0.2796	0.031	0.155
1241.72	TOTAL	1055.5	111.75	12.42	62.09

APPENDIX III. Collision Risk Modelling

Table 9 - Estimated gannet density (birds/km^2) for Round 1 and 2 offshore wind farms. Wind farm status is indicated as A – Approved, O – Operational, S – Submitted or U - Under construction (as of Feb 2012). Cell colour indicates the gannet data source: white – site specific boat survey (single estimate available), orange – site specific boat survey (range of monthly estimates provided, peak value used), yellow – boat survey of overlapping or adjacent site used, red – boat survey from more distant site used, pink – aerial survey of much larger area within which wind farm located. Aerial density estimates shown have been adjusted to correspond to boat density estimates (i.e. boat density = 7*aerial density; see text for details). Wind farms are not named due to agreed data usage restrictions

Wind Farm Index	Status	January	February	March	April	May	June	July	August	September	October	November	December
1	A	0.000	0.000	0.060	0.000	0.060	1.000	0.490	0.000	0.160	0.120	0.000	0.000
2	A	0.006	0.000	0.150	0.200	1.310	0.630	0.370	0.750	2.460	0.004	0.000	0.000
3	A	0.000	0.000	0.130	0.000	0.060	2.690	0.490	0.000	0.160	0.120	2.590	0.000
4	O	0.006	0.000	0.150	0.200	1.310	0.630	0.370	0.750	2.460	0.040	0.000	0.000
5	O	0.000	0.000	0.000	0.190	0.000	0.000	0.000	0.000	0.480	0.380	0.630	0.000
6	O	0.000	0.000	0.000	0.000	0.000	0.120	0.910	0.250	0.910	0.870	0.370	0.130
7	O	0.000	0.000	0.000	0.000	1.070	0.134	0.068	0.273	0.273	0.433	0.000	0.000
8	O	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.090	0.000
9	O	0.000	0.000	0.220	0.190	2.710	0.210	1.060	0.200	0.480	0.380	0.000	0.000
10	O	0.000	0.000	0.220	0.100	2.700	1.420	1.200	0.110	0.100	2.500	0.000	0.000
11	O	0.000	0.000	0.150	0.200	0.210	0.630	0.370	0.750	2.460	0.040	0.000	0.000
12	O	0.000	0.000	0.150	0.200	0.700	0.630	0.370	0.750	2.460	0.040	0.000	0.000
13	O	0.010	0.427	8.400	0.160	1.260	1.820	3.080	1.750	0.580	1.980	1.190	0.000
14	O	0.000	0.070	0.000	0.000	0.000	0.599	1.650	1.654	0.273	0.070	0.150	0.070
15	O	0.000	0.070	0.000	0.000	0.000	0.182	0.042	0.035	0.273	0.070	0.150	0.070
16	O	0.013	0.040	0.017	0.000	0.170	0.230	0.240	0.730	0.000	2.469	0.000	0.250
17	O	0.006	0.000	0.150	0.200	1.310	0.630	0.370	0.750	2.460	0.040	0.000	0.000
18	S	0.000	0.160	0.160	0.000	0.100	0.180	0.720	0.210	0.460	0.310	0.000	0.040
19	S	0.480	0.061	1.200	0.160	0.180	0.260	0.440	0.250	0.580	1.980	0.170	0.000
20	S	0.020	0.160	0.160	0.020	0.130	0.290	0.880	0.930	0.250	0.610	0.390	0.040
21	U	0.000	0.008	0.003	0.000	0.030	0.009	0.053	0.028	0.252	0.064	0.000	0.280
22	U	0.000	0.011	0.118	0.130	0.118	0.086	0.236	0.236	0.337	0.228	0.084	0.000
23	U	0.000	0.000	0.220	0.100	2.710	0.210	1.060	0.200	0.480	0.170	0.000	0.000
24	U	0.000	0.000	0.000	0.000	0.150	0.120	0.360	0.120	0.240	0.480	0.000	0.000
25	U	0.042	0.000	1.050	0.000	0.170	4.410	0.240	5.250	0.000	0.028	0.000	0.000
26	U	0.000	0.000	0.910	0.020	0.420	18.830	1.210	0.000	0.250	0.610	0.238	0.000
27	U	0.002	0.013	0.070	0.635	1.837	0.492	0.268	0.559	0.274	0.025	0.000	0.000

Table 10 - Wind farm and gannet parameters used in collision risk modelling. Site specific details have not been included due to agreed data usage restrictions

Wind farm parameter	**Value (range)**
Wind farm maximum dimension (km)	1.5 - 33.2
Wind farm latitude (degrees)	51.4 - 54.8
Number of turbines	22 – 206
Turbine radius (m)	40 - 70
Number of blades / turbine	3
Rotation speed (rpm)	12.1 – 16.7
Proportion of time operational (yr $^{-1}$)	0.85
Maximum blade width (m)	4.2
Blade pitch (degrees)	20
Percentage gannets flying at rotor height (Cook *et al.* 2011)	14
Gannet body length (m)	0.935
Gannet wingspan (m)	1.725
Gannet flight speed (ms^{-1})	14.9
Percentage nocturnal activity	25

Table 11 - Estimated adult gannet collision mortality for Round 1 and 2 offshore wind farms, using avoidance rate of 98%

Wind Farm Index	January	February	March	April	May	June	July	August	September	October	November	December	Total
1	0.0	0.0	2.2	0.0	0.4	35.3	15.4	0.0	5.1	3.5	6.2	0.0	68.0
2	0.3	0.0	6.5	11.7	71.4	25.1	14.0	32.0	108.5	0.2	0.0	0.0	269.5
3	0.0	0.0	4.5	0.0	0.4	90.8	14.7	0.0	4.8	3.4	5.9	0.0	124.5
4	0.1	0.0	1.6	2.8	17.0	6.0	3.3	7.6	25.9	0.4	0.0	0.0	64.7
5	0.0	0.2	0.1	2.4	35.0	1.0	19.6	3.5	5.0	4.0	0.0	0.1	71.0
6	0.0	0.0	0.0	0.0	0.0	0.4	5.7	1.5	3.2	8.9	0.8	0.9	21.3
7	0.0	0.0	0.0	0.0	13.4	1.5	0.0	0.0	2.6	5.6	0.0	0.0	23.1
8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.2
9	0.0	0.0	2.8	2.8	38.7	1.2	10.2	2.5	5.8	4.6	0.0	0.0	68.5
10	0.0	0.0	2.4	1.3	33.3	6.9	9.9	1.2	1.0	26.0	0.0	0.0	82.1
11	0.0	0.0	1.7	3.1	19.9	6.7	3.7	8.5	28.8	0.5	0.0	0.0	73.1
12	0.0	0.0	1.7	3.1	19.9	6.7	3.7	8.5	28.8	0.5	0.0	0.0	73.1
13	0.8	5.4	1.5	2.4	1.0	3.6	3.7	2.7	5.1	22.1	19.0	0.1	67.3
14	0.0	0.8	0.0	0.0	0.0	2.0	0.4	0.3	2.6	0.8	1.7	0.8	9.3
15	0.0	0.6	0.0	0.0	0.0	1.4	0.3	0.2	1.9	0.6	1.3	0.6	6.8
16	0.5	1.4	0.7	0.0	6.4	7.6	0.0	0.0	0.0	94.9	0.0	8.7	120.1
17	0.2	0.0	6.2	11.0	67.4	23.7	13.2	30.2	102.4	1.8	0.0	0.0	256.2
18	0.0	12.8	15.6	0.0	0.0	4.3	35.0	9.5	12.4	24.7	0.0	2.1	116.5
19	28.4	3.6	84.5	12.5	9.1	11.7	17.7	8.6	26.9	116.3	10.1	0.0	329.4
20	1.9	15.0	17.8	2.5	0.0	8.0	49.5	48.9	7.8	56.8	7.4	2.5	218.0
21	0.0	0.1	0.0	0.0	0.2	0.0	0.6	0.5	3.3	0.8	0.0	0.6	6.1
22	0.0	0.6	7.6	9.1	6.8	4.4	10.3	7.1	14.8	12.2	4.6	0.0	77.5
23	0.0	0.0	15.6	8.1	213.8	6.5	56.2	14.0	32.2	11.3	0.0	0.0	357.8
24	0.0	0.0	0.0	0.0	0.0	1.1	6.7	2.1	2.5	14.7	0.0	0.0	27.0
25	2.9	0.0	3.5	0.0	13.0	2.7	13.9	1.6	0.0	93.7	6.5	13.4	151.3
26	0.3	0.0	7.2	0.9	3.2	4.4	27.8	7.2	6.6	20.5	8.5	0.2	86.7
27	0.0	0.2	0.8	10.2	27.4	5.4	2.8	6.5	3.3	0.3	0.0	0.0	56.9
Total	35.4	40.6	184.6	83.7	597.8	268.6	338.3	204.8	441.6	529.0	72.0	29.9	2826.2

Table 12 - Estimated immature gannet collision mortality for Round 1 and 2 offshore wind farms, using avoidance rate of 98%

Wind Farm Index	January	February	March	April	May	June	July	August	September	October	November	December	Total
1	0.0	0.0	0.0	0.0	2.2	8.8	6.6	0.0	0.8	0.6	0.9	0.0	19.9
2	0.0	0.0	1.6	0.0	14.6	16.7	11.0	15.0	27.1	0.0	0.0	0.0	86.1
3	0.0	0.0	0.0	0.0	2.1	22.7	6.3	0.0	0.8	0.5	0.9	0.0	33.3
4	0.0	0.0	0.4	0.0	3.5	4.0	2.6	3.6	6.5	0.0	0.0	0.0	20.6
5	0.0	0.0	0.0	0.0	5.2	2.0	14.8	0.8	0.7	0.3	0.0	0.0	23.8
6	0.0	0.0	0.0	0.0	0.0	1.5	8.6	2.2	8.6	1.5	3.1	0.4	25.7
7	0.0	0.0	0.0	0.0	5.0	0.8	1.2	4.5	1.4	0.4	0.0	0.0	13.2
8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.7
9	0.0	0.0	0.2	0.0	5.8	2.3	7.7	0.6	0.8	0.3	0.0	0.0	17.6
10	0.0	0.0	0.2	0.0	5.0	13.4	7.5	0.3	0.1	1.7	0.0	0.0	28.1
11	0.0	0.0	0.4	0.0	4.1	4.5	2.9	4.0	7.2	0.0	0.0	0.0	23.2
12	0.0	0.0	0.4	0.0	4.1	4.5	2.9	4.0	7.2	0.0	0.0	0.0	23.2
13	0.0	0.0	0.0	0.0	0.7	3.5	4.6	3.9	3.0	3.6	0.0	0.0	19.3
14	0.0	0.0	0.0	0.0	0.0	1.1	0.3	0.4	1.3	0.1	0.0	0.0	3.3
15	0.0	0.0	0.0	0.0	0.0	0.8	0.2	0.3	1.0	0.1	0.0	0.0	2.4
16	0.0	0.0	0.0	0.0	2.4	4.3	12.6	36.0	0.0	6.1	0.0	0.0	61.4
17	0.0	0.0	1.5	0.0	13.8	15.8	10.4	14.2	25.6	0.2	0.0	0.0	81.5
18	0.0	0.0	0.2	0.0	11.9	17.2	52.5	14.3	33.6	4.0	0.0	1.0	134.7
19	0.0	0.0	2.6	0.0	6.6	11.2	21.7	12.3	15.8	18.9	0.0	0.0	89.2
20	0.0	0.0	0.6	0.0	17.8	32.1	74.2	73.4	21.2	9.2	29.5	1.2	259.2
21	0.0	0.0	0.0	0.0	0.3	0.1	0.3	0.0	0.2	0.0	0.0	0.3	1.3
22	0.0	0.0	0.2	0.0	2.5	2.5	8.7	10.7	7.6	2.0	0.0	0.0	34.1
23	0.0	0.0	1.0	0.0	32.0	12.7	42.4	3.3	4.4	0.7	0.0	0.0	96.5
24	0.0	0.0	0.0	0.0	6.8	4.4	10.0	3.1	6.7	2.4	0.0	0.0	33.5
25	0.0	0.0	0.1	0.0	4.8	1.5	11.8	2.4	0.0	15.3	0.0	0.0	35.9
26	0.0	0.0	0.2	0.0	2.3	4.2	34.0	10.4	3.9	3.3	0.0	0.1	58.4
27	0.0	0.0	0.2	0.0	5.6	3.6	2.2	3.1	0.8	0.0	0.0	0.0	15.5
Total	0.0	0.0	9.9	0.0	159.0	196.4	358.0	222.8	186.3	71.3	35.1	3.1	1241.7

Table 13 - Proportions of total annual estimated mortality for adult and immature gannets by wind farm status

Wind farm status	Proportion of mortality	
	Adult	Immature
Approved	0.16	0.11
Operational	0.33	0.28
Submitted	0.24	0.39
Under construction	0.27	0.22