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Review of the potential of seabird colony monitoring to inform monitoring programmes for consented offshore wind farm projects

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EXECUTIVE SUMMARY

1. To inform the consenting process, the potential impacts on seabird populations of the key effects associated with renewable energy developments are assessed through an Environmental Impact Assessment (EIA). When preparing applications for Nationally Significant Infrastructure Projects (NSIPs) in England or Wales, or for equivalent national developments or major developments in Scotland and Northern Ireland, developers are legally required as part of the Habitats Regulations Assessment or Appraisal (HRA) process to consider if the project is likely to affect European protected sites.

2. If Likely Significant Effects (LSEs) on the features of a European protected site cannot be ruled out after HRA screening, the HRA report provided with the application should enable the competent authority to then carry out an Appropriate Assessment (AA). The purpose of the AA is to ascertain whether or not there are any Adverse Effects On Integrity (AEOI) on the relevant sites. These assessments determine whether populations within the site may be negatively affected by a development, for seabirds, typically through the use of demographic models.

3. The demographic models used as part of assessments require baseline information on the abundance and demographic parameters of the populations concerned. However, it is unclear the extent to which the demographic data used in these models relate to the specific sites considered in assessments.

4. It is also unknown whether existing monitoring of seabird populations at a colony level has sufficient statistical power to detect changes of the magnitudes predicted in response to the effects associated with offshore wind farms.

5. We reviewed AAs that have been carried out in relation to 21 consented, but not yet fully operational, offshore wind farms, to identify seabird populations of Special Protection Areas (SPAs) where LSEs in relation to offshore wind farms could not be ruled out by HRA screening prior to AA. Consequently, the populations selected reflect those for which monitoring might be required to detect the impacts associated with new offshore wind farms as they become operational.

6. We identified 16 species within 38 SPAs for which LSEs in relation to offshore wind farms could not be ruled out during HRA screening prior to AA.

7. Using monitoring data collected by the Seabird Monitoring Programme (SMP) and the Retrapping Adults for Survival (RAS) scheme, we identified sites within SPAs where baseline data are available to inform the monitoring programmes for the offshore wind farms at which LSEs could not be ruled out and highlight where key gaps remain.

8. We carried out a power analysis in order to determine the extent to which changes in abundance, breeding success and survival could be detected in populations given current levels of monitoring.

9. Drawing from the values considered as part of AAs, we considered impacts equivalent to a reduction in abundance of up to 50% over 25 years, in addition to any underlying population trends; a reduction in breeding success of up to 50% over 25 years, again after any pre-existing trends in breeding success were accounted for; and, a reduction in annual adult survival of up to 5% following the construction of an offshore wind farm.

10. There are significant gaps in the monitoring of populations within SPAs for which AAs were carried out. This is likely to present difficulties to developers, governmental advisors and regulators due to a lack of baseline data against which to assess predicted impacts. Furthermore, it will make determining whether any subsequent changes recorded at these sites can be attributed to the effects associated with offshore wind farms extremely challenging.

11. Limitations in the spatial and temporal coverage of existing monitoring mean that existing data typically lack sufficient statistical power to detect the changes that may be associated with the effects of offshore wind farms. This raises the risk of type II errors – incorrectly failing to reject the null hypothesis that a wind farm is not having an impact on a population. Whilst a wind farm may not be affecting a population, if data lack sufficient power to detect a change of the magnitude expected, it may not be possible to conclude this.

12. In relation to abundance, it was possible to detect a reduction in population size over 25 years, relative to the population size in the absence of a wind farm, of 25% for a single species at a single site, Northern Gannet at Flamborough and Filey Coast SPA. It was possible to detect a decline of 50%, relative to the population size in the absence of a wind farm, in eight species in up to three SPAs.

13. Productivity tended to be more widely monitored than abundance and data tended to have greater power to detect changes related to the effects of offshore wind farms. It was possible to detect a decline in breeding success of 25% in seven species at up to four SPAs and a decline of 50% in nine species at up to 10 SPAs. Where species had consistent annual breeding success (e.g. Northern Fulmar, Northern Gannet and auks) it was shown that monitoring breeding success picked up more subtle population-level changes than could be detected through monitoring abundance. However, in the case of species with highly variable annual breeding success (e.g. large gulls and terns), it was easier to detect population level changes through monitoring abundance.

14. At present, adult survival is the most poorly monitored of the parameters we considered. If between five and ten years pre-construction monitoring data could be collected, it would be possible to detect changes in adult survival that would result in more subtle population-level impacts than could be detected through monitoring abundance. Such data would also be of value in relation to the assessment of the cumulative effects of multiple wind farms, as they would account for the background mortality attributable to existing projects.

15. Our analyses highlight the potential to improve both monitoring and the analysis of data which are collected as part of monitoring. We make 16 recommendations to suggest how current seabird monitoring and associated analyses could be enhanced to support monitoring of the impacts associated with offshore wind farms:

I. It is vital that the current national seabird census 'Seabirds Count' is supported. This will provide vital baseline information about the population status of the UK's seabirds. Amongst other uses, this will provide robust baseline population estimates for species and sites at which LSEs in relation to offshore wind farms cannot be ruled out by the HRA screening process.
II. Existing monitoring of abundance by the SMP should be expanded so that abundance of all species within SPAs where LSEs could not be ruled out by the HRA screening process is monitored at least once every two years.

III. In the case of terns, it is important to monitor populations at an ecologically coherent scale (e.g. Cook et al., 2011), as well as at an individual colony level. In some cases (e.g. in relation to those breeding within the Forth Islands SPA), this may include birds that nest outside a SPA.
IV. Consideration (within a modelling framework) of the anthropogenic and environmental factors influencing seabird population processes at a sites and/or colony level will increase the power of monitoring data to detect any changes attributable to the presence of an offshore wind farm by accounting for some of the unexplained variation in existing data sets.
V. Existing monitoring of breeding success by the SMP should be expanded and, consideration by scheme organisers should be given to use of data from the BTO/JNCC Nest Record Scheme.
VI. Monitoring of breeding success in burrow-nesting seabirds like Manx Shearwater and Atlantic Puffin should be expanded to selected other sites as a priority. This will give us valuable demographic data for species whose abundance can be hard to monitor.
VII. It is important the monitoring of breeding success in Northern Gannet, Northern Fulmar and auk species is expanded to selected other sites. The changes that can be detected in breeding success would result in more subtle population-level impacts than those that could be detected through monitoring of abundance.
VIII. Expanding monitoring of breeding success for gulls, skuas and terns may help us to understand how any impacts at a population-level, in response to the effects associated with offshore wind farms, are driven by changes in breeding success and/or adult survival. This could be achieved through the use of population models incorporating any recorded offshore wind farm impacts on breeding success and/or survival.
IX. The SMP database should include a clear site definitions and structure, showing linkages between sites covered for abundance and demographic monitoring data, e.g. plots/sub-sites monitored for breeding success and adult survival should be linked to the sites monitored for abundance. The number of nests monitored for breeding success should be directly relatable to the whole colony count, e.g. it must be possible to express the nests monitored for breeding success as a proportion of the whole colony. If this proportion exceeds 1, a clear reason must be given for this (e.g. a re-nesting attempt by a pair which had previously failed).
X. Existing monitoring of adult survival, through the SMP key site programme and RAS studies must be maintained and expanded. The value of such studies could be further enhanced through the ringing of chicks as, over time, this would provide a better understanding of the age structure of populations.
XI. Additional RAS studies (including studies based on colour-ringing) should be instigated in order to measure changes in adult survival in response to effects associated with offshore wind farms. These should be instigated towards the end of the consenting process for the relevant wind farms at the latest and ideally, as part of baseline data collection, in order to ensure sufficient data can be collected.
XII. Species such as Northern Gannet, Black-legged Kittiwake and large gulls, which are believed to be particularly sensitive to offshore wind farms with respect to collision, should be prioritised for adult survival studies.
XIII. As pre-construction data are likely to be limited, where possible, studies should include approaches, such as colour-ringing, which are likely to maximise recapture/re-sighting rates.
XIV. Consideration should be given to the potential for added value for any capture-mark recapture study; for example, the widespread deployment of geolocators in order to understand wintering areas for key species and, the potential for interaction with wind farms outside the breeding season.
XV. By monitoring multiple parameters within the same site, we are likely to increase the power of the data to detect any changes which may be associated with offshore wind farms.

Consequently, careful consideration should be given to identifying instances where abundance, adult survival and breeding success can feasibly be collected together in order to enable integrated population monitoring for species at individual sites. Such an approach is likely to be most feasible for Northern Gannet, Black-legged Kittiwake and the large gulls.

XVI. In order to monitor seabird populations within colonies at the level required in order to detect any effects associated with offshore wind farms, it is vital that sufficient support, both in terms of finance and infrastructure, is provided to those who collect the data (many of whom currently do so on a voluntary basis).

1. INTRODUCTION

1.1 Background

Offshore wind farms may potentially have a number of negative effects on seabird populations. These include displacement from preferred foraging areas, the risk of collision with turbines and the wind farm acting as a barrier to migrating or commuting birds (Drewitt & Langston, 2006; Everaert & Stienen, 2007; Furness *et al.*, 2013; Garthe & Hüppop, 2004; Krijgsveld *et al.*, 2011; Masden *et al.*, 2010; Vanermen *et al.*, 2015).

To inform the consenting process, the potential impacts of the key effects associated with developments are assessed through an Environmental Impact Assessment (EIA) in relation to baseline populations at site, local, regional and national levels. When preparing applications for Nationally Significant Infrastructure Projects (NSIPs) in England or Wales, or for equivalent national developments or major developments in Scotland and Northern Ireland, developers are legally required to consider if the project is likely to affect European sites by providing a Habitats Regulations Assessment or Appraisal (HRA). HRA is an iterative process and the emphasis is on understanding the potential for Likely Significant Effects (LSEs) and avoiding Adverse Effects On Integrity (AEOI) on relevant sites. If LSEs on features of European site, either alone or in combination with other plans or projects, cannot be ruled out after HRA screening, the HRA report provided with the application should enable the competent authority to then carry out an Appropriate Assessment (AA). The purpose of the AA is to ascertain whether or not there are any AEOI on the relevant sites. Under the EC Birds Directive, sites are classified as Special Protection Areas (SPAs) based on the size of the population of a species, or suite of species, that they hold and must be maintained in a favourable condition (i.e. species' population sizes should not decline below those set out in the conservation objectives for the site). For the purposes of the HRA, the potential for an offshore wind farm to negatively impact the condition of features of a protected site is typically assessed using demographic modelling, for example Population Viability Analysis (PVA) or Potential Biological Removal (PBR) (Cook & Robinson, 2017; Cook & Robinson, 2015; O'Brien *et al.*, 2017), informed by mortality rates predicted from collision risk models (Masden & Cook, 2016) and/or predicted displacement rates.

Constructing realistic population models is the first step towards reliably assessing how infrastructure developments, such as offshore wind farms, may impact the population trends of different species (Cook & Robinson, 2017; Freeman *et al.*, 2014). The construction of these models requires the individual demographic rates that influence the size of a population to be quantified. Thus there is a specific need for robust baseline data on both the breeding success and adult survival rates of seabird species which can be explored both spatially (e.g. ecological coherent regions or even at a colony level) and temporally (over a number of years).

Post-consent monitoring requirements within the licence conditions agreed with the regulatory authority are largely designed to detect any unforeseen impacts and validate predictions made in an EIA or HRA. With respect to birds, monitoring programmes have focused on at-sea surveys as a means to provide the data required to assess displacement or barrier effects and collision risk assessments through modelling (Marine Management Organisation, 2014). However, there has been little attempt to validate whether demographic rates for species at protected sites identified through the assessment process may have actually been impacted by key effects. Furness & Wanless (2014) nevertheless highlighted that, for Northern Gannet *Morus bassanus,* strategic monitoring of adult survival rates for a range of colonies at varying distances (as a proxy of likely impact) from offshore wind farms could provide evidence as to whether collision mortality did or did not cause impacts on the species' adult survival.

The Seabird Monitoring Programme (SMP)[1], which is overseen by the Joint Nature Conservation Committee (JNCC) on behalf of a wider partnership, collects annual data on the breeding numbers and breeding success of seabirds across the UK, to enable their conservation status to be assessed. The scheme was set up in 1986 and aims to collect long-term, annual data at four key sites – the Isle of May (eastern Scotland), Fair Isle (northern Scotland), Canna (West Scotland) and Skomer (Wales) – for which financial assistance is provided, complemented by sample data collection at other colonies across UK, undertaken by the partnership organisations and volunteers. Monitoring of adult survival/return rates is also undertaken at the SMP key sites with data from the Isle of May and Skomer contributing to formal JNCC reporting on UK seabird populations to government (e.g. Newell *et al.*, 2016, Stubbins *et al.*, 2016). Wider adult survival monitoring is achieved by volunteers contributing to the Retrapping Adults for Survival (RAS) scheme[2], coordinated by the British Trust for Ornithology (BTO) and to which the data from Canna also contribute (Swann *et al.*, 2016). The SMP key sites also provide data on additional parameters, such as phenology and chick diet, which are more time consuming to measure, but provide important information on the state of seabird populations and the health of the wider marine environment[3]. Data collection methods are provided in Walsh *et al.* (1995), but work associated with current redevelopment of the SMP database, being undertaken on behalf of JNCC by the BTO, has identified the need to update this document to reflect emerging methods. Work on behalf of JNCC has also assessed the feasibility of Integrated Population Monitoring of Britain's Seabirds, bringing together the SMP's population monitoring and demographic monitoring (Robinson & Ratcliffe, 2010) and existing demographic monitoring of seabirds using ringing (Robinson & Baillie, 2012).

Monitoring the populations and demography of seabird colonies can inform both impact assessments, where it is necessary to understand baseline population sizes, and post-consent monitoring, where it is necessary to validate the predictions made during the impact assessment process. Despite this, requirements for population-level monitoring are rarely included as part of license conditions. Therefore, there is an urgent need to assess the extent to which existing seabird colony monitoring in the UK meets the requirements of the renewables industry. This project aims to meet that need and to make recommendations on: (i) the requirements for developing existing population and demographic monitoring of seabirds to support the needs of the renewables industry and (ii) the short-, medium- and long-term priorities for such monitoring.

1.2 Aims

This project aims to bring together information from these existing reviews and on key seabird species and sites identified through the consenting process for offshore wind farms to:

i. Identify SPAs and feature species potentially affected by developments;

ii. Summarise existing demographic monitoring and the availability of baseline information on the annual abundance, breeding success and adult survival rates of seabird species to inform population models at a site-specific and regional level;

iii. Determine where sufficient statistical power exists to detect changes of the magnitudes described in appropriate assessments;

iv. Provide recommendations as to the potential for improved monitoring to provide for the above.

[1] http://jncc.defra.gov.uk/page-1550)
[2] (https://www.bto.org/volunteer-surveys/ringing/surveys/ras)
[3] (http://jncc.defra.gov.uk/page-6285)

2. METHODS

2.1 Identifying relevant SPAs and feature species

Protected site breeding populations of seabirds may potentially be impacted by the effects associated with offshore wind farms through impacts on individual birds' fitness, i.e. their survival or breeding success. As part of the consenting process for offshore wind farms, any LSEs on features of a European site and/or a European marine site should be identified by developers through an HRA, such that decision-makers may make an AA, if required.

A list of all the relevant wind farms in the UK was derived from the 4COffshore website[4] based on the following categories: Partial Generation/Under Construction, Under Construction, Pre-Construction, Consent Authorised. Excluded categories were: Decommissioned, Concept/Early Planning, Development Zone, Dormant, Cancelled. Those which were operational (Fully Commissioned) were also disregarded on the basis that it would be too late to propose colony-based monitoring which could detect possible wind farms impacts. However, voluntary, strategic monitoring in relation to these sites would still be of value.

AAs for these offshore wind farms were then located. AAs for Scottish developments, as under taken by Marine Scotland, were sourced from the Scottish Government's website[5]. AAs undertaken for English developments, by the Department of Energy and Climate Change (DECC), now the Department for Business, Energy and Industrial Strategy (BEIS), were located on the National Infrastructure Planning website[6].

For each offshore wind farm, we extracted information on those breeding seabird species features of SPAs for which LSEs in relation to offshore wind farms could not be ruled out by HRA screening prior to AA, either as a result of the project alone or due to in combination effects with other plans and projects. We have therefore not included all the European qualifying features for all listed SPAs in this work.

We have not made any distinction between breeding seabird species which qualify as SPA features in their own right and those which are only named as part of a breeding seabird assemblage. Similarly, we have not differentiated as to whether a LSE was identified for a particular wind farm alone or whether it was in combination with other plans and projects. In addition to seabird species (for which abundance and productivity are monitored by the SMP), we also include breeding Red-throated Diver *Gavia stellata,* to highlight a significant gap in monitoring. As the focus of this work was on monitoring during the breeding season, we did not consider wintering features of SPAs for which an LSE could not be ruled out by HRA screening (e.g. Common Eider *Somateria mollissima* within the Ythan Estuary, Sands of Forvie and Meikle Loch SPA in relation to the Aberdeen Bay Offshore Wind Farm). We also present by species, the offshore wind farm for which a LSE had been identified and the associated effect.

Note, throughout the report, we refer to the present names of SPAs where these may have changed to reflect boundary amendments.

[4] https://www.4coffshore.com/windfarms/windfarms.aspx?windfarmId=UK36
[5] http://www.gov.scot/Topics/marine/Licensing/marine/scoping
[6] https://infrastructure.planninginspectorate.gov.uk/

2.2 Summarising existing demographic monitoring and the availability of data to inform population models

2.2.1 Abundance

In the UK, the abundance of seabirds at breeding colonies is monitored as part of the SMP and reported as the number of Apparently Occupied Nests or Territories (AONs/AOTs). Although it is possible to report UK abundance trends for 18 species (see Table 2.2.1), not all registered colonies are covered on an annual basis. We reviewed the latest SMP data to summarise the availability of abundance data for those species features of SPAs for which LSEs associated with offshore wind farms could not be ruled out by the HRA screening process (see section 2.1). We determined the extent to which existing SMP data can be used to provide baseline population estimates for the colonies and SPAs concerned. It should be noted that a single SPA may contain multiple colonies (or sub-colonies). The focus of our analysis for abundance, and subsequently also for breeding success and survival, is on whether a change can be detected at those colonies that are monitored within SPAs.

2.2.2 Breeding success

In addition to abundance, breeding success data (the number of chicks fledged in relation to the total number of monitored nests) are also collected as part of the SMP. At present, it is possible to report UK breeding trends for 20 species based on the data collected across colonies (See Table 2.2.1) but, as for abundance, breeding success data are not collected from all registered colonies on an annual basis. As above, we reviewed the latest SMP data to summarise the availability of breeding success data for those species features of SPAs for which LSEs associated with offshore wind farms could not be ruled out by the HRA screening process (section 2.1).

2.2.3 Adult survival

Adult return rates, which can be analysed in order to estimate survival, are monitored at only two of the SMP key sites (e.g. Newell et al., 2016; Swann et al., 2016) and for a total of eight species (Table 2.2.1). Outside the key sites, wider adult survival monitoring is also achieved by volunteers contributing to the RAS scheme (Horswill *et al.*, 2016). We summarise the availability of data from both the key sites and RAS scheme for those species features of SPAs for which LSEs associated with offshore wind farms could not be ruled out by HRA screening prior to AA (section 2.1).

Table 2.2.1 Demographic parameters of seabird species monitored by the Seabird Monitoring Programme (SMP) and the extent to which they feed into national indicators; - = no data, (✓) = data collected but not possible to generate UK trend and ✓ = UK trend data available (based on results presented in the most recent SMP report available from http://jncc.defra.gov.uk/page-3201).

Species	Abundance	Breeding success	Survival /return rates
Red-throated Diver	-	-	-
Northern Fulmar	✓	✓	-
Manx Shearwater	(✓)	(✓)	✓[S]
European Storm-petrel	(✓)	(✓)	-
Leach's Storm-petrel	(✓)	(✓)	-
Northern Gannet	✓	✓	-
Great Cormorant	✓	(✓)	-
European Shag	✓	✓	✓[IOM]
Arctic Skua	✓	✓	-
Great Skua	(✓)	✓	-
Black-legged Kittiwake	✓	✓	✓[S,IOM]
Black-headed Gull	✓	✓	-
Mediterranean Gull	(✓)	(✓)	-
Common Gull	(✓)	✓	-
Lesser Black-backed Gull	✓	✓	✓[S]
Herring Gull	✓	✓	✓[S]
Great Black-backed Gull	✓	✓	-
Little Tern	✓	✓	-
Sandwich Tern	✓	✓	-
Common Tern	✓	✓	-
Roseate Tern	✓	✓	-
Arctic Tern	✓	✓	-
Common Guillemot	✓	✓	✓[S,IOM]
Razorbill	✓	✓	✓[S,IOM]
Black Guillemot	✓	✓	-
Atlantic Puffin	(✓)	✓	✓[S,IOM]

Key sites: [S] = Skomer; [IOM] = Isle of May.

2.3 Assessing the statistical power to detect changes in demographic parameters

2.3.1 Abundance

As many colonies are not monitored on an annual basis by the SMP, and there is annual variation around mean population trends at individual colonies (Hatch 2003), it may be difficult to detect meaningful population changes attributable to an offshore wind farm at a colony level. In order to understand this, we carried out power analyses as follows.

Analyses were undertaken for those species features of SPAs for which LSEs associated with offshore wind farms could not be ruled out by the HRA screening process and for which regular (at least five years data collected since 2000) monitoring data were available from SMP (section 3.1). Using a

Generalized Linear Model (GLM) with a negative binomial error structure, we modelled the change in population size for each individual colony over the time period for which data were available (Figure 2.3.1). We then used this model to predict the likely change in population size, in the absence of a wind farm, over the subsequent 25 years (the expected lifespan on an offshore wind farm). We then applied a wind farm effect of a gradual reduction in population size of between 0 and 50% at 5% intervals over the lifetime of the wind farm (Figure 2.3.1). These ranges were considered comparable to those predicted in PVAs referred to in the AAs. For example, in combination reductions in populations, beyond those predicted in the absence of a wind farm, of up to 50% over 25 years were considered for auks in relation to the Hornsea Two Offshore Wind Farm. Differences in methodology, e.g. the use of Potential Biological Removal (O'Brien et al., 2017), mean that comparable figures are not available in relation to other wind farms. We simulated the annual variation in mean trend required to achieve these declines by sampling from the residuals of the original GLM. For every species, colony and impact level, we repeated this process for over 1,000 iterations. For each iteration, we fitted two models. The first modelled population size in relation to time. As we considered that the divergence between the populations with and without an impact as likely to increase over time, for the second model we fitted an interaction between time and the presence of a wind farm. The power of the data to detect a population impact associated with a wind farm was taken to be the proportion of iterations in which the second model better explained the data (assessed using the change in Akaike Information Criterion, AIC). In order to aid comparison with analyses of power to detect changes in breeding success and adult survival, results are presented as the counterfactual of the impacted and un-impacted populations, derived by comparing the population size after 25 years for each level of impact to the population size after 25 years assuming no wind farm impact (Cook & Robinson, 2017).

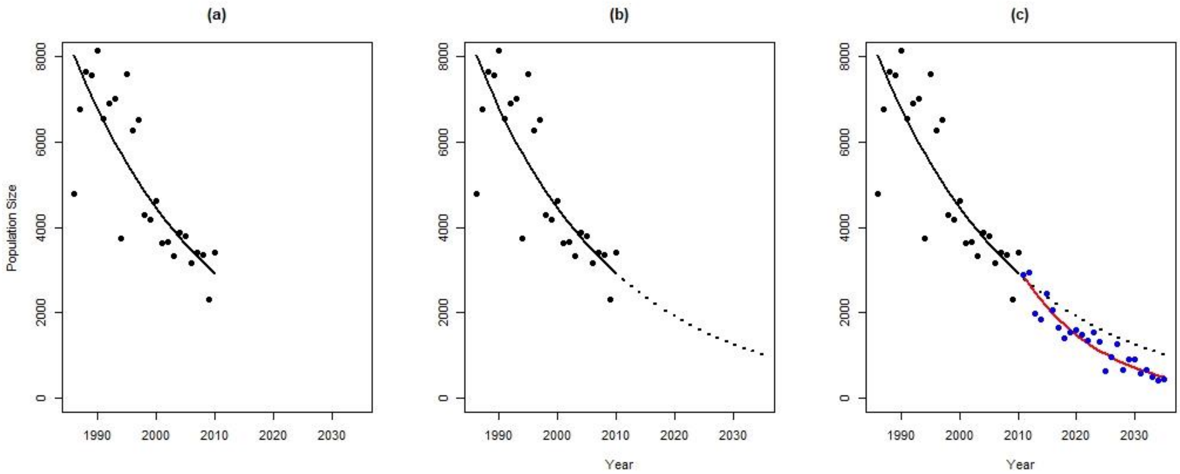

Figure 2.3.1 Determining the power of existing monitoring to detect the changes in abundance which may be associated with an offshore wind farm: (a) the change in abundance between 1986 and 2010 was modelled (solid black line = fitted values, black dots = actual values); (b) this model was then projected over 25 years, reflecting the lifespan of a wind farm (dotted line = projected values for 2010-2035); (c) a wind farm effect, in this case a 50% reduction over 25 years was applied (red line). Annual variation around these projected values was then simulated by sampling from the model residuals (blue dots). The process was repeated over 100 iterations and power was taken to be the proportion of iterations in which the inclusion of an interaction between time and the presence of a wind farm improved model performance. The example presented relates to Black-legged Kittiwake on the Isle of May.

In order to understand how monitoring frequency may affect our ability to detect population impacts, we repeated this analysis assuming a 25% and 50% reduction in population size over 25 years as a result of the impact associated with a wind farm. We considered post-construction monitoring of population size at intervals of 2, 3, 4, 5, 6, 7, 8, 9 and 10 years and, how these intervals affected our ability to detect impacts of a 25% or 50% reduction in population size over 25 years associated with a wind farm. Finally, as it may be desirable to detect a trend likely to result in a significant impact at a population-level as early as possible, we assessed how many years of successive annual counts were required before changes of a magnitude likely to result in a 25% or 50% reduction over 25 years could be detected.

All analyses were carried out in R (R Core Team 2014) and negative binomial GLMs were fitted using the MASS package (Venables & Ripley 2002).

2.3.2 Breeding success

As is the case with the abundance data, inter-annual variability in breeding success and differences in monitoring strategies may affect our ability to detect meaningful changes in breeding success at a colony level which can be attributed to the effects of wind farms. Consequently, for species features of SPAs for which LSEs associated with offshore wind farms could not be ruled out by the HRA screening process, and for which breeding success was regularly monitored as part of SMP (at least five years data collected since 2000), we carried out power analyses.

Following Cook *et al.* (2014), we converted annual estimates of breeding success into an average success rate per egg, based on data presented in Robinson (2017). We then modelled this using a GLM with a binomial error distribution and used the output from these models to estimate a mean breeding success per colony in each year. As was the case with the abundance data, we then projected trends in breeding success forward through time before applying a wind farm effect. We allowed for annual variation around the mean trend in breeding success by sampling from the model residuals. We repeated this process over 1,000 iterations, fitting two models: one with and one without a wind farm effect, each time. We considered that the presence of a wind farm was likely to cause a step change in the predicted level of breeding success; consequently, for the second model we included the presence or absence of the wind farm as a factor. As was the case with abundance, the power of the data to detect changes in breeding success attributable to the wind farm was taken to be the proportion of iterations in which the inclusion of a wind farm effect improved the model.

Finally, in order to put these changes in breeding success into the context of changes in the population size resulting from the effects of an offshore wind farm, for each species we constructed a simple Leslie Matrix Model for each species using the demographic data presented in Horswill and Robinson (2015). Using a matched runs approach, we then modelled the change in population size over 25 years for populations with, and without, the impact associated with an offshore wind farm applied (Cook & Robinson, 2017). As for the PVAs that informed the AA for offshore wind farms in the Firth of Forth, we considered that this reduction would apply to populations for the whole period over which the wind farm was operational, reflecting a reduced carrying capacity following the potential loss of a key foraging habitat. For the Firth of Forth PVA, a reduction in breeding success of up to 20% was considered[7]. However, given that there are wide uncertainties about how the presence of an offshore wind farm may affect seabird breeding success and the potential for impacts to be highly site specific (e.g. the impact of a wind farm on a key foraging area may be more significant than one in an area used for commuting), we felt it appropriate to consider a wider range of potential impacts, in this case a reduction in breeding success of 0-50%. In order to put the results

[7] https://www2.gov.scot/Resource/0046/00460542.pdf

into context, for each level of impact, we estimated the counterfactual of population size after 25 years. These results were then compared to the minimum changes that were detectable by monitoring abundance only.

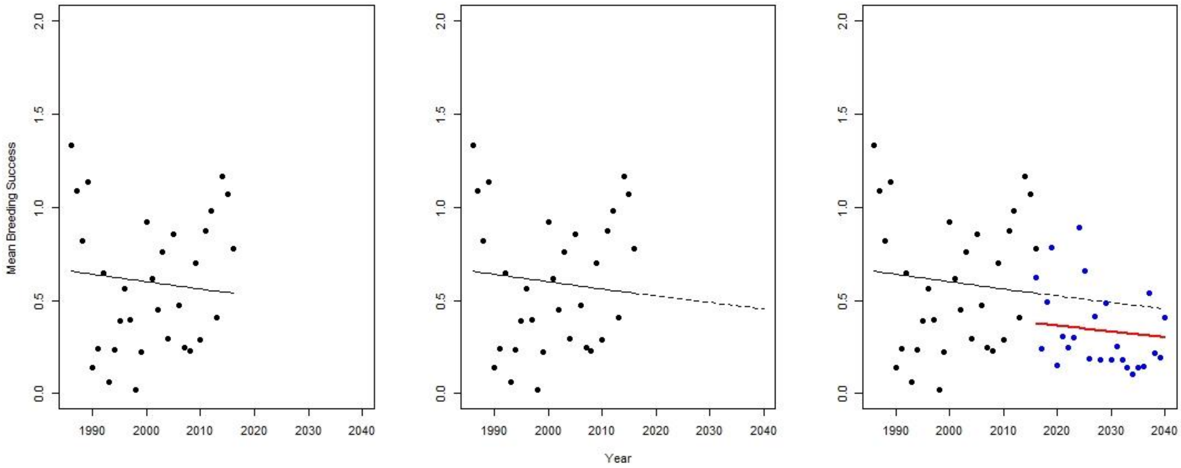

Figure 2.3.2 Determining the power of existing monitoring to detect the changes in breeding success which may be associated with a wind farm: (a) the change in breeding success between 1986 and 2010 was modelled (solid black line fitted values, black dots, actual values); (b) this model was then projected over 25 years, reflecting the lifespan of a wind farm (dotted line projected values for 2010-2035); (c) a wind farm effect, in this case a 25% reduction once the wind farm was operational was applied (red line). Annual variation around these projected values was then simulated by sampling from the model residuals (blue dots). The process was repeated over 100 iterations and power was taken to be the proportion of iterations in which the inclusion of a wind farm factor improved model performance. The example presented relates to Black-legged Kittiwake on the Isle of May.

2.3.3 Adult survival

A recent study by Horswill *et al.* (2018) sought to identify optimal strategies for obtaining robust estimates of adult survival in seabirds through the RAS scheme. This study suggested that in order to obtain robust estimates of adult survival, a mark-recapture study should operate for a minimum of 10 years, mark a minimum of 200 birds each year and obtain a recapture rate of at least 0.6. Based on the recommendations from this study, we carried out power analyses in order to understand what the minimum monitoring requirements needed to detect changes in adult survival associated with the effects of an offshore wind farm were.

Using the adult survival rates for each species recommended in Horswill and Robinson (2015), and recommendations for minimum sampling effort on based on typical seabird mark-recapture studies (Horswill *et al.* 2015, 2018), we simulated recapture histories for birds marked in the 10 years prior to wind farm construction and the 10 years following wind farm construction. We assumed 200 birds were marked each year and that there was a recapture rate of at least 0.6. Following the construction of the wind farm, we applied a wind farm effect of a reduction in annual adult survival rates of 0-0.05. These ranges were considered comparable to those declines used in the PVAs that informed the AA for offshore wind farms in the Firth of Forth (Freeman et al., 2014). We then analysed our simulated recapture histories using Cormack-Jolly-Seber (CJS) models in the R package "marked" (Laake *et al.* 2013). We fitted two CJS models, one assuming constant adult survival and one assuming a wind farm impact. We considered that the presence of a wind farm was likely to result in a step change in the adult survival rate. Consequently, we included the presence or absence of a wind farm as a factor. These models are more computationally intensive than the GLMs described above, consequently it was not possible to use 1000 iterations, as above. For each level of impact, we thus repeated this process over 100 iterations and the power to detect a change in adult survival was taken to be the proportion of simulations in which the model with a wind farm effect better explained the simulated data than the model assuming constant adult survival, as assessed using AIC.

If there was no study underway to estimate adult survival at a seabird breeding colony prior to the consenting process for an offshore wind farm, it is unlikely to be realistic to collect 10 years of capture-mark-recapture data in advance of the construction of the offshore wind farm. Consequently, we consider alternative scenarios in which 3 or 5 years of capture-mark-recapture data were collected prior to construction. In order to account for the reduced data availability, we considered alternative strategies in relation to the total number of birds marked and the recapture rate of these birds. The scenarios considered for the collection of capture-mark-recapture data are presented in Table 2.3.1.

As for breeding success, to put the changes in adult survival that could be detected following different monitoring strategies into the context of changes that could be detected through monitoring abundance alone, we again used simple Leslie Matrix Models to estimate the counterfactual of population size for each level of offshore wind farm impact. We then compared the impacts that could be detected through the monitoring of adult survival, to the levels that could be detected through monitoring abundance only.

Table 2.3.1 Scenarios considered for capture-mark-recapture studies at seabird breeding colonies that were used to generate estimates of adult survival.

Scenario	**Number of years pre-construction data collection**	**Number of years post-construction data collection**	**Number of birds marked each year**	**Recapture probability**
Baseline scenario (Horswill *et al.* 2018)	10	10	200	0.6
Scenario 1	5	10	200	0.6
Scenario 2	5	10	400	0.6
Scenario 3	5	10	400	0.8
Scenario 4	3	10	200	0.6
Scenario 5	3	10	400	0.6
Scenario 6	3	10	400	0.8

3. RESULTS

3.1 Identifying relevant SPAs and feature species

We reviewed a total of 21 AAs. These identified 16 breeding seabird species within 38 SPAs for which LSEs could not be ruled out by the HRA screening process (Figure 3.1.1). The complete list of all 21 not yet fully operational offshore wind farms in the UK considered in this report is shown in Appendix 1, Table A1, along with the stage of the planning /development. For each of these offshore wind farms, the qualifying features of SPAs for which a LSE could not be ruled out by the HRA screening process are identified in Table A2; these are the species and site combinations then considered in the simulations. We also summarise, by species, those SPAs for which a LSE was identified and which type of effect was predicted to arise (collision, displacement and/or barrier effects) (Table A3).

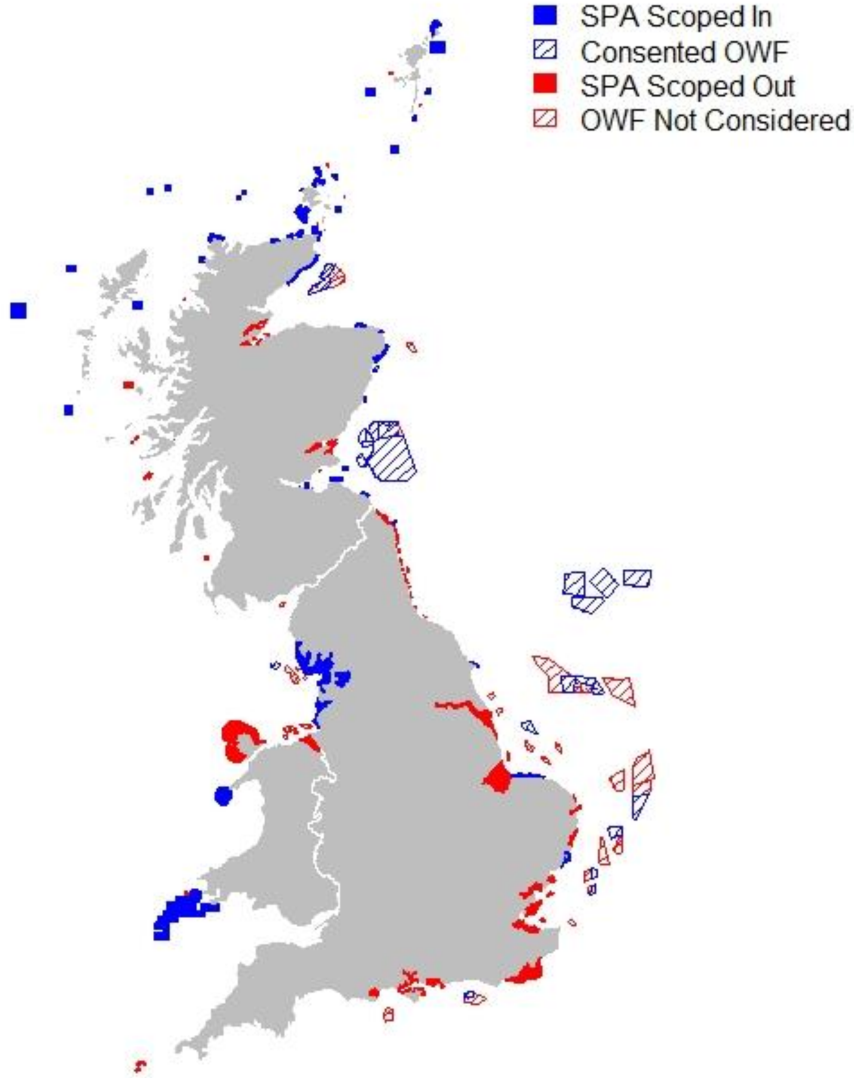

Figure 3.1.1 Special Protection Areas with breeding seabird features at which Likely Significant Effects in relation to offshore wind farm could not be ruled out by the HRA screening process.

3.2 Summarising existing demographic monitoring and the availability of data to inform population models

Colony specific estimates of abundance and breeding success from the SMP were obtained for the species and sites summarised in Table A2 through a data request to JNCC. Data were available up to, and including, the 2015 breeding season.

Table A4 summarises the data available on these parameters for the species and sites for which a LSE could not be ruled out by the HRA screening process.

3.3 Species results

3.3.1 Northern Fulmar *Fulmarus glacialis*

Abundance

Assessments for 12 offshore wind farm projects concluded that LSEs could not be ruled out by the HRA screening process for populations of Northern Fulmar at a total of 25 SPAs (Figure 3.3.1, Table A2). We identified nine colonies within four of these 25 SPAs at which the species' abundance has been regularly monitored by the SMP, which may be sufficient to establish baseline population estimates against which any impacts arising as a consequence of the development of offshore wind farms could be assessed. Of these colonies, six were within the Forth Islands SPA (Bass Rock, Craigleith, Fidra, Inchmickery, Isle of May and the Lamb) and the remaining three represented the Copinsay SPA, the Fowlsheugh SPA and the Mingulay and Berneray SPA.

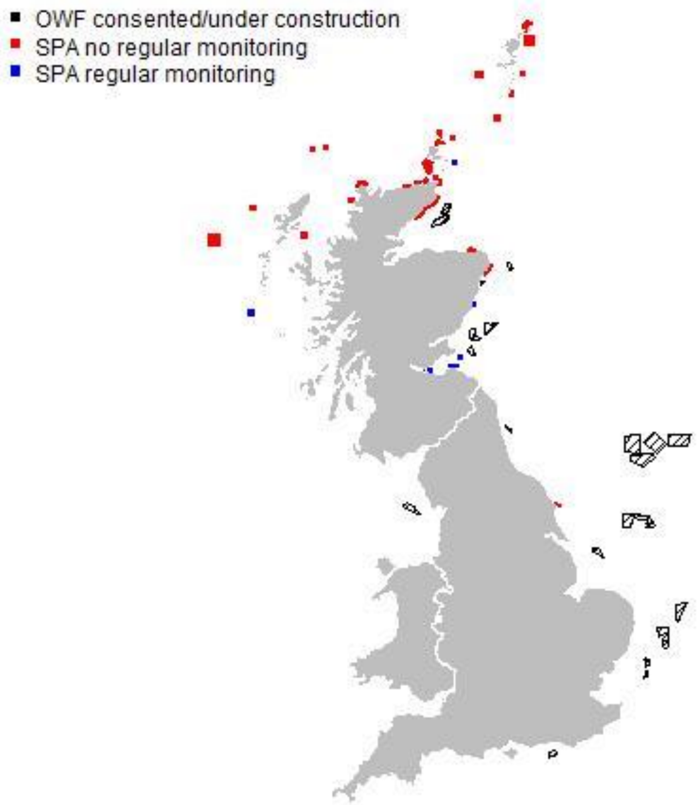

Figure 3.3.1 Special Protection Areas for Northern Fulmar at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of abundance.

It was not possible to detect a wind farm effect leading to a reduction in population size of 50% over 25 years at four (Bass Rock, Copinsay, Inchmickery and The Lamb) of the nine colonies for which regular monitoring data were available (Figure 3.3.2). It was possible to detect changes of 30% or more at two colonies – the Isle of May (part of the Forth Islands SPA) and Mingulay (part of the Mingulay and Berneray SPA). It was also possible to detect changes of 50% at the Fidra and Craigleith colonies (both part of the Forth Islands SPA) and the Fowlsheugh RSPB colony (part of the Fowlsheugh SPA).

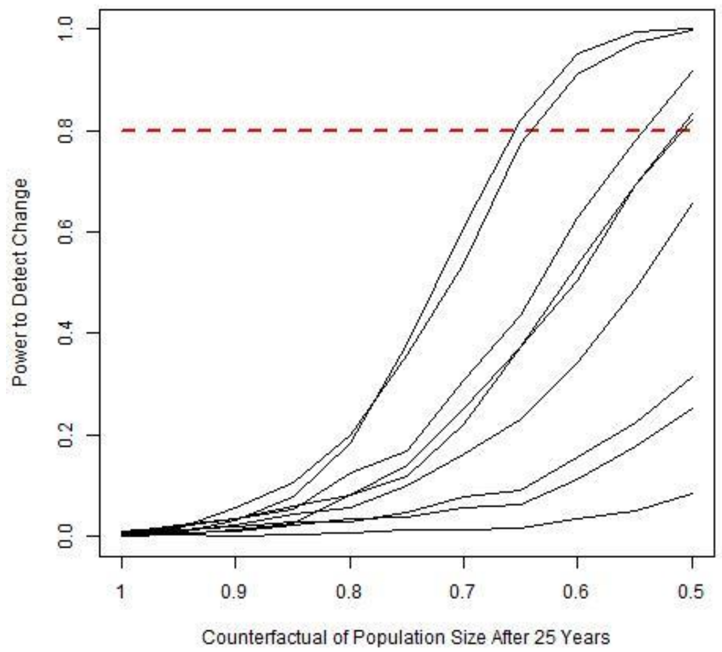

Figure 3.3.2 Power to detect a reduction of 0-50% in population size at nine Northern Fulmar breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm. Red line indicates power to detect change of 0.8, conventionally taken to be the point at which there is sufficient power to detect a significant change.

At present, it would not be possible to detect an impact resulting in a wind farm reducing population size by 25% or less at monitored Northern Fulmar colonies (Figure 3.3.3). An impact likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 17 years at Mingulay, 19 years at the Isle of May, 23 years at Fidra, and 25 years at Craigleith and Fowlsheugh. In order to detect these impacts, monitoring would have to be carried out annually at Craigleith, Fidra and Fowlsheugh, biennially on the Isle of May and at least every five years at Mingulay (Figure 3.3.3).

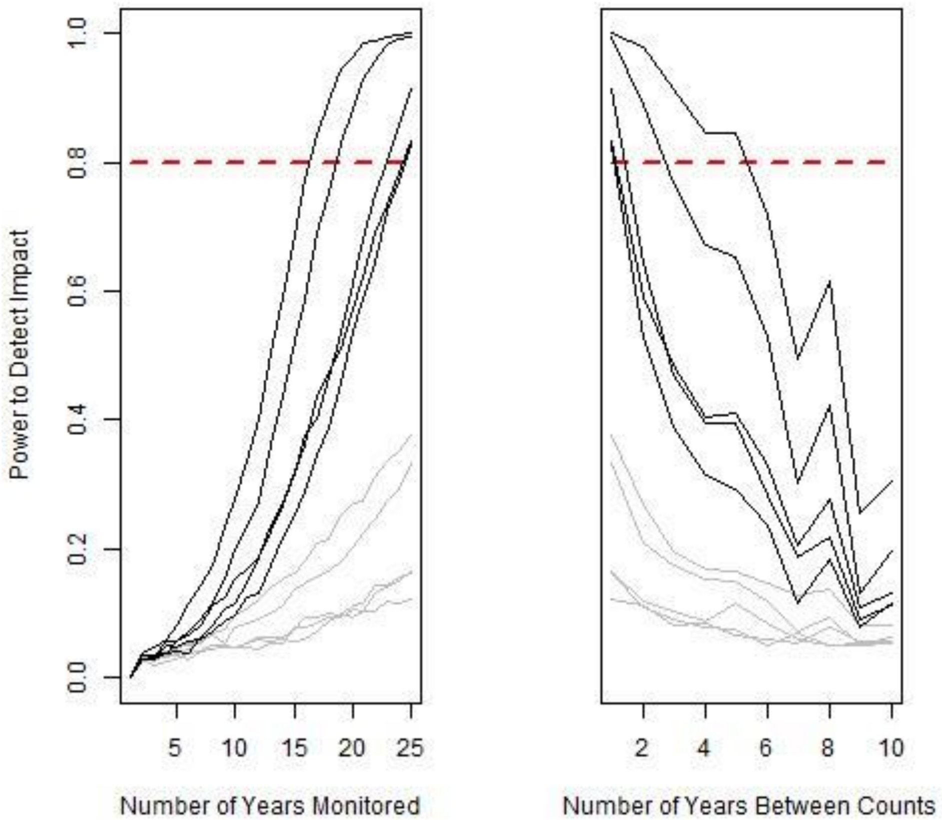

Figure 3.3.3 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Northern Fulmar breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right). Red line indicates power to detect change of 0.8, conventionally taken to be the point at which there is sufficient power to detect a significant change.

Breeding success

We identified 11 SPAs for which offshore wind farm assessments concluded that LSEs on Northern Fulmar populations could not be ruled out by the HRA screening process and, within which, there has been regular monitoring of the species' breeding success through the SMP (Figure 3.3.4) – Fair Isle SPA, Flamborough Head and Filey Coast SPA, Foula SPA, Handa SPA, Hermaness, Valla Field and Saxa Vord SPA, Noss SPA, Hoy SPA, Rousay SPA, Sumburgh Head SPA, Troup, Pennan and Lion's Heads SPA and the Forth Islands SPA. Since 2010, monitoring programmes for Northern Fulmar breeding success have also been established within the Mingulay and Berneray, Copinsay and West Westray SPAs. Provided these are continued, in the future it may be possible to use the data collected at these sites to monitor changes in breeding success associated with offshore wind farms.

Across these colonies, the coefficient of variation in annual breeding success was relatively low (mean CV at each colony 24.8, SD 6.6), indicating relatively consistent levels of breeding success between years. This is reflected in a high power to detect changes in breeding success that may be associated with the impacts arising from offshore wind farms. The minimum change in breeding success that could be detected given current monitoring efforts varies between colonies (Figure 3.3.5). At Flamborough Head and Filey Coast SPA, there is sufficient power to detect a decline in breeding success of no less than 15%; within the Fair Isle SPA, a decline of no less than 25% could be detected. Within the Sumburgh Head and Handa Island SPAs, declines in breeding success of no less than 30% could be detected. Declines in breeding success of no less than 35% could be detected at the Foula, Hermaness, Saxa Vord and Valla Field, and Troup, Pennan and Lion's Heads SPAs. Declines in breeding success of no less than 45% could be detected within the Noss and Forth Islands SPAs and a decline of no less than 50% could be detected within the Rousay SPA.

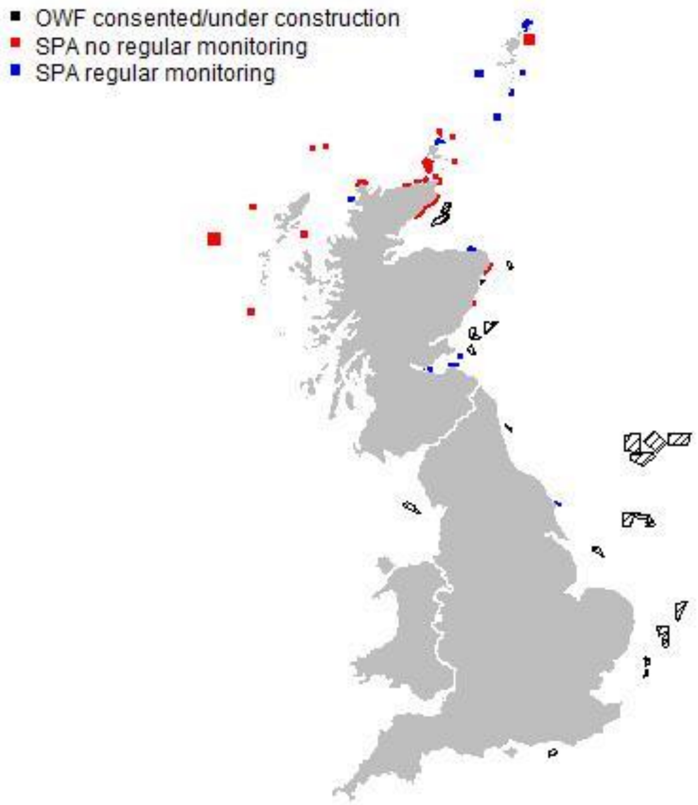

Figure 3.3.4 Special Protection Areas for Northern Fulmar at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

Estimates of both breeding success and population size were only available from the Isle of May, within the Forth Islands SPA. The minimum change in breeding success that could be detected at the Isle of May was a 45% decrease (Figure 3.3.5). Over 25 years, this would lead to a counterfactual of population size of 0.66. In comparison, through monitoring abundance, the minimum change that could be detected would result in a counterfactual of population of 0.55 after 25 years (Figure 3.3.6). If changes in Northern Fulmar populations are driven by declines in breeding success, monitoring annual breeding success at a colony level is more likely to pick up the impacts associated with the wind farm than monitoring abundance.

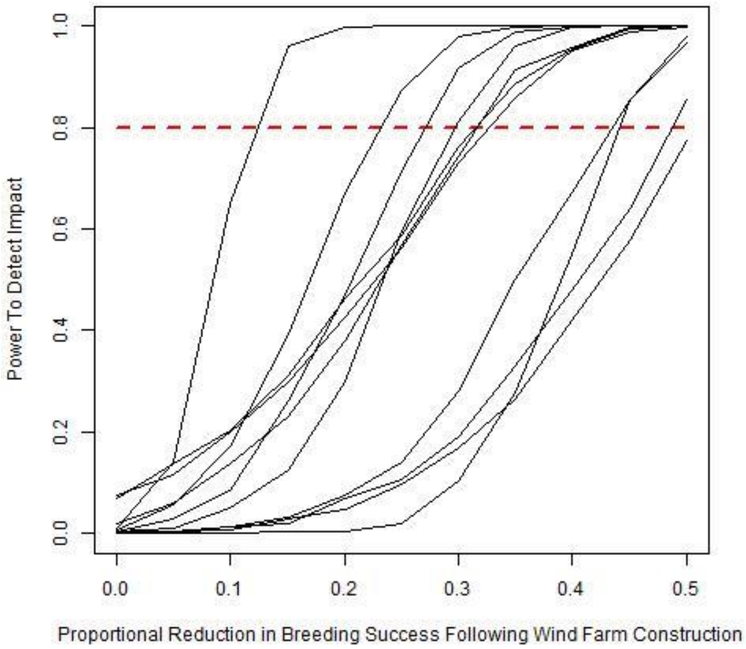

Figure 3.3.5 Power to detect a reduction in breeding success in Northern Fulmars of 0-50% in colonies within SPAs where a LSE associated with an offshore wind farm could not be ruled out by the HRA screening process. Red line indicates power to detect change of 0.8, conventionally taken to be the point at which there is sufficient power to detect a significant change.

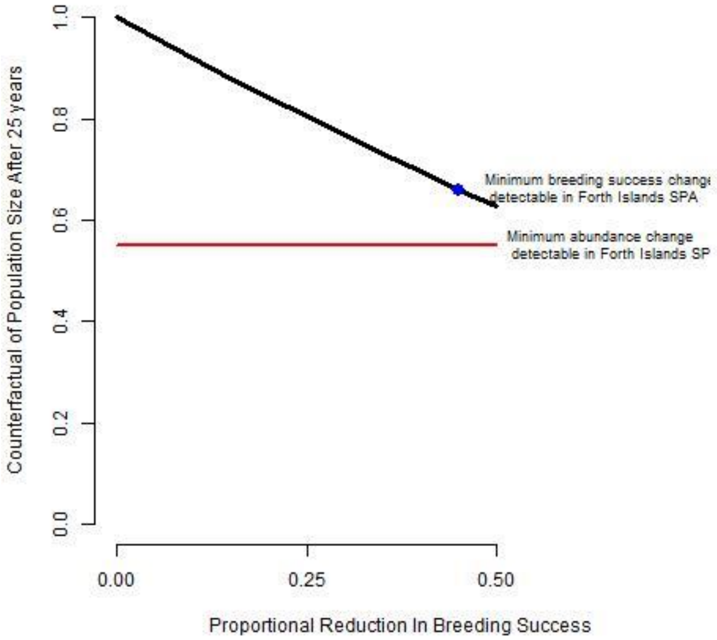

Figure 3.3.6 Population-level consequences of a reduction in breeding success in Northern Fulmar as assessed using the counterfactual of population size after 25 years.

Adult survival

We were unable to identify any SPAs for which offshore wind farm assessments concluded that LSEs on Northern Fulmar populations could not be ruled out by the HRA screening process, at which the species' adult survival is monitored on an annual basis (Figure 3.3.7). If a capture-mark-recapture scheme were set up at a Northern Fulmar colony following the recommendations set out in Horswill *et al.* (2018) it is likely that a reduction in the adult survival rate of 0.010 could be detected in the 10 years following wind farm construction (Figure 3.3.8). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Northern Fulmars, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.025 could be detected over the 10 years following wind farm construction.

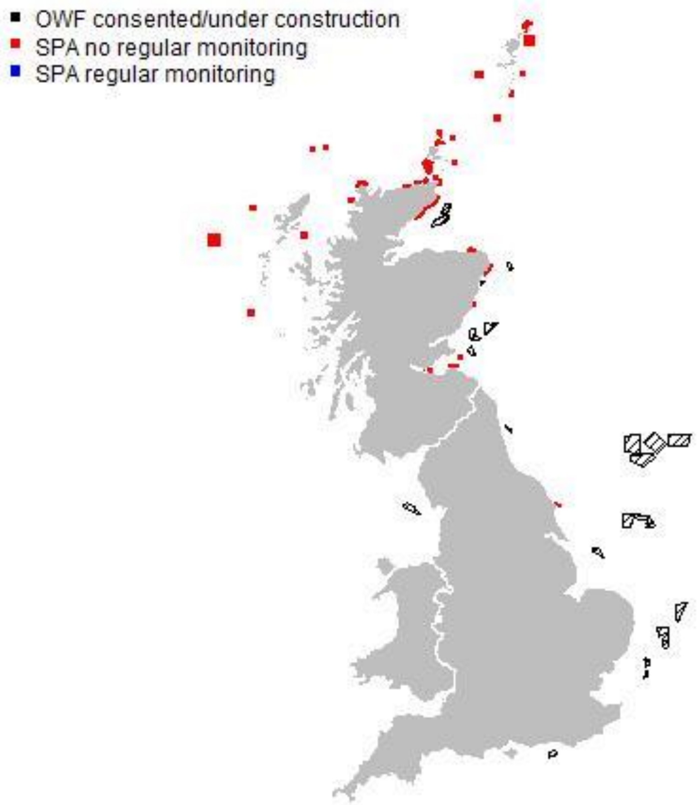

Figure 3.3.7 Special Protection Areas for Northern Fulmar at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for five or 10 years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone. Even with a capture-mark-recapture study established just three years prior to wind farm construction, it would be possible to detect a change in adult survival likely to result in a population-level impact equivalent to that which could be detected through the long-term monitoring of abundance at the Mingulay and Berneray SPA, and greater than that which could be detected through the monitoring of abundance at the Fowlsheugh or Forth Islands SPAs (Figure 3.3.9). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any change in abundance.

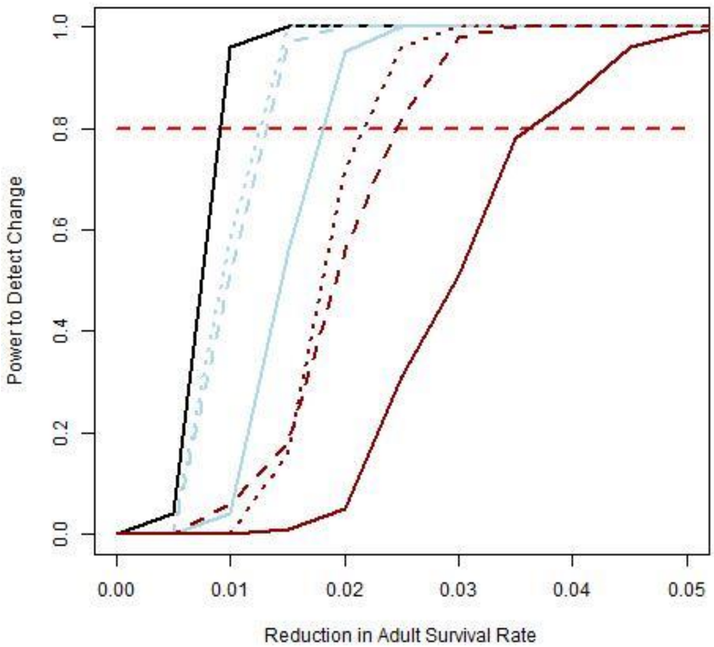

Figure 3.3.8 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Northern Fulmar. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, dashed lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8. Red line indicates power to detect change of 0.8, conventionally taken to be the point at which there is sufficient power to detect a significant change.

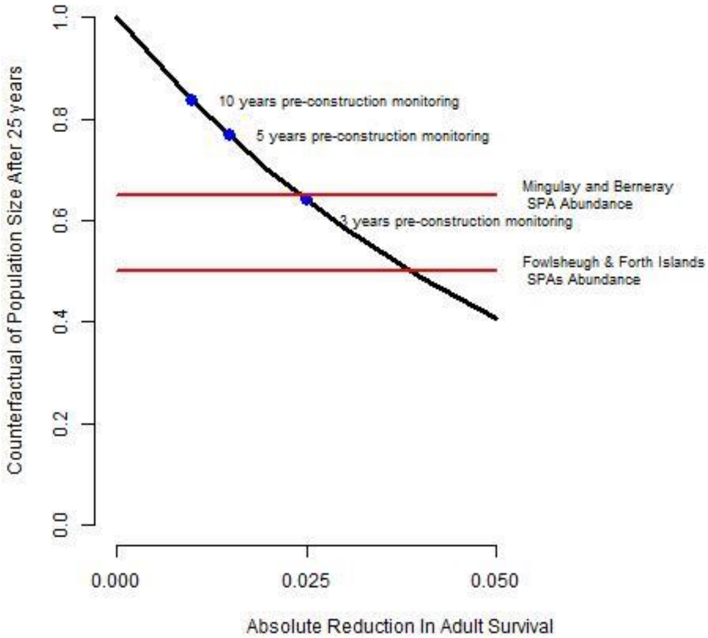

Figure 3.3.9 Population-level consequences of a reduction in adult survival in Northern Fulmar as assessed using the counterfactual of population size after 25 years.

3.3.2 Manx Shearwater *Puffinus puffinus*

Abundance

LSEs on populations of Manx Shearwaters could not be ruled out by the HRA screening process at four colonies within three SPAs – Copeland Islands SPA (not shown), Skomer and Skokholm within the Skomer, Skokholm and Seas off Pembrokeshire SPA and Bardsey within the Aberdaron Coast and Bardsey Island SPA - in the assessment of one offshore wind farm project (Figure 3.3.10, Table A3). We did not identify any colonies within these SPAs at which regular abundance monitoring may enable baseline population estimates to be established against which any impacts arising as a consequence of the development of offshore wind farms could be assessed.

Figure 3.3.10 Special Protection Areas for Manx Shearwater at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

Breeding success

Regular monitoring of the breeding success of Manx shearwaters has been carried out at two sites – Bardsey within the Aberdaron Coast and Bardsey Island SPA and at Skomer within the Skomer, Skokholm and Seas off Pembrokeshire SPA – where offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.11). We also noted that regular monitoring of Manx Shearwater breeding success has recently been established at Skokholm (three years data so far), which will improve our ability to detect impacts of offshore wind farms on the breeding success of this species within the Skomer, Skokholm and Seas off Pembrokeshire SPA.

Figure 3.3.11 Special Protection Areas for Manx Shearwater at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

The coefficients of variation in annual breeding success at Skomer and Bardsey are relatively low (22.9 and 8.6 respectively), indicating relatively consistent levels of breeding success. This is reflected in a high power to detect changes in breeding success that may arise as a result of the effects associated with offshore wind farms (Figure 3.3.12). At Bardsey, a decline in breeding success of no less than 20% could be detected given existing levels of monitoring and at Skomer a decline of no less than 35% in breeding success could be detected. Changes in breeding success of these magnitudes would equate to counterfactuals of population size after 25 years of 0.84 and 0.73 respectively (Figure 3.3.13).

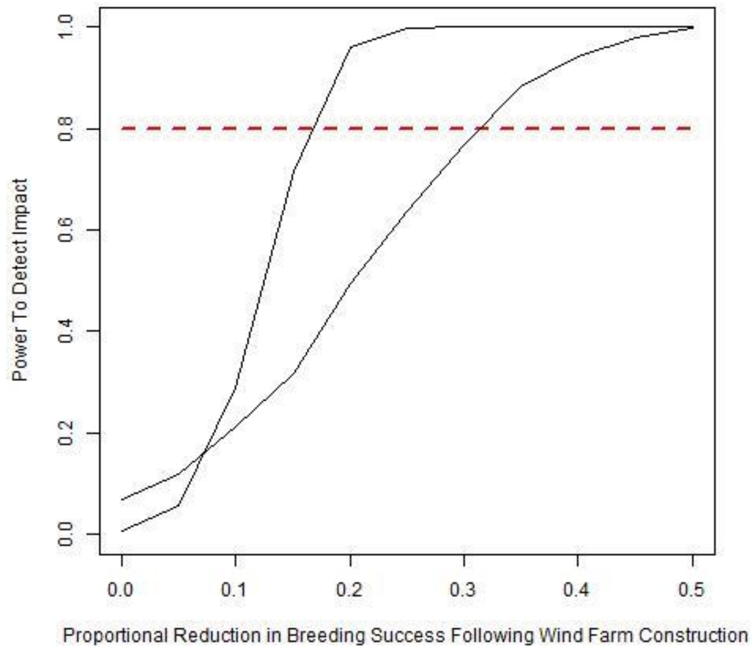

Figure 3.3.12 Power to detect a reduction in breeding success in Manx Shearwaters of 0-50% in colonies within SPAs where a LSE associated with an offshore wind farm could not be ruled out by the HRA screening process.

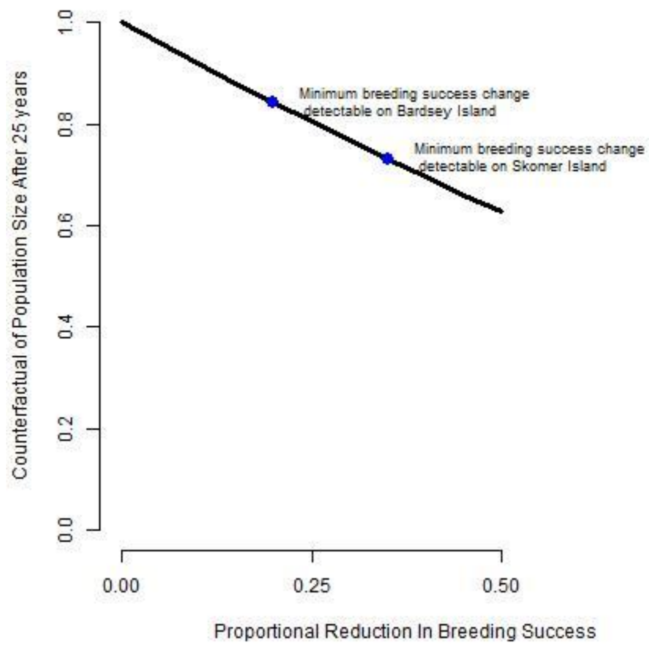

Figure 3.3.13 Population-level consequences of a reduction in breeding success in Manx Shearwater as assessed using the counterfactual of population size after 25 years.

Adult survival

Adult survival data for Manx shearwater are currently collected on Skomer, within the Skomer, Skokholm and Seas off Pembrokeshire SPA, one of the SMP key sites (Figure 3.3.14). If a capture-mark-recapture scheme were set up in either the Skokholm or Bardsey colonies following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.15). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Manx Shearwaters, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of 0.02 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

Figure 3.3.14 Special Protection Areas for Manx Shearwater at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If it were only possible to collect three years of mark-recapture data prior to the construction of a wind farm, it would be possible to detect a reduction in adult survival of no less than 0.030. This would lead to a counterfactual of population size of 0.62 over 25 years (Figure 3.3.16). If it were possible to collect five years of data, it would be possible to detect a reduction in adult survival of no less than 0.020, leading to a counterfactual of population size of 0.73. For 10 years of pre-construction mark-recapture data, it would be possible to detect a change in adult survival of 0.015, leading to a counterfactual of population size after 25 years of 0.79.

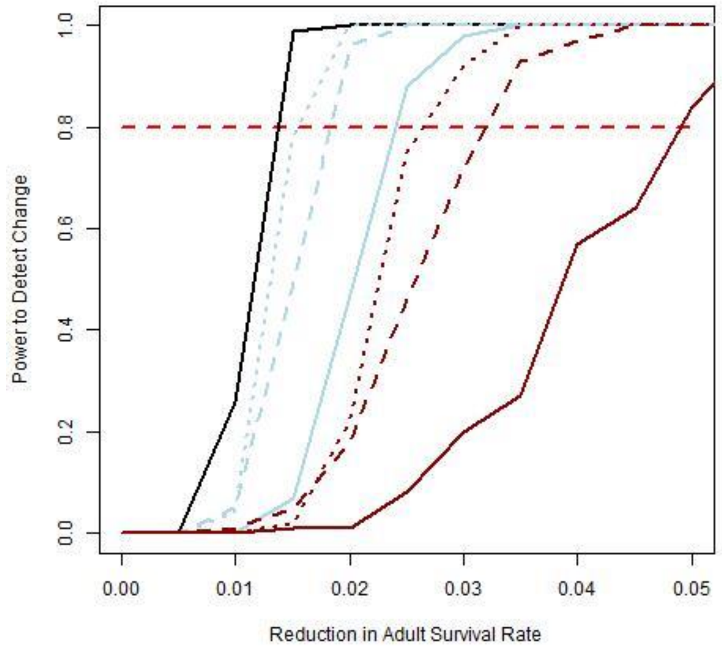

Figure 3.3.15 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Manx Shearwater. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, dashed lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

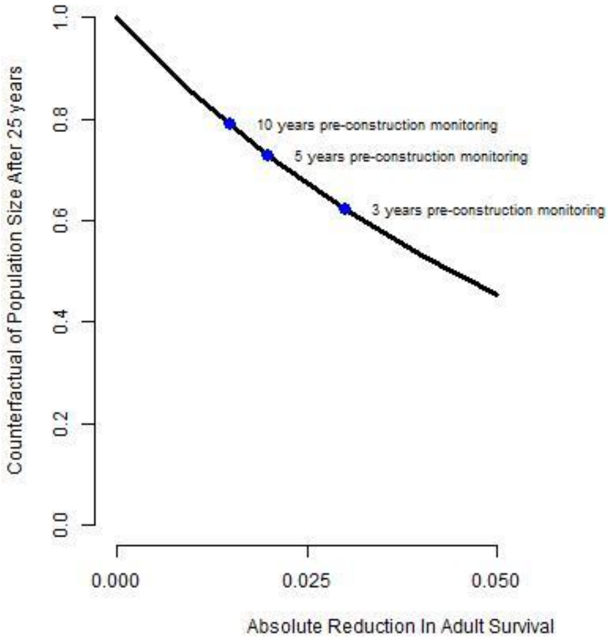

Figure 3.3.16 Population-level consequences of a reduction in adult in Manx Shearwater as assessed using the counterfactual of population size after 25 years.

3.3.3 Northern Gannet *Morus bassanus*

Abundance

Assessments for 12 offshore wind farm projects concluded that LSEs on populations of Northern Gannets could not be ruled out by the HRA screening process at a total of eight SPAs (Figure 3.3.17, Table A3). We identified two SPAs – Fair Isle SPA and Flamborough Head and Filey Coast SPA – where there is regular monitoring of Northern Gannet abundance. At present, there is sufficient power to detect a reduction in gannet population size of 25% over 25 years at Flamborough Head and Filey Coast SPA (Figure 3.3.18) and 35% over 25 years at Fair Isle SPA as a result of the effects associated with an offshore wind farm.

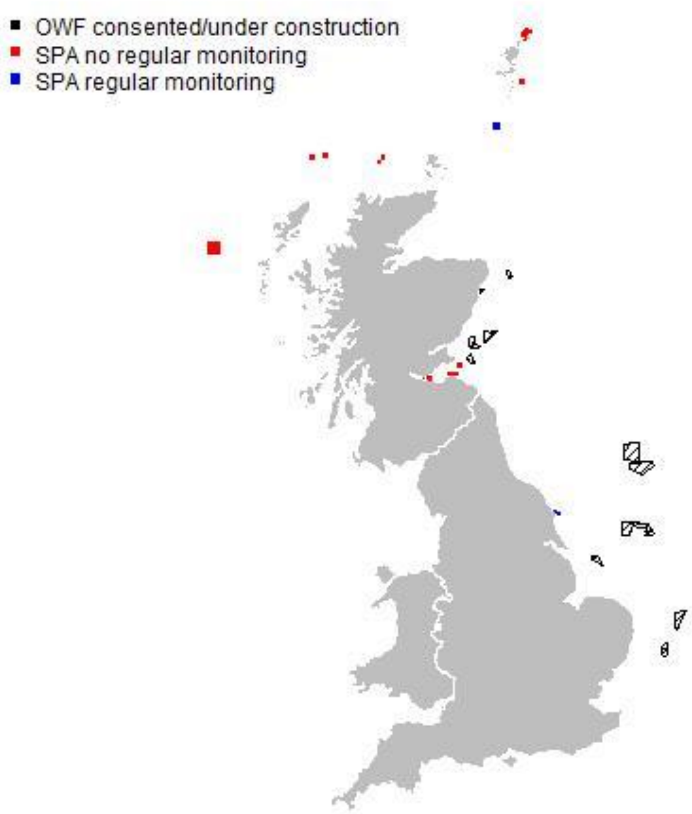

Figure 3.3.17 Special Protection Areas for Northern Gannet at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

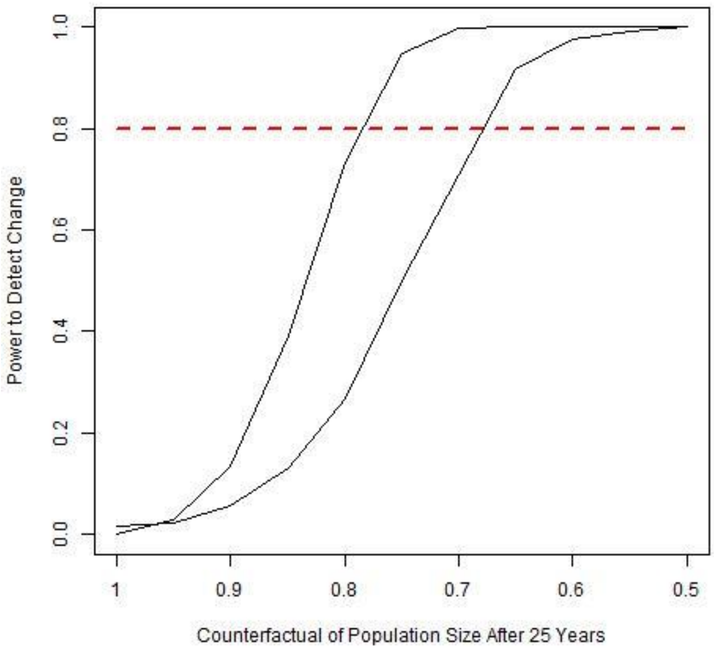

Figure 3.3.18 Power to detect a reduction of 0-50% in population size at Northern Gannet breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

With present monitoring, an effect likely to result in a 25% reduction in population size over 25 years could be detected within 21 years at the Flamborough Head and Filey Coast SPA, but not at the Fair Isle SPA (Figure 3.3.19). An effect likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 12 years at the Flamborough Head and Filey Coast SPA and within 17 years at the Fair Isle SPA. In order to detect a 25% reduction in abundance over 25 years at the Flamborough Head and Filey Coast SPA, monitoring would have to be carried out on an annual basis (Figure 3.3.19), but a 50% reduction could be detected assuming monitoring was carried out every seven years. At the Fair Isle SPA, a 50% reduction in population size could be detected assuming monitoring was carried out every three years.

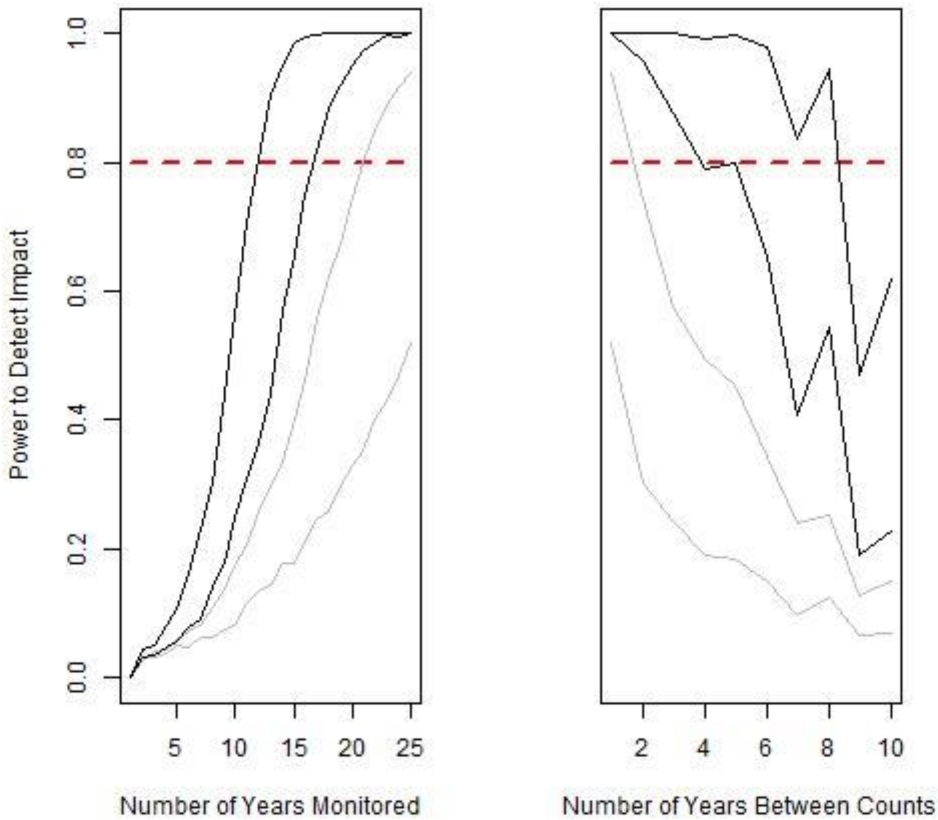

Figure 3.3.19 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Northern Gannet breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

__Breeding success__

The breeding success of Northern Gannets has been monitored by the SMP at four of the eight SPAs for which offshore wind farm assessments concluded that LSEs for the species could not be ruled out by the HRA screening process – Fair Isle, Flamborough Head and Filey Coast, Noss and Hermaness, Saxa Vord and Valla Field (Figure 3.3.20). The magnitude of changes which could be detected given current monitoring efforts varied between SPAs. At present, the minimum decrease in breeding success associated with a wind farm effect that could be detected at the Flamborough Head and Filey Coast SPA is 10%. At the Fair Isle and Noss SPAs, the minimum decrease in breeding success associated with a wind farm effect that could be detected is 15% and, at the Hermaness, Saxa Vord and Valla Field SPA it is 25% (Figure 3.3.21).

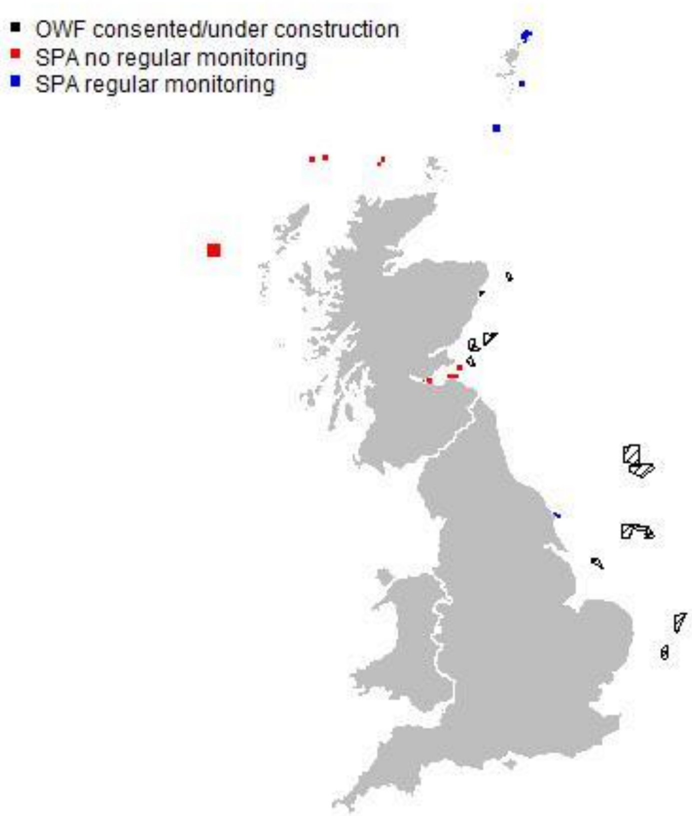

Figure 3.3.20 Special Protection Areas for Northern Gannet at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

At present, there is no regular monitoring of abundance at two of the breeding colony SPAs at which breeding success is monitored – the Noss SPA and Hermaness, Saxa Vord and Valla Field SPA. However, at both Fair Isle and Flamborough Head and Filey Coast, it is possible to detect changes in breeding success that would result in more subtle changes at a population-level than could be detected by monitoring abundance only (Figure 3.3.22).

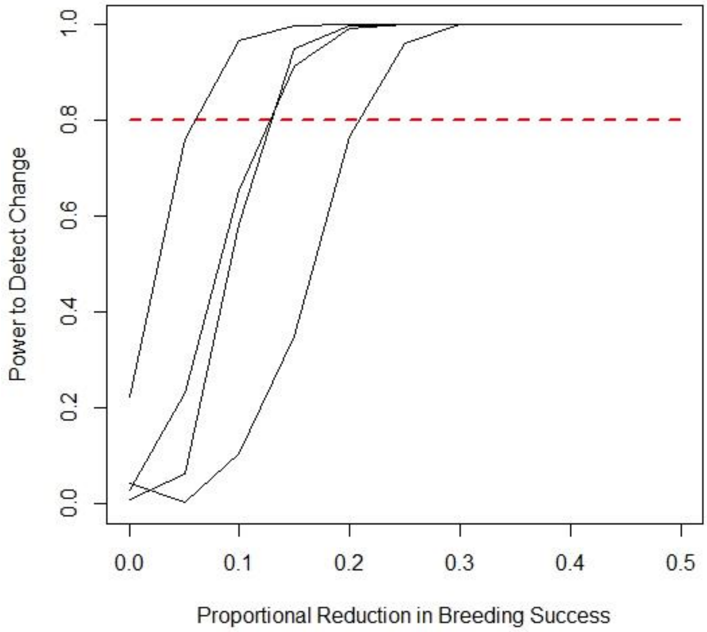

Figure 3.3.21 Power to detect a reduction in breeding success associated with an offshore wind farm in Northern Gannets at Fair Isle, Noss, Flamborough Head and Filey Coast and Hermaness, Saxa Vord and Valla Field SPAs.

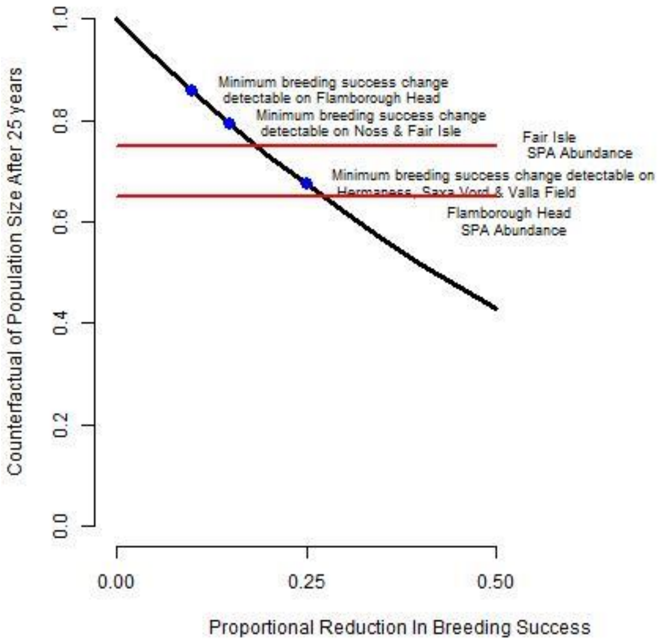

Figure 3.3.22 Counterfactual of population size after 25 years for Northern Gannet relative to changes in annual breeding success associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Red lines indicate changes in abundance that could be detected given current monitoring effort.

Adult survival

Monitoring of Northern Gannet adult survival rates in relation to the effects associated with offshore wind farms has been identified as a research priority (Furness & Wanless, 2014). However, we were unable to identify any SPAs for which offshore wind farm assessments concluded that LSEs on Northern Gannet populations could not be ruled out at a population level by HRA screening prior to AA, at which the species' adult survival has been monitored on an annual basis (Figure 3.3.23). If a capture-mark-recapture scheme were set up at a Northern Gannet colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.010 could be detected in the 10 years following wind farm construction (Figure 3.3.24). However, recognising that it is unlikely that 10 years pre-construction survival data could be collected for Northern Gannets, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.020 could be detected over the 10 years following wind farm construction.

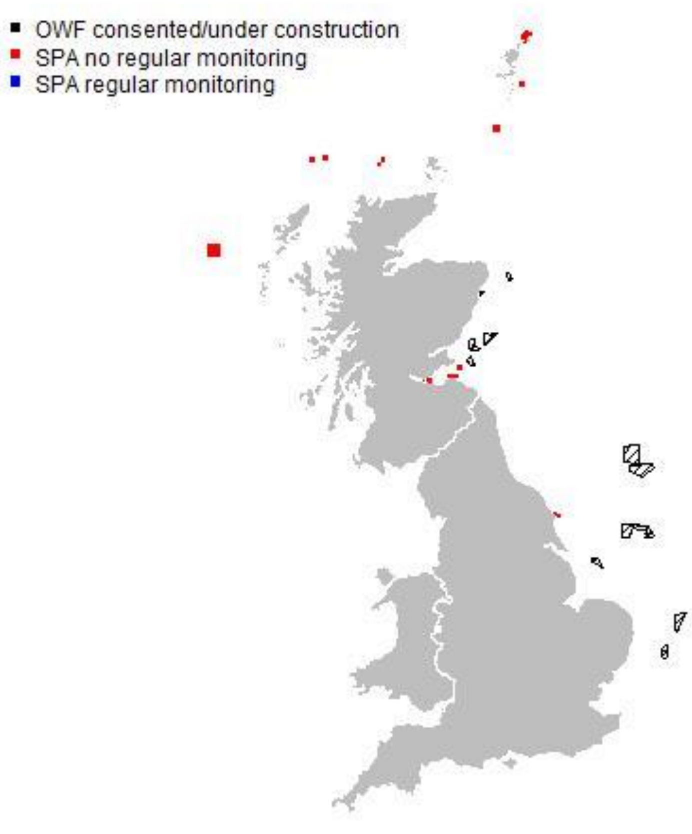

Figure 3.3.23 Special Protection Areas for Northern Gannet at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for five or 10 years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone. Even with a capture-mark-recapture study established three years prior to wind farm construction, it would be possible to detect a change in adult survival likely to result in a population-level impact greater than that which could be detected through the monitoring of abundance at Flamborough Head and Filey Coast (Figure 3.3.25). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any impact on abundance.

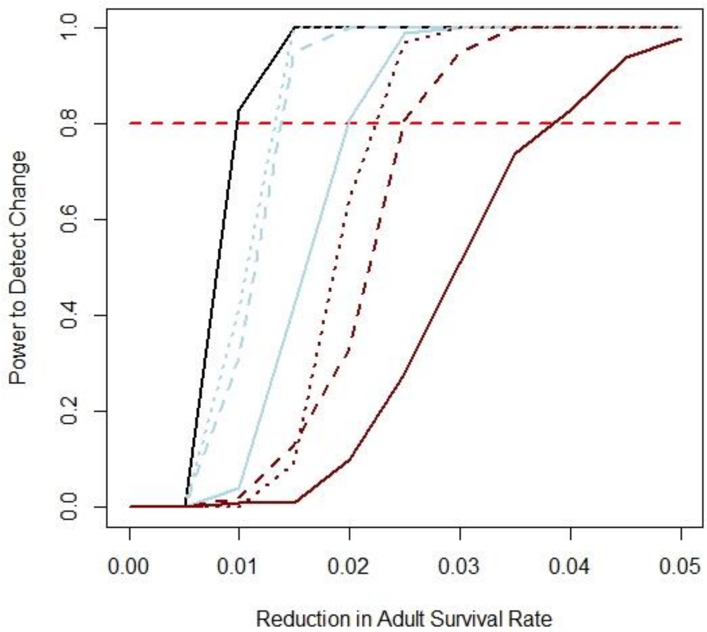

Figure 3.3.24 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Northern Gannet. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

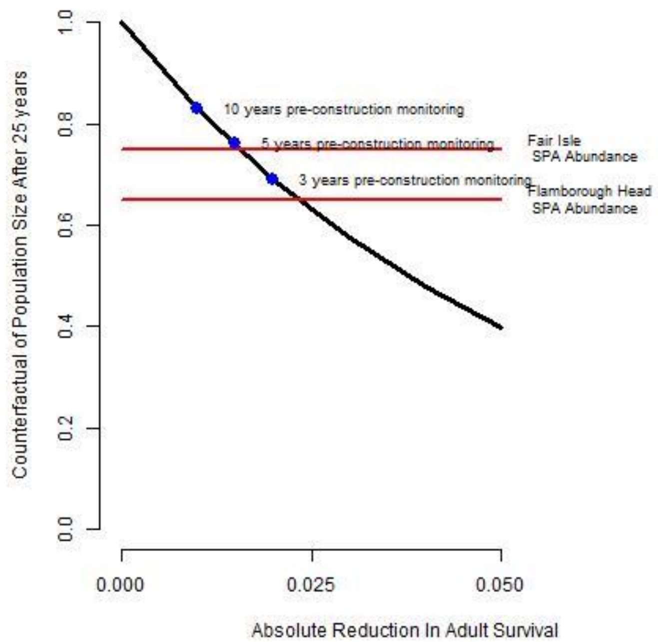

Figure 3.3.25 Counterfactual of population size after 25 years for Northern Gannet relative to changes in adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection. Red lines indicate changes in abundance that could be detected given current monitoring effort.

3.3.4 European Shag *Phalacrocorax aristotelis*

Abundance

LSEs on European Shag populations, arising as a result of the effects associated with the Aberdeen Bay Offshore Wind Farm, could not be ruled out at a population level by HRA screening prior to AA within the Buchan Ness to Collieston SPA (Figure 3.3.26, Table A3). Monitoring of abundance within the SPA has been intermittent with counts undertaken during seven years between 1986 and 2013. This is reflected in limited power to detect any meaningful population changes that might be associated with offshore wind farms within this SPA (Figure 3.3.27). The power to detect a reduction in population size arising as a result of the effect of the offshore wind farm of 50% over 25 years was only 0.37.

Figure 3.3.26 Special Protection Areas for European Shag at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

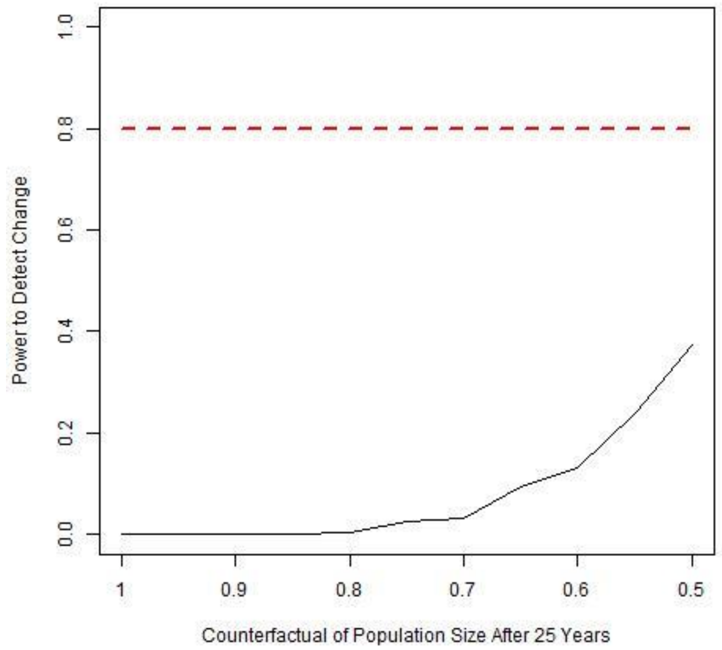

Figure 3.3.27 Power to detect a reduction of 0-50% in population size at European Shag breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Breeding success

Breeding success in European Shags is not currently monitored within the Buchan Ness to Collieston Coast SPA (Figure 3.3.28). Consequently, it would not be possible to detect any changes in breeding success associated with an offshore wind farm at this site.

Figure 3.3.28 Special Protection Areas for European Shag at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

Adult survival

We were unable to identify any colonies within the Buchan Ness to Collieston SPA at which adult survival in European Shags is monitored on an annual basis (Figure 3.3.29). If a capture-mark-recapture scheme were set up at a European Shag colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.30). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for European Shag, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.020 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.035 could be detected over the 10 years following wind farm construction.

Figure 3.3.29 Special Protection Areas for European Shag at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If mark-recapture data were available for the 10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.015 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.77 (Figure 3.3.31). If only five years data could be collected prior to the construction of a wind farm, then a reduction in the adult survival rate of no less than 0.020, resulting in a counterfactual of population size of 0.70, could be detected, assuming the number of birds marked per year were increased to 400 and the recapture rate was increased to 0.8. However, if only three years pre-construction data could be collected, the minimum change that could be detected is a reduction in adult survival of 0.035, which would result in a counterfactual of population size after 25 years of 0.54.

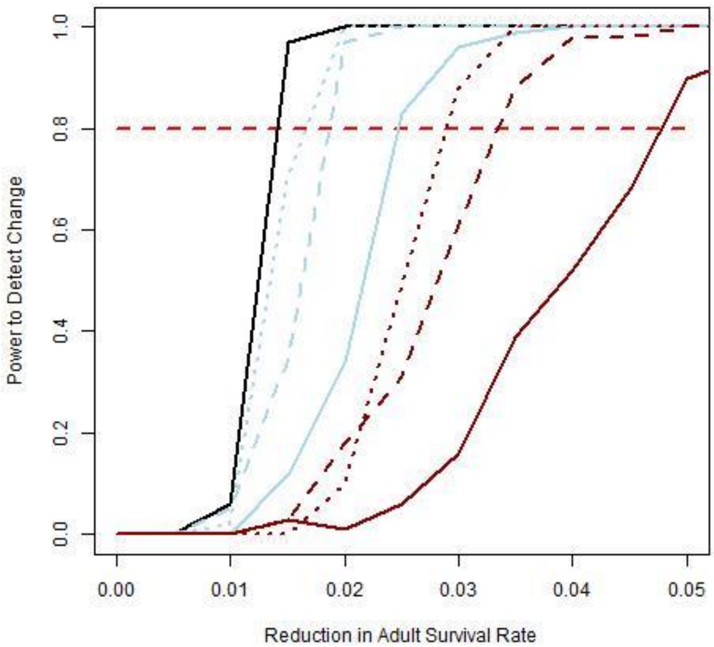

Figure 3.3.30 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for European Shag. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

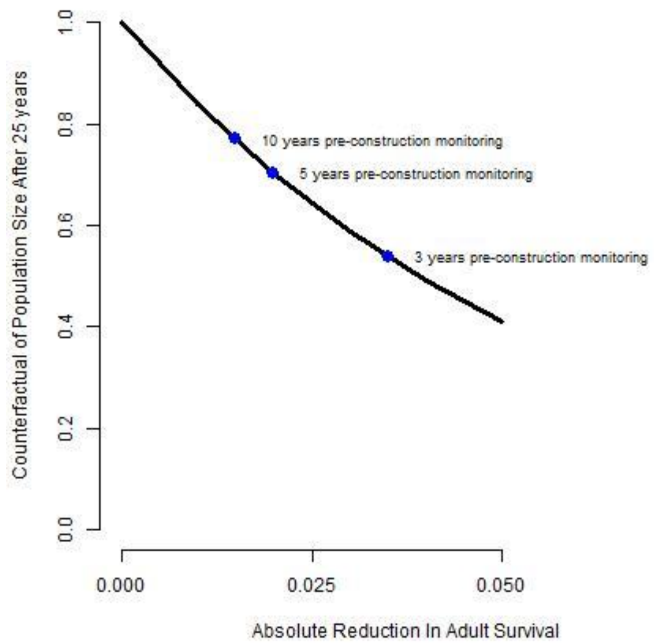

Figure 3.3.31 Counterfactual of population size after 25 years for European Shags relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.5 Black-legged Kittiwake *Rissa tridactyla*

Abundance

Assessments for 19 offshore wind farm projects concluded that LSEs on populations of Black-legged Kittiwake could not be ruled out at a population level by HRA screening prior to AA at a total of 16 SPAs (Figure 3.3.32, Table A3). We identified seven colonies within three of these SPAs at which there has been regular monitoring of black-legged kittiwake abundance by the SMP – St. Abbs Head SPA, Farne Islands SPA and the Isle of May, Bass Rock, Craigleith, Fidra and the Lamb (within the Forth Islands SPA).

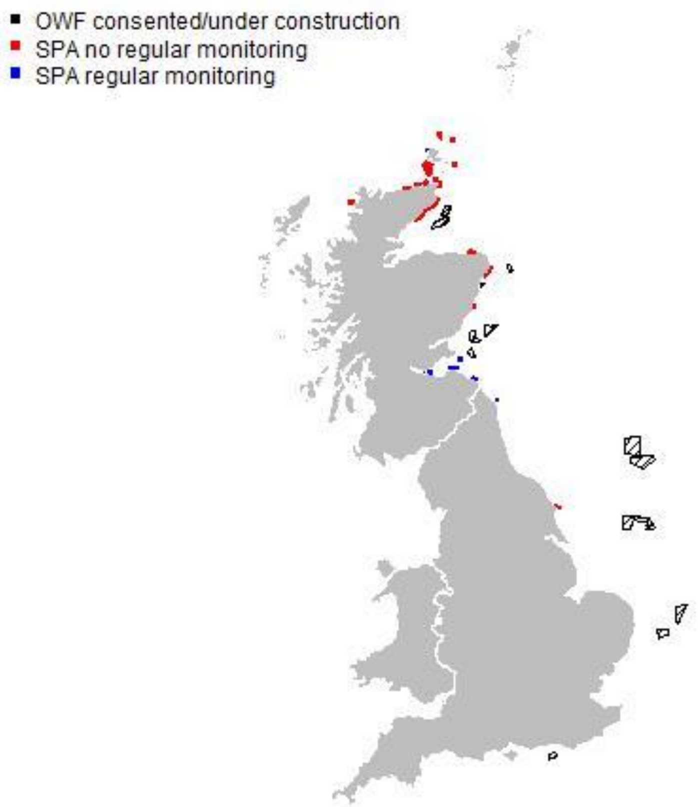

Figure 3.3.32 Special Protection Areas for Black-legged Kittiwake at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

Based on existing monitoring, it would be possible to detect reductions in population size attributable to offshore wind farms of no less than 40% on the Farne Islands SPA and at Fidra and the Isle of May within the Forth Islands SPA (Figure 3.3.33). This change would be equivalent to a counterfactual of population size of 0.6 25 years after wind farm construction. It would also be possible to detect reductions in population size of no less than 50% (equivalent to a counterfactual of 0.5) at the St Abb's Head SPA and at Craigleith in the Forth Islands SPA.

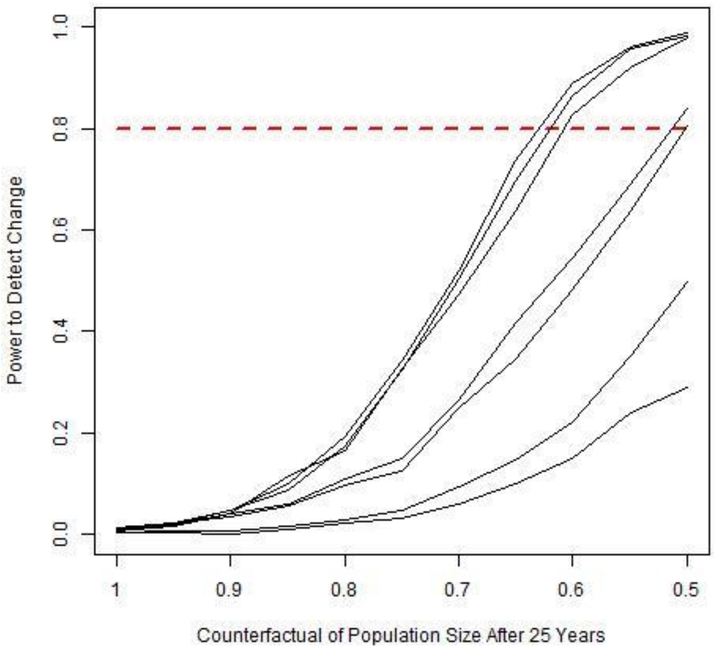

Figure 3.3.33 Power to detect a reduction of 0-50% in population size at Black-Legged Kittiwake breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

At present, it would not be possible to detect an impact resulting in a wind farm reducing population size by 25% or less at monitored Black-legged Kittiwake colonies (Figure 3.3.34). An effect likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 20 years at the Farne Islands and on the Isle of May and, within 21 years at Fidra. In order to detect these impacts, monitoring would have to be carried out annually at Fidra and biennially on the Isle of May and the Farne Islands (Figure 3.3.34).

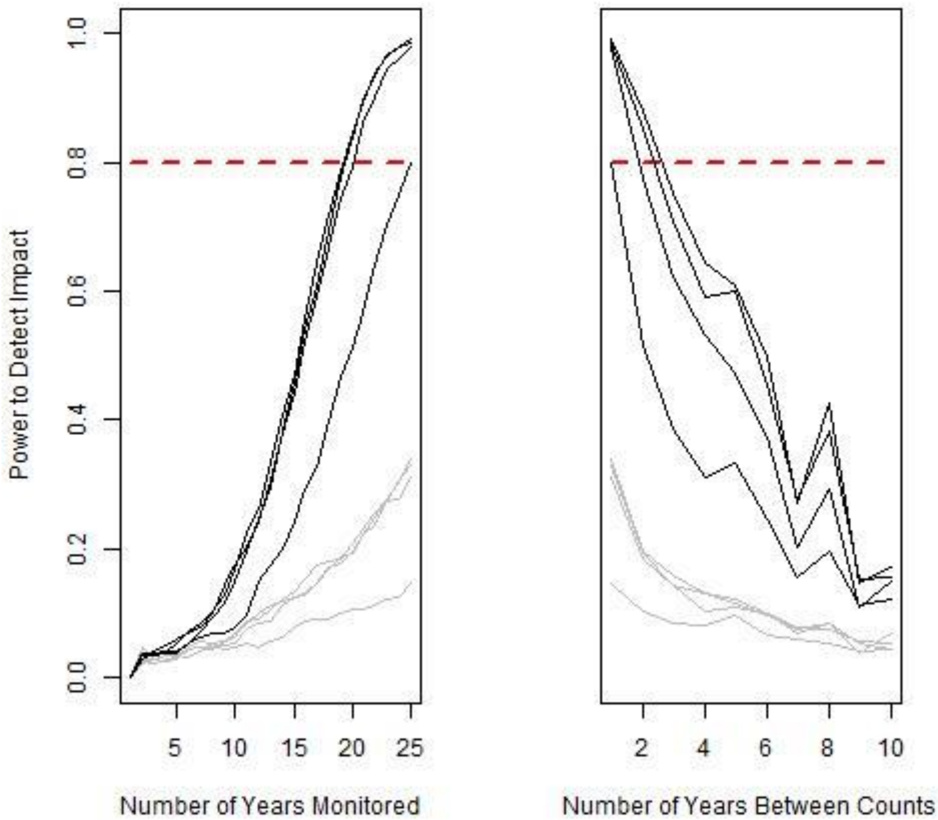

Figure 3.3.34 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Black-legged Kittiwake breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

Breeding success

We identified eight SPAs for which offshore wind farm assessments concluded that LSEs on black-legged kittiwake populations could not be ruled out by the HRA screening process and, at which there is regular monitoring of the species' breeding success – Buchan Ness to Collieston SPA, Farne Islands SPA, Flamborough Head and Filey Coast SPA, Fowlsheugh SPA, Hand Island SPA, Marwick Head SPA, St Abb's Head SPA and the Isle of May within the Forth Islands SPA (Figure 3.3.35). Monitoring has also recently been established (in 2010) at the Troup, Pennan and Lion's Heads SPA and at West Westray SPA but, at present, the time-series is not long enough to reliably assess the power to detect effects of offshore wind farms on kittiwake breeding success. As additional years are added to these data, this is likely to change.

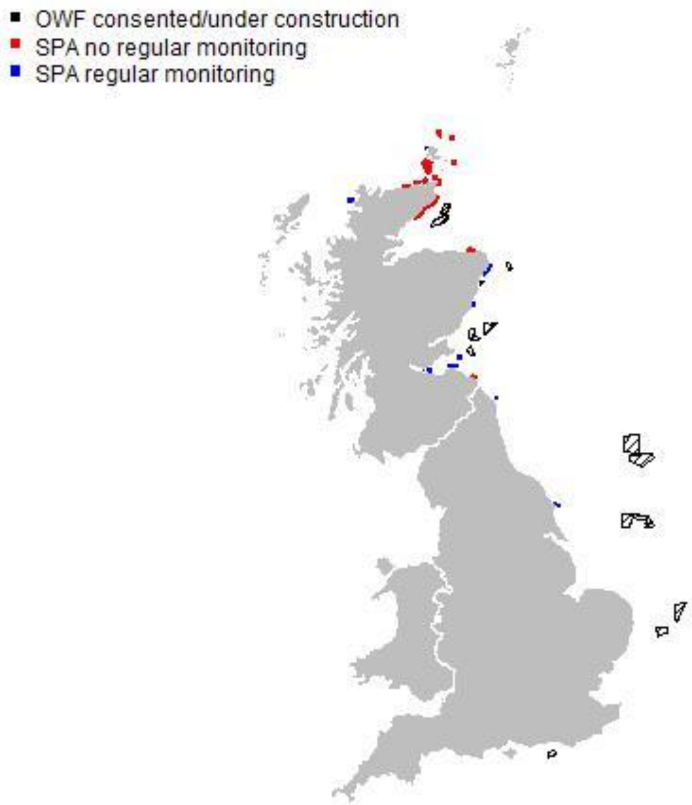

Figure 3.3.35 Special Protection Areas for Black-legged Kittiwake at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

The coefficient of variation for annual breeding success at each colony is relatively high (mean 48.5 SD 8.7). This is reflected in the fact that power to detect meaningful changes in breeding success at each colony is relatively low. At present, the only colonies at which it would be possible to detect a reduction of no less than 50% in response to the effects associated with an offshore wind farm if the Farne Islands SPA (Figure 3.3.36). The maximum level of power with which a reduction in breeding success of 50% could be detected at Flamborough Head and Filey Coast SPA was only 0.67. At Fowlsheugh SPA, Handa Island SPA and the Isle of May within the Forth Islands SPA, power to detect a reduction in breeding success of 50% was between 0.2 and 0.4. At the remaining colonies, the power to detect a 50% decline in breeding success was less than 0.4. This may reflect breeding failures at these colonies in recent years, particularly in relation to Marwick Head.

The only colony at which it was possible to detect changes in abundance and breeding success in black-legged kittiwake was the Farne Islands SPA. In this instance, the minimum population-level impact that could be detected by monitoring breeding success is greater than the minimum impact that could be detected through monitoring abundance (Figure 3.3.37). The coefficient of variation of breeding success at the Farne Islands is 38.4, potentially signifying strong annual variation in breeding success, making it more difficult to detect meaningful changes.

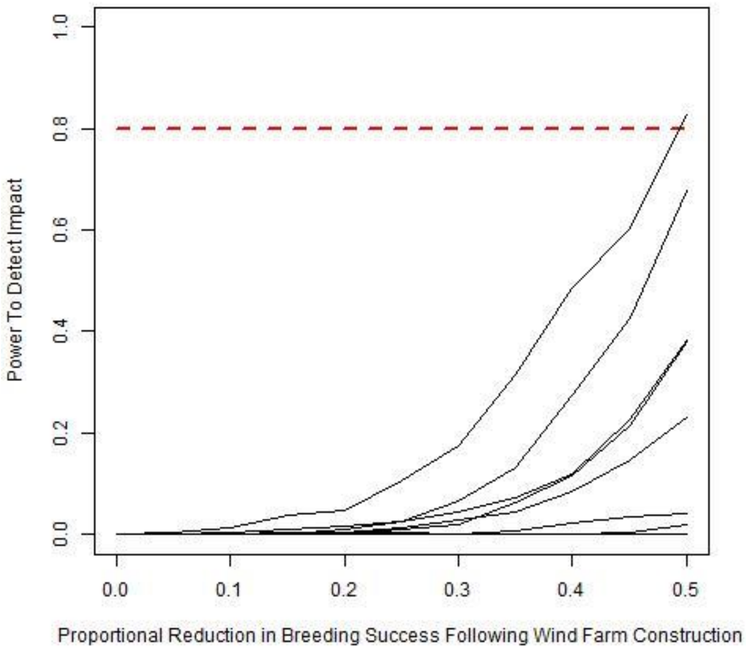

Figure 3.3.36 Power to detect a reduction of 0-50% in population size at Black-legged Kittiwake breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

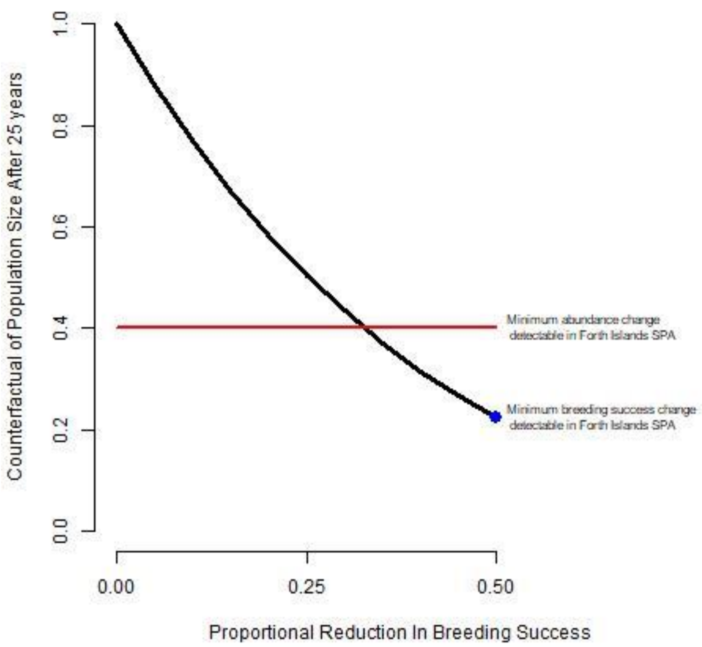

Figure 3.3.37 Counterfactual of population size after 25 years for Black-legged Kittiwake relative to changes in annual breeding success associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Red lines indicate changes in abundance that could be detected given current monitoring effort.

Adult survival

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Black-legged Kittiwake populations could not be ruled out by the HRA screening process, the adult survival of the species is only monitored on the Isle of May, within the Forth Islands SPA, one of the SMP key sites (Figure 3.3.38). If capture-mark-recapture schemes were set up at other Black-legged Kittiwake colonies following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.39). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Lesser Black-backed Gull, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.020 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

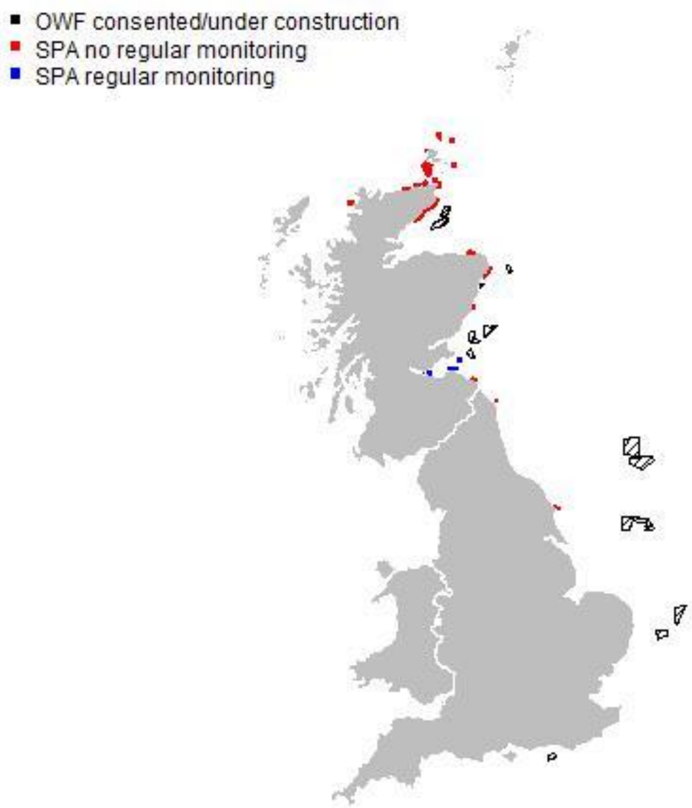

Figure 3.3.38 Special Protection Areas for Black-legged Kittiwake at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for five or 10 years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone. Even with a capture-mark-recapture study established three years prior to wind farm construction, it would be possible to detect a change in adult survival likely to result in a population-level impact equivalent to that which could be detected through the monitoring of abundance within the Farne Islands and Forth Islands SPAs, and greater than that which could be detected through the monitoring of abundance at St Abb's Head SPA (Figure 3.3.40). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any change in abundance.

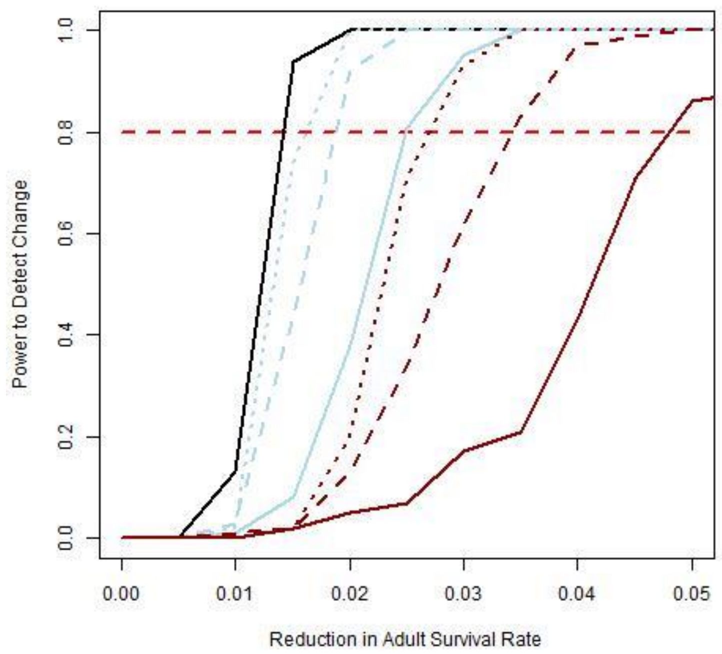

Figure 3.3.39 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Black-legged Kittiwake. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

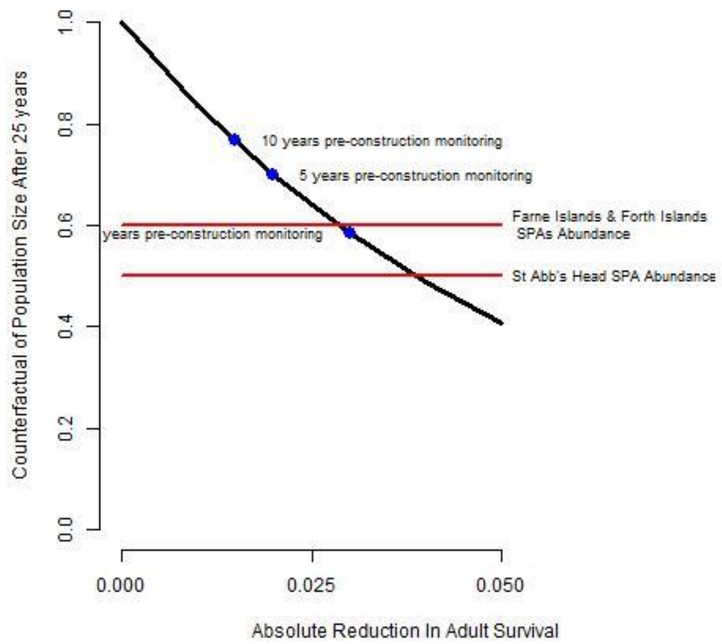

Figure 3.3.40 Counterfactual of population size after 25 years for Black-legged Kittiwake relative to changes in adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection. Red lines indicate changes in abundance that could be detected given current monitoring effort.

3.3.6 Lesser Black-backed Gull *Larus fuscus*

Abundance

Assessments for 10 offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA on populations of Lesser Black-backed Gulls at a total of five SPAs (Figure 3.3.41, Table A3).

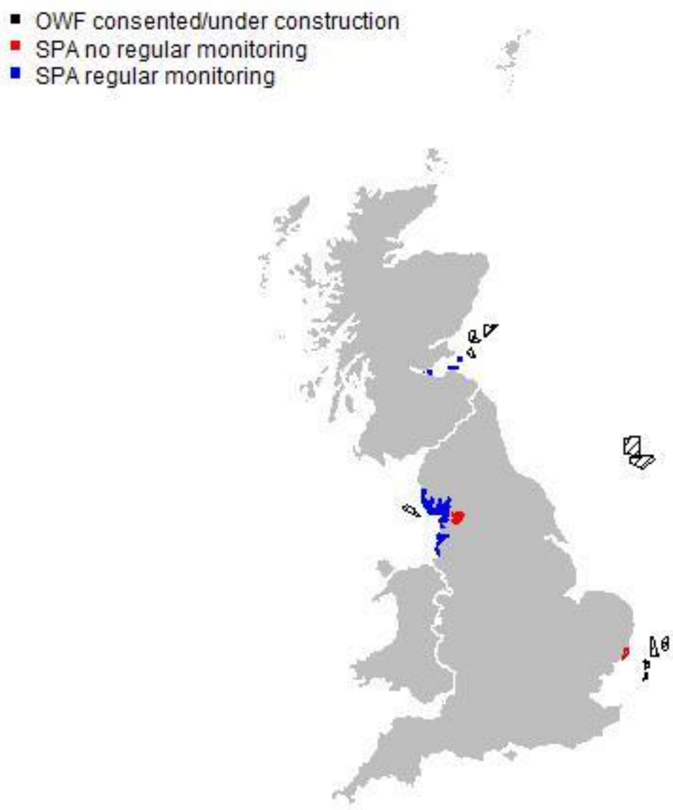

Figure 3.3.41 Special Protection Areas for Lesser Black-backed Gulls at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

Regular monitoring of the abundance of Lesser Black-backed Gulls has been undertaken by the SMP at eight colonies within four of these SPAs – The Ribble Estuary SPA, South Walney within the Morecambe Bay & Duddon Estuary SPA and Bass Rock, Fidra, Inchmickery and the Isle of May within the Forth Islands SPA and Havergate Islands and Orford Ness within the Alde-Ore Estuary SPA. On the Isle of May, it is possible to detect an effect leading to the population being 45% lower than it would have been in the absence of the wind farm (Figure 3.3.42). It was not possible to detect a wind farm effect at the remaining colonies although, the power to detect an impact leading to the population being 50% of what it would have been in the absence of a wind farm was 0.77 at South Walney.

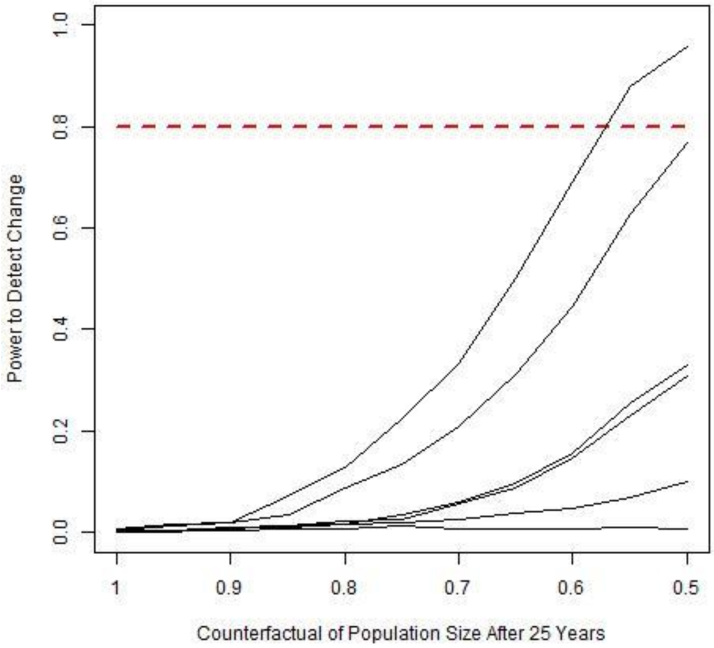

Figure 3.3.42 Power to detect a reduction of 0-50% in population size at Lesser Black-backed Gull breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

At present, it would not be possible to detect an impact resulting in a wind farm reducing population size by 25% or less at monitored Lesser Black-backed Gull colonies (Figure 3.3.43). An effect likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 22 on the Isle of May. However, to detect such an impact, monitoring would have to be carried out on an annual basis (Figure 3.3.43)

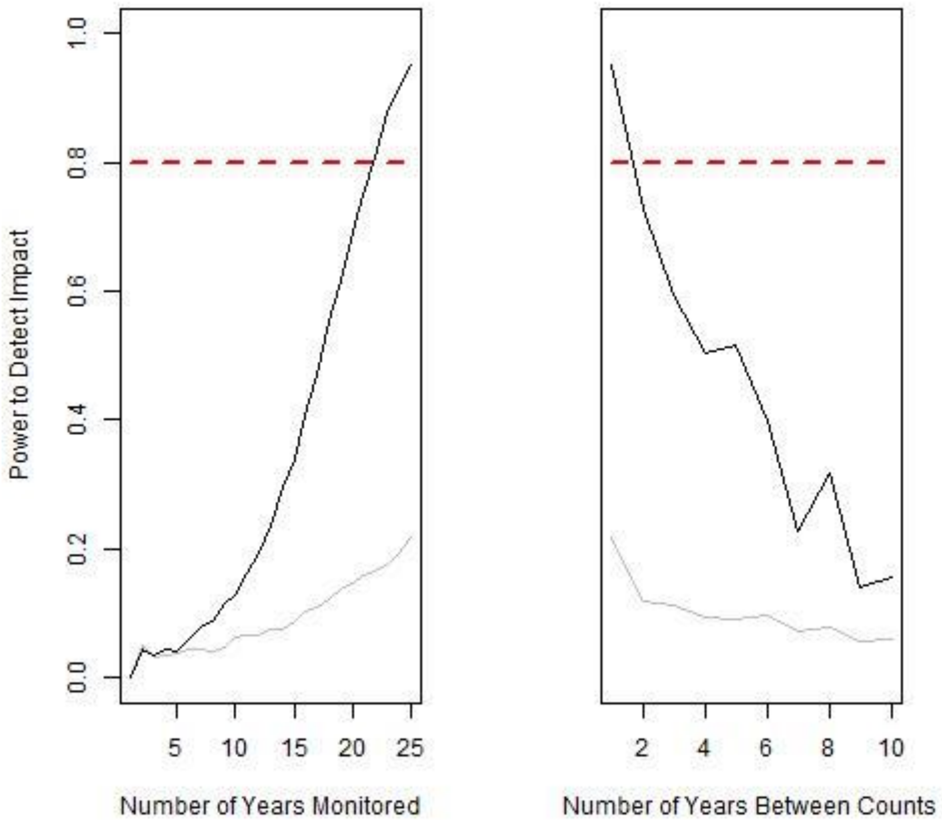

Figure 3.3.43 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Lesser Black-backed Gull breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

Breeding success

Breeding success in Lesser Black-backed Gulls is currently monitored at two sites where offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process – on the Isle of May within the Forth Islands SPA and Havergate Island within the Alde-Ore Estuary SPA (Figure 3.3.44). Monitoring of breeding success on Havergate Island has been running since 2009 and there is wide variability between years (coefficient of variation for mean breed success 104) (Figure 3.3.45). This is reflected in a low power to detect the changes that may be associated with offshore wind farms. In contrast, monitoring on the Isle of May is better established with a lower coefficient of variation for annual breeding success (34), reflected in greater power to detect changes in breeding success. At present, a reduction of 50% or more in breeding success, as a result of effects associated with a wind farm, could be detected on the Isle of May. There has also been some intermittent (1-2 years) monitoring of Lesser Black-backed gulls at South Walney, within the Morecambe Bay & Duddon Estuary SPA, and within the Ribble Estuary SPA. Were these monitoring programmes to become established, it may be possible to detect effects associated with offshore wind farms on lesser black-backed gull breeding success at these sites.

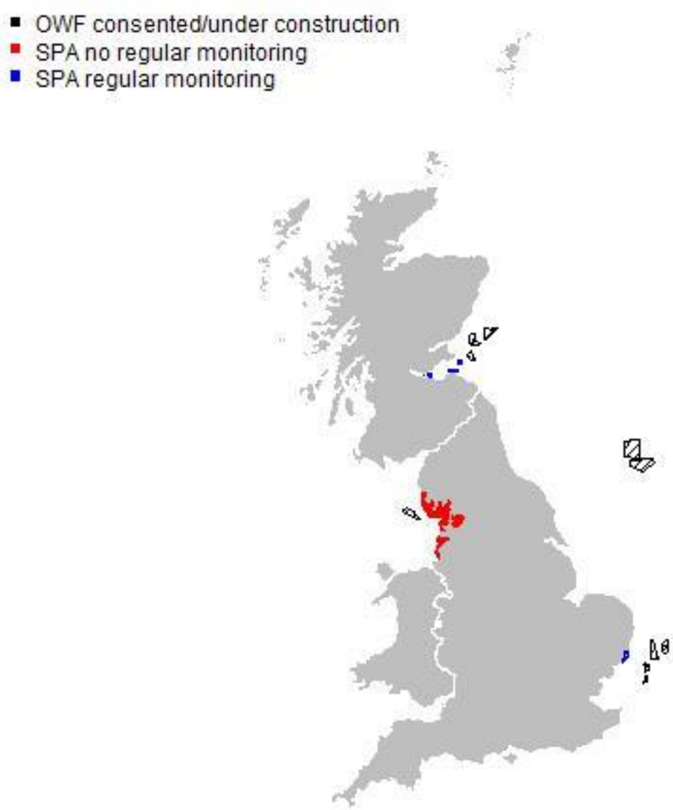

Figure 3.3.44 Special Protection Areas for Lesser Black-backed Gulls at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

The only colony at which it was possible to detect changes in abundance and breeding success in Lesser Black-backed Gull was the Forth Islands SPA. In this instance, the minimum population-level impact that could be detected by monitoring breeding success is greater than the minimum impact that could be detected through monitoring abundance (Figure 3.3.46). The coefficient of variation of breeding success at the Farne Islands is 34, potentially signifying strong annual variation in breeding success, making it more difficult to detect meaningful changes.

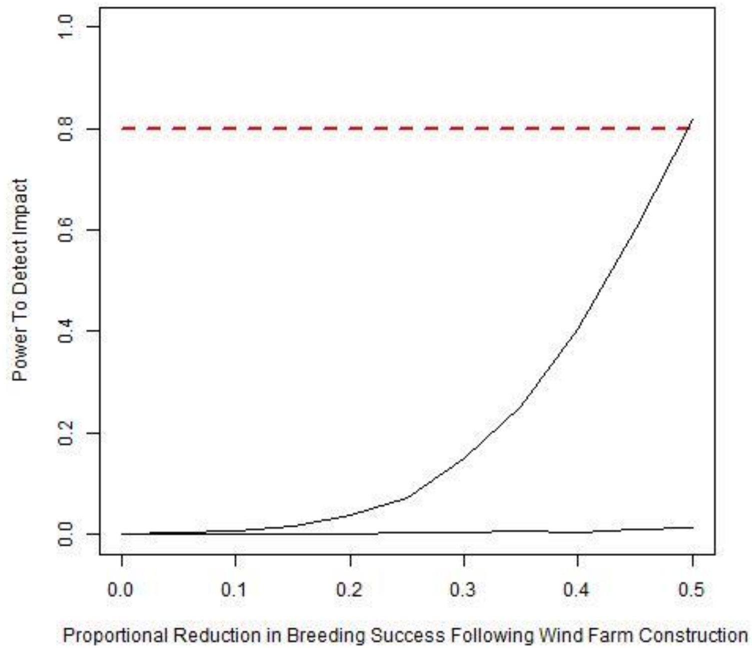

Figure 3.3.45 Power to detect a reduction of 0-50% in population size at Lesser Black-backed Gull breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

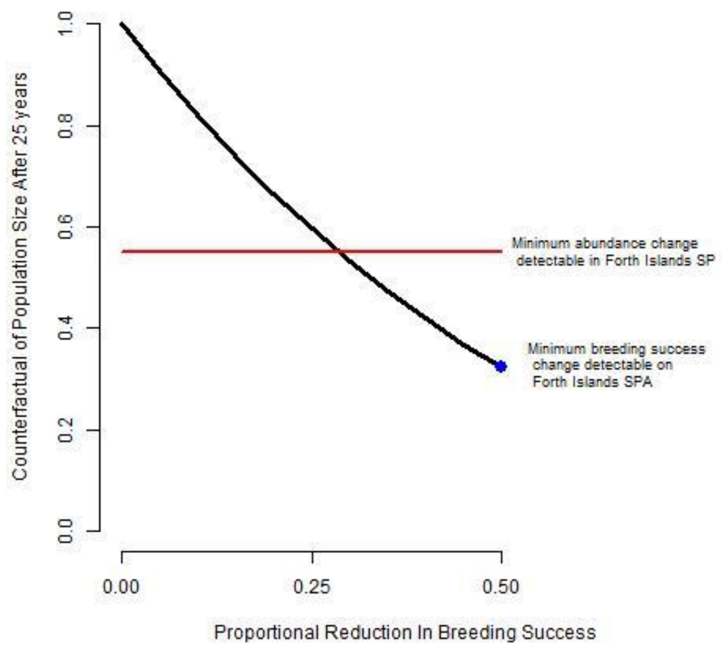

Figure 3.3.46 Counterfactual of population size after 25 years for Lesser Black-backed Gull relative to changes in annual breeding success associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Red lines indicate changes in abundance that could be detected given current monitoring effort.

Adult survival

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Lesser Black-backed Gull populations could not be ruled out by the HRA screening process, the adult survival of the species is only monitored at Orfordness, within the Alde-Ore Estuary SPA (Figure 3.3.47). If capture-mark-recapture schemes were set up at other Lesser Black-Backed Gull colonies following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.48). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Lesser Black-backed Gull, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.025 could be detected over the 10 years following wind farm construction.

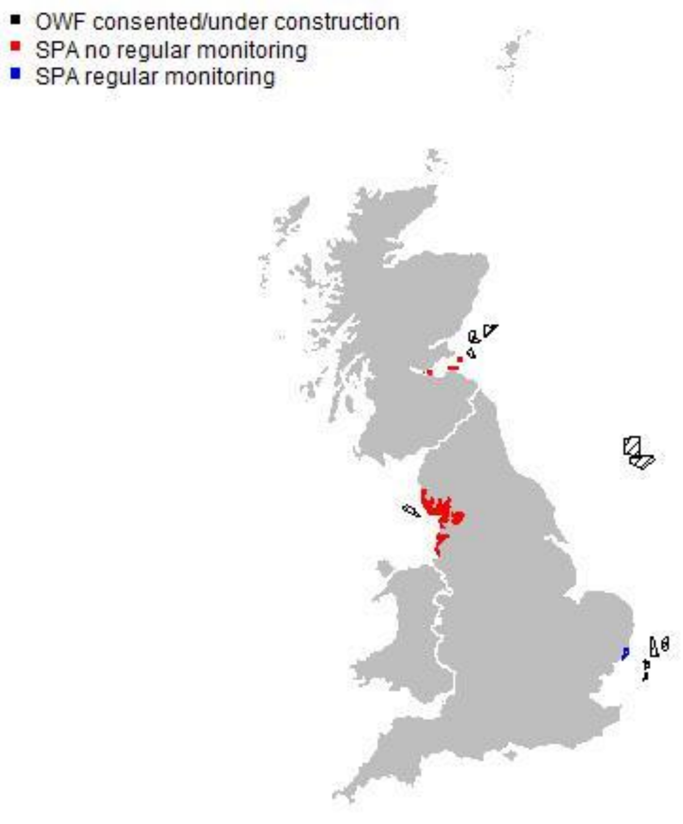

Figure 3.3.47 Special Protection Areas for Lesser Black-backed Gull at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for at least three years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone (Figure 3.3.49). However, if it were not possible to collect at least 10 years data prior to construction of the wind farm, requirements in terms of both sample size and recapture rate would increase (Figure 3.3.49).

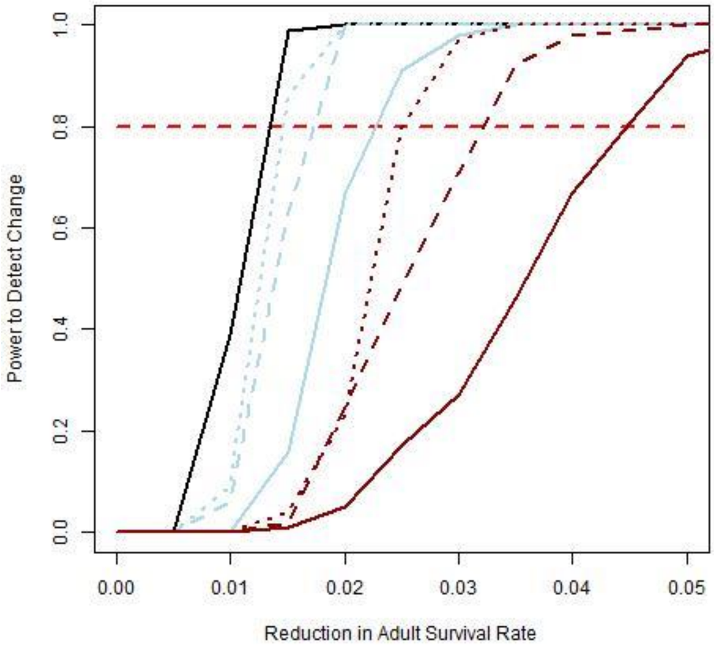

Figure 3.3.48 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Lesser Black-backed Gull. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

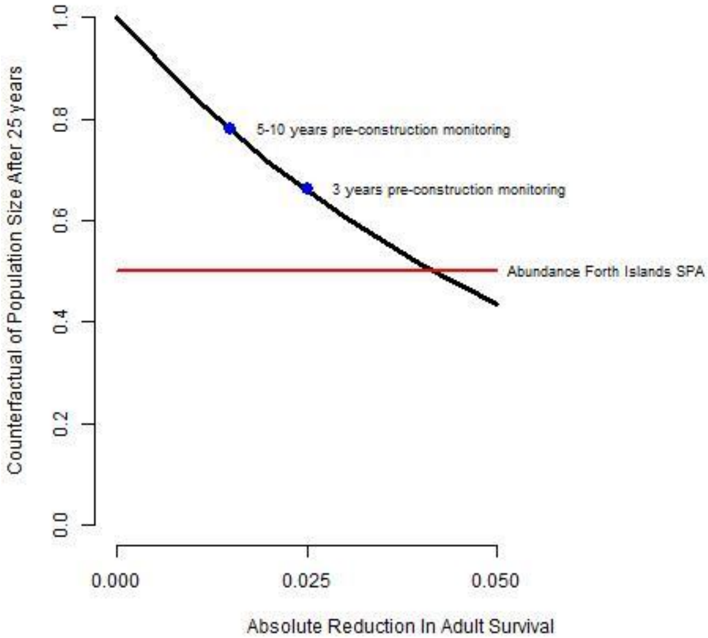

Figure 3.3.49 Counterfactual of population size after 25 years for Lesser Black-backed Gull relative to changes in adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection. Red lines indicate changes in abundance that could be detected given current monitoring effort.

3.3.7 Herring Gull *Larus argentatus*

Abundance

Assessments for 11 offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Herring Gull at a total of nine SPAs (Figure 3.3.50, Table A3).

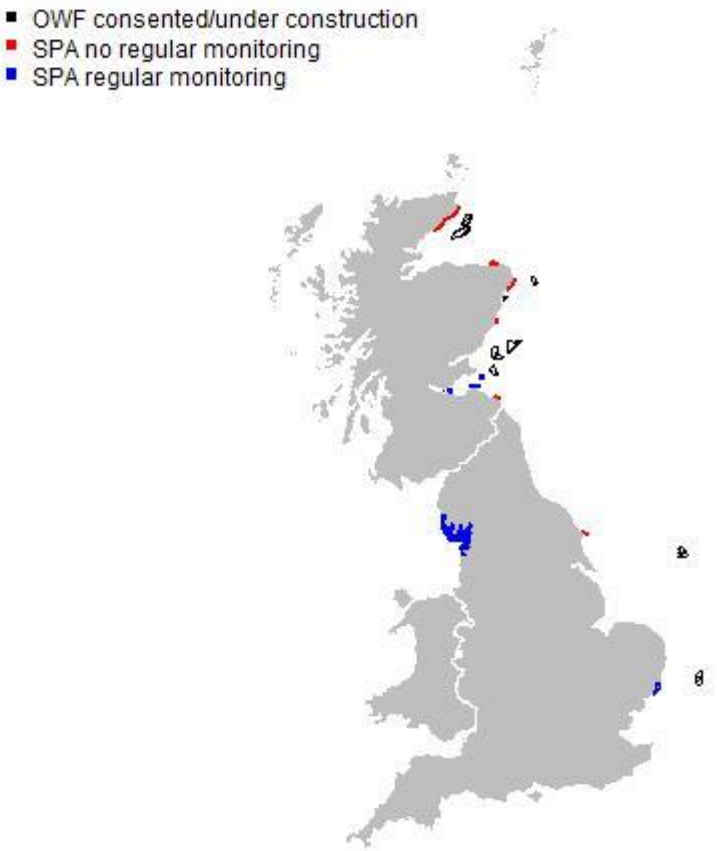

Figure 3.3.50 Special Protection Areas for Herring Gulls at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

Regular monitoring of the abundance of Herring Gulls has been undertaken by the SMP at eight colonies within three of these SPAs – Bass Rock, Fidra, Inchmickery, The Lamb and the Isle of May within the Forth Islands SPA, South Walney within the Morecambe Bay and Duddon Estuary SPA and Havergate Island and Orford Ness within the Alde-Ore Estuary SPA. On the Isle of May, it is possible to detect an effect leading to the population being 35% lower than it would have been in the absence of the wind farm and at South Walney it was possible to detect an effect leading to the population being 50% lower (3.3.51). It was not possible to detect a wind farm effect leading to less than a 50% reduction in population size at the remaining colonies.

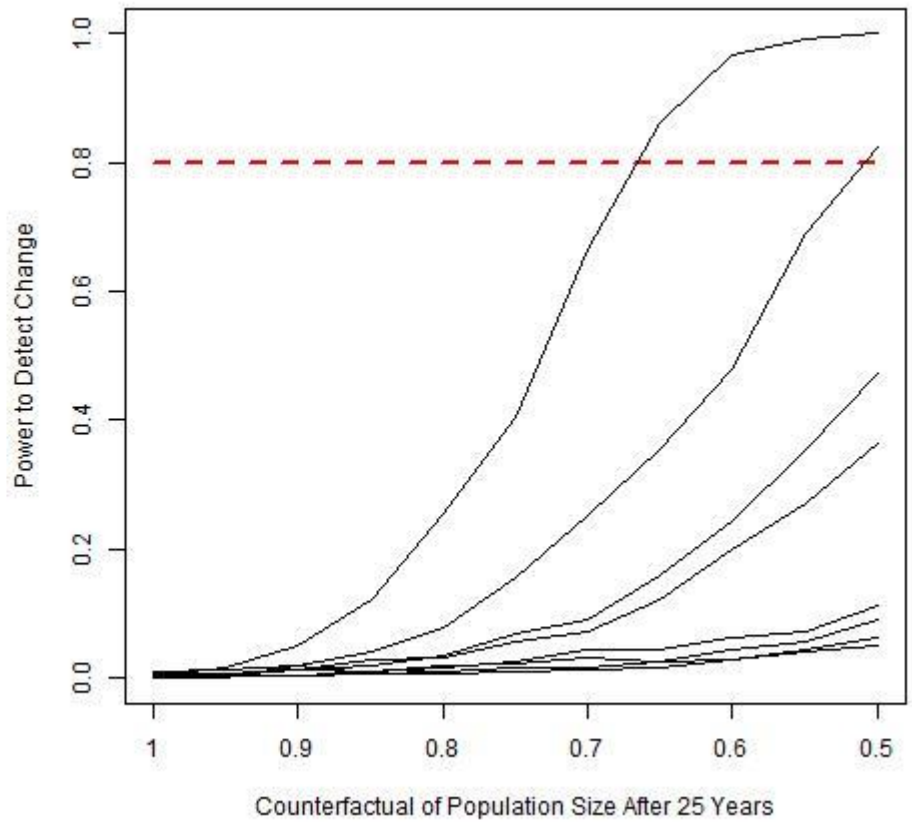

Figure 3.3.51 Power to detect a reduction of 0-50% in population size at Herring Gull breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

At present, it would not be possible to detect an impact resulting in a wind farm reducing population size by 25% or less at monitored Herring Gull colonies (Figure 3.3.52). An effect likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 17 years on the Isle of May. However, this would rely on monitoring being carried out on a biennial basis (Figure 3.3.52).

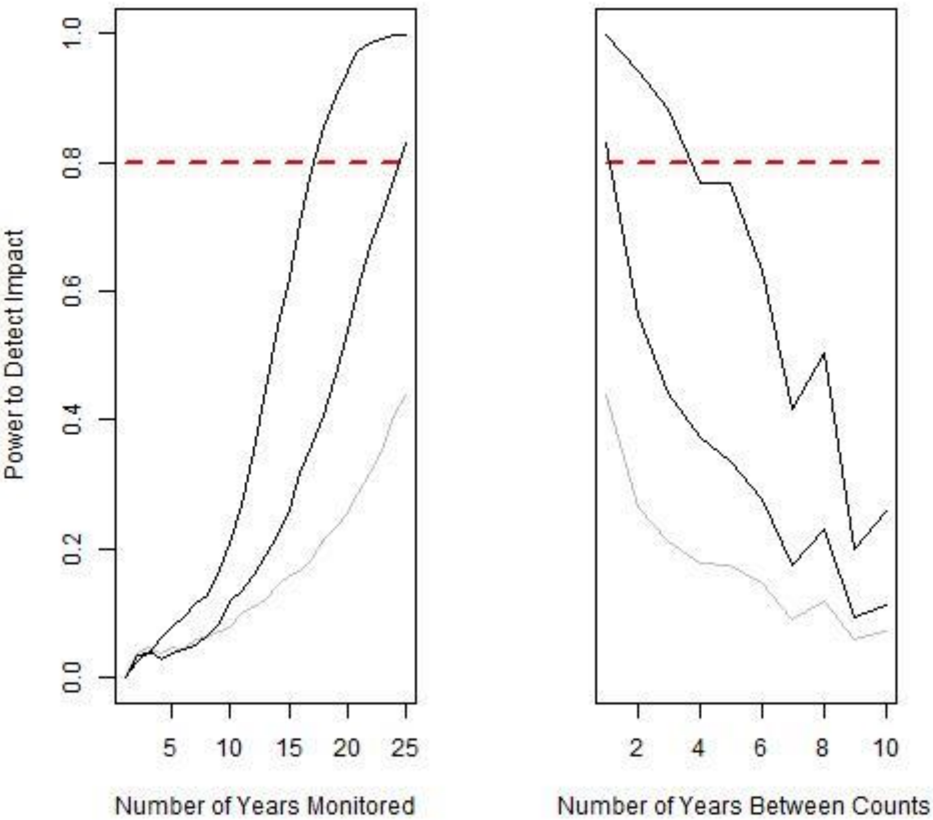

Figure 3.3.52 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Herring Gull breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

Breeding success

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Herring Gull populations could not be ruled out by the HRA screening process, breeding success is monitored by the SMP on Havergate Island within the Alde-Ore Estuary SPA, on the Isle of May within the Forth Islands SPA, and in the Flamborough Head and Filey Coast SPA (Figure 3.3.53). Within the Flamborough Head and Filey Coast SPA, monitoring of breeding success has taken place over the last seven years, during which time it has been relatively consistent (coefficient of variation 21.5). As a consequence, there is relatively high power to detect reductions of 5% or more in breeding success arising from the effect of offshore wind farms (Figure 3.3.54). However, given the relatively short time-span of the data, this should be treated with caution as the true annual variation in breeding success may not have been fully captured. Similarly, at Havergate Island monitoring of Herring Gull Breeding success has taken place in five out of the previous seven years. However, in this instance there is more substantial inter-annual variation in breeding success (coefficient of variation 94.3), meaning that reductions in breeding success of 50%, or more, cannot be detected at this site. In contrast, monitoring of herring gull breeding success on the Isle of May is more established, with 19 years of data. Whilst there is some variation in annual rates of breeding success (coefficient of variation 31.4), there is sufficient power in these data to detect reductions in breeding success, as a result of effects associated with an offshore wind farm, of 25% or more.

In addition to Havergate Island and Flambourgh Head and Bempton Cliffs, monitoring of Herring Gull breeding success has recently been established (in 2013) at the Fowlsheugh RSPB reserve. Assuming these monitoring programmes are continued, they may enable any changes in breeding success associated with offshore wind farms to be detected in coming years.

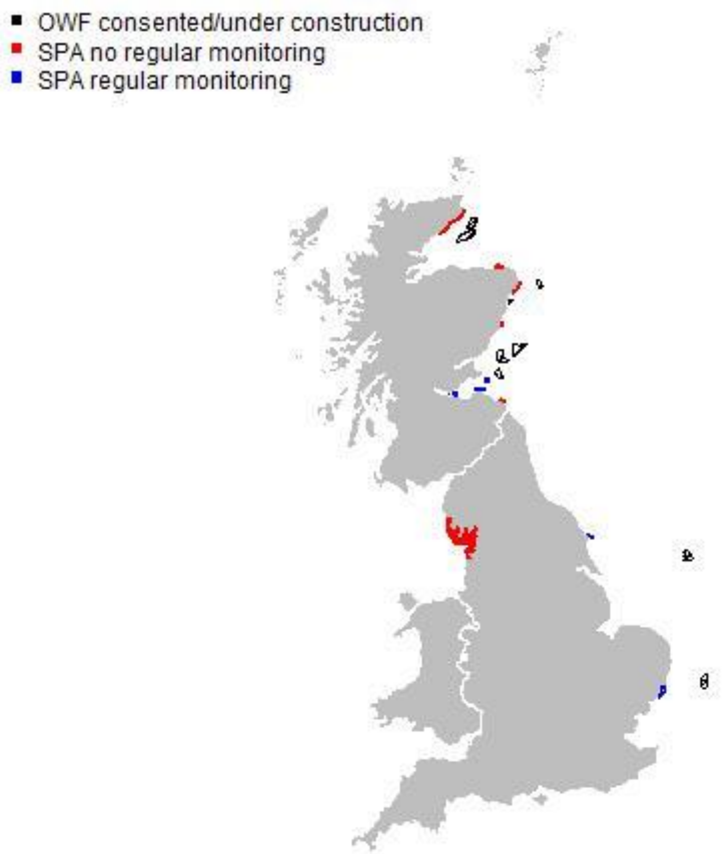

Figure 3.3.53 Special Protection Areas for Herring Gulls at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

The only colony at which it was possible to detect changes in abundance and breeding success in Herring Gull was the Forth Islands SPA. In this instance, the minimum population-level impact that could be detected by monitoring breeding success is greater than the minimum impact that could be detected through monitoring abundance (Figure 3.3.55). The coefficient of variation of breeding success at the Farne Islands is 31.4, potentially signifying strong annual variation in breeding success, making it more difficult to detect meaningful changes.

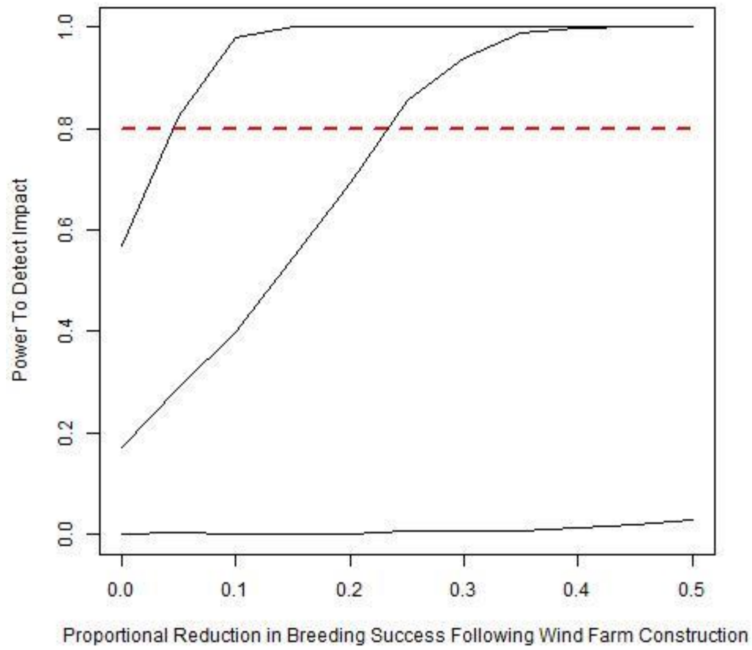

Figure 3.3.54 Power to detect a reduction of 0-50% in population size at Herring Gull breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

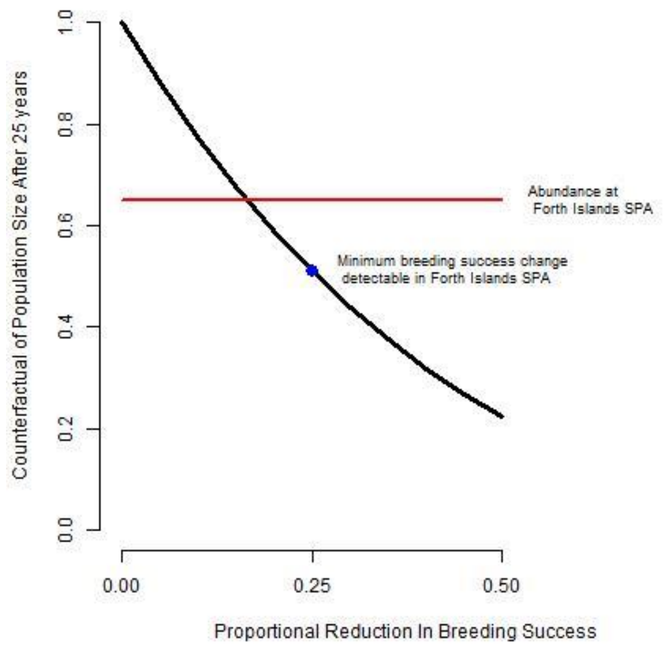

Figure 3.3.55 Counterfactual of population size after 25 years for Herring Gull relative to changes in annual breeding success associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Red lines indicate changes in abundance that could be detected given current monitoring effort.

Adult survival

We were unable to identify any SPAs for which offshore wind farm assessments concluded that LSEs on Herring Gull populations could not be ruled out by the HRA screening process, at which the species' adult survival has been monitored on an annual basis (Figure 3.3.56). If a capture-mark-recapture scheme were set up at a Herring Gull colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.020 could be detected in the 10 years following wind farm construction (Figure 3.3.57). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Herring Gull, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.020 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

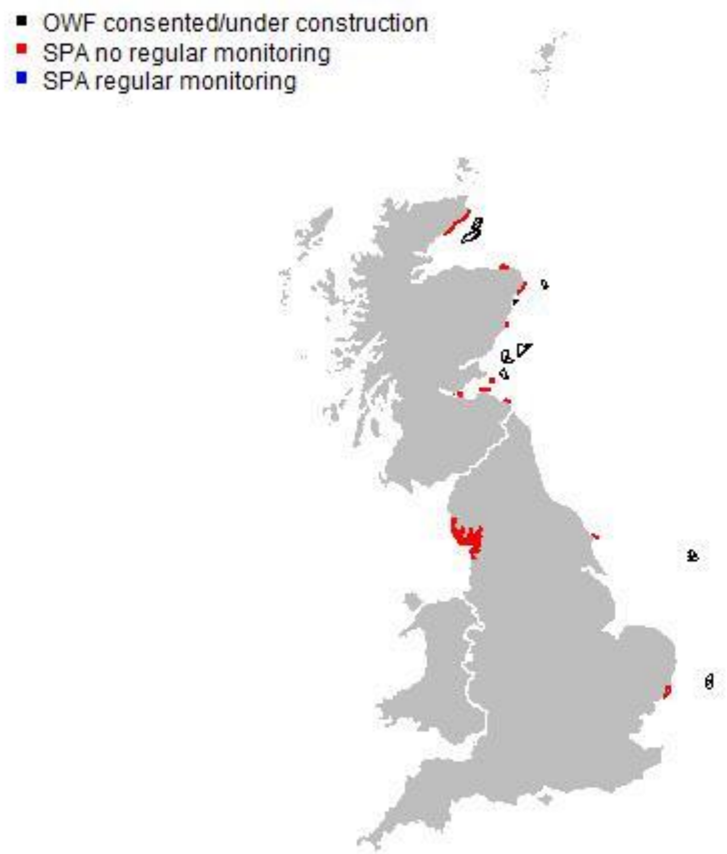

Figure 3.3.56 Special Protection Areas for Herring Gull at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for five or 10 years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone. Even with a capture-mark-recapture study established three years prior to wind farm construction, it would be possible to detect a change in adult survival likely to result in a population-level impact equivalent to that which could be detected through the monitoring of abundance within the Forth Islands SPAs (Figure 3.3.58). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any change in abundance.

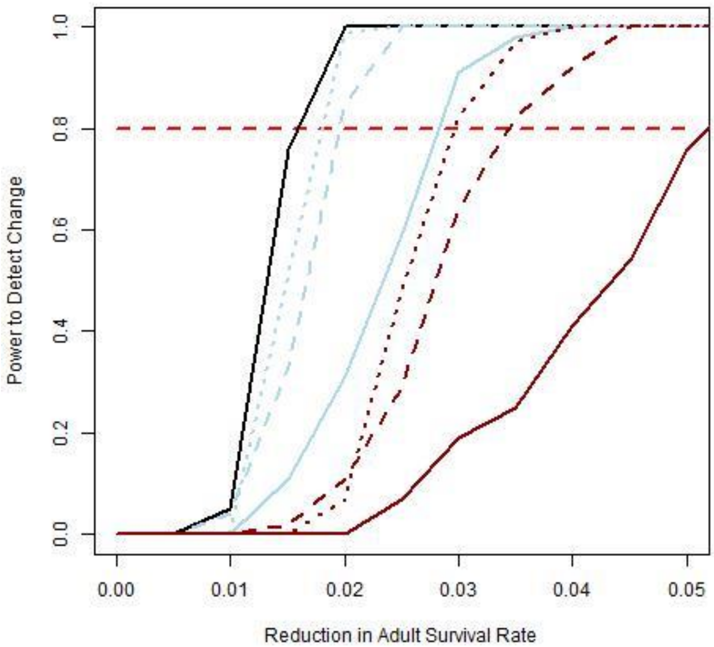

Figure 3.3.57 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Herring Gull. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

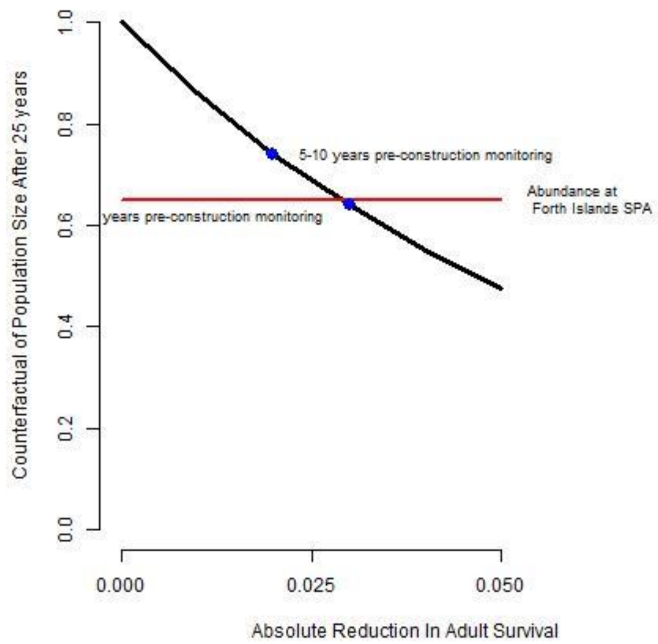

Figure 3.3.58 Counterfactual of population size after 25 years for Herring Gull relative to changes in adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection. Red lines indicate changes in abundance that could be detected given current monitoring effort.

3.3.8 Great Black-backed Gull *Larus marinus*

Abundance

Assessments for three offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Great Black-backed Gull at two SPAs (Figure 3.3.59, Table A3).

Figure 3.3.59 Special Protection Areas for Great Black-backed Gulls at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

Monitoring of the abundance of Great Black-backed Gulls is undertaken by the SMP at 12 sites within one of these sites, the Hoy SPA, and at the East Caithness Cliffs SPA. However, these sites are small and, typically, consist of low numbers of pairs (<10). Consequently, there is insufficient statistical power to detect changes in abundance that may be associated with offshore wind farms within these sites (Figure 3.3.60).

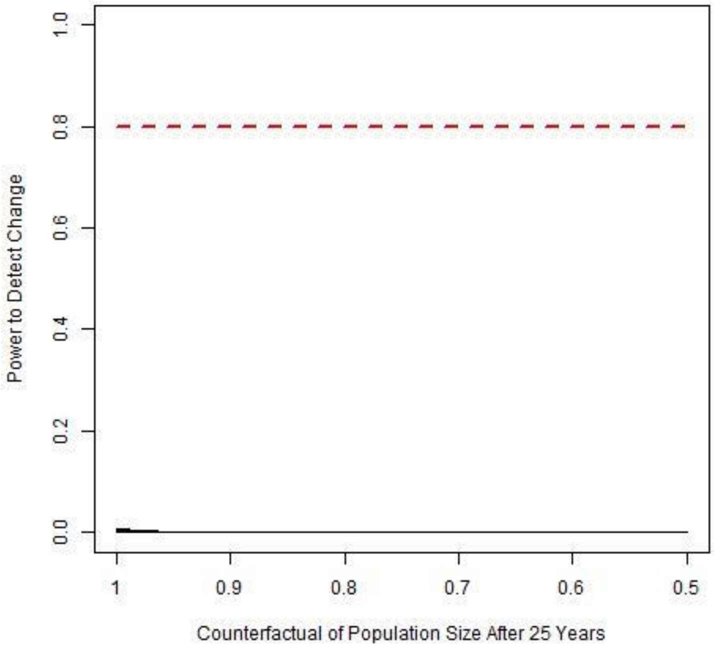

Figure 3.3.60 Power to detect a reduction of 0-50% in population size at Great Black-backed Gull breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Breeding success

At present, the breeding success of great black-backed gulls is not monitored by the SMP at either of the SPAs at which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.61).

Figure 3.3.61 Special Protection Areas for Great Black-backed Gull at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

Adult survival

At present, Great Black-backed Gull adult survival is not monitored at any sites within SPAs at which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.62). If a capture-mark-recapture scheme were set up at a Great Black-backed Gull colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.010 could be detected in the 10 years following wind farm construction (Figure 3.3.63). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Great Black-backed Gull, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.020 could be detected over the 10 years following wind farm construction.

Figure 3.3.62 Special Protection Areas for Great Black-backed Gull at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If mark-recapture data were available for the 10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.010 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.88 (Figure 3.3.64). If only five years data could be collected prior to the construction of a wind farm, then a reduction in the adult survival rate of no less than 0.015, resulting in a counterfactual of population size of 0.83, could be detected, assuming the number of birds marked per year were increased to 400 and the recapture rate was increased to 0.8. However, if only three years pre-construction data could be collected, the minimum impact that could be detected is a reduction in adult survival of 0.025, which would result in a counterfactual of population size after 25 years of 0.73.

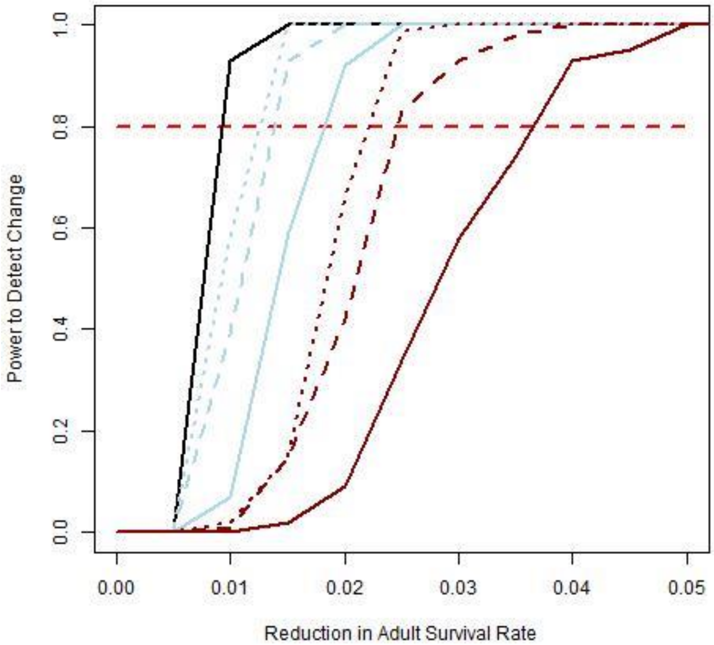

Figure 3.3.63 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Great Black-backed Gull. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

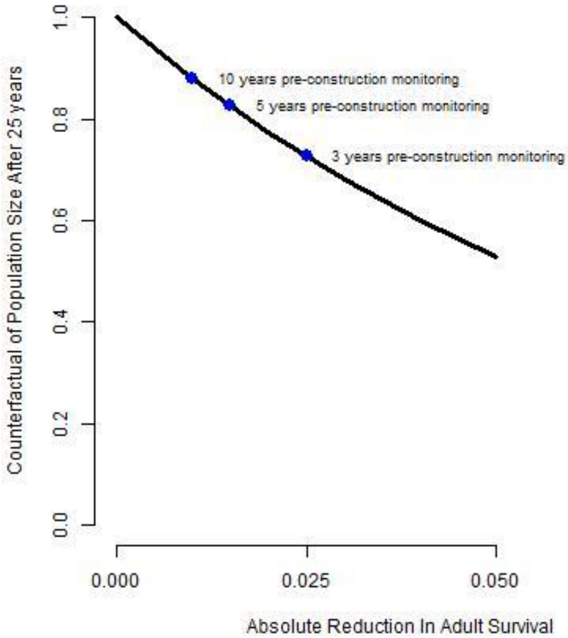

Figure 3.3.64 Counterfactual of population size after 25 years for Great Black-backed Gulls relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.9 Sandwich Tern *Sterna sandvicensis*

Abundance

Assessments for two offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Sandwich Terns at three SPAs (Figure 3.3.65, Table A3). Of these, Sandwich Terns do not appear to have been present within the Loch of Strathbeg SPA since 2000, with the population apparently having moved to the Ythan Estuary, Sands of Forvie and Meikle Loch SPA (A. Webb *pers. comm.*).

Figure 3.3.65 Special Protection Areas for Sandwich Terns at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

There was insufficient statistical power to detect a decline of no less than 50% within the Scolt Head and Blakeney Point colonies within the North Norfolk Coast SPA (Figure 3.3.66). However, these colonies may act as a single population. If we consider the cumulative abundance across both colonies, there is sufficient statistical power to detect a reduction in population size of no less than 30%, over 25 years, in comparison to what the population size would have been in the absence of a wind farm. Within the Sands of Forvie colony, part of the Ythan Estuary, Sands of Forvie and Meikle Loch SPA, there is sufficient power to detect a decline of no less than 10% over 25 years, relative to what would have been observed in the absence of a wind farm. However, as was the case with the North Norfolk Coast population, this colony may be part of a larger population which includes the, now defunct, Loch of Strathbeg colony. By considering only the Sands of Forvie population, we may not be accurately reflecting the true variation in population size. This may lead to an over-estimate of our ability to detect a change at a population-level resulting from the effects associated with an offshore wind farm. If we consider the birds from the Loch of Strathbeg colony and the Sands of

Forvie colony to be a single population, power to detect a population-level change as a consequence of the effects of an offshore wind farm drops substantially, with the power to detect a decline of 50%, relative to a population not affected a wind farm, of only 0.73.

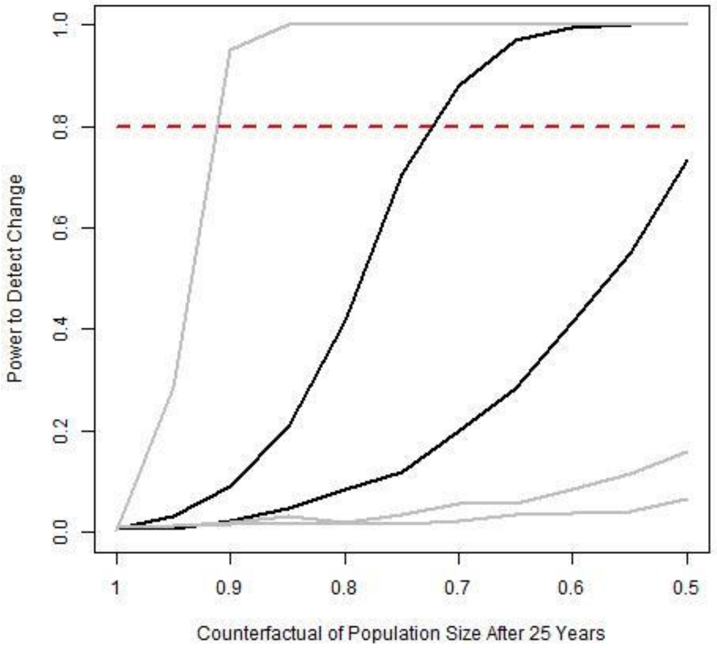

Figure 3.3.66 Power to detect a reduction of 0-50% in population size at Sandwich Tern breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm. Grey line indicate individual colonies, black lines indicate data from adjacent colonies combined.

At present, it would not be possible to detect an impact resulting in a wind farm reducing population size by 25% or less at monitored Sandwich Tern colonies (Figure 3.3.67). An effect likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 16 years on the North Norfolk Coast. However, this is reliant on monitoring being carried out at least once every five years (Figure 3.3.67).

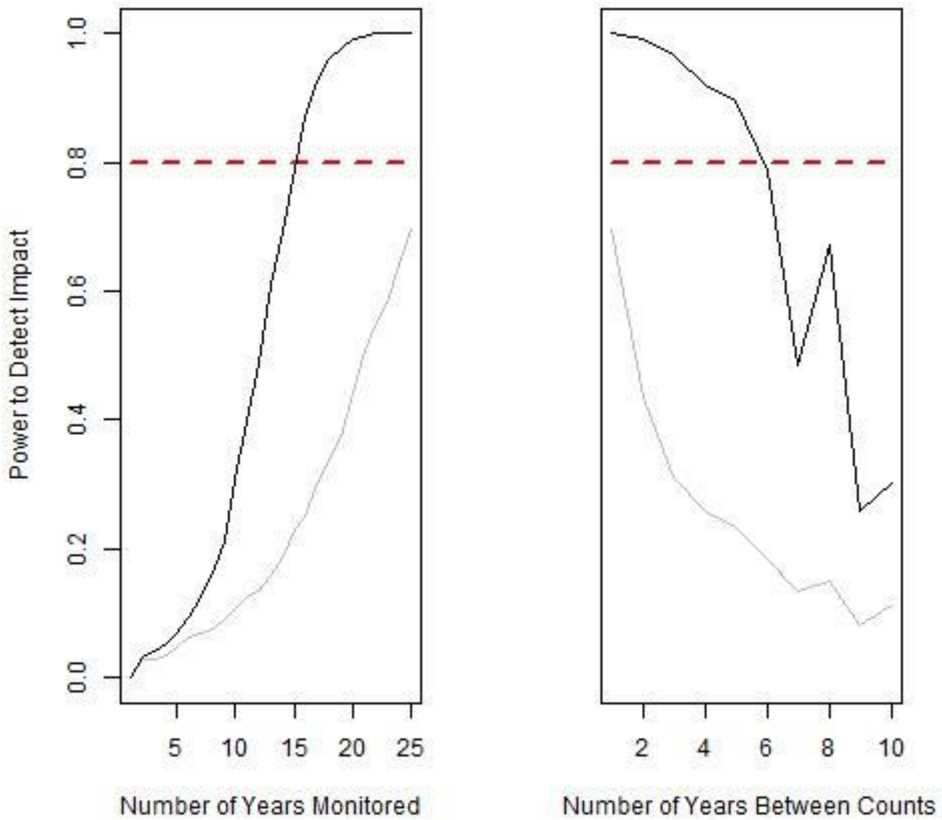

Figure 3.3.67 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Sandwich Tern breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

Breeding success

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Sandwich Tern populations could not be ruled out by the HRA screening process, the breeding success of the species is monitored at Sands of Forvie within the Ythan Estuary, Sands of Forvie and Meikle Loch SPA and at Blakeney Point and Scolt Head within the North Norfolk Coast SPA (Figure 3.3.68). There is insufficient power to detect a decline in breeding success in response to the effects associated with an offshore wind farm at any individual colony (Figure 3.3.69). As birds at Blakeney Point and Scolt Head may operate as a single population, breeding success estimates from these colonies were combined. Having done this, there was still insufficient power to detect a reduction in breeding success of no less than 50%. However, power to detect changes of this magnitude had increased to 0.71. Combining estimates of breeding success from the Sands of Forvie and Loch of Strathbeg populations did not have a similar effect.

Figure 3.3.68 Special Protection Areas for sandwich Tern at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

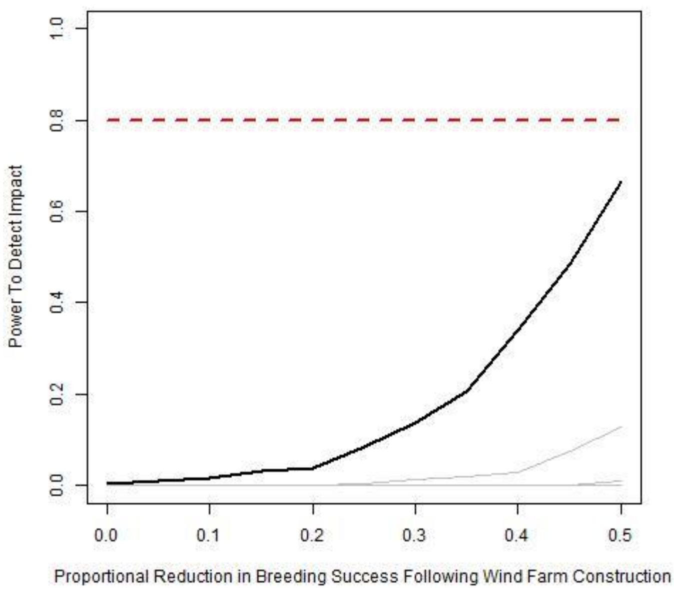

Figure 3.3.69 Power to detect a reduction of 0-50% in breeding success at Sandwich Tern breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm. Grey line indicate individual colonies, black lines indicate data from adjacent colonies combined.

Adult survival

At present, Sandwich Tern adult survival is not monitored at any SPAs for which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.70). If a capture-mark-recapture scheme were set up at a Sandwich Tern colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.71). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Sandwich Tern, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.025 could be detected over the 10 years following wind farm construction.

Figure 3.3.70 Special Protection Areas for Sandwich Tern at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for five or 10 years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone (Figure 3.3.72). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any change in abundance.

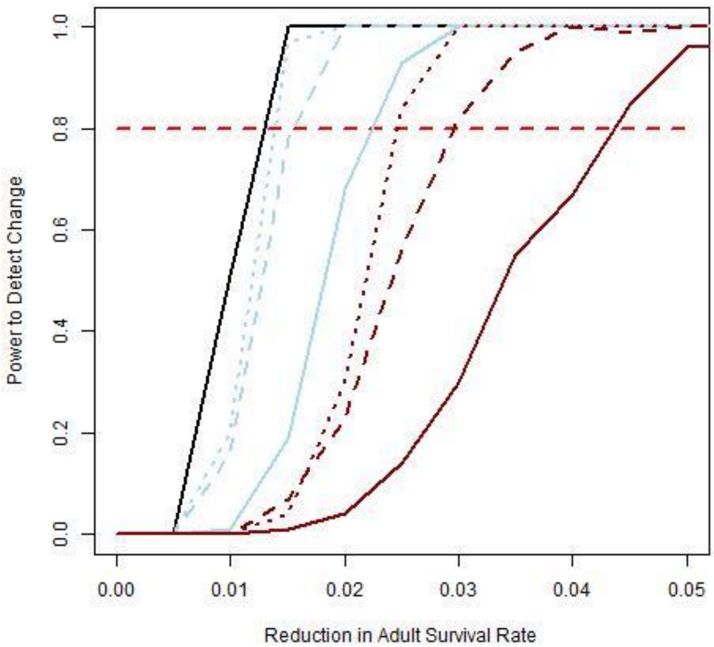

Figure 3.3.71 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Sandwich Tern. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

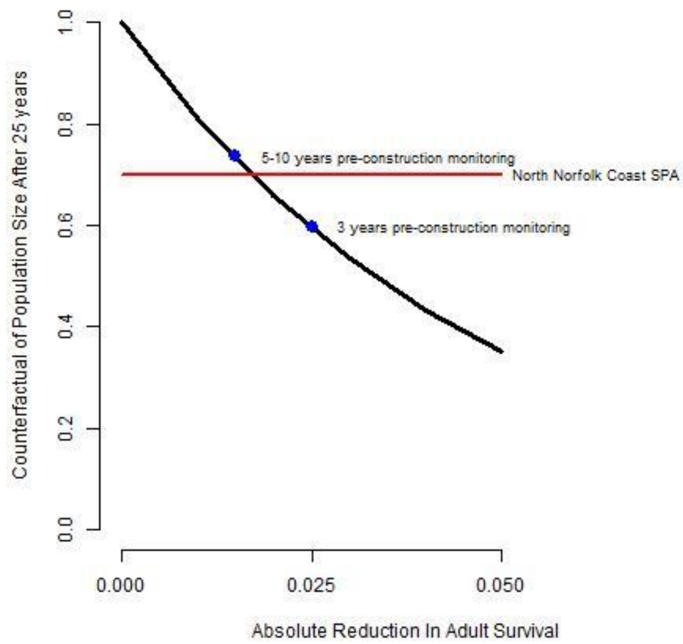

Figure 3.3.72 Counterfactual of population size after 25 years for Sandwich Terns relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.10 Common Tern *Sterna hirundo*

Abundance

Assessments for five offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Common Tern at two SPAs (Figure 3.3.73, Table A3). Common Tern populations are currently monitored by the SMP at the Sands of Forvie, within the Ythan Estuary, Sands of Forvie and Meikle Loch SPA and at Long Craig and the Isle of May within the Forth Islands SPA. At present, there is insufficient statistical power to detect declines of no less than 50% in response to the effects of offshore wind farms (Figure 3.3.74). As was the case with Sandwich Terns, it is likely that Common Terns within the Forth Islands SPA form a single population. Consequently, we consider cumulative abundance within both the Long Craig and Isle of May colonies. This resulted in a marginal improvement in power to detect changes in population size. However, this was still well below the level needed to detect changes of no less than 50% in response to the construction of an offshore wind farm.

Figure 3.3.73 Special Protection Areas for Common Terns at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

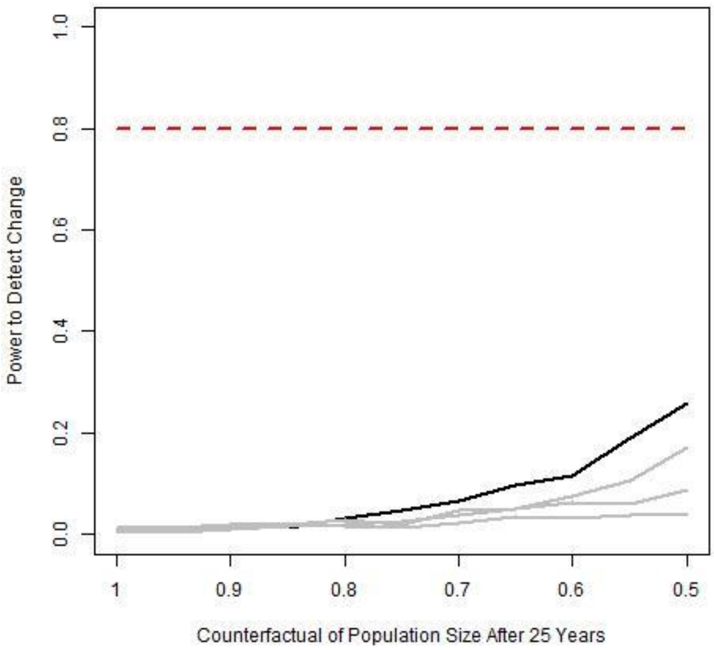

Figure 3.3.74 Power to detect a reduction of 0-50% in population size at Common Tern breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm. Grey line indicate individual colonies, black lines indicate data from adjacent colonies combined.

Breeding success

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Common Tern populations could not be ruled out by the HRA screening process, the breeding success of the species is monitored at the Sands of Forvie colony within the Ythan Estuary, Sands of Forvie and Meikle Loch SPA and at Long Craig and at the Isle of May within the Forth Islands SPA (Figure 3.3.75). At present, there is insufficient power to detect changes in breeding success within either the Sands of Forvie, or the Isle of May colonies. Within the Long Craig colony, the power to detect a reduction in breeding success of 50%, as a consequence of the effects of an offshore wind farm, is 0.79 (Figure 3.3.76).

Figure 3.3.75 Special Protection Areas for Common Tern at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

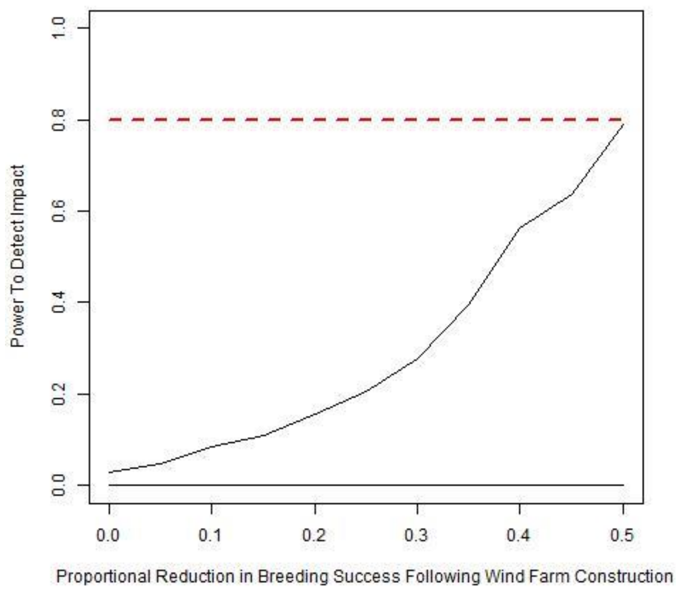

Figure 3.3.76 Power to detect a reduction of 0-50% in breeding success at Common Tern breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Adult survival

At present, Common Tern adult survival is not monitored at any SPAs for which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.77). If a capture-mark-recapture scheme were set up at a Common Tern colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.78). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Common Tern, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

Figure 3.3.77 Special Protection Areas for Common Tern at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If mark-recapture data were available for the five10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.015 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.75 (Figure 3.3.79). However, if only 3 years pre-construction data could be collected, the minimum change that could be detected is a reduction in adult survival of 0.03, which would result in a counterfactual of population size after 25 years of 0.68.

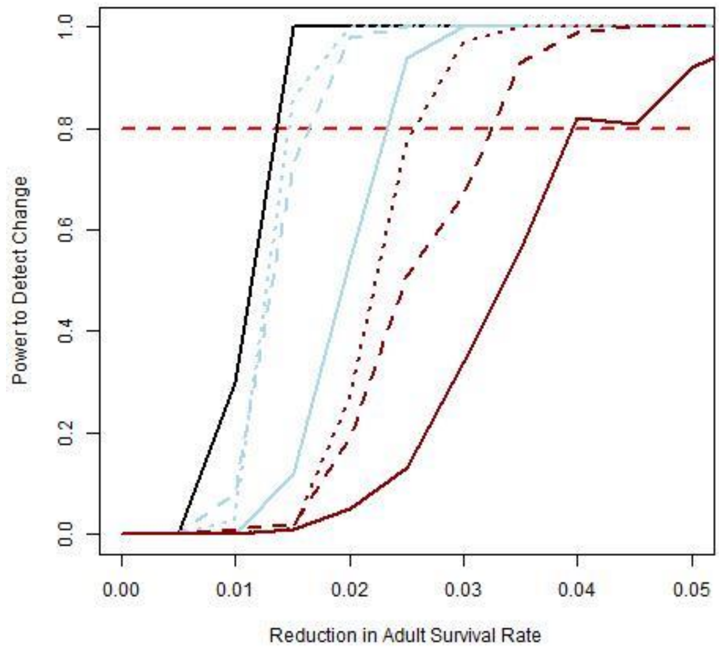

Figure 3.3.78 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Common Tern. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8

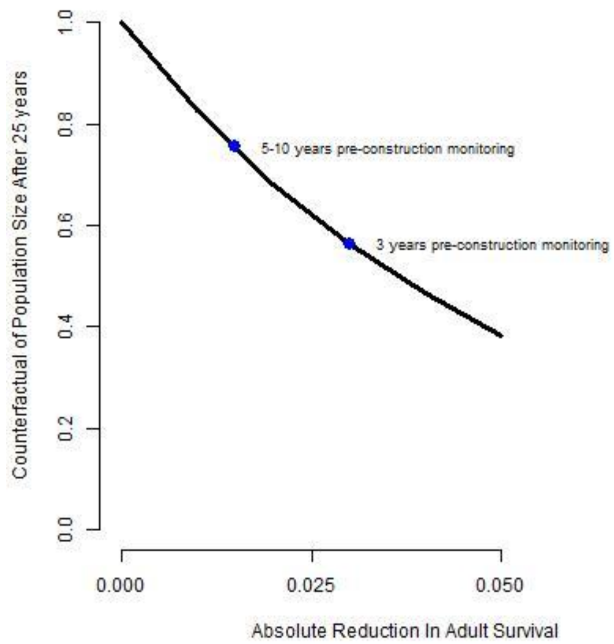

Figure 3.3.79 Counterfactual of population size after 25 years for Common Terns relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.11 Arctic Tern *Sterna paradisaea*

Abundance

Assessments for four offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Arctic Terns at one SPA, the Forth Islands SPA (Figure 3.3.80, Table A3). At present, Arctic Tern abundance is only monitored by the SPA at one site, the Isle of May, within this SPA. However, at present, data have insufficient power with which to detect population declines of no less than 50% in response to the effects associated with offshore wind farms (Figure 3.3.81).

Figure 3.3.80 Special Protection Areas for Arctic Terns at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

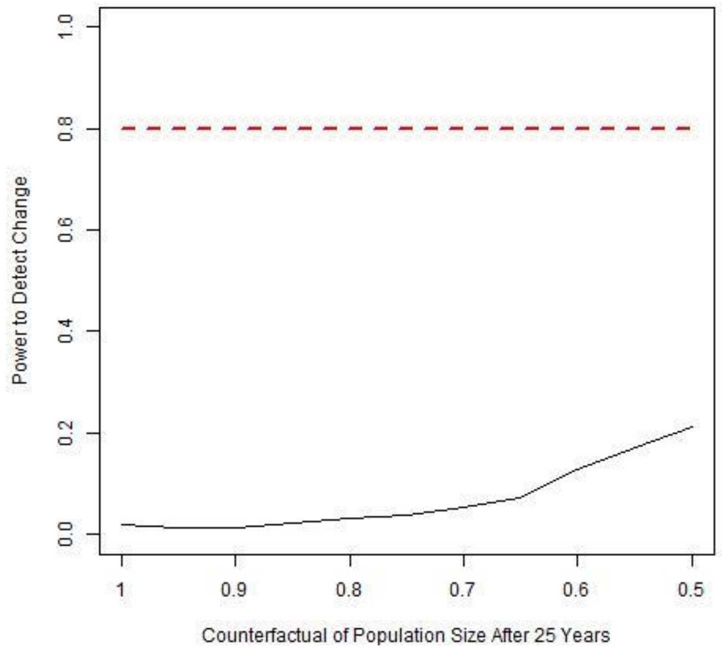

Figure 3.3.81 Power to detect a reduction of 0-50% in population size at Arctic Tern breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Breeding success

As with abundance, at present, at SPAs for which offshore wind farm assessments concluded that LSEs on Arctic tern populations could not be ruled out by the HRA screening process, the breeding success of the species is only monitored on the Isle of May, within the Forth Islands SPA (Figure 3.3.82). However, annual breeding success is highly variable (coefficient of variation 98.6). Consequently, it is not possible to detect declines in breeding success at this colony of no less than 50% (Figure 3.3.83).

Figure 3.3.82 Special Protection Areas for Arctic Tern at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

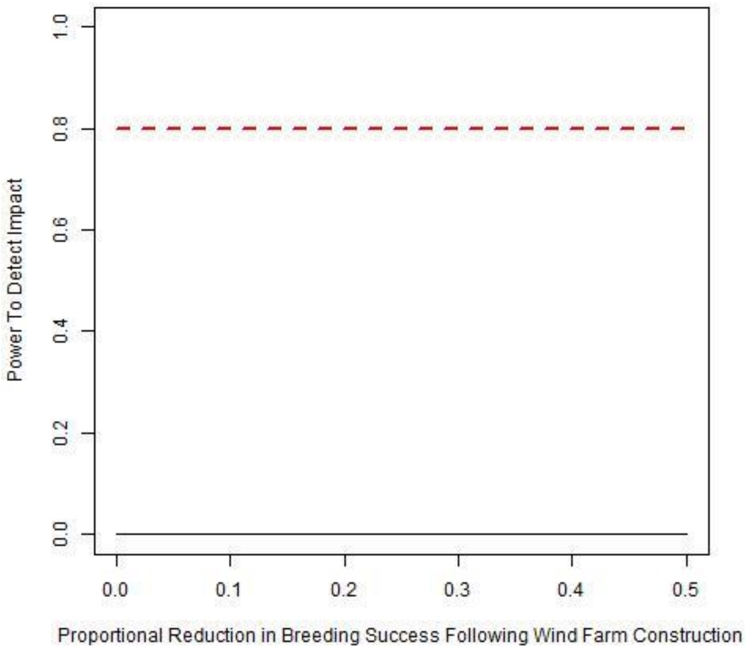

Figure 3.3.83 Power to detect a reduction of 0-50% in breeding success at Arctic Tern breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Adult survival

At present, Arctic Tern adult survival is not monitored at any SPAs for which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.84). If a capture-mark-recapture scheme were set up at an Arctic Tern colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.020 could be detected in the 10 years following wind farm construction (Figure 3.3.85). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Arctic Tern, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.020 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

Figure 3.3.84 Special Protection Areas for Arctic Tern at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If mark-recapture data were available for the five10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.020 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.65 (Figure 3.3.86). However, if only three years pre-construction data could be collected, the minimum change that could be detected is a reduction in adult survival of 0.030, which would result in a counterfactual of population size after 25 years of 0.52.

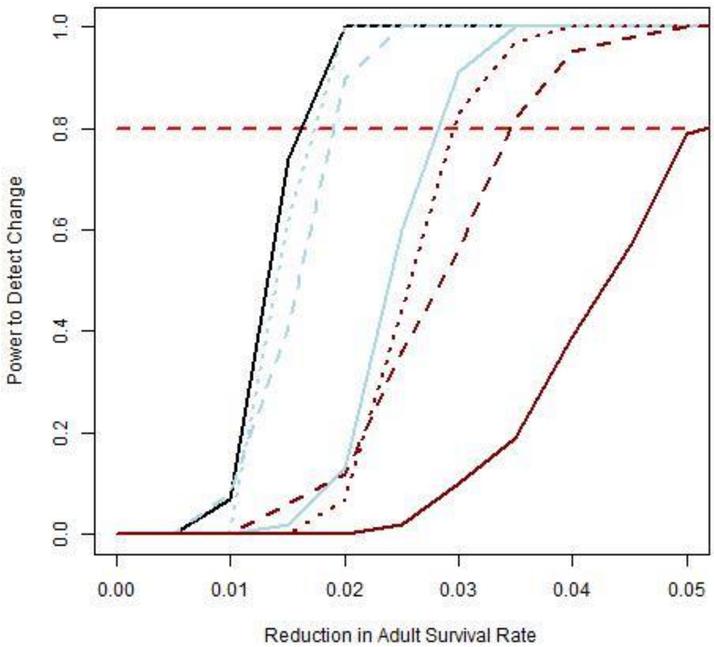

Figure 3.3.85 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Arctic Tern. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

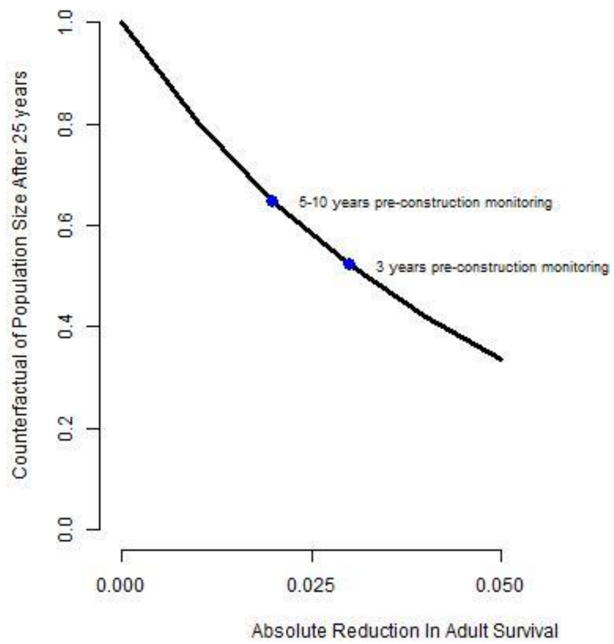

Figure 3.3.86 Counterfactual of population size after 25 years for Arctic Terns relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.12 Arctic Skua *Stercorarius parasiticus*

Abundance

Assessments for four offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Arctic Skuas at one SPA – the Hoy SPA in Orkney (Figure 3.3.87, Table A3). We did not identify any colonies within this SPA at which regular abundance monitoring by the SMP would enable us to establish baseline population estimates against which any impacts arising as a consequence of the development of offshore wind farms could be assessed.

Figure 3.3.87 Special Protection Areas for Arctic skuas at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

Breeding success

Arctic Skua breeding success is monitored at the Hoy RSPB reserve within the Hoy SPA. At present breeding success has been monitored in four years since 2000 with a maximum of six nests monitored in any single year. However, such data are insufficient to detect any changes which may be associated with offshore wind farms on Arctic Skuas within this SPA. Consequently, at present, monitoring of Arctic Skua breeding success is not regular enough to detect the changes that may be associated with offshore wind farms at an SPA level in Arctic Skuas (Figure 3.3.88)

Figure 3.3.88 Special Protection Areas for Arctic Skua at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

Adult survival

At present, Arctic Skua adult survival is not monitored at any SPAs for which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (Figure 3.3.89). If a capture-mark-recapture scheme were set up at an Arctic Skua colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.90). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Arctic Skua, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.025 could be detected over the 10 years following wind farm construction.

Figure 3.3.89 Special Protection Areas for Arctic Skua at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If mark-recapture data were available for the 10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.015 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.75 (Figure 3.3.91). If only five years data could be collected prior to the construction of a wind farm, then a reduction in the adult survival rate of no less than 0.020, resulting in a counterfactual of population size of 0.68, could be detected, assuming the number of birds marked per year were increased to 400 and the recapture rate was increased to 0.8. However, if only three years pre-construction data could be collected, the minimum change that could be detected is a reduction in adult survival of 0.025, which would result in a counterfactual of population size after 25 years of 0.62.

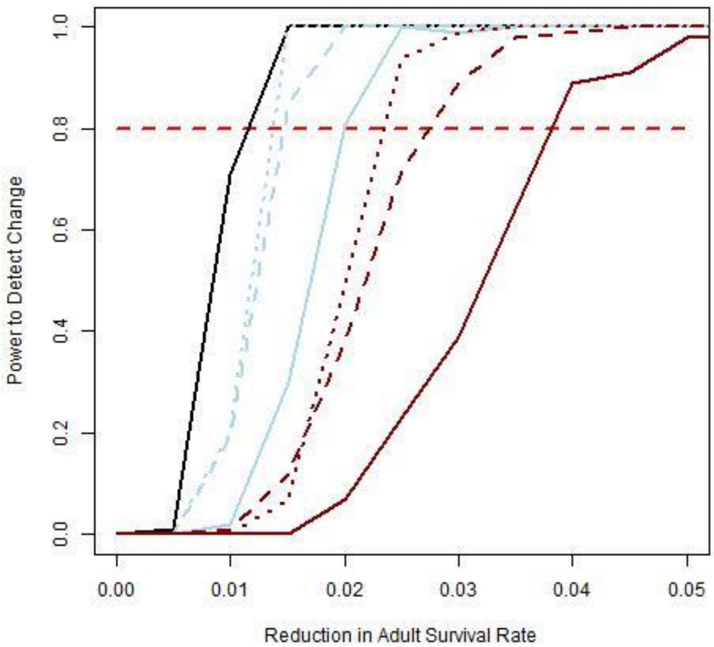

Figure 3.3.90 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Arctic Skua. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

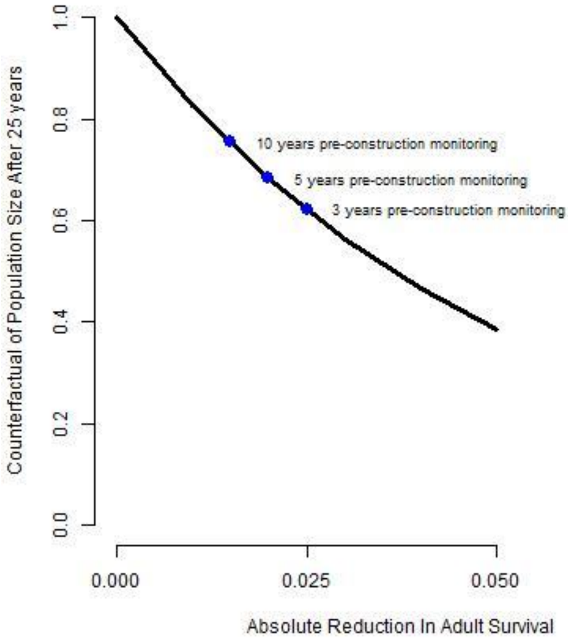

Figure 3.3.91 Counterfactual of population size after 25 years for Arctic Skuas relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.13 Great Skua *Stercorarius skua*

Abundance

Assessments for two offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Great Skua at two SPAs (Figure 3.3.92, Table A3).

Figure 3.3.92 Special Protection Areas for Great Skuas at which LSEs could not be ruled out in relation to effects associated with offshore wind farms.

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Great Skua populations could not be ruled out by the HRA screening process, the species' populations are only regularly monitored within the Handa SPA. However, population estimates show strong inter-annual variation, meaning that current data lack sufficient statistical power to detect changes that may be associated with offshore wind farms (Figure 3.3.93).

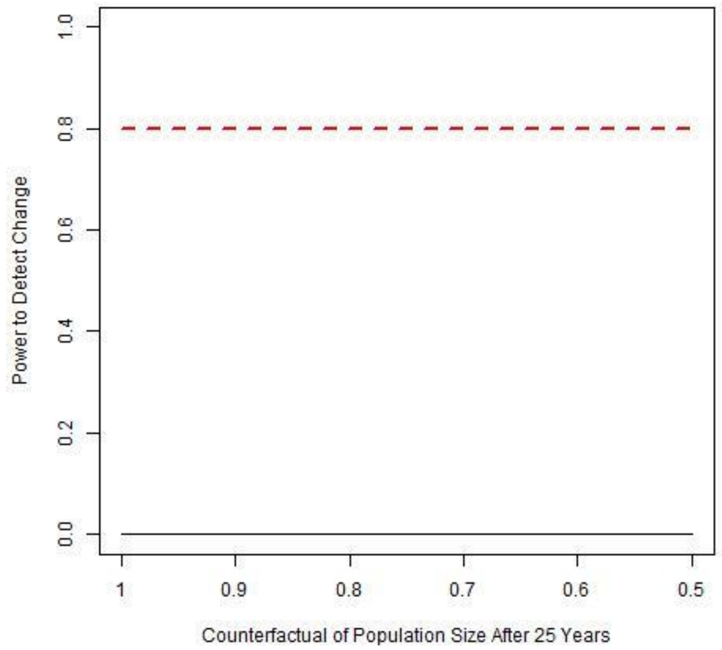

Figure 3.3.93 Power to detect a reduction of 0-50% in population size at Great Skua breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Breeding success

As with abundance, at present, at SPAs for which offshore wind farm assessments concluded that LSEs on Great Skua populations could not be ruled out by the HRA screening process, the species' breeding success is only monitored at the Handa SPA (3.3.94). There is strong inter-annual variation in breeding success within this SPA (coefficient of variation 62.6). Consequently, it is not possible to detect reductions in breeding success, associated with the effect of offshore wind farms, of 50% or less at this colony (Figure 3.3.95).

Figure 3.3.94 Special Protection Areas for Great Skua at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

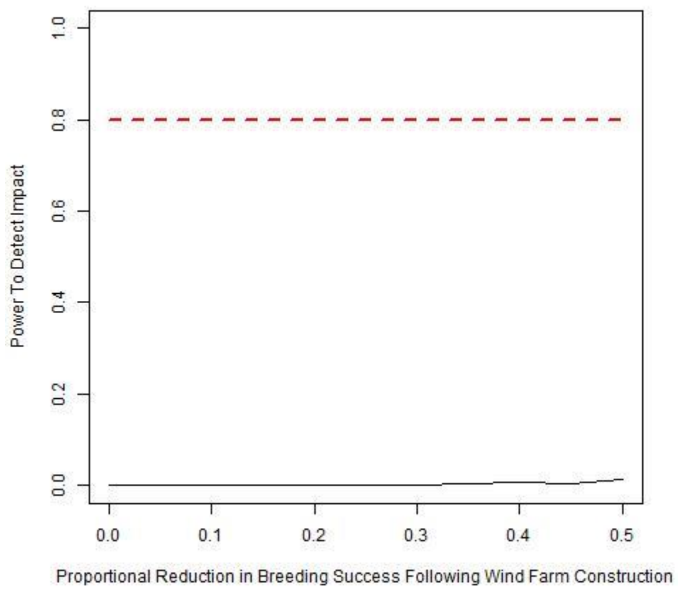

Figure 3.3.95 Power to detect a reduction of 0-50% in population size at Great Skua breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Adult survival

At present, Great Skua adult survival is not monitored at any SPAs for which offshore wind farm assessments concluded that LSEs on the species' populations could not be ruled out by the HRA screening process (3.3.96). If a capture-mark-recapture scheme were set up at an Great Skua colony following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.97). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Great Skua, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.020 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

Figure 3.3.96 Special Protection Areas for Great Skua at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

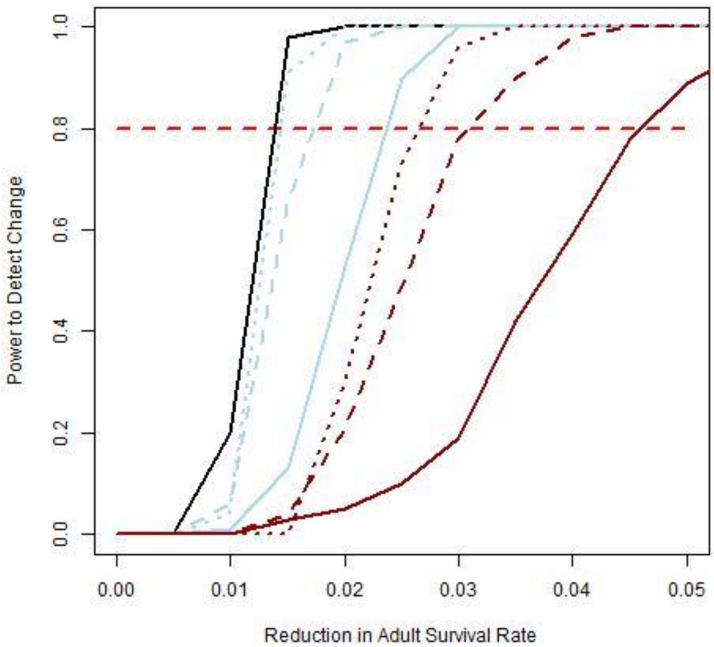

Figure 3.3.97 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Great Skua. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

If mark-recapture data were available for the 10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.015 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.79 (Figure 3.3.98). If only five years data could be collected prior to the construction of a wind farm, then a reduction in the adult survival rate of no less than 0.020, resulting in a counterfactual of population size of 0.73, could be detected, assuming the number of birds marked per year were increased to 400 and the recapture rate was increased to 0.8. However, if only three years pre-construction data could be collected, the minimum change that could be detected is a reduction in adult survival of 0.030, which would result in a counterfactual of population size after 25 years of 0.63.

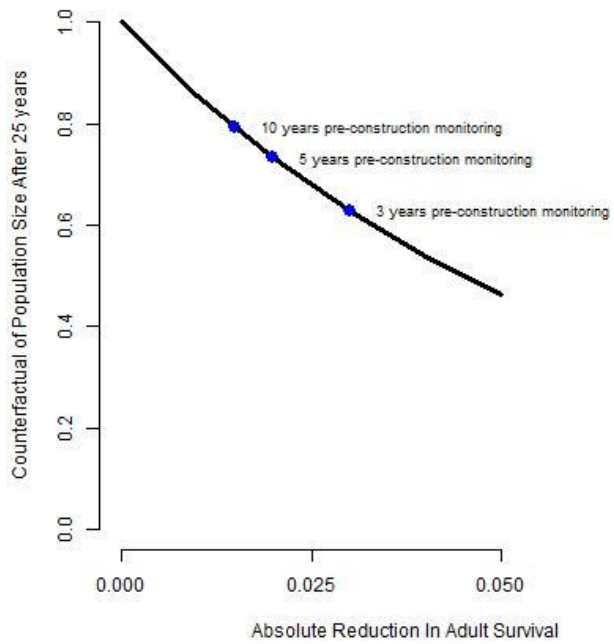

Figure 3.3.98 Counterfactual of population size after 25 years for Great Skuas relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.14 Common Guillemot *Uria aalge*

Abundance

Assessments for 15 offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Common Guillemot at a total of 19 SPAs (Figure 3.3.99, Table A3). At present, the abundance of Common Guillemots is monitored at six colonies within two of these SPAs – the Farne Islands SPA and Fidra, The Lamb, The Bass Rock, Craigleith and the Isle of May within the Forth Islands SPA. At present, there is sufficient power to detect a decline attributable to the effects associated with an offshore wind farm of no less than 30% on the Farne Islands and no less than 40% on the Isle of May (3.3.100). There is insufficient statistical power to detect declines of no less than 50% at the smaller Common Guillemot colonies within the Forth Islands SPA.

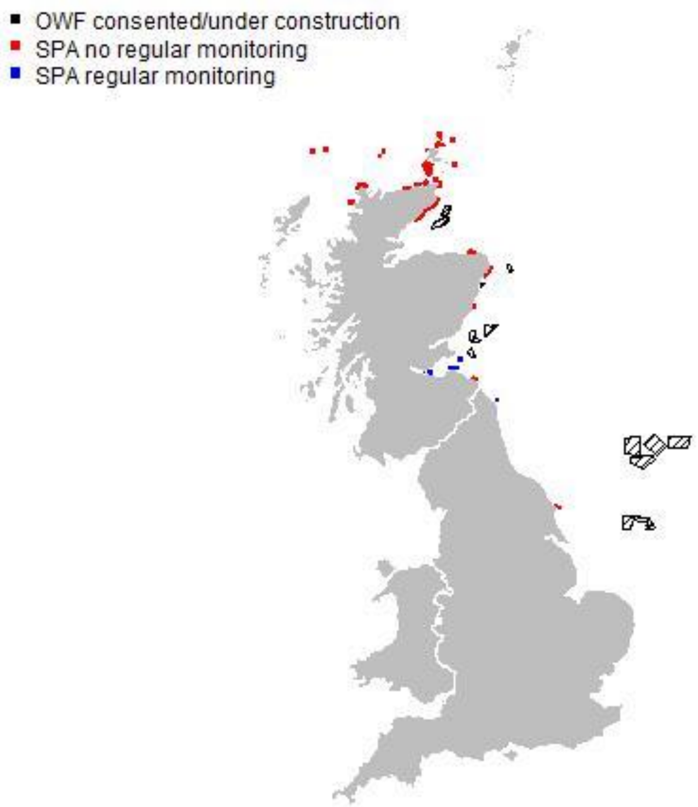

Figure 3.3.99 Special Protection Areas for Common Guillemots at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

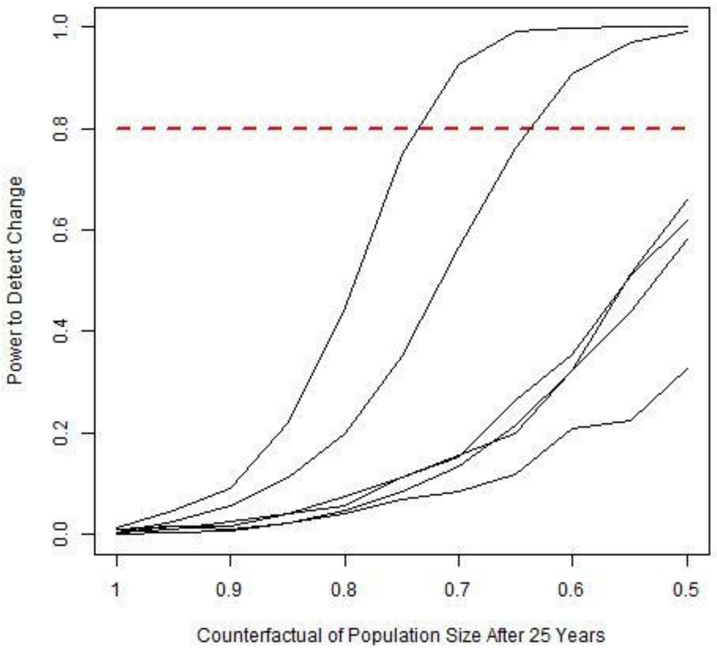

Figure 3.3.100 Power to detect a reduction of 0-50% in population size at Common Guillemot breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

At present, it would not be possible to detect an impact resulting in a wind farm reducing population size by 25% or less at monitored Common Guillemot (Figure 3.3.101). An effect likely to lead to a 50% reduction over 25 years as a result of the effects associated with an offshore wind farm could be detected within 15 years at the Farne Islands and within 19 years on the Isle of May. In order to detect these impacts, monitoring would have to be carried out every three years on the Isle of May and every six years on the Farne Islands (Figure 3.3.101).

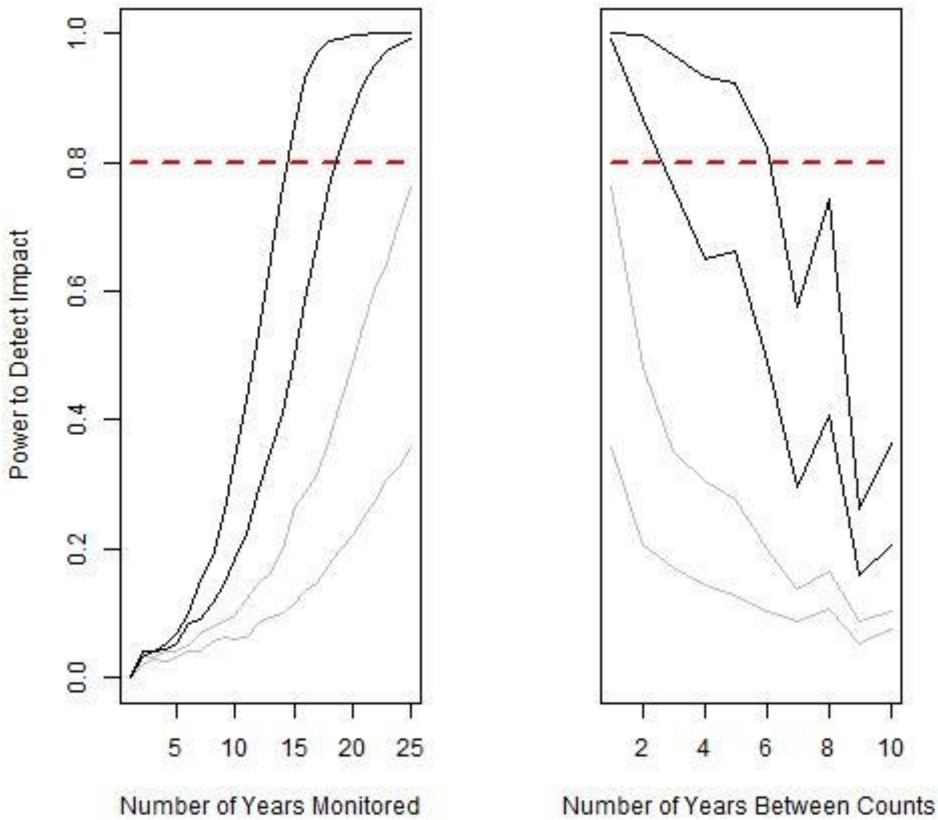

Figure 3.3.101 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Common Guillemot breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

Breeding success

We identified five SPAs for which offshore wind farm assessments concluded that LSEs on Common Guillemot populations could not be ruled out by the HRA screening process and, within which, there is regular monitoring of the species' breeding success (Figure 3.3.102). In the future, recently established monitoring within the Copinsay SPA may also enable changes in breeding success to be detected at this site.

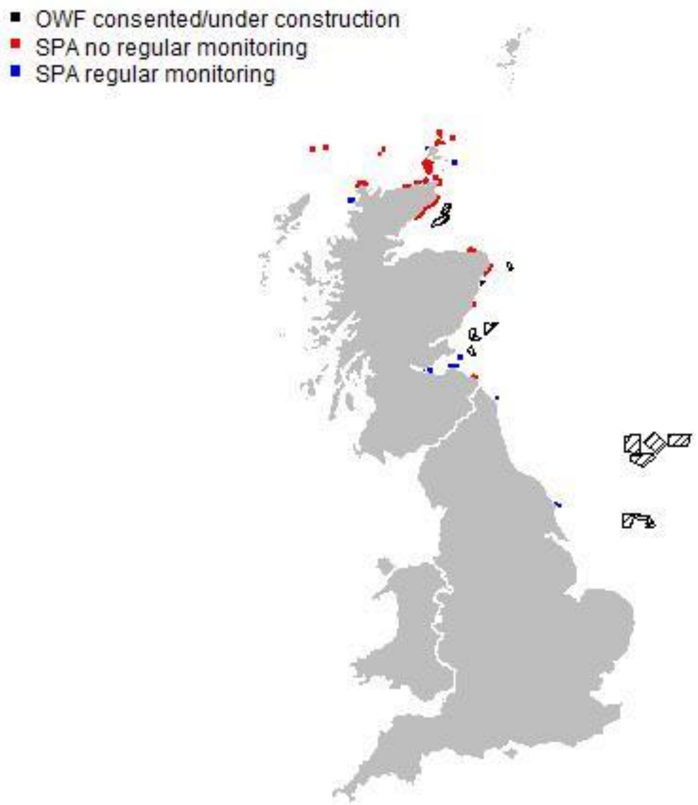

Figure 3.3.102 Special Protection Areas for Common Guillemot at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

There is relatively little annual variation in the levels of breeding success recorded within the Farne Islands, Flamborough Head and Filey Coast and Forth Islands SPAs (coefficients of variation of 17.9, 16.0 and 17.9 respectively). This is reflected in a relatively high power to detect reductions in breeding success that may be attributable to offshore wind farms (Figure 3.3.103). At Flamborough Head and Filey Coast SPA and the Isle of May, within the Forth Islands SPA, there is sufficient power to detect a 10% reduction in breeding success as a result of the effects associated with an offshore wind farm. On the Farne Islands, there is sufficient power to detect a 20% decline in breeding success. There is more variation in the annual recorded breeding success within the Handa SPA (coefficient of variation 38.3). However, there is still sufficient power to detect a decline in breeding success of no less than 40%. However, very low breeding success in recent years within the Marwick Head SPA has made it difficult to detect any effects that may be associated with offshore wind farms at this site.

The only colonies at which it was possible to detect changes in abundance and breeding success in Common Guillemot were the Farne Islands SPA and the Isle of May, within the Forth Islands SPA. In both cases, the minimum population-level impact that could be detected by monitoring breeding success is greater than the minimum impact that could be detected through monitoring abundance (Figure 3.3.104).

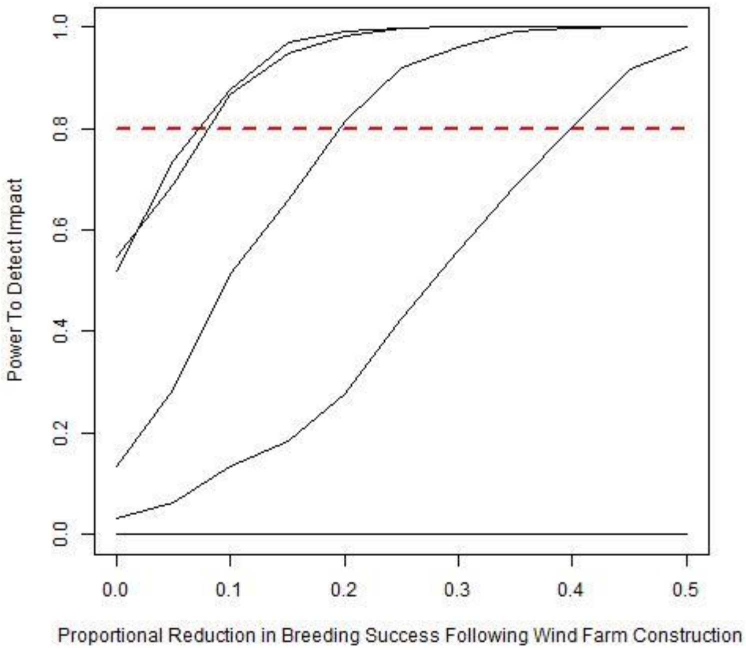

Figure 3.3.103 Power to detect a reduction of 0-50% in breeding success at Common Guillemot breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

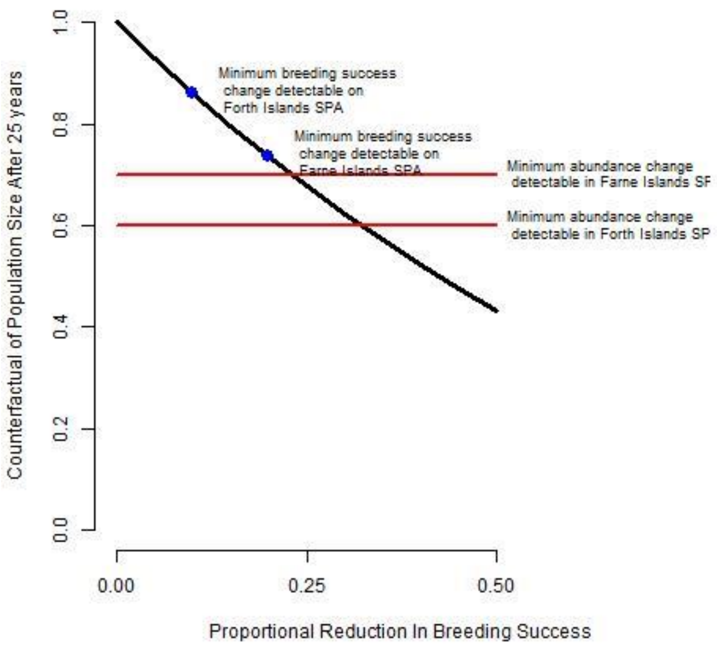

Figure 3.3.104 Counterfactual of population size after 25 years for Common Guillemots relative to changes in annual breeding success associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Red lines indicate changes in abundance that could be detected given current monitoring effort.

Adult survival

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Common Guillemot populations could not be ruled out by the HRA screening process, the adult survival of the species is only monitored on the Isle of May, within the Forth Islands SPA, one of the SMP key sites (Figure 3.3.105). If a capture-mark-recapture scheme were set up at other Common Guillemot colonies following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.01 could be detected in the 10 years following wind farm construction (Figure 3.3.106). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Common Guillemot, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.020 could be detected over the 10 years following wind farm construction.

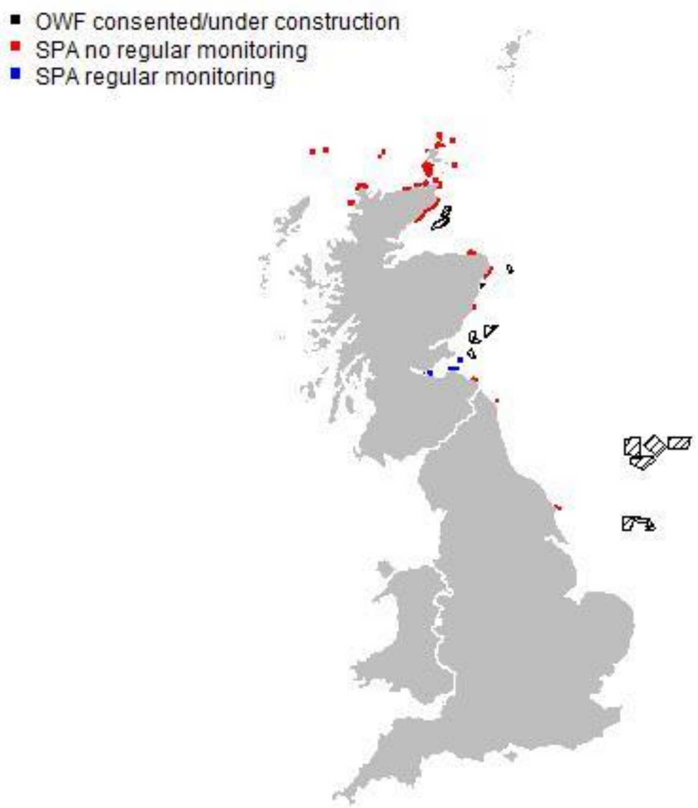

Figure 3.3.105 Special Protection Areas for Common Guillemot at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for 5 or 10 years pre-construction, it would be possible to detect changes in the adult survival rate of a magnitude likely to result in a more subtle population-level impact than could be detected through the monitoring of abundance alone. Even with a capture-mark-recapture study established three years prior to wind farm construction, it would be possible to detect a change in adult survival likely to result in a population-level impact equivalent to that which could be detected through the monitoring of abundance on the Farne Islands, and greater than that which could be detected through the monitoring of abundance on the Isle of May (Figure 3.3.107). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any change in abundance.

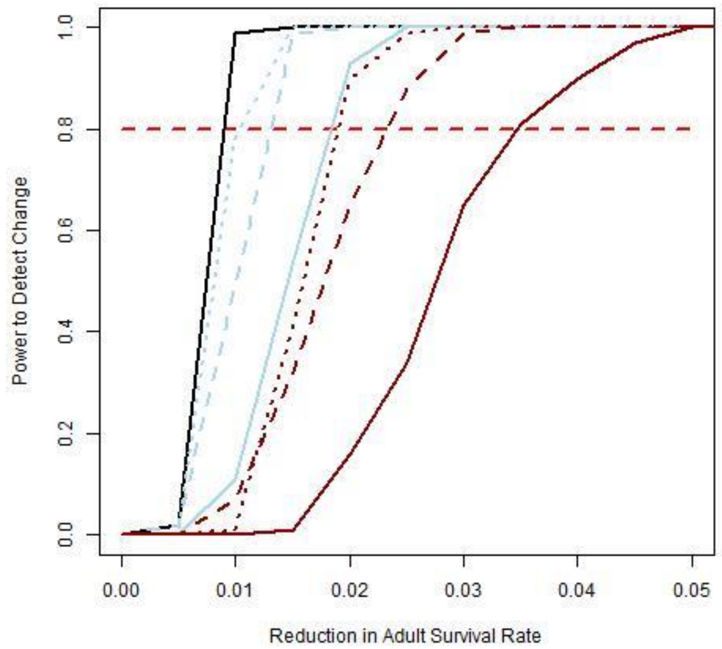

Figure 3.3.106 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Common Guillemot. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

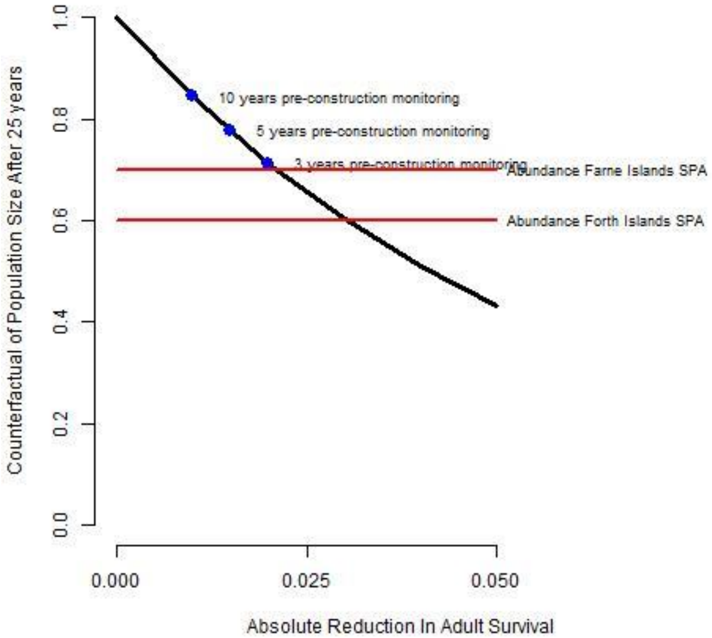

Figure 3.3.107 Counterfactual of population size after 25 years for Common Guillemots relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.15 Razorbill *Alca torda*

Abundance

Assessments for 14 offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Razorbill at a total of 10 SPAs (Figure 3.3.108, Table A3). At present, Razorbill abundance is monitored at six sites within two SPAs – Bass Rock, Craigleith, Fidra, The Lamb and the Isle of May within the Forth Islands SPA and, the Farne Islands SPA. At present, there is sufficient power to detect an impact attributable to an offshore wind farm which leads to a reduction in population size of 40% over 25 years on the Isle of May, 45% on Fidra and 50% on the Farne Islands (Figure 3.3.109).

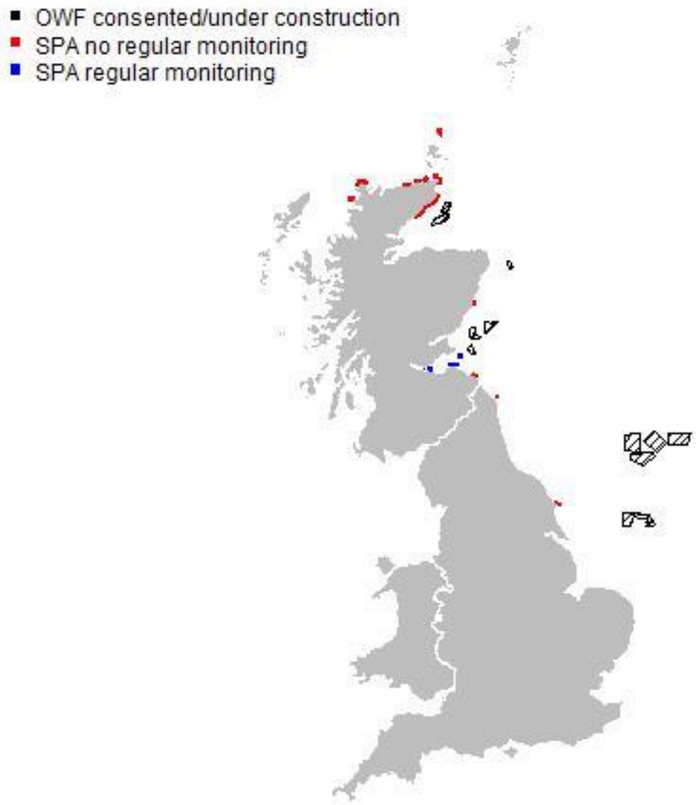

Figure 3.3.108 Special Protection Areas for Razorbills at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

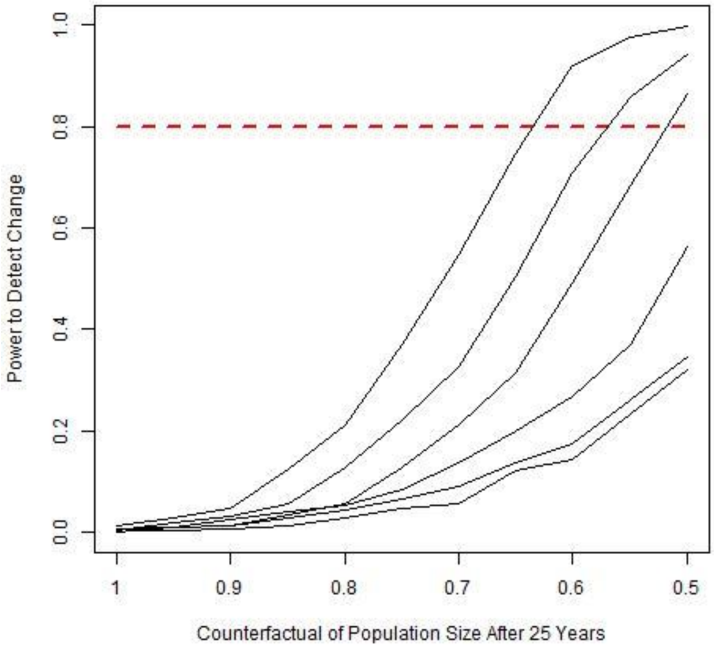

Figure 3.3.109 Power to detect a reduction of 0-50% in population size at Razorbill breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

At present, it would not be possible to detect a reduction in population size of 25% or less at any monitored Razorbill colonies (Figure 3.3.110). An effect likely to lead to a 50% reduction in population size over 25 years could be detected within 19 years on the Isle of May, 22 years on Fidra and 24 years on the Farne Islands. In order to detect these impacts, monitoring would have to be carried out annually at Fidra and the Farne Islands and biennially on the Isle of May (Figure 3.3.10).

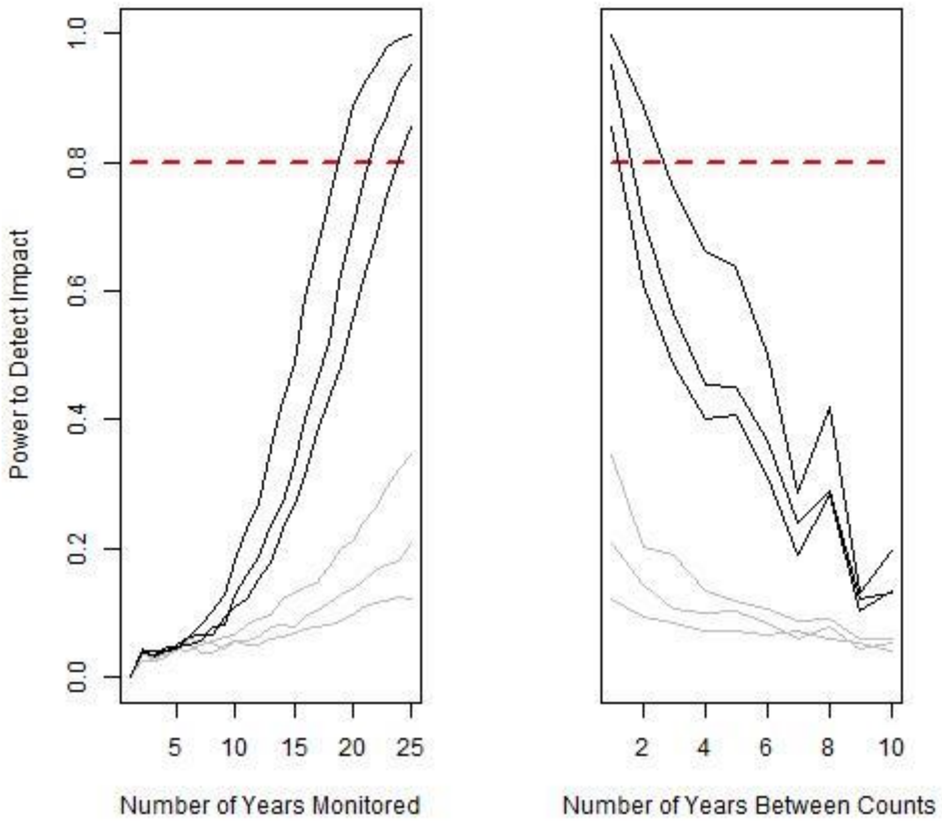

Figure 3.3.110 Number of years before an impact equivalent to a 25% (grey) or 50% (black) reduction in population size over 25 years could be detected at Razorbill breeding colonies assuming annual monitoring (left). Impact of count regularity on power to detect a 25% (grey) or 50% (black) reduction in population size over 25 years as a result of an effect from an offshore wind farm (right).

Breeding success

At present, Razorbill breeding success is monitored at three sites within SPAs at which LSEs associated with offshore wind farms could not be ruled out by the HRA screening process – Isle of May within the Forth Islands SPA, the Farne Islands SPA and Flamborough Head and Filey Coast SPA (Figure 3.3.111). There is relatively little annual variation in the levels of breeding success recorded at these sites (coefficients of variation of 15.5, 15.9 and 7.5 respectively). This is reflected in a high power to detect changes in breeding success, with sufficient power to detect a reduction of 15% at the Flamborough Head and Filey Coast SPA, 30% on the Isle of May and 40% on the Farne Islands (Figure 3.3.112). If recent monitoring carried out in West Westray and St Abb's Head SPAs were continued, it may be possible to detect changes in breeding success in Razorbills at these sites as well.

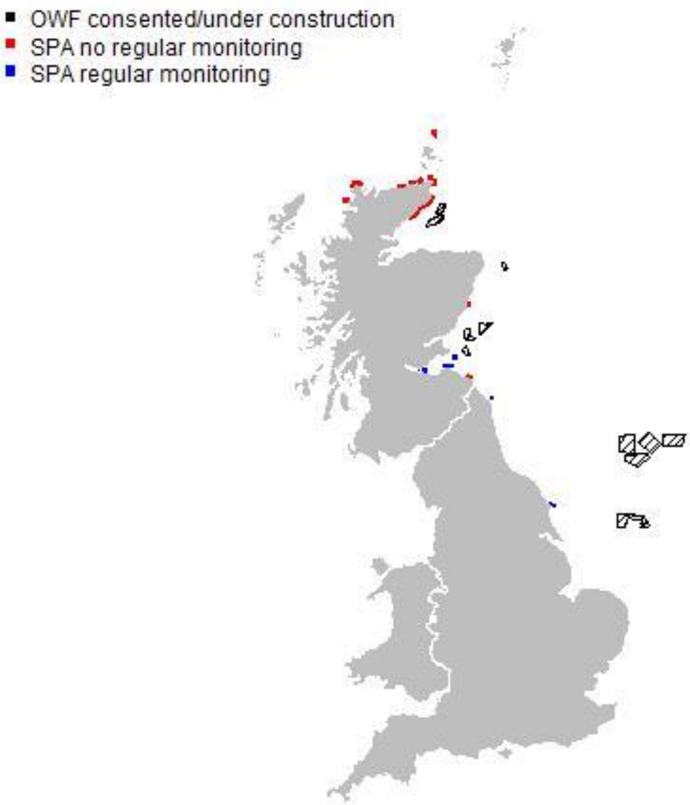

Figure 3.3.111 Special Protection Areas for Razorbill at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

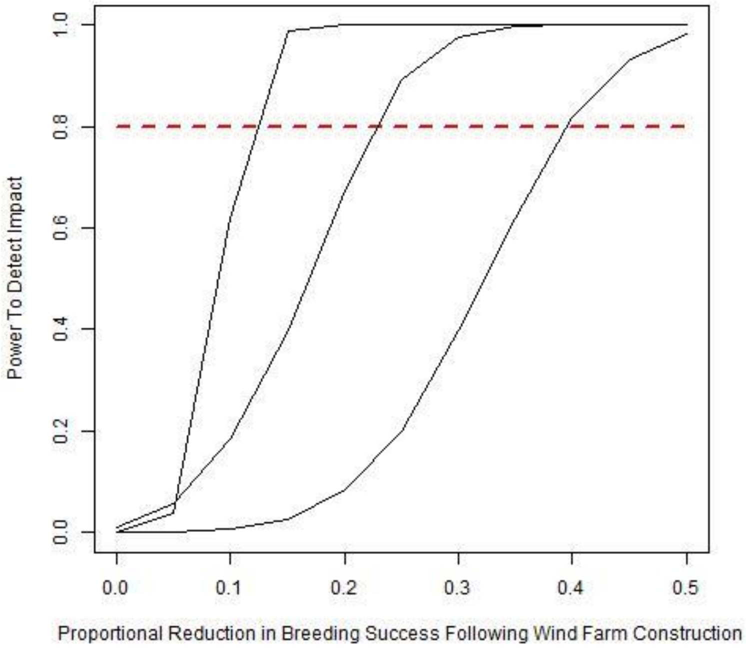

Figure 3.3.112 Power to detect a reduction of 0-50% in breeding success at Razorbill breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

The only colonies at which it was possible to detect changes in abundance and breeding success in Razorbills were the Farne Islands SPA and the Isle of May, within the Forth Islands SPA. In both cases, the minimum population-level impact that could be detected by monitoring breeding success is marginally greater than the minimum impact that could be detected through monitoring abundance (Figure 3.3.113).

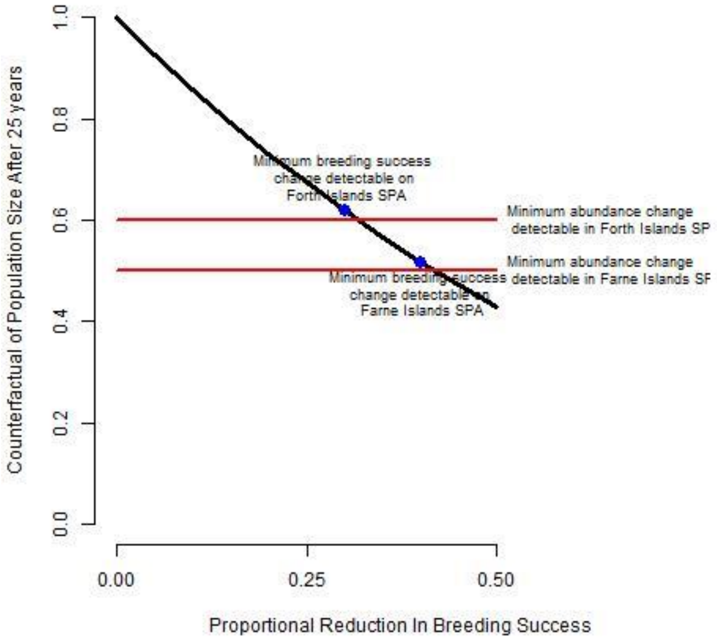

Figure 3.3.113 Counterfactual of population size after 25 years for Razorbills relative to changes in annual breeding success associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Red lines indicate changes in abundance that could be detected given current monitoring effort.

Adult survival

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Razorbill populations could not be ruled out by the HRA screening process, the adult survival of the species is only monitored on the Isle of May, within the Forth Islands SPA, one of the SMP key sites (Figure 3.3.105). If a capture-mark-recapture scheme were set up at other Razorbill colonies following the recommendations set out in Horswill *et al.* (2018) it is likely that a reduction in adult survival of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.115). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Common Guillemot, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.030 could be detected over the 10 years following wind farm construction.

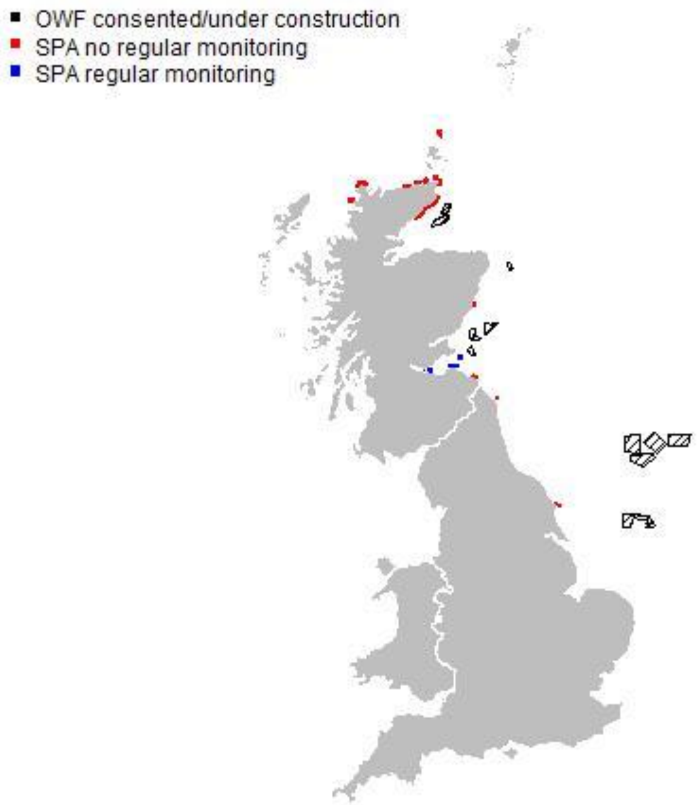

Figure 3.3.14 Special Protection Areas for Razorbill at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

With capture-mark-recapture studies running for 5 or 10 years pre-construction, it would be more possible to detect a population-level impact through monitoring of changes in adult survival rate than through the monitoring of abundance alone. Even with a capture-mark-recapture study established three years prior to wind farm construction, it would be possible to detect a population-level impact through monitoring of adult survival on the Farne Islands (Figure 3.3.116). It is also worth highlighting that significant changes in adult survival could be detected at an earlier stage than any change in abundance.

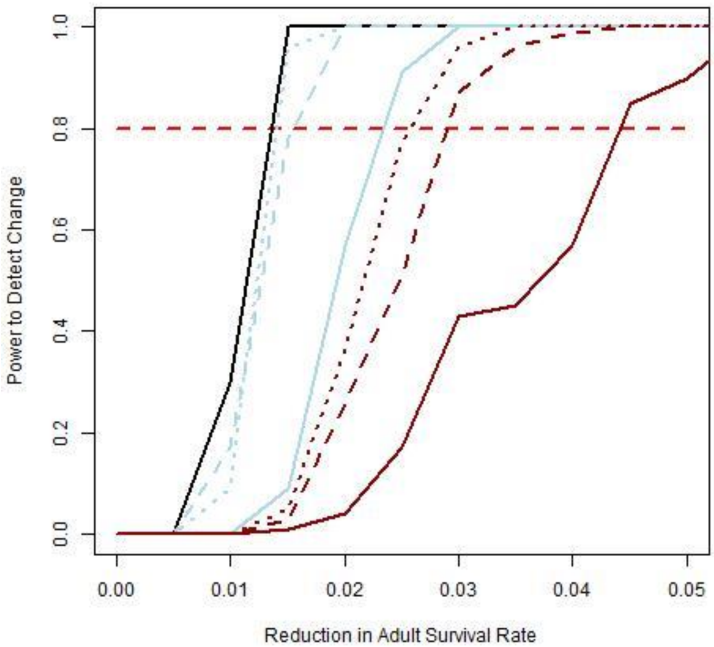

Figure 3.3.115 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Razorbill. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

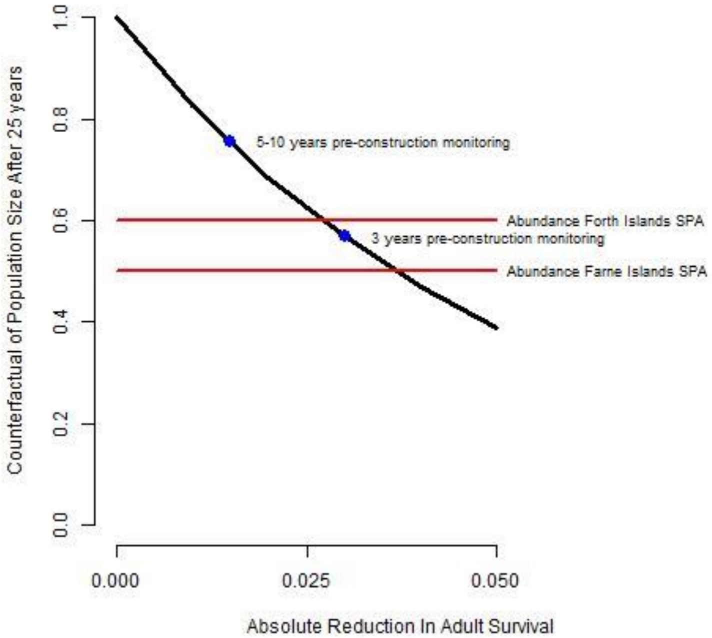

Figure 3.3.116 Counterfactual of population size after 25 years for Razorbills relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

3.3.16 Atlantic Puffin *Fratercula arctica*

Abundance

Assessments for 10 offshore wind farm projects concluded that LSEs could not be ruled out at a population level by HRA screening prior to AA for populations of Atlantic Puffin at a total of nine SPAs (Figure 3.3.117, Table A3). At present, Atlantic Puffin populations are only regularly monitored by the SMP at one of these SPAs, the Forth Islands SPA. Here, five sub-colonies have been counted regularly since 2000 – Craigleith, Fidra and smaller populations on The Lamb, the Bass Rock and Inchmickery. The largest colony, on the Isle of May, has only been counted three times since 2000. This is likely to reflect the difficulty in counting burrow-nesting seabirds.

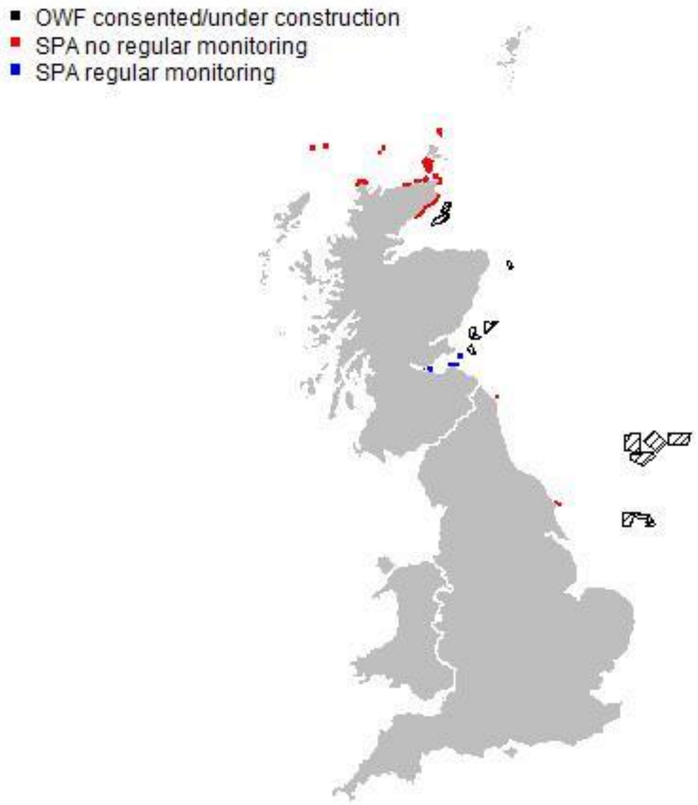

Figure 3.3.117 Special Protection Areas for Atlantic Puffins at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report.

At present, there is insufficient statistical power to detect a reduction in population size of 50% as the result of impacts arising from an offshore wind farm at any of the colonies regularly monitored within the Forth Islands SPA (Figure 3.3.118). This is likely to reflect strong annual variation in colony size at Fidra and Craigleith (e.g. from 1,466 Apparently Occupied Burrows at Fidra in 2003 to only 51 in 2006) and the small size of the remaining populations (e.g. only two Apparently Occupied Burrows on the Bass Rock in 2015).

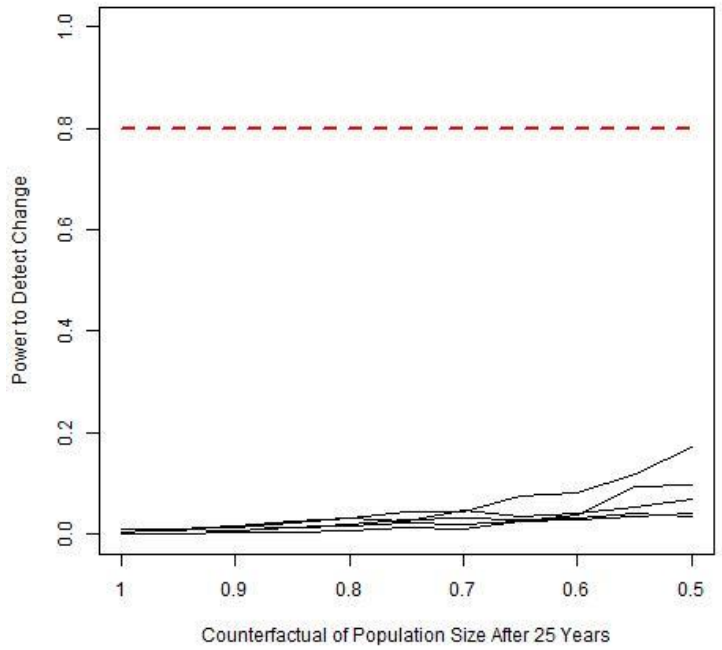

Figure 3.3.118 Power to detect a reduction of 0-50% in population size at Atlantic Puffin breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

Breeding success

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Atlantic Puffin populations could not be ruled out by the HRA screening process, the breeding success of the species is monitored at two colonies, the Farne Islands SPA and the Isle of May within the Forth Islands SPA (Figure 3.3.119). There is relatively little annual variation in breeding success at these colonies (coefficients of variation 22.8 and 16.5 respectively). This is reflected in a relatively high power to detect declines in breeding success that may be associated with the effects of offshore wind farms (Figure 3.3.120). On the Isle of May there is sufficient power to detect a 15% decline in breeding success attributable to offshore wind farms, while on the Farne Islands there is sufficient power to detect a 20% decline in breeding success. Changes in breeding success of these magnitudes would equate to counterfactuals of population size after 25 years of 0.79 and 0.73 respectively (Figure 3.3.121).

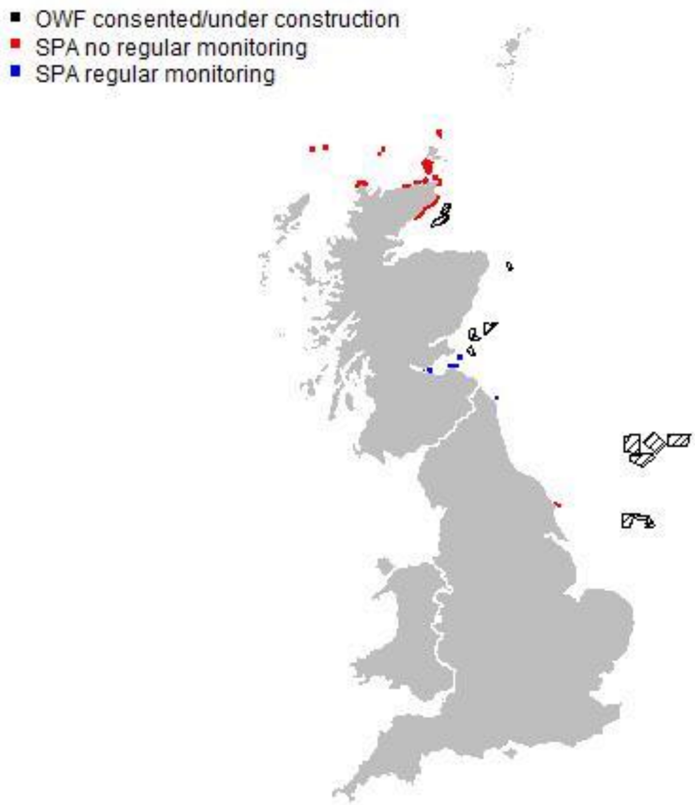

Figure 3.3.119 Special Protection Areas for Atlantic Puffin at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of breeding success.

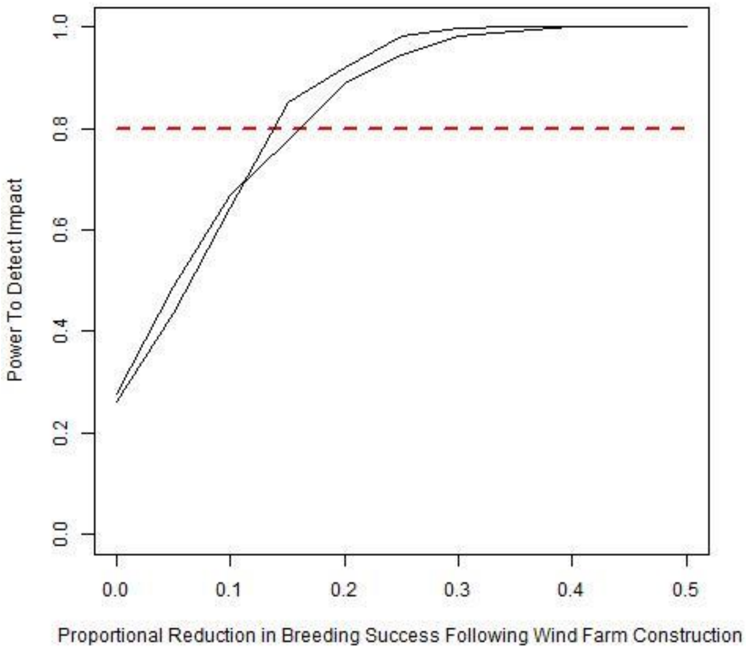

Figure 3.3.120 Power to detect a reduction of 0-50% in breeding success at Atlantic Puffin breeding colonies within SPAs over 25 years as a consequence of the effect arising from an offshore wind farm.

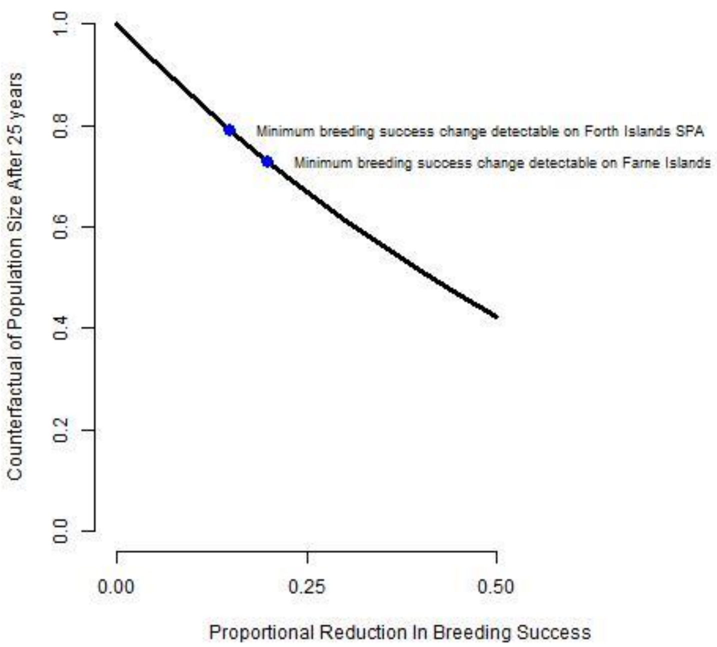

Figure 3.3.121 Power to detect a reduction in breeding success in Atlantic Puffin of 0-50% in colonies within SPAs where a LSE associated with an offshore wind farm could not be ruled out by the HRA screening process.

Adult survival

At present, at SPAs for which offshore wind farm assessments concluded that LSEs on Atlantic Puffin populations could not be ruled out by the HRA screening process, the adult survival of the species is monitored on the Isle of May, one of the SMP key sites (Figure 3.3.122). If capture-mark-recapture schemes were set up at additional Atlantic Puffin colonies following the recommendations set out in (Horswill *et al.* 2018) it is likely that a reduction in the adult survival rate of 0.015 could be detected in the 10 years following wind farm construction (Figure 3.3.123). However, recognising that it is unlikely that 10 years pre-construction adult survival data could be collected for Atlantic Puffin, if five years pre-construction data could be collected and, 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival of no less than 0.015 could be detected over the 10 years following wind farm construction. If only three years of pre-construction data could be collected then, assuming 400 birds were marked per year with a recapture rate of 0.8, a reduction in adult survival rate of no less than 0.025 could be detected over the 10 years following wind farm construction.

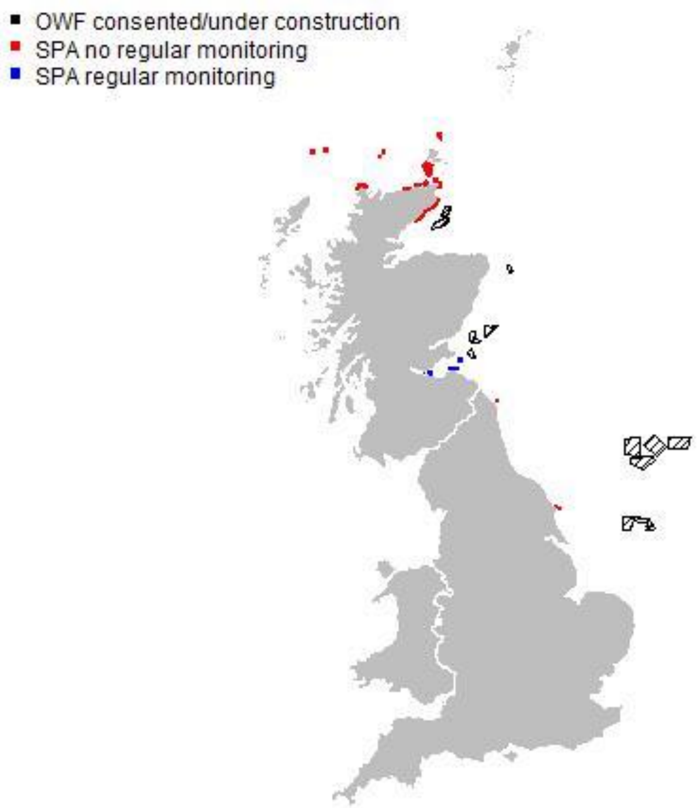

Figure 3.3.122 Special Protection Areas for Atlantic Puffin at which Likely Significant Effects could not be ruled out by Habitats Regulations Assessment or Appraisal screening for the offshore wind farms considered in this report and where there is regular monitoring of adult survival.

If mark-recapture data were available for the five10 years prior to the construction of the wind farm, then assuming 200 birds had been marked each year and a recapture rate of 0.6 was maintained, a reduction in the adult survival rate, as a result of effects associated with a wind farm, of 0.015 could be detected. Over 25 years, this would result in a counterfactual of population size of 0.76 (Figure 3.3.124). However, if only three years pre-construction data could be collected, the minimum impact that could be detected is a reduction in adult survival of 0.025, which would result in a counterfactual of population size after 25 years of 0.63.

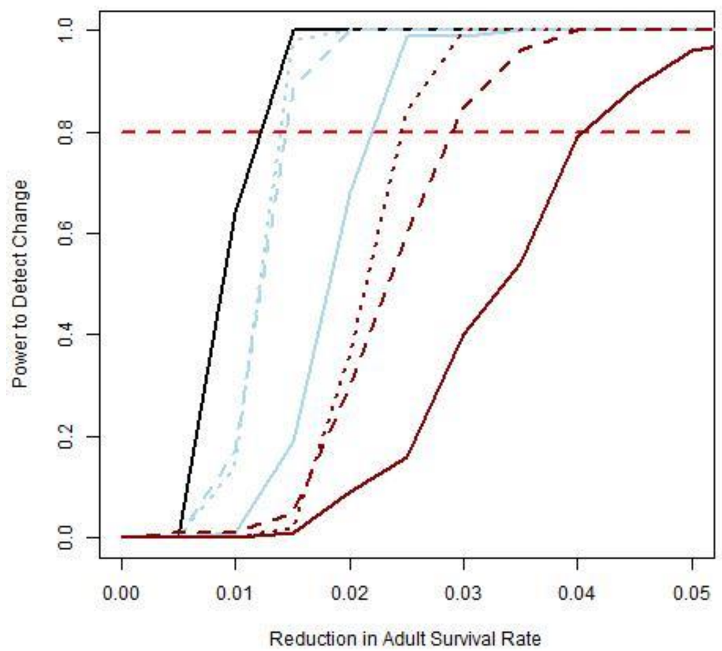

Figure 3.3.123 Power to detect changes in adult survival rate in response to the effects of an offshore wind farm for Atlantic Puffin. Black lines indicate the capture-mark-recapture scenario set out in (Horswill *et al.* 2018) with 10 years pre- and post-wind farm construction monitoring of adult survival with 200 birds marked per year and a recapture rate of 0.6. Blue lines indicate scenarios with 5 years pre- and 10 years post-wind farm construction monitoring, red lines indicate scenarios with 3 years pre- and 10 years post-wind farm construction monitoring. Solid lines indicate 200 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.6, broken lines indicate 400 birds marked per year with a recapture rate of 0.8.

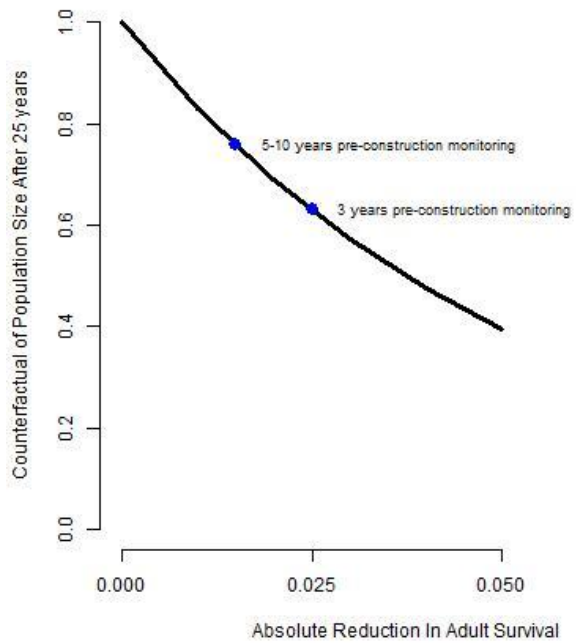

Figure 3.3.124 Counterfactual of population size after 25 years for Atlantic Puffins relative to changes in annual adult survival associated with an offshore wind farm. Blue dots reflect the minimum changes in breeding success that could be detected within each SPA given current monitoring effort. Blue dots reflect the minimum changes in adult survival that could be detected assuming 3, 5 or 10 years of pre-construction capture-mark-recapture data collection and 10 years of post-construction data collection.

4. DISCUSSION AND RECOMMENDATIONS

4.1 Discussion

Predicting the potential impacts of developments on birds is a core component of the consenting process for renewable energy projects, including offshore wind farms. Consequently, there is presently considerable research aimed at building the evidence base on key effects, such as collision, displacement and barrier effects (Dierschke *et al.*, 2016; Skov *et al.*, 2018; Thaxter *et al.*, 2018; Vanermen *et al.*, 2015). Ornithological monitoring programmes for consented projects have also been focussed on directly monitoring the possible impacts at wind farm sites and research has been undertaken to consider the power of these programmes to detect change (Maclean *et al.*, 2013; Vanermen *et al.*, 2015). The number of consented and operational wind farms in the North Sea, in particular, is increasingly rapidly. As a consequence, there is increasing concern that the cumulative effect of these multiple wind farms might have a significant impact on species at a population level (e.g. Brabant et al., 2015; Busch & Garthe, 2017). This increases the importance of understanding how individual colony populations respond to the effects associated with offshore wind farms. For example, a recent analysis highlights the potential for the cumulative collision risk associated with wind farms in the North Sea to negatively affect the Helgoland population of Black-legged Kittiwake (Busch & Garthe, 2017). However, despite population modelling being used to predict likely consequences associated with the construction and operational phases of an offshore wind farm, little consideration has been given to the quality/representativeness of the underlying demographic rates collected and, in particular, whether, given existing levels of monitoring at seabird colonies, it is possible to detect impacts of the magnitude predicted in relation to offshore wind farms, and validate predictions, such as those made by Busch & Garthe (2017).

This report focuses on the monitoring of breeding seabird features of SPAs and the recommendations which follow are informed by the analysis of data collected for these species. However, a key knowledge gap remains in relation to potential impacts on wintering features of SPAs, for example, seaduck and divers. Whilst such species were beyond the scope of this work, there is clearly a need to develop methodologies to better understand the impact that offshore wind farms may have on these species (e.g. Mendel et al., 2019) and, understand the linkages between breeding populations and the offshore wind farms that they may interact with during the winter.

Seabird monitoring in the UK has been delivered through a combination of periodic censuses (Cramp *et al.*, 1974; Lloyd *et al.*, 1991; Mitchell *et al.*, 2004) alongside annual monitoring of abundance, breeding and adult survival rates through the SMP and RAS scheme. This review aimed to assess the power of present monitoring of these key parameters to detect changes that might be associated with the effects of offshore wind farms.

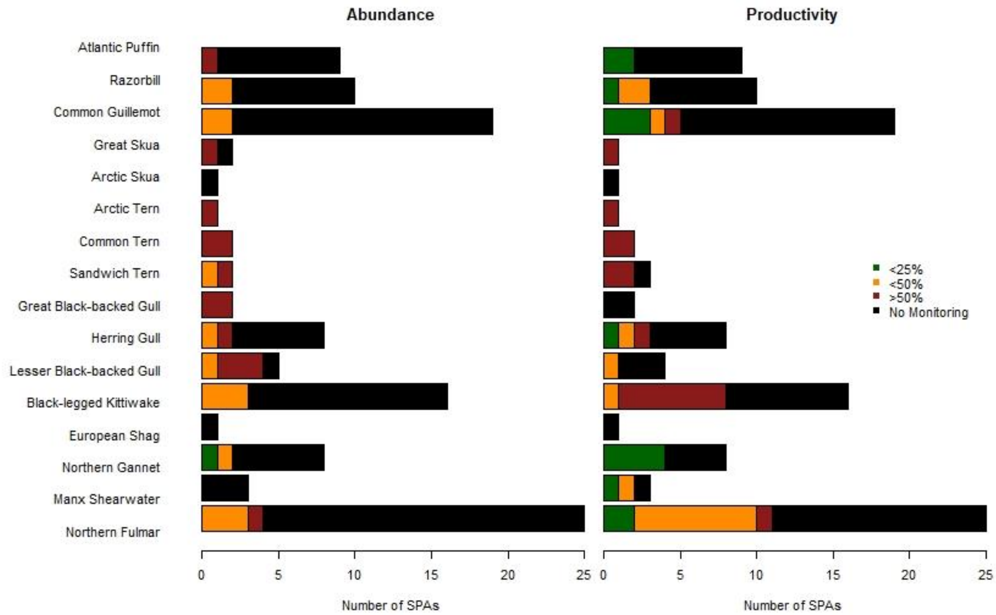

Figure 4.1.1 Number of colonies for each species at which the minimum change in abundance or productivity that can be detected is 25% or less, 25-50%, more than 50% or, at which there is no regular monitoring in place.

It should be highlighted that the analyses presented above, with respect to seabird abundance and breeding success, assess the power of existing monitoring to detect changes which are in addition to any underlying trends in the data. For the vast majority of species/SPAs at which LSEs associated with offshore wind farms could not be ruled out by the HRA screening process, present levels of monitoring are insufficient to detect the colony level changes that may be expected in response to the effects predicted in relation to offshore wind farms, reflecting a need to increase the frequency of monitoring for abundance and the frequency and effort for monitoring breeding success and survival (Figure 4.1.1 & Table 4.1.1). At present, it would only be possible to detect a reduction in colony size of 25% over 25 years, after current population trends are accounted for, in a single species at a single site – Northern Gannet within the Flamborough Head and Filey Coast SPA. Even in scenarios where the effects associated with offshore wind farms are predicted to lead to a 50% reduction in colony size over 25 years, after current population trends are accounted for, it would only be possible to detect these impacts in eight out of the 16 species for which LSEs could not be ruled out by the HRA screening process. The difficulty in detecting impacts on the abundance of seabirds which may be associated with offshore wind farms is likely to reflect a number of factors, including: the regularity of counts; the extent of variation around the underlying population trend; and, limitations in our understanding of the processes affecting population trends.

Table 4.1.1 Number of Special Protections Areas (SPAs) for each species at which LSEs associated with the impacts of offshore wind farms could not be ruled out by the HRA screening process and at which current monitoring programmes have sufficient power to detect a 25% or 50% decline in abundance and/or breeding success, in addition to any underlying trends in the data, which may be attributable to the effects associated with offshore wind farms. Values in italics reflect *the number of colonies at which it was possible to detect a change of the given magnitude/the total number of colonies for which data were available*. Values in bold reflect **the total number of SPAs for which change of a given magnitude could be detected in at least one colony.**

Species	Number of SPAs where LSEs could not be ruled out	Abundance		Breeding Success	
		25%	50%	25%	50%
Northern Fulmar	25	*0/9*	*5/9*	*2/11*	*10/11*
		0	**3**	**2**	**10**
Manx Shearwater	3	*0/0*	*0/0*	*1/2*	*2/2*
		0	**0**	**1**	**2**
Northern Gannet	8	*1/2*	*2/2*	*4/4*	*4/4*
		1	**2**	**4**	**4**
European Shag	1	*0/1*	*0/1*	*0/0*	*0/0*
		0	**0**	**0**	**0**
Black-legged Kittiwake	16	*0/7*	*5/7*	*0/8*	*1/8*
		0	**3**	**0**	**1**
Lesser Black-backed Gull	5	*0/8*	*1/8*	*0/2*	*1/2*
		0	**1**	**0**	**1**
Herring Gull	9	*0/8*	*2/8*	*2/3*	*2/3*
		0	**2**	**2***	**2***
Great Black-backed Gull	2	*0/13*	*0/13*	*0/0*	*0/0*
		0	**0**	**0**	**0**
Sandwich Tern	3	*0/2***	*1/2***	*0/2***	*0/2***
		0	**1**	**0**	**0**
Common Tern	2	*0/3*	*0/3*	*0/3*	*0/3*
		0	**0**	**0**	**0**
Arctic Tern	1	*0/1*	*0/1*	*0/1*	*0/1*
		0	**0**	**0**	**0**
Arctic Skua	1	*0/0*	*0/0*	*0/0*	*0/0*
		0	**0**	**0**	**0**
Great Skua	2	*0/0*	*0/0*	*0/1*	*0/1*
		0	**0**	**0**	**0**
Common Guillemot	19	*0/6*	*2/6*	*3/5*	*4/5*
		0	**2**	**3**	**4**
Razorbill	10	*0/6*	*3/6*	*1/3*	*3/3*
		0	**2**	**1**	**3**
Atlantic Puffin	9	*0/5*	*0/5*	*2/2*	*2/2*
		0	**0**	**2**	**2**

*At one site, monitoring has only recently been established and annual variation in breeding success may not be accurately characterised. Consequently, it is necessary to treat assessments of power to detect impacts on this species at this site with caution.
**For Sandwich Terns, values reflect the combination of multiple sites in order to better reflect the population status of the species, which resulted in a higher power to detect changes.

In simulations where colonies were not monitored on at least a biennial basis, power to detect any change attributable to an offshore wind farm dropped substantially. Given the magnitude of impacts predicted in relation to offshore wind farms, insufficient statistical power to detect a reduction in abundance of 25% or less is likely to result in an increased likelihood of a type II error – incorrectly rejecting the null hypothesis that the wind farm has not affected the population.

For several species, breeding success tended to be more widely monitored than abundance (Table 4.1.1). As seabirds are long-lived species, changes in demographic parameters, like adult survival and breeding success, may manifest themselves before they are reflected by changes in colony size (Cook *et al.*, 2014). Focussing on breeding success, we found evidence to support this in some species, but not others. In species such as Northern Gannet, Northern Fulmar and Manx Shearwater, where annual measurements of breeding success are relatively consistent year on year, there was a relatively high power to detect reductions in breeding success that may be attributable to the effects associated with offshore wind farms (Figure 4.1.2). At a colony level, the population level changes that could be detected through monitoring breeding success were more subtle than those could be detected through monitoring abundance. However, for other species, including terns and gulls, high intra-annual variability in breeding success meant that detecting changes that may be attributable to offshore wind farms was more difficult. Indeed, for the Forth Islands SPA, where comparison could be made between the changes in abundance and breeding success that could be detected in Black-legged Kittiwake, Lesser Black-backed Gull and Herring Gull, the minimum population level changes that could be detected through monitoring breeding success exceeded those could be detected through monitoring abundance. In the case of terns, which are known to move colonies between years, considering populations at a regional, rather than colony-specific level appeared to improve power to detect changes breeding success. However, this was still not at a level beyond which could be detected through looking at abundance.

Our analyses highlighted that, with appropriate monitoring, it is possible to detect relatively small changes in adult survival rate for all species (i.e. a reduction of between 0.010 and 0.020). The population-level consequences of such reductions, over the 25 year lifespan of an offshore wind farm, would be more subtle than could be detected through monitoring abundance. At present, survival is the most poorly monitored of the parameters we considered, with data from only five species at three sites where LSEs in relation to offshore wind farms could not be ruled out by the HRA screening process – Black-legged Kittiwake, Common Guillemot and Razorbill on the Isle of May within the Forth Islands SPA, Manx Shearwater on Skomer, within the Skomer, Skokholm and Seas off Pembrokeshire SPA and Lesser Black-backed Gull within the Alde-Ore Estuary SPA. This is likely to reflect the significant effort required to attach rings and subsequently recatch/resight the marked population. However, our analyses also suggest that, with appropriate ringing and recapture/resighting effort, it would be possible to detect a change in adult survival with 3-5 years pre-construction data; this indicates that, whilst far from ideal, if RAS-type studies could be established towards the end of the consenting process it would still be possible to detect changes in adult survival likely to result in a significant impact at a population-level. Such studies would provide valuable data in relation to baseline survival rates against which any impacts associated with offshore wind farm(s) could be assessed at a colony level. The value of RAS-type studies could be further enhanced if data collection were extended to chicks, enabling the estimation of age-specific survival rates (Horswill et al., 2018). This is important as, when assessing the impacts of an offshore wind farm at a site-level, those impacts must be apportioned to adult and immature birds separately. At present, this can prove challenging because, in many seabird species, it can be hard to distinguish between adult and immature birds. If we were able to estimate age-specific survival rates for immature seabirds, it may be possible to make inferences about the age-structure of the population.

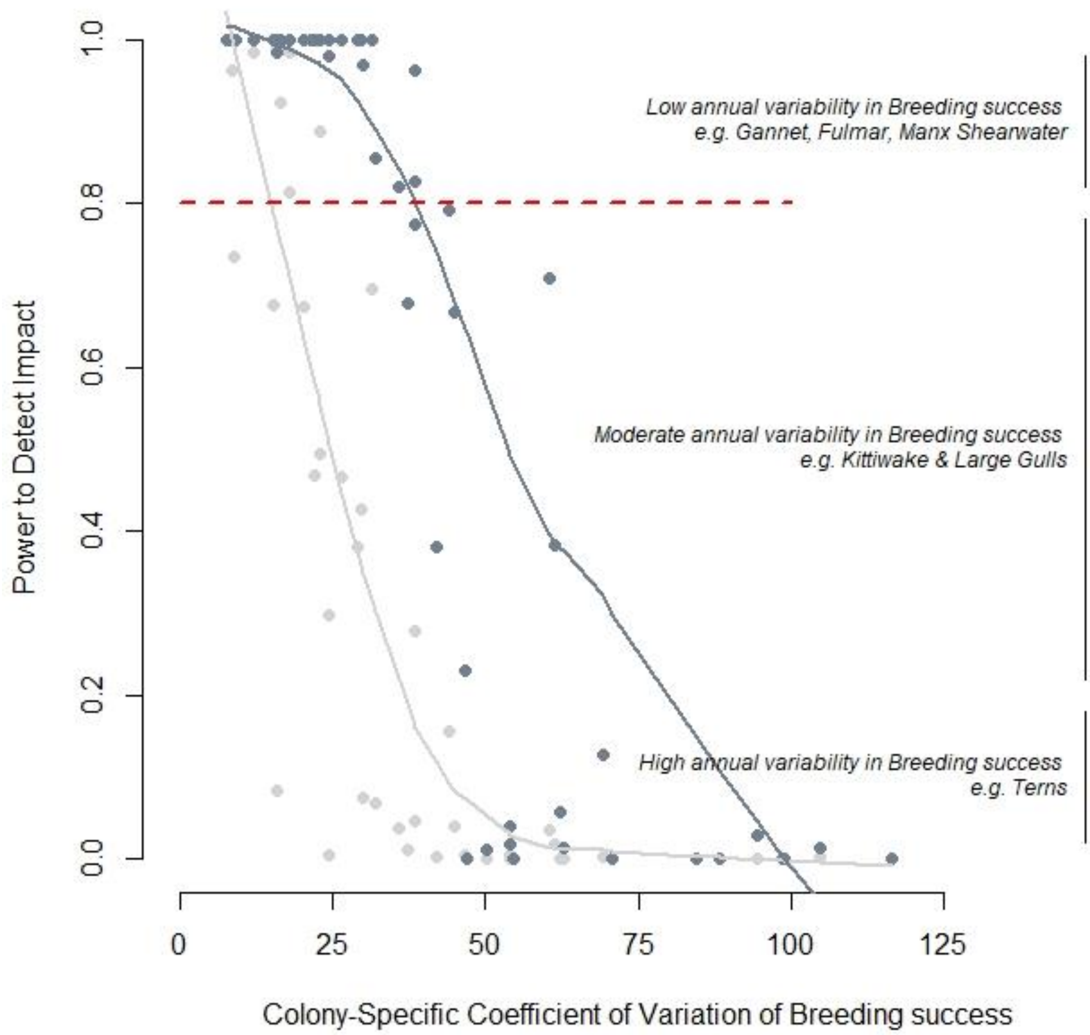

Figure 4.1.2 Power to detect a 25% (light grey line & points) or 50% (dark grey line & points) change in breeding success in relation to annual variability in breeding success.

There is substantial underlying annual variation in the existing monitoring data that are available to quantify the seabird impacts associated with offshore wind farms. Such variation is likely to contribute to the low power of existing monitoring data to detect, and quantify, any impacts attributable to an offshore wind farm that might be additional to existing trends. However, by accounting for some of this variation, through the inclusion of additional covariates in models, quantifying any impacts may become easier (Fewster *et al.*, 2000). For example, an analysis of Black-legged Kittiwake populations on the Isle of May demonstrated that the inclusion of an environmental covariate (sea surface temperature) improved the ability of models to detect impacts of fisheries on demographic parameters (Frederiksen *et al.*, 2004). As multiple pressures are acting on seabird populations (Burthe *et al.*, 2014), if we are able to better account for some of these in the modelling process, we may be able to better quantify the impacts that offshore wind farms, and other anthropogenic activities, have on seabird populations. As approaches such as the use of camera systems to record collisions (e.g. Skov et al., 2018) and spatial modelling of survey data to investigate potential displacement (e.g. Mendel et al., 2019) are developed, there is a need to link these impacts back to the populations concerned. This could be achieved through tools such as tracking (e.g. Thaxter et al., 2018) which enable impacts on individual birds to be put in the context of the population as a whole, offering a more comprehensive understanding of wind farm impacts. This challenge is likely to become more pressing as the number, and size, of offshore wind farms increases in coming years (Busch & Garthe, 2016, 2017).

Our analyses highlight the potential to improve both monitoring and the analysis of data which are collected as part of monitoring. Whilst our analyses focus on the potential impacts of offshore wind farms, the conclusions are applicable more generally, e.g. to the assessment of the impacts of wave and tidal power on seabirds.

4.2 Recommendations

It is clear that, if we are to further increase the likelihood of detecting population-level impacts of the magnitudes predicted in the AAs for offshore wind farms, improved monitoring, at both a spatial and temporal scale, is required. Within many of the SPAs where LSEs could not be ruled out by the HRA screening process, we identified significant gaps in existing monitoring effort (Table A4). This means that establishing baseline population abundance estimates for the species concerned at the sites affected is extremely challenging.

These recommendations are based on an analysis of populations where LSEs could not be ruled out at a population level by HRA screening prior to AA in relation to offshore wind farms which are under construction or, for which planning consent has been authorised. However, they are also applicable to situations where LSEs cannot be ruled out for sites and species in relation to future developments.

I. **It is vital that the current national seabird census 'Seabirds Count' is supported. This will provide vital baseline information about the population status of the UK's seabirds. Amongst other uses, this will provide robust baseline population estimates for species and sites at which LSEs in relation to offshore wind farms cannot be ruled out by the HRA screening process.**

For many species, there is broad inter-annual variability between counts. Where colonies are not monitored on a regular basis, this can make detecting significant effects associated with offshore wind farms very difficult. This is particularly true for species like terns, where individuals may move between different sites within the same region from year to year. Ensuring annual counts of key colonies can be logistically challenging, particularly when these are in remote locations and/or hard to reach islands. However, our analyses suggest that having biennial, rather than annual counts of key sites is unlikely to have a detrimental effect on our ability to detect meaningful population changes in most species. In the case of terns, our analyses suggest that monitoring at a regional scale is likely to improve our ability to detect impacts at a population-level.

II. **Existing monitoring of abundance by the SMP should be expanded so that abundance of all species within SPAs where LSEs could not be ruled out by the HRA screening process is monitored at least once every two years.**
III. **In the case of terns, it is important to monitor populations at an ecologically coherent scale (e.g. Cook et al., 2011), as well as at an individual colony level. In some cases (e.g. in relation to those breeding within the Forth Islands SPA), this may include birds that nest outside a SPA.**

Species demography and population trends are likely to be influenced by a range of anthropogenic and environmental factors (Burthe *et al.*, 2014; Frederiksen *et al.*, 2004). The power to detect any changes attributable to offshore wind farms could be improved by better understanding how these factors influence population processes at a site and/or colony level. This could be achieved through a modelling approach (e.g. Fewster *et al.*, 2000). Better understanding of these processes may also enable the wider development of more advanced models for use in the consenting process (Freeman *et al.,* 2014).

IV. **Consideration (within a modelling framework) of the anthropogenic and environmental factors influencing seabird population processes at a sites and/or colony level will increase the power of monitoring data to detect any changes attributable to the presence of an offshore wind farm by accounting for some of the unexplained variation in existing data sets.**

As may be expected, monitoring demographic parameters may enable more subtle population-level impacts to be detected. However, this varies by species. In species such as Northern Gannet, Northern Fulmar, Manx Shearwater and the auks, there is relatively high power to detect changes in breeding success that would lead to more subtle population-level impacts than could be detected by monitoring abundance. This is particularly important for burrow-nesting species like Manx Shearwater and Atlantic Puffin, whose ecology makes obtaining robust population estimates particularly challenging. However, there is also value in monitoring breeding success in species such as large gulls and Black-legged Kittiwake, where annual estimates are more variable. In addition to allowing us to obtain baseline estimates of productivity, and the annual variability around those estimates, this may help us to understand the extent to which any impacts observed at a population-level are driven by impacts on breeding success and/or adult survival.

V. **Existing monitoring of breeding success by the SMP should be expanded and, consideration by scheme organisers should be given to use of data from the BTO/JNCC Nest Record Scheme.**

VI. **Monitoring of breeding success in burrow-nesting seabirds like Manx Shearwater and Atlantic Puffin should be expanded to selected other sites as a priority. This will give us valuable demographic data for species whose abundance can be hard to monitor.**

VII. **It is important the monitoring of breeding success in Northern Gannet, Northern Fulmar and auk species is expanded to selected other sites. The changes that can be detected in breeding success would result in more subtle population-level impacts than those that could be detected through monitoring of abundance.**

VIII. **Expanding monitoring of breeding success for gulls, skuas and terns may help us to understand how any impacts at a population-level, in response to the effects associated with offshore wind farms, are driven by changes in breeding success and/or adult survival. This could be achieved through the use of population models incorporating any recorded offshore wind farm impacts on breeding success and/or survival.**

There is considerable variability in the recorded levels of breeding success for many species. This variability is likely to affect our ability to detect changes in breeding success that may be related to the impacts associated with an offshore wind farm. This variability may be reduced by ensuring a higher proportion of nests within a colony are monitored on an annual basis and by ensuring that the plots in which breeding success is monitored are consistent between years. We sought to examine the relationship between the proportion of a colony in which breeding success was monitored and the annual variability in breeding success. However, inconsistencies in the breeding success data and the abundance data meant that we were unable to do this. There were a number of other cases in which the number of nests where breeding success was monitored exceeded the total number of nests within the colony. There are likely to be good reasons for this. For example, discrepancies between the site boundaries in which abundance and breeding success are monitored or nests which are monitored failing prior to the whole colony count being undertaken. However, it is important that, going forwards, there is greater integration between the abundance and breeding success monitoring (and, indeed, adult survival monitoring).

IX. **The SMP database should include a clear site definitions and structure, showing linkages between sites covered for abundance and demographic monitoring data, e.g. plots/sub-sites monitored for breeding success and adult survival should be linked to the sites monitored for abundance. The number of nests monitored for breeding success should be directly relatable to the whole colony count, e.g. it must be possible to express the nests monitored for breeding success as a proportion of the whole colony. If this proportion exceeds 1, a clear reason must be given for this (e.g. a re-nesting attempt by a pair which had previously failed).**

As might be expected, monitoring of adult survival can lead to the detection of changes likely to lead to population-level impacts that would not be apparent through the monitoring of abundance. Furthermore, there is the possibility to detect those changes earlier than would be the case through any changes that could be detected through monitoring abundance, meaning the data have the potential to act as an early-warning signal for what is happening at a population-level. However, such data are likely to require substantial effort and investment to collect. In the small number of instances where there is ongoing monitoring of adult return rates (which can be used to estimate adult survival rates), it is important that this effort is maintained. Whilst ideally, 10 years of pre-construction data would be available (Horswill *et al.*, 2018), our analyses demonstrate that studies with at least 5 years pre-construction data can still provide useful data. The typical time period between consent for an offshore wind farm being authorised and construction beginning appears to be 3-5 years (e.g. 5 years London Array, 3 years East Anglia One and Walney Extension[8]). Assuming that programmes for data collection can be put in place towards the end of the consenting process, by which stage the species and sites for which LSEs cannot be ruled by the HRA screening process out are likely to have been agreed between regulators and developers, valuable pre-construction data can be collected. Such data would be of immense value in establishing baseline survival rates, against which the impacts associated with offshore wind farms could be assessed at both an individual site and cumulative scale. Previous authors (Furness & Wanless, 2014) have highlighted the importance of gathering better adult survival data for the Northern Gannet and, our analyses further highlight the value of this, both for Northern Gannet and other species, especially Black-legged Kittiwake. RAS studies that use colour-ringing may be of particular value in generating recapture/resighting data for adult survival analysis. This is particularly true for species like large gulls and terns, which are highly visible and therefore, may generate large numbers of resightings. This is particularly important as large gulls are amongst the species perceived to be at greatest risk from offshore wind farms (Furness *et al.*, 2013; Garthe & Huppop, 2004) and, for which existing monitoring is particularly lacking.

X. **Existing monitoring of adult survival, through the SMP key site programme and RAS studies must be maintained and expanded. The value of such studies could be further enhanced through the ringing of chicks as, over time, this would provide a better understanding of the age structure of populations.**

XI. **Additional RAS studies (including studies based on colour-ringing) should be instigated in order to measure changes in adult survival in response to effects associated with offshore wind farms. These should be instigated towards the end of the consenting process for the relevant wind farms at the latest and ideally, as part of baseline data collection, in order to ensure sufficient data can be collected.**

[8] https://www.4coffshore.com/

XII. **Species such as Northern Gannet, Black-legged Kittiwake and large gulls, which are believed to be particularly sensitive to offshore wind farms with respect to collision, should be prioritised for adult survival studies.**

XIII. **As pre-construction data are likely to be limited, where possible, studies should include approaches, such as colour-ringing, which are likely to maximise recapture/re-sighting rates.**

XIV. **Consideration should be given to the potential for added value for any capture-mark recapture study; for example, the widespread deployment of geolocators in order to understand wintering areas for key species and, the potential for interaction with wind farms outside the breeding season.**

XV. **By monitoring multiple parameters within the same site, we are likely to increase the power of the data to detect any changes which may be associated with offshore wind farms. Consequently, careful consideration should be given to identifying instances where abundance, adult survival and breeding success can feasibly be collected together in order to enable integrated population monitoring for species at individual sites. Such an approach is likely to be most feasible for Northern Gannet, Black-legged Kittiwake and the large gulls.**

At present, much of the seabird monitoring in the UK relies on volunteer effort. If seabird monitoring is going to be expanded in the way that is needed in order to understand how offshore wind farms may affect populations, it is vital that volunteers are given the support and infrastructure needed to achieve this. This includes ensuring that the IT infrastructure to collate these data is properly maintained and providing financial support in order to enable regular visits to colonies and the purchase of equipment (e.g. colour rings).

XVI. **In order to monitor seabird populations within colonies at the level required in order to detect any effects associated with offshore wind farms, it is vital that sufficient support, both in terms of finance and infrastructure, is provided to those who collect the data (many of whom currently do so on a voluntary basis).**

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[9] https://www.gov.uk/government/publications/uk-offshore-energy-strategic-environmental-assessment-research-projects

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APPENDIX 1

Table A1 List of offshore wind farms considered in this report, according to the stage of planning/development, leasing round and web links to Appropriate Assessments (AAs). Construction status of wind farms correct at time of analysis (27 November 2018).

Offshore Wind Farm (ordered clockwise starting due north	**Stage**[1]	**Leasing round**[2]	**AA document**
Dounreay Tri	Consent Authorised	SFD	https://www2.gov.scot/Resource/0051/00515496.pdf
Beatrice (BOWL)	Under Construction	STW1	https://www2.gov.scot/Resource/0044/00446505.pdf
Moray (MORL)	Consent Authorised	R3	https://www2.gov.scot/Resource/0044/00446526.pdf
Aberdeen Offshore Wind Farm (European Offshore Wind Deployment Centre)	Under Construction	D	https://www2.gov.scot/Resource/0041/00417113.pdf
Kincardine Offshore Windfarm Project	Consent Authorised	SFD	https://www2.gov.scot/Resource/0053/00535406.pdf
Seagreen Alpha	Consent Authorised	R3	https://www2.gov.scot/Resource/0046/00460542.pdf
Seagreen Bravo	Consent Authorised	R3	https://www2.gov.scot/Resource/0046/00460542.pdf
Inch Cape	Consent Authorised	STW1	https://www2.gov.scot/Resource/0046/00460542.pdf
Neart na Gaoithe	Consent Authorised	R3	https://www2.gov.scot/Resource/0046/00460542.pdf
Dogger Bank Creyke Beck A	Consent Authorised	R3	http://www.forewind.co.uk/uploads/Habitats%20Regulations%20Assessment.PDF
Dogger Bank Creyke Beck B	Consent Authorised	R3	http://www.forewind.co.uk/uploads/Habitats%20Regulations%20Assessment.PDF
Dogger Bank Teesside A	Consent Authorised	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010051/EN010051-002090-Habitats%20Regulations%20Assessment.pdf
Sofia (formerly Dogger Bank Teesside B)	Consent Authorised	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010051/EN010051-002090-Habitats%20Regulations%20Assessment.pdf

Offshore Wind Farm (ordered clockwise starting due north	Stage[1]	Leasing round[2]	AA document
Hornsea Project One	Under Construction	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010033/EN010033-002059-Hornsea%20Offshore%20Wind%20Farm%20Final%20EA%20including%20HRA%20TA%20and%20AIUGI.pdf
Hornsea Project Two	Pre-Construction	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010053/EN010053-002079-Habitats%20Regulation%20Assessment
Triton Knoll	Consent Authorised	R2	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010005/EN01000five000014-Habitats%20Regulations%20Assessment.pdf
East Anglia One	Under Construction	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010025/EN01002five000008-Habitat%20Regulations%20Assessment%20(HRA).pdf
East Anglia Three	Consent Authorised	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010056/EN010056-002381-East%20Anglia%20THREE%20Habitats%20Regulations%20Assessment%20Dated%207%20August%202017.pdf
Galloper	Partial Generation/Under Construction	R2.5	https://infrastructure.planninginspectorate.gov.uk/document/1814936
Rampion	Partial Generation/Under Construction	R3	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010032/EN010032-001702-Rampion%20Environmental%20Assessment%20Report.pdf
Walney Extension	Partial Generation/Under Construction	R2.5	https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010027/EN010027-000013-Habitats%20Regulations%20Assessment.pdf

[1] https://www.4coffshore.com/windfarms/windfarms.aspx?windfarmId=UK36;

[2] Leasing rounds: R1= Round 1 (Crown Estate), R2 =Round 2 (Crown Estate), R2.5 =Round 2.5 (Crown Estate), R3 = Round 3 (Crown Estate), SFD = Scottish Floating Demonstration, STW1 = Scottish Territorial Waters 1, D= Demonstration.

Table A2 Species features of Special Protection Areas (SPAs)[1] for which a Likely Significant Effect (LSE) could not be ruled out by the Habitats Regulations Assessment or Appraisal (HRA) screening process for offshore wind farms considered in this report (as shown in grey shading).

Offshore Windfarms[2]	SPA	Qualifying features for which a LSE was identified (BOU order)																									
		Red throated Diver	European Storm Petrel	Leach's Storm Petrel	Northern Fulmar	Manx Shearwater	Northern Gannet	European Shag	Great Cormorant	Black-legged Kittiwake	Black-headed Gull	Common Gull	Great Black-backed Gull	Herring Gull	Lesser Black-blacked Gull	Sandwich Tern	Little Tern	Roseate tern	Common Tern	Arctic Tern	Arctic Skua	Great Skua	Common Guillemot	Razorbill	Black Guillemot	Atlantic Puffin
Dounreay Tri	Buchan Ness to Collieston Coast																									
	Calf of Eday																									
	Cape Wrath																									
	Copinsay																									
	East Caithness Cliffs																									
	Fair Isle																									
	Fetlar																									
	Flannan Isles																									
	Forth Islands																									
	Foula																									
	Fowlsheugh																									
	Handa																									
	Hermaness, Saxa Vord and Valla Field																									
	Hoy																									
	Marwick Head																									
	Mingulay and Berneray																									
	North Caithness Cliffs																									
	North Rona and Sula Sgeir																									
	Noss																									
	Rousay																									
	St Kilda																									
	Sule Skerry and Sule Stack																									

Offshore Windfarms[2]	SPA	Qualifying features for which a LSE was identified (BOU order)																								
		Red throated Diver	European Storm Petrel	Leach's Storm Petrel	Northern Fulmar	Manx Shearwater	Northern Gannet	European Shag	Great Cormorant	Black-legged Kittiwake	Black-headed Gull	Common Gull	Great Black-backed Gull	Herring Gull	Lesser Black-blacked Gull	Sandwich Tern	Little Tern	Roseate tern	Common Tern	Arctic Tern	Arctic Skua	Great Skua	Common Guillemot	Razorbill	Black Guillemot	Atlantic Puffin
Dounreay Tri (cont'd)	Sumburgh Head																									
	Shiant Isles																									
	Troup, Pennan and Lion's Head																									
	West Westray																									
Beatrice (BOWL)	East Caithness Cliffs																									
	North Caithness Cliffs																									
	Hoy																									
Moray (MORL)	East Caithness Cliffs																									
	North Caithness Cliffs																									
	Hoy																									
Aberdeen Offshore Wind Farm	Buchan Ness to Collieston Coast																									
	Caithness and Sutherlands Peatlands																									
	Fair Isle																									
	Forth Islands																									
	Foula																									
	Fowlsheugh																									
	Hermaness, Saxa Vord and Valla Field																									
	Hoy																									
	Loch of Strathbeg																									
	Noss																									
	Orkney Mainland Moors																									
	Otterswick and Graveland																									
	Ronas Hill- North Roe and Tingon																									
	Ythan Estuary, Sands of Forvie and Meikle Loch																									

Offshore Windfarms[2]	SPA	Qualifying features for which a LSE was identified (BOU order)																								
		Red throated Diver	European Storm Petrel	Leach's Storm Petrel	Northern Fulmar	Manx Shearwater	Northern Gannet	European Shag	Great Cormorant	Black-legged Kittiwake	Black-headed Gull	Common Gull	Great Black-backed Gull	Herring Gull	Lesser Black-blacked Gull	Sandwich Tern	Little Tern	Roseate tern	Common Tern	Arctic Tern	Arctic Skua	Great Skua	Common Guillemot	Razorbill	Black Guillemot	Atlantic Puffin
Kincardine Offshore Windfarm Project	Buchan Ness to Collieston Coast				■					■				■									■			
	Flamborough Head and Filey Coast						■																			
	Forth Islands				■																					■
	Fowlsheugh				■					■				■									■	■		
	Troup, Pennan and Lion's Head				■					■				■									■			
Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Buchan Ness to Collieston Coast				■					■				■									■			
	Fowlsheugh				■					■				■									■	■		
	Forth Islands				■		■			■				■	■				■ [3]	■ [3]			■	■		■
	St Abb's Head to Fast Castle									■				■									■	■		
Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B	Forth Islands				■		■			■					■								■	■		■
	Farne Islands									■													■			■
	Flamborough Head and Filey Coast						■			■													■	■		■
	Flamborough Head and Filey Coast pSPA				■		■			■													■	■		
Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)	Farne Islands									■													■	■		■
	Flamborough Head and Filey Coast						■			■													■	■		■
	Flamborough Head and Filey Coast pSPA						■			■													■	■		■
	Forth Islands						■			■													■	■		■
	Fowlsheugh									■																
Hornsea Project One	Flamborough Head and Filey Coast						■			■				■									■	■		
	Flamborough Head and Filey Coast pSPA				■		■			■				■									■	■		■
Hornsea Project Two	Flamborough Head and Filey Coast									■																
	Flamborough Head and Filey Coast pSPA				■		■			■													■	■		■
	Forth Islands				■					■																
	Fowlsheugh				■																					

		Qualifying features for which a LSE was identified (BOU order)																								
Offshore Windfarms[2]	SPA	Red throated Diver	European Storm Petrel	Leach's Storm Petrel	Northern Fulmar	Manx Shearwater	Northern Gannet	European Shag	Great Cormorant	Black-legged Kittiwake	Black-headed Gull	Common Gull	Great Black-backed Gull	Herring Gull	Lesser Black-blacked Gull	Sandwich Tern	Little Tern	Roseate tern	Common Tern	Arctic Tern	Arctic Skua	Great Skua	Common Guillemot	Razorbill	Black Guillemot	Atlantic Puffin
Triton Knoll	Flamborough Head and Filey Coast																									
	North Norfolk Coast																									
East Anglia One	Alde-Ore Estuary																									
	Flamborough Head and Filey Coast																									
East Anglia Three	Alde-Ore Estuary																									
	Flamborough Head and Filey Coast																									
	Flamborough Head and Filey Coast pSPA																									
Galloper	Alde-Ore Estuary																									
Rampion	Flamborough Head and Filey Coast																									
Walney Extension	Aberdaron Coast and Bardsey Island																									
	Bowland Fells																									
	Copeland Islands																									
	Morecambe Bay & Duddon Estuary																									
	Ribble and Alt Estuaries																									
	Skomer, Skokholm and the Seas off Pembrokeshire																									

[1] Note, here and elsewhere in this report, we refer to the present names of SPAs where these may have changed to reflect boundary amendments;

[2] The exception to this was Aberdeen Offshore Wind farm, where the ES had not contained sufficient information to make an assessment and for which further information was required for a selected number of species to inform the AA;

[3] Neart na Gaoithe and Inch Cape only.

Table A3 Species features of Special Protection Areas for which a Likely Significant Effect (LSE) could not be ruled out by the Habitats Regulations Assessment or Appraisal (HRA) screening process for offshore wind farms considered in this report and the associated effects for which LSEs were predicted.

Species (BOU order)	Offshore Wind Farm[1]	**LSE**			
		Collision	**Displacement**	**Barrier**	**Habitat loss**
Red-throated diver	Aberdeen Offshore Wind Farm	Y	Y		
Northern Fulmar	Dounreay Tri	Y	Y		
	Beatrice (BOWL)	Y	Y		
	Moray (MORL)	Y	Y		
	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project	Y	Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y	Y		
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B	Y	Y	Y	
	Hornsea Project One	Y			
	Hornsea Project Two		Y		
Manx Shearwater	Walney Extension		Y		
Northern Gannet	Dounreay Tri	Y	Y		
	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project	Y	Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y	Y		
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B	Y	Y		Y
	Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)	Y	Y		Y
	Hornsea Project One	Y			
	Hornsea Project Two	Y	Y		
	Triton Knoll	Y			

Species (BOU order)	Offshore Wind Farm[1]	LSE			
		Collision	Displacement	Barrier	Habitat loss
Northern Gannet (cont'd)	East Anglia One	Y			
	East Anglia Three	Y			
	Rampion	Y			
European Shag	Aberdeen Offshore Wind Farm	Y			
Black-legged Kittiwake	Dounreay Tri	Y	Y		
	Beatrice (BOWL)	Y	Y		
	Moray (MORL)	Y	Y		
	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project	Y	Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y	Y		
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B	Y			
	Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)	Y			
	Hornsea Project One	Y			
	Hornsea Project Two	Y			
	Triton Knoll	Y			
	East Anglia One	Y			
	East Anglia Three	Y			
	Rampion	Y			
Great Black-backed Gull	Dounreay Tri	Y			
	Beatrice (BOWL)	Y			
	Moray (MORL)	Y			
Herring Gull	Dounreay Tri	Y			
	Beatrice (BOWL)	Y			

Species (BOU order)	Offshore Wind Farm[1]	LSE			
		Collision	Displacement	Barrier	Habitat loss
Herring Gull (cont'd)	Moray (MORL)	Y			
	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project	Y	Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y			
	Hornsea Project One	Y	Y		
	East Anglia One	Y			
Lesser Black-backed Gull	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y			
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B	Y			
	Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)				
	East Anglia One	Y			
	East Anglia Three	Y			
	Galloper	Y			
	Walney Extension	Y			
Sandwich Tern	Aberdeen Offshore Wind Farm	y			
	Triton Knoll	Y			
Common Tern	Aberdeen Offshore Wind Farm	Y			
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y	Y		
Arctic Tern	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe	Y	Y		
Arctic Skua	Beatrice (BOWL)	Y			
	Moray (MORL)	Y			
Great Skua	Dounreay Tri	Y	Y		
	Beatrice (BOWL)	Y			
	Moray (MORL)	Y			

Species (BOU order)	Offshore Wind Farm[1]	LSE			
		Collision	**Displacement**	**Barrier**	**Habitat loss**
Common Guillemot	Dounreay Tri	Y	Y		
	Beatrice (BOWL)		Y		
	Moray (MORL)		Y		
	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project	Y	Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe		Y		
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B		Y		
	Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)		Y		
	Hornsea Project One		Y		
	Hornsea Project Two		Y		
Razorbill	Dounreay Tri	Y	Y		
	Beatrice (BOWL)		Y		
	Moray (MORL)		Y		
	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project		Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe		Y		
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B		Y		
	Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)		Y		
	Hornsea Project One		Y		
	Hornsea Project Two		Y		
Atlantic Puffin	Dounreay Tri	Y	Y		
	Beatrice (BOWL)		Y		
	Moray (MORL)		Y		

Species (BOU order)	Offshore Wind Farm[1]	LSE			
		Collision	Displacement	Barrier	Habitat loss
Atlantic Puffin (cont'd)	Aberdeen Offshore Wind Farm	Y			
	Kincardine Offshore Windfarm Project		Y		
	Seagreen Alpha, Seagreen Bravo, Inch Cape and Neart na Gaoithe		Y		
	Dogger Bank Creyke Beck A and Dogger Bank Creyke Beck B		Y		
	Dogger Bank Teesside A and Sofia (formerly Dogger Bank Teesside B)		Y		
	Hornsea Project One		Y		
	Hornsea Project Two		Y		

[1] For the Aberdeen Offshore Wind farm, the Environmental Statement had not contained sufficient information to make an assessment and further information was required for a selected number of species to inform the AA.

Table A4 Species features of Special Protection Areas for which a Likely Significant Effect (LSE) could not be ruled out by the Habitats Regulations Assessment or Appraisal (HRA) screening process for offshore wind farms considered in this report and the existing monitoring of abundance (A), breeding success (B) and survival (S) of these species' populations.

	Northern Fulmar	Manx Shearwater	Northern Gannet	European Shag	Black-legged Kittiwake	Lesser Black-backed Gull	Herring Gull	Great Black-backed Gull	Sandwich Tern	Common Tern	Arctic Tern	Arctic Skua	Great Skua	Common Guillemot	Razorbill	Atlantic Puffin
Aberdaron Coast and Bardsey Island		B														
Alde-Ore Estuary						B,S	A,B									
Bowland Fells																
Buchan Ness to Collieston Coast				A	B											
Calf of Eday																
Cape Wrath																
Copinsay	A													B		
East Caithness Cliffs								A								
Fair Isle	B		A,B													
Farne Islands					A,B									A,B	B	B
Fetlar																
Flamborough Head and Filey Coast	B		A,B		B		B							B	B	
Flannan Isles																
Forth Islands	A,B				A,B,S	A,B	A,B			A,B	A,B			A,B,S	ABS	A,B
Foula	B															
Fowlsheugh	A				B											
Handa	B				B								A,B	B		
Hermaness, Saxa Vord and Valla Field	B		B													
Hoy	A							A								
Loch of Strathbeg																
Marwick Head					A,B									B		
Mingulay and Berneray	A															
Morecambe Bay and Duddon Estuary						A										

	Northern Fulmar	Manx Shearwater	Northern Gannet	European Shag	Black-legged Kittiwake	Lesser Black-backed Gull	Herring Gull	Great Black-backed Gull	Sandwich Tern	Common Tern	Arctic Tern	Arctic Skua	Great Skua	Common Guillemot	Razorbill	Atlantic Puffin
North Caithness Cliffs																
North Norfolk Coast									A,B							
North Rona and Sula Sgeir																
Noss	B		B													
Ribble and Alt Estuaries						A										
Rousay	B															
Skomer, Skokholm and the Seas off Pembrokeshire		B,S														
St Abb’s Head					A											
St Kilda																
Sumburgh Head	B															
Sule Skerry and Sule Stack																
The Shiant Isles																
Troup, Pennan and Lion’s Heads	B															
West Westray																
Ythan Estuary, Sands of Forvie and Meikle Loch									A,B	A,B						