From:	Brown, Emma (NE)
To:	Hornsea Project Three
Subject:	Hornsea Project Three Deadline 4: NE Response to Further Examiners" Questions.
Date:	15 January 2019 21:21:54
Attachments:	13483_MB0138 Furness et al 2013.pdf EN 10080 NE Hornsea Project Three Deadline 4 Submission - Response to further ExA Questions.pdf

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Good Evening,

Please find attached Natural England's Deadline 4 Response to the Further Examiners' Questions and Requests for information.

Along with this I am attaching 'Furness et al, 2013' which we refer to in our response.

In addition to this, we had hoped to provide a pdf copy of 'Roberts et al. 2016' as requested by the Examiner, but unfortunately it is too large to email across, even as a compressed file (17MB). Please note that this document is freely available on the Natural England website at http://publications.naturalengland.org.uk/publication/5970080978960384 and as such I would be grateful if you could consider accessing this document to submit into the examination as per the request.
Failing this I try and find an alternative means to submit it.

Finally, I just wanted to highlight that we have identified that a number of the documents the Examiner has requested are subject to copyright and as such we have been advised that we cannot submit them, given that all submissions are to be published on your website.

Kind regards,

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Please note I currently work Monday - Thursday

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THE PLANNING ACT 2008

THE INFRASTRUCTURE PLANNING (EXAMINATION PROCEDURE) RULES 2010

HORNSEA PROJECT THREE OFFSHORE WIND FARM

Planning Inspectorate Reference: EN010080

NATURAL ENGLAND

WRITTEN SUBMISSION FOR DEADLINE 4
Natural England's Response to the Examining Authority's Further Written Questions and Requests for Information

Dated 15th January 2019

1. Alternatives and design flexibility

Ref:	Question to	Question

2. Ecology – Offshore

Ref:	Question to	Questions	
		Ornithology	
Q2.2.1	Applicant, Natural England (NE)	Please produce a draft Statement of Common Ground (SoCG) for ornithology at Deadline 6 that includes but is not limited to the following headings: Baseline Characterisation Collision Risk Model Band Model Options Maximum Likelihood Estimates Nocturnal Activity Factors Avoidance Rates Flight Height Estimation	Natural England notes this request and will seek to progress this with the applicant. Natural England also notes that the applicant submitted a ‘ Appendix 16 - Ornithology roadmap with Natural England for the examination phase’ [REP3-025]. We did not have the opportunity to input into on comment on this roadmap prior to its submission. Overall we consider this to be a helpful document, but would welcome the inclusion of the requests/and requirements of other interested parties and the Examining Authority to give a clear indication of the information that will be provided at each stage. To aid the development of the roadmap, Natural England has provided the Applicant with a list of outstanding data

		Flight Speed Estimation Biological Seasons Migratory Species Predicted Displacement Mortality Likely Significant Effects In Combination Screening Population Viability Assessment Impact Apportioning Where you cannot reach agreement you should state that your position is final and will not be resolved.	requirements. A copy of which is provided in Annex 2 below which we provided to the Applicant on 9/1/2019.
Q2.2.2	NE, Royal Society for the Protection of Birds (RSPB)	Notwithstanding the use of two out of four cameras, do you agree that the digital aerial survey data forms an adequate ornithological baseline for the months where data were collected over two separate years?	Data collected to inform baseline characterisation should allow the production of the most reliable, precise and accurate estimates possible of bird numbers at the Hornsea Three project site. In regards precision, survey designs should carefully consider the target precision and devise sampling strategies that collect sufficient samples to deliver this for all focal species and for all relevant seasons. As previously noted in our written representations (REP1-211, section 2.20) the Applicant indicated during the Evidence Working Group process that the proposed coverage of 10% would be sufficient for achieving a CV of 16% or better for abundance estimates. Thus indicating that the target precision is a CV of 16%. Natural England has not seen information on the precision of the estimates for the final digital

			aerial survey dataset (using just two cameras) as the Applicant has not presented these data in the Application documents, but the levels of precision achieved for the population estimates that Natural England have seen via Evidence Working Group reports have typically been considerably lower, with CVs greater than 16% (and in many cases much higher). Natural England are not in a position to comment as to whether the analysis of all four cameras would achieve the target level of precision. However, in principle, if the analysis of the data from the all four cameras demonstrates that the target levels of precision in the derived population estimates are attained (for focal species in all relevant seasons) then the digital aerial surveys for the months where data were collected over two separate years forms an adequate baseline to quantify bird abundance and distribution. Natural England also advise that site-specific flight height data for those species at risk of collision should be obtained as part of the baseline characterisation. The original intention was to utilise flight height data collected as part of the digital aerial surveys for Hornsea Three, however it came to light during the data analysis period that there were errors within the flight height calculation method used by the Digital aerial surveyor, and therefore could not be used to provide flight height data. See APP-109, sections 1.3.4.4-1.3.4.5 (6.5.5.3 ES Volume 5 - 5.3 - Collision Risk Modelling). This eventuality was beyond the control of the applicant, however, for completeness it should be noted that the digital aerial data collected for Hornsea Three does not form a complete ornithological baseline, as there is a lack of site specific flight height data.

Q2.2.4	NE	It was highlighted at ISH2 that you have assessed the likely significant impact of other offshore wind farm (OWF) projects with less than two years baseline data. How were you able to advise on potential adverse effects on European sites under these circumstances? If you were able to do this for previous projects why are you unable to provide the necessary advice in this instance?	Natural England's advice to developers when planning baseline surveys for offshore wind farms or extension projects, and for decision-makers to determine whether surveys have been carried out to an acceptable standard is that, as a minimum, offshore ornithology survey programmes should provide 24 consecutive months of full survey coverage. This is because there can be considerable variability in the numbers of birds present in an offshore area between months and between years, and therefore characterising the use of a project area by a species requires multiple years of data. Prior to the commencement of the Hornsea Project Three baseline surveys, Natural England advised the Applicant that two complete years of monthly surveys was the minimum required for quantifying the ornithological baseline for the Project. Previously, Natural England has been required to provide advice on potential ornithological impacts for a limited number of Offshore Wind Farms (OWFs) where a developer has collected/analysed, less than 24 consecutive months of baseline survey data. In each case, Natural England raised concerns that a baseline of less than 24 months was not adequate for these OWFs, and would make it difficult for us to draw any conclusions regarding EIA or HRA based on these data However, Natural England made a case specific assessment for each of these OWFs, taking account of:

			• The pattern and extent of missing data. In the majority of cases the collection of less than 24 months of data was not a planned survey decision – missing coverage generally occurred as a result of poor weather conditions preventing individual surveys from taking place, or being completed, and in many instances additional surveys were then undertaken to compensate for a missed survey (for example 2 surveys taken during the month following a missed survey. Missing survey coverage did not typically affect a block of consecutive months or a complete season for a key species; • The applicability of additional sources of distribution and abundance data available. Where additional contemporaneous and/or historical datasets were available Natural England agreed these could either be included in the assessment, or used to contextualise the OWF survey data and assessment conclusions • The species and SPAs involved and the level of impact that would be required to reach AEoSI. On the whole when Natural England provided advice on these cases, none of the species and SPAs included in the assessments had ALREADY reached a point where Natural England could not rule out AEoSI (at an in-combination level, based on previous OWF assessments); In these cases Natural England felt able to advise that either: it was satisfied that reliance on the available data did not present a risk of underestimating the project's effects; or that even with the most precautionary assumptions regarding potential impact levels, that the impact would not reach a level that would be considered AEoSI for the species and SPAs in question.

			As the level of potential impact to seabird populations has increased, and our understanding of seabird abundance and distribution data has evolved, the need for adequate evidence to inform assessments has been further emphasised. In considering the specific case of Hornsea Project Three, based on the impact assessment presented by the Applicant, and the data and information provided in the application documents, Natural England are unable to advise on potential adverse effects on European sites for some species and SPAs. While our concerns about the Applicant's assessment are not exclusively as a result of the incomplete baseline data, this is a significant component, and Natural England initially highlighted to the Applicant in March 2016 that the incomplete baseline would increase the risk that we would not be able to reach conclusions regarding the impact assessment. In regards Hornsea Project Three the following applies: • The pattern and extent of missing data. The incomplete baseline dataset is not the result of individual survey dates being missed due, for example, to poor weather conditions. It has arisen from a decision taken by the developer to not collect 24 months of survey data for Hornsea Project Three. Furthermore, the original intention of the Applicant was to only collect 12-18 months of data, and Natural England were asked to comment on assessments at PEIr that were based on only 11 months of survey data.

			A result of the planned nature of the missing survey months (rather than a non-systematic pattern of missing data resulting from surveys missed due to poor weather, for example), is that there is a block of consecutive months where there is only one year of survey data. For some species, this coincides with the whole of a particular assessment season being un-surveyed (for example, the pre-breeding seasons for gannet, kittiwake and razorbill) While we have advised in previous cases that two complete years of baseline survey data are required, our advice regarding the methodologies for baseline surveys has evolved over time as more empirical evidence has become available – for example, regarding the levels of inter-annual variability in bird numbers at offshore sites, distribution patterns, behaviour, survey methods. In particular there is now a considerable body of evidence from offshore sites regarding the spatial and temporal variability in seabird numbers, including variability between years. This includes information from the Hornsea Zone that indicates that there is significant inter-annual variability in numbers of the key receptor species that we are interested in at Hornsea Project Three. • <u>The applicability of additional sources of distribution and abundance data available</u>. While there are historical datasets available for the Hornsea Zone, that can provide useful contextual information and could have been used to inform the survey design for Hornsea Project Three, Natural England do not consider that these datasets can be integrated with the Hornsea Three baseline data, as the Applicant has done.

			Natural England has set out the reasons for this in REP1-211. These include the incompatibility of the datasets dues to different survey platforms being used (boat-based versus digital aerial), spatial coverage, age of data and sample size issues. Since the Applicant's assessment is based on survey data that integrates historical data with the Hornsea Project Three data, Natural England is unable to provide advice based on these assessments. • The species and SPAs involved and the level of impact that would be required to reach AEoSI. Finally, in the case of Hornsea Project Three Natural England is in a position where we consider cumulative and in-combination totals of key seabird species to have reached levels which we consider to be significant (and in some cases have been unable to advice no AEoSI) and therefore it is increasingly important that subsequent project assessments that contribute to the in-combination totals are as precise and accurate as possible, with every effort taken to reduce the uncertainty in the impact assessment. Accurate information about the cumulative and in-combination impacts will be important to inform consideration of mitigation (or where applicable compensation) measures.
Q2.2.6	RSPB, NE	The Data Hierarchy Report [APP-110] indicates that more limited variation in bird density was observed between December and March.	The Data Hierarchy report presents a number of partial datasets relevant to the 'data hierarchy' method described (Natural England to do accept this method, please refer to extensive comments in regards the data hierarchy method in our Relevant Rep [RR-097] and Written Rep [REP1-213, REP1-211].)

		Is it reasonable to assume that missing data for this period would have less impact on the confidence in the modelling than missing data from other months? As the principal ornithological issue relates to the effect of the project on the breeding bird assemblage at the Flamborough and Filey Coast Special Protection Area (SPA), why does it matter if there is missing data between December and March? In practical terms, how is the additional information you are seeking likely to alter the conclusions of the Environmental Statement (ES) and Habitat Regulations Assessment (HRA)?	The report presents, for each focal species some or all of the following: bird densities, flying bird densities, population estimates and mean of seasonal peaks derived from the digital aerial survey data, and from some (but not all) boat based survey data sets. The report does not present a comparison of within month (or season) inter-annual variation, indeed it would not be possible to compare levels of inter-annual variation for Dec-March using the digital aerial data, as only one year of data exists. While it may be possible to conduct an analysis of the level of inter-annual variation in bird densities derived from boat based survey data collected in the Hornsea Zone, and compare Dec-March to other months, this is not something presented within the report. As such, Natural England does not agree that referenced report demonstrates that ‘more limited variation in bird density was observed between December and March’. Confidence in modelling outputs is closely linked to the level of uncertainty in key modelling inputs, missing data increases uncertainty. In the case of collision risk modelling, one of the key input parameters is the density of birds in flight, and collision risk is calculated on a monthly basis (using monthly densities). As such we have substantially reduced confidence in the modelled outputs for the months where only one year of data exists (Dec-March). Whether these months are more or less important in the final assessment of impact depends on the population scale/legislation (see below). In the case of displacement assessment (presumed to be included as ‘modelling’), the impact of having four consecutive months missing (irrespective of when they fall in the year) as opposed to

			four non-consecutive months results in a greater impact to the confidence of the assessment. This is due to the use of seasonal mean of peaks in the calculation of displacement figures, and the need to have sufficient data points within a ‘season’ to make the analysis meaningful. The principle ornithological issues relate to the effect of the project on the qualifying features (breeding gannet, guillemot, kittiwake, razorbill and the breeding seabird assemblage) of FFC SPA. It should be noted that Natural England advise that March is considered part of the breeding season for Kittiwake, Gannet and Guillemot. The seabird populations breeding at FFC SPA are protected throughout the year, and hence a proportion of the potential effects of Hornsea Project Three are apportioned to these populations throughout the year. It is true to say that in the non-breeding months the level of apportioning is substantially lower (than in the breeding season), and therefore the relative contribution of these months to the overall impact is decreased. The principle ornithological issues are at an in-combination level and so while the contribution of the non-breeding season may be less than the breeding season, it is the combination of all seasons and projects that need to be considered. Conducting robust in-combination (or cumulative) assessments is extremely challenging, as more and more uncertainties combine. One of our key aims is to increase certainty (and hence decrease the need for precautionary assessments), and the introduction of additional, avoidable uncertainty (e.g. due to the planned failure to collect sufficient data to inform the baseline characterisation) is unacceptable.

			In practical terms, the effect of the missing data on the impact assessment will depend on the nature of the impact being assessed. In the case of displacement impacts the assessment depends on being able to select a peak abundance estimate for each season, and there being at least two years of data for a particular season to allow a mean peak across the years to be calculated. This is clearly not possible where there are only data for a particular season from one year. Even where there are data from some months within a defined season across two consecutive years, there is the possibility that the peak count may have occurred in the missing month/s and so the calculation of a mean peak of counts will be an under-estimate. For collision estimates the effect of the missing data is difficult to predict as collision risk assessment relies on having density data from 24 months to use in the calculation of collision values. Missing months could result in higher or lower collision predictions compared to a full 24 month dataset. As such, it is not possible to comment on how the conclusions may be altered within the ES and HRA if the additional data had been collected (i.e. 24 months of consecutive survey data, notwithstanding queries over the precision of the data/the analysis of all four cameras). The collection of the additional data may result in an increase or decrease in the calculated collision risk and displacement effects, and may result in an alteration to the conclusions (either of greater or lesser impact).
Q2.2.8	NE	Given your stated position in relation to the baseline characterisation and the fact that you are unable to conclude beyond reasonable	Although it is acceptable to discuss compensation measures in principle and without prejudice prior to an Appropriate Assessment (AA) or Habitats Regulations Assessment (HRA), it is important to recognise that **Compensatory measures** can only be *formally* considered after a negative assessment under regulation 63* and

		scientific doubt that the integrity of European sites would not be affected by the proposal, please suggest any feasible compensation measures that would be needed for Flamborough and Filey Coast SPA.	where in the absence of alternatives and there are imperative reasons of overriding public interest (regulation 64*), and the competent authority is minded to approve the plan or project (*Regulation 25 and 26 in Offshore Regs). In this scenario it is the duty of the relevant Secretary of State to secure such compensatory measures as is necessary to ensure the overall coherence of Natura 2000 is protected. The relevant SNCB(s) role is to advise on the effectiveness of the proposed compensatory measures and whether they are likely to achieve the objectives. Compensation should not be confused with mitigation measures which aim to avoid or reduce the extent of harm and form part of the plan or project and/or are directly connected with its implementation. Compensatory measures therefore need to be independent of the proposed project. In order to ensure the overall coherence of the Natura 2000 Network (and comply with EC guidance), Natural England advises that: - It should be possible to draw on empirical evidence to demonstrate a reasonable expectation of success within a reasonable timeframe. - There should be a clear plan for undertaking the compensation and subsequent management to ensure that objectives are met. - Compensation should be in comparable proportions to those habitats and species that are adversely affected. They should be within the same biogeographical region in the territory of the same Member State and should provide functions comparable to those that had justified the selection criteria of the original site.

			- Compensatory measures should be completed and land designated (where applicable) before work on the consented plan or project commences. It should be noted that there are very few cases that have reached the IROPI stage within the marine environment, and of those cases there is limited commonality with this project. As such Natural England is unable to provide examples of suitable compensatory measures at this stage. Natural England reference a report commissioned and published by CEFAS titled: ‘ Evidence Review to support the identification of potential conservation measures for selected species of seabirds’ (MacArthur Green 2013) (submitted with this response). The report seeks to identify measures that could be implemented either at protected sites or elsewhere with a view to informing considerations around the mitigation or compensation of predicted impacts from offshore marine developments. It should be noted that compensatory measures were not discussed as part of the Evidence Plan Process and therefore this matter is yet to be explored with the applicant. An additional factor for consideration in relation to this particular project is that deficiencies in the baseline data mean that it is difficult to have certainty over the nature and extent of the impacts and consequently it will be difficult to determine the nature and extent of the compensatory measures that may be required.

			Natural England are happy to engage in informal discussions regarding compensatory measures at this stage, but in the absence of previous examples to draw upon we would look to the applicant to propose options supported by empirical evidence as a starting point.
Q2.2.12	NE	In your Deadline 3 submission [REP3-075], you highlighted flight speeds that are not in accordance with Skov et al (2018), Alerstam et al (2007) and Pennycuick (1997). You go on to state that the flight heights were markedly higher in Johnston et al. (2014). Please provide copies of these papers if you wish to rely upon them as evidence.	Natural England note that Skov et al. (2018) is already in the Hornsea Three Examination library [REP1-149]. Natural England do not have permission to make papers that are subject to copyright restrictions publically available. Some of the documents requested may be available as open source documents. Natural England has provided full reference details for the requested papers below for the ExA: Alerstam, T., Rosén, M., Bäckman, J., Ericson, P.G.P. and Hellgren, O. (2007). Flight speeds among bird species: allometric and phylogenetic effects. PLoS Biology 5(8): e197. doi: 10.1371/journal.pbio.0050197. Johnston, A., Cook, A.S.C.P., Wright, L.J., Humphreys, E.M. and Burton, N.H.K. (2014a). Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines. Journal of Applied Ecology, 51: 31-41. Johnston, A., Cook, A.S.C.P., Wright, L.J., Humphreys, E.M. and Burton, N.H.K. (2014b). corrigendum. Journal of Applied Ecology, 51, doi: 10.1111/1365-2664.12260.

			Pennycuick, C.J. (1997). Actual and "optimum" flight speeds: field data reassessed. Journal of Experimental Biology, 200, 2355-2361. Please also note that REP3-075 states that "*Flight heights of birds were markedly higher in the Skov et al. (2018) study than the Johnston et al (2014) modelled distributions.*", not that "flight heights were markedly higher in Johnston et al. (2014)". Natural England note that this statement refers to flight height data rather than flight speeds.
Q2.2.14	RSPB, NE	The Applicant has advised that the nocturnal activity factors (NAF) historically used for collision risk modelling are not taken directly from Garthe and Hüppop (2004) but are instead based on an incorrect representation of the scores by Band (2012). The Applicant goes on to state that Band (2012) recommends that empirical data should be used when available, as has been the case for gannet and kittiwake. Please comment on these views and the empirical robustness of the studies that	The nocturnal activity factors historically used for collision risk modelling (CRM) are taken directly from Garthe and Huppop. For example a factor of 2 has been typically assigned to gannet and 3 for kittiwake for CRM and these are the same factors given in Garthe and Hüppop (2004). A recent review of the potential vulnerability of seabirds to marine renewable energy developments by Wade et al 2016 considered that nocturnal activity factors of 2 for gannet and 3 for kittiwake following Garthe and Hüppop (2004), King et al 2009, and Furness et al 2013 remained appropriate for the assessment of renewable impacts from collisions. However, while the Band (2012) model requires users to input a factor of 1 to 5 to represent nocturnal activity levels, Band (2012) translates these factors to levels of flight activity relative to daytime flight activity that are respectively 0%, 25%, 50%, 75% and 100% of daytime activity.

		were used to justify the use of different NAF by the Applicant, as set out in [REP1-188]. Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.	Band recommends that "*Flight activity estimates should allow both for daytime and night-time activity. Daytime activity should be based on field survey. Night-time flight activity should be based if possible on night-time survey; if not on expert assessment of likely levels of nocturnal activity*." It is therefore correct that Band (2012) recommends that night-time survey data (or other records of nocturnal activity) should be used if available and Natural England agrees with this recommendation. Band (2012) recommends that "where there is no night-time survey data available, or other records of nocturnal activity, for the species in question, (or for other sites if not at this site), it should be assumed that the Garthe and Hüppop/ King et al 1-5 rankings apply. Band acknowledges that the translation of the factors to percentages of daytime activity is simplistic and may be precautionary. In REP1-188 the Applicant has presented actual percentages of nocturnal activity relative to daytime activity rather than factors (1-5) for gannet and kittiwake. Further the Applicant has presented different percentages for the breeding and non-breeding seasons. The Applicant uses 8% nocturnal activity relative to daytime activity for the breeding season and 3% for the non-breeding season for gannet and 20% for the breeding season and 17% for the no-breeding season for kittiwake. These figures are taken from Furness et al (2018) for gannet [REP1-143] and MacArthur Green (2018) for kittiwake.

			Natural England does not agree that the empirical data on nocturnal activity for gannet and kittiwake that the Applicant has used is sufficient to justify the NAFs proposed by the Applicant for CRM as set out in REP1-188. Natural England also note that the NAFs presented in REP1-188 are not the same as those used for the collision risk assessments in the Applicant's ES and RIAA which are presented in [APP-109]. Sections 3.9-3.13 of REP1-211 and Q1.2.59 of ExA Q1 sets out our views on the robustness of the studies used to derive NAFs in REP1-188 (Garthe and Hüppop (2004) / Furness et al. (2018) / MacArthur Green (2018), but some additional key points are: **Source Data** • The percentages of night-time flight activity relative to daytime flight activity presented in REP1-188 (Furness et al 2018, MacArthur Green 2018) have been derived from an analysis of data from a number of different tagging studies for gannet and kittiwake; • The original tagging studies were mostly not designed to measure nocturnal activity levels, and information on nocturnal activity is not always presented in the source papers cited in REP1-143 (Furness et al 2018) and MacArthur Green (2018) – or if it is presented it is not in a format applicable to the calculation of nocturnal activity levels for CRM; • The nocturnal activity factors presented in Garthe and Huppop (2004), were derived from consideration of empirical data from tracking studies – some of which are the same studies

			that have been used to derive the nocturnal activity percentages presented in REP1-188; **Interpretation of data and sources of variability** • Nocturnal activity levels are not measured directly in the tagging studies. In order to derive information on nocturnal activity levels (which Furness et al 2018 define as flight activity), REP1-143 (Furness et al 2018, cited in REP1-188) makes assumptions about how parameters derived from tags on birds translate into flight activity. • The types of tags used varies across the studies as do the parameters that can be used to derive flight activity information. For example, some studies used internal and external temperature monitors – where for example temperature is used to indicate whether a bird is sitting on the water or not or has ingested food, others used accelerometer data to estimate flying activity, others salt-water immersion sensors to indicates periods resting on water etc. Different models, methods and assumptions need to be made to derive estimates of flight behaviour from the tag data. • There are also differences in sample sizes and location of colonies between the studies etc. Therefore there are a number of sources of variability and uncertainty in the measures of percentage night-time activity levels presented in REP1-188. These account for some of the differences in nocturnal activity rates between different publications (e.g. between REP1-188 and MacArthur Green (2018)*) as different datasets are included;

			• Table 1 in Furness et al 2018 [REP1-143] presents "*Flight activity from sunset to sunrise as % of flight activity during day*" derived from 11 publications and it is an average of these percentages that is used to denote nocturnal activity levels for CRM in REP1-188 for gannet. However it is not clear where the % figures in Table 1 come from or how they have been calculated as they are not generally presented in the publications cited. For example, according to Table 1, night time flight activity was 20.9% of the daytime levels based on the Garthe et al (1999) study. Garthe et al (1999) does not include this percentage. Figure 3 in Garthe et al (1999) shows the diel pattern of activity of tagged birds which includes percentage of time birds were flying. Based on Figure 3, flight activity from sunset to sunrise as a % of flight activity during day appears to be greater than 25% whether calculated using all activity data (including time birds spent at the colony), or if calculated using only the data for when birds were at sea. • Further, Figure 3 in Garthe et al (1999) shows that birds were in flight less during the period during the core daylight hours away from sunrise and sunset (when at sea surveys typically take place) and therefore calculating nocturnal flight activity from sunset to sunrise as a % of flight activity during the day should be higher if compared to activity in these core daytime hours. This is also evident from Figures 2 and 3 in REP1-143 (Furness et al 2018) where activity levels were generally lower in the middle of the day. This is relevant because the percentage nocturnal activity used in collision risk modelling (e.g. at Hornsea Three) is applied relative to the activity level measured during day-time by the snapshot of birds in flight from the digital aerial surveys. If a digital aerial survey

			records 100 birds of which 30 are in flight, then applying a nocturnal activity percentage of 8% (as proposed in REP1-188 for gannet in the breeding season) translates into 2 birds at night. This means that CRM will be applied to 30 birds in the daytime and 2 during the night – i.e. 2% of the birds recorded at sea on surveys, which given that night-time includes periods of twilight has the potential to be underestimating nocturnal activity levels. It is therefore Natural England's view that there is considerable variability and uncertainty about the appropriate activity level to use in CRM when applied relative to a daytime activity level that is estimated from an at sea survey. The calculated empirical NAFs presented within REP1-188 (Furness et al 2018 and MacArthur Green 2018) do not present any variability measure or confidence intervals to reflect this. We consider that appropriate nocturnal activity factors to use for gannet are 1-2 (ie 0-25% of daytime activity as measured from an at-sea survey) and 2-3 for kittiwake (i.e. 25-50% of daytime activity as measured from an at-sea survey), and consider that these rates are likely to better reflect the variability in nocturnal activity than the single figures proposed by the Applicant. Furthermore, we do not consider that there is sufficient evidence to apply different rates to the Hornsea Three data for the breeding season and non-breeding seasons for kittiwake and gannet as proposed in REP1-188. Band (2012) points out that "*Levels of activity may vary from season to season, and activity at sea may in any case differ from the levels of activity in breeding colonies for which the rankings have been formulated. Some species are particularly active during dawn and*

			dusk or extended twilight periods, or in locations where there is ambient windfarm lighting. When expressing the output of the collision risk assessment, the uncertainty surrounding flight activity should reflect the degree of confidence (or lack of confidence) in the flight activity information." *MacArthur Green (2018). Norfolk Vanguard Offshore Wind Farm Appendix 13.1 Offshore Ornithology Technical Appendix. [Online]. Available at: https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010079/EN010079-001547-Appendix%2013.01%20Ornithology%20Technical%20Appendix.pdf (Accessed October 2018).
Q2.2.17	NE, Applicant	In ISH2, NE highlighted the fact that tagging studies show different activity levels during the day. Please can NE provide copies of the relevant publications and a table that summarises daytime activity levels for all the species for which you have identified a likely significant effect. Please can the Applicant provide information on when the DAS was undertaken that	Natural England do not have permission to make papers that are subject to copyright restrictions publically available. However, Natural England has provided full reference details for papers that provide information on daytime activity levels for gannet and kittiwake, as these are the two species that are subject to collision risk modelling, and for which the Applicant has used different values for NAFs compared to those advised by Natural England. Natural England are not able to provide a table that summarises daytime activity levels for all species requested, as to provide specific numbers would require access to the original data sources, but we have signposted particular figures within the papers that show daytime activity levels and the variability in these. Furness et al (2018). This is available in the Examination Library [REP1-143]. Figure 2 shows flight activity of gannets in relation to

		includes the transect start and finish times.	hour of the day using datasets from a variety of tracking studies at different times of the year. Garthe, S., Grémillet, D., Furness, R.W., 1999. At-sea activity and foraging efficiency inchick-rearing northern gannets (Sula bassana): a case study in Shetland. Mar. Ecol. Prog. Ser. 185, 93–99. Figure 3 shows the diel pattern for the activity of 3 gannet fitted with data-loggers from colony at Hermaness, Shetland. Activity is split into swimming, flying and staying in the colony across the 24hr cycle. Warwick-Evans, V., Atkinson, P.W., Walkington, I., Green, J.A., 2017. Predicting the impacts of windfarms on seabirds: an individual based model. J. Appl. Ecol. http://dx.doi.org/10.1111/1365-2664.12996. Figure 3 (a) shows behaviour budgets across the 24hr period for gannet during early chick-rearing from the colony on Les Etacs, Alderney, Channel Islands Daunt, F., Benvenuti, S., Harris, M.P., Dall'Antonia, L., Elston, D.A., Wanless, S. 2002. Foraging strategies of the black-legged kittiwake Rissa tridactyla at a North Sea colony: evidence for a maximum foraging range. Mar Ecol Prog Ser. Vol. 245: 239–247.

			Figure 3 shows the daily activity patterns in terms of travelling flight, foraging flight, presence on sea surface and nest attendance for nine kittiwake from the Isle of May during the breeding season.
Q2.2.18	RSPB, NE	Cook et al (2018) recommends avoidance rates for kittiwake and lesser black-backed gull that are different to those proposed by JNCC et al (2014) and the RSPB. Please comment on the results of the additional modelling, its empirical basis and the implications for the ES and HRA as set out by the Applicant in Appendix 10 at Deadline 1[REP1-188].	Cook et al (2018) recommend avoidance rates (Basic Band Model) of 0.998 for lesser black-backed gull and 0.992 for kittiwake. Cook et al (2014) recommend avoidance rates of 0.995 for lesser black-backed gull and 0.992 for kittiwake. JNCC et al (2014) advise that an avoidance rate of 0.995 (+/- 0.001) and 0.989 (+/-0.002) are used for lesser black-backed gull and kittiwake respectively with the Basic Band Model (Band 2012). Cook et al (2014) is the report of work carried out by the British Trust for Ornithology in collaboration with the Environmental Research Institute for a Marine Scotland Science project to derive avoidance rates for use in collision risk models at offshore windfarms. Cook et al (2018) is a paper based on the Cook et al (2014) work that was published in the journal Marine Environmental Research, and so is based on the same empirical data and analyses of avoidance rates. Cook et al (2014) concluded that whilst it was possible to derive a species-specific within-windfarm avoidance rate for lesser black-backed gull (0.998), it was based on limited data and Cook et al (2014) recommended that the within-windfarm avoidance rate for “large gulls” (a dataset that included the lesser black-backed gull

			data) was more appropriate for use for this species – i.e. a total avoidance rate of 0.995 for use with the basic Band model. Subsequently, Cook et al (2018) recommended use of a 0.998 avoidance rate for lesser black-backed gull. This rate is not based on any additional data or analysis compared to Cook et al (2014) and therefore since the lesser black-backed gull species specific within windfarm avoidance rate was based on data from only two sites, and has lower confidence associated with it compared to the “large gull” avoidance rate (which had a higher confidence level), Natural England continue to recommend an avoidance rate of 0.995 for use with the Basic Band Model (as per recommendations in Cook et al 2014 and JNCC et al 2014). Cook et al (2014) concluded that it was not possible to derive species-specific within-windfarm avoidance rates for black-legged kittiwake. However, on the basis of black-legged kittiwake having similar flight characteristics to black-headed and common gulls (which contributed the majority of records for the small gulls group), the within-windfarm avoidance rates derived for the small gulls group was considered appropriate for this species. A total avoidance rate of 0.992 was therefore recommended for the basic Band model which is the “small gull” avoidance rate. JNCC et al (2014) did not agree with the use of the small gull avoidance rate for kittiwake as considered appropriate by Cook et al (2014, 2018). The rationale for this is set out in JNCC et al (2014), but in summary it was considered that the more marine nature of the species compared to the other small gulls meant that there was considerable uncertainty around appropriate avoidance rates to use for kittiwake and that as a result a more precautionary approach

			should be adopted and the more generic “all gull” category avoidance rate of 0.989 should be used. The implications of using an avoidance rate of 0.992 over an avoidance rate of 0.989 with the Basic Band Model (i.e. the difference between the Applicant’s approach and Natural England advice for kittiwake) is that the number of predicted collisions will be around 27% lower with the 0.992 avoidance rate compared to a 0.989 avoidance rate. Therefore the effect of applying a 0.992 avoidance rate across all projects would result in the predicted impact both at HRA and EIA being 27% lower than if an avoidance rate of 0.989 were used. The implications of using an avoidance rate of 0.998 rather than 0.995 with the Basic Band Model would be a 60% reduction in the number of predicted collisions. The effect of applying a 0.998 avoidance rate across all projects would result in the predicted impact both at EIA being 60% lower than if an avoidance rate of 0.995 were used.
Q2.2.36	NE	In [REP3-075] you stated that no clear audit trail is present showing how the figures presented in [REP1-148] were derived. Could you confirm if the type of information you are requesting here has been made available for the	This type of information has not been made available for the cumulative/in-combination assessments for previous windfarm projects. Natural England also note that previous windfarm projects have either not presented revised collision figures for windfarms as per REP3-075, or Natural England has not accepted such changes to the collision figures.

		cumulative/in combination assessments for previous offshore wind farm projects?	
		Benthic Ecology	
Q2.2.42	Applicant, NE	Please produce a draft Statement of Common Ground for benthic ecology at Deadline 6 that includes but is not limited to the following headings: Baseline Characterisation Biotope Classification Sandwave Levelling Cable Burial and Protection Micro-Siting Potential Biogenic and Geogenic Reefs Markham's Triangle pMCZ Where you cannot reach agreement you should state that your position is final and will not be resolved.	Natural England notes this request and will seek to progress this with the applicant.
Q2.2.43	NE	The tables you submitted at Deadline 3 [REP3-076] contain hyperlinks to information on SAC sub-	Please see Annex 1. (NB. The Wash and North Norfolk Coast SAC is the only site listed within [REP3-076] with sub features).

		features that are not accessible to the ExA. Please provide updated tables showing the definitive list of sub-features in plain text.	
Q2.2.45	NE	Given your stated position in relation to the baseline characterisation and the fact that you are unable to conclude beyond reasonable scientific doubt that the integrity of European sites would not be affected by the proposal, please suggest any feasible compensation measures that would be needed for the North Norfolk Sandbanks and Saturn Reef SAC and The Wash and North Norfolk Coast SAC.	Although it is acceptable to discuss compensation measures in principle and without prejudice prior to an Appropriate Assessment (AA) or Habitats Regulations Assessment (HRA), it is important to recognise that **Compensatory measures** can only be *formally* considered after a negative assessment under regulation 63* and where in the absence of alternatives and there are imperative reasons of overriding public interest (regulation 64*), and the competent authority is minded to approve the plan or project.(*Regulation 25 and 26 in Offshore Regs). In this scenario it is the duty of the relevant Secretary of State to secure such compensatory measures as is necessary to ensure the overall coherence of Natura 2000 is protected. The relevant SNCB(s) role is to advise on the effectiveness of the proposed compensatory measures and whether they are likely to achieve the objectives. Compensation should not be confused with mitigation measures which aim to avoid or reduce the extent of harm and form part of the plan or project and/or are directly connected with its implementation. Compensatory measures therefore need to be independent of the proposed project. In order to ensure the overall coherence of the Natura 2000 Network (and comply with EC guidance), Natural England advises that:

			- It should be possible to draw on empirical evidence to demonstrate a reasonable expectation of success within a reasonable timeframe. - There should be a clear plan for undertaking the compensation and subsequent management to ensure that objectives are met. - Compensation should be in comparable proportions to those habitats and species that are adversely affected. They should be within the same biogeographical region in the territory of the same Member State and should provide functions comparable to those that had justified the selection criteria of the original site. - Compensatory measures should be completed before work on the consented plan or project commences. It should be noted that there are very few cases that have reached the IROPI stage within the marine environment. Of those cases there is limited commonality with this project, and there are no examples in within Offshore SACs. As such Natural England and JNCC are unable to provide specific examples of suitable compensatory measures at this stage. An additional factor for consideration in relation to this particular project is that deficiencies in the baseline data mean that it is difficult to have certainty over the nature and extent of the impacts and consequently it will be difficult to determine the nature and extent of the compensatory measures that may be required. Natural England are happy to engage in informal discussions regarding compensatory measures at this stage, but in the absence of previous examples to draw upon we would look to the applicant

			to propose options supported by empirical evidence as a starting point.
Q2.2.47	NE, MMO	If the Secretary of State were to conclude that there may be harm to the Cromer Shoal Chalk Beds MCZ and/or the Markham's Triangle pMCZ, what measures of equivalent environmental benefit to the harm that might be caused could be provided?	Although there is guidance on the process for assessing the impacts on Marine Conservation Zones, there is currently no Government guidance in relation to Measures of Equivalent Environmental Benefit (MEEB) and there and there have been no other cases that have reached this stage. Therefore, should the SoS conclude that MEEB are required, this case would be precedent setting. In the absence of guidance/experience to draw upon, we would recommend that discussions relating to MEEB include input from the SNCBs, Regulatory Agencies (i.e. MMO and BEIS) and Defra.
Q2.2.48	NE	You questioned the conclusions of the MCZ assessment for the Cromer Shoal Chalk Beds in [REP1-125] and believe there is sufficient uncertainty to have limited confidence in the Stage 1 conclusion that there would be no significant risk to delivering the site conservation objectives. The Applicant maintains in [REP2-004] that a Stage 2 assessment is not required due to the "very small proportion of designated	In their MCZ assessment for Cromer Shoal Chalk Beds the applicant has assessed trenching as a worst case scenario. Natural England is of the view that there is limited certainty regarding the scale of the impacts associated with the use of HDD and associated construction of 8, 50x20m cofferdams within the MCZ to conclude that the trenching option would be the worst case. From our experience of similar developments which have used the HDD method, the sediment that is excavated from the exit pits becomes destabilised such that when it is returned it either leaves a depression or is elevated compared with the surrounding habitat. This change in sediment composition may alter the biological communities thus not maintaining the habitats and hindering the conservation objectives for the site. Based on the current assessment there is no certainty of the depth of the sand at this location and depth of any excavation required for the exit pits so potentially there may be interest features affected beyond those

		features affected". The Applicant also highlights the fact that the majority of impacts would be temporary and reversible and that longer lasting effects would affect a very small (i.e. <0.02%) proportion of the Subtidal Sand feature of the MCZ and only where cable protection is required. What are your views on these matters?	considered in the current assessment. Additionally there is likely to be compaction of sediment within the cofferdam areas from use and storage of associated equipment and/or the location and impacts from storing excavated sediment. It is Natural England's view that the combination of these impacts associated with the HDD option have the potential to impact on different features of the site in different ways to that of the trenching option which has been assessed. Consequently as the preferred option, we believe that this option warrants consideration within the MCZ assessment. In relation to the MCZ assessment as it stands, given our uncertainty that the HDD impacts have been captured within the parameters of the WCS assessment we advise that a more precautionary approach is taken at this time in order to future proof the project and avoid delays at the time of construction. Natural England would also highlight that the impacts should be considered at a feature level, and that there should be consideration of all attributes of that feature. As set out in Natural England PEIr response (point 2.3.15) and highlighted during the evidence plan process, the Humber gateway cable installation demonstrates that impacts to some features are unlikely to be reversible.
Q2.2.50	NE, MMO	Paragraph 2.87 of [REP2-004] states that a Cable Burial Risk Assessment would be produced post consent and paragraph 2.88 goes on to state that this would be	The Applicant's proposal at Para 2.87, would essential mean that the full assessment of impacts associated with cable installation would be considered and addressed post consent. Whilst this approach may be acceptable outside of designated sites, based on Natural England's experience of previous projects we would no longer consider this a suitable option for addressing impacts on

		secured as part of the Cable Burial Plan through Schedule 11, Condition 13(1)(h) (generation assets DML) and Schedule 12, Condition 14(1)(h) (transmission assets DML) of the dDCO. You highlighted the lack of adequate sampling along the inshore cable corridor re-route in relation to MPAs in ISH2 and the need for an early Cable Burial Risk Assessment to avoid problems that have arisen elsewhere. Please elaborate on the problems that have occurred elsewhere. What practical steps could be taken to avoid such problems in this project? How could adequate mitigation be secured through the dDCO?	designated sites, and we would expect to see a cable burial risk assessment based on data from a recent comprehensive geotechnical survey campaign to underpin the assessment of impacts on designated sites. We advise that this evidence is required prior to consent to address scientific doubt in relation to the Habitat Regulations Assessment/ MCZ Assessment. This should in part avoid the issues that have occurred during several cable installation operations which have caused not only delays to the project, cost considerable amount of money and resources, but still resulted in significant environmental damage. Most notably of these is Race Bank OWF which is also located within the Wash and North Norfolk Coast SAC. Race Bank example: Background Race Bank followed the approach that is currently proposed for Hornsea Project 3 (i.e. a high level assessment at the consenting stage with a commitment to undertake detailed surveys post consent and agree appropriate mitigation prior to construction). As has been the case with many projects, once the geotechnical and geophysical investigations had taken place and a contractor was on board, it was apparent that the installation techniques assessed within their environmental statement would not appropriate. The cable installation techniques consented were either no longer feasible, or required significant seabed preparation activities. This

			lead to a further 32 consultations with Natural England (via the MMO) over an 18 month period which were required to discharge the marine licence conditions. This also included (but not exclusively) new applications and variations for repeated sandwave levelling, use of mass flow excavator, and cable protection, all of these had additional protracted documentation sign off processes including separate Appropriate Assessment (AAs) due to the potential to significantly impact the features of the designated sites. Options to take the project forwards needed to take into account NEs ongoing advice since 2009 that no cable protection should be placed in the Wash and North Norfolk Coast SAC, due to the likely hindrance of the conservation objectives for the site and risk of Adverse Effect on Integrity. Furthermore the discussions around potential solutions were constrained by the original consent resulting in novel (untested) and expensive resolutions being found. The result of this was that even prior to the commencement of any cable installation works, the project design had dramatically changed from the original consent, and the resulting impacts were demonstrably greater than those assessed in the Appropriate Assessment at the time of consent (2012). Example outcome: Even with the geotechnical information to inform the post consent discussions, unexpected issues arose during the installation and

			additional marine licences/variations were required. This included a requirement to increase the amount of sandwave levelling by 7 times the amount permitted in the additional post consent marine licence, and the length of cable requiring levelling increased by over 30 times the length, than that assessed in the associated AA. There were also impacts that were never anticipated including dredging below the seabed level which has resulted in impacts from which the site has not yet recovered. Additionally, despite use of several different techniques it was not possible to reach sufficient burial depth for around 12,370m of export cable. Of this area 9,072m was believed to be able to meet burial depth with use of a mass flow excavator (MFE) and that cable protection was only required along 1,022m of export cable and not within the Wash and North Norfolk Coast SAC. However, the cable remediation works were not fully effective in burying the cables. In May 2018 NE and MMO were informed that further applications would be required to deploy cable protection including deployment of further cable protection within the Race Bank sandbank and along the export cable, but outside of The Wash and North Norfolk Coast. (A separate Marine Licence Application for cable remediation and cable protection works within The Wash and North Norfolk Coast SAC is currently under the consideration of the MMO). In conclusion, the approach of providing a high level assessment at the application phase with a view to addressing the impacts post consent (used in this example and others) has proved highly

			problematic and has resulted in significant cost both environmentally and economically (i.e. to the project). This has served to highlight the importance of assessing the impacts as fully as possible at the application stage. The piecemeal changes made to the cable installation for Race Bank we piecemeal, and as such, the impacts were never satisfactorily assessed at a combined level, and as a result the impacts to the site have not been fully captured within the parameters of the individual appropriate assessments, and therefore the true extent of the impact has never been fully captured. **Mitigation** **Natural England advises it is vital that prior to consent, the worst case scenario is fully assessed, based on detailed geotechnical and geophysical data.** **The Race Bank project highlights that even when this data is available the challenging environmental conditions result in the cable installation operations rarely occurring as predicted. Therefore it is also important to build these lessons learnt into the worst case scenario assessment.** Natural England is of the view that there is insufficient data and/or information for Hornsea Project 3 to support the conclusions of their RIAA (and to a lesser extent their MCZ assessment).

			Additionally, based on our experience of other projects (including Race Bank cable installation), we do not feel that the WCS has been considered. . At this time, we do not believe that there are sufficient mitigation measures that could be secured in dDCO to remove the risk Adverse Effect on Integrity for the Wash and North Norfolk Coast and North Norfolk Coast Sandbanks and Saturn Reef SAC beyond reasonable scientific doubt. <ins>For your information - timeframes</ins> The time taken to sign off the Race Bank pre-construction document ranged from 1 to 11 months. With 13 of 30+ documents taking longer than the 4 months suggested by the licence conditions. Many of the documents signed off quickly were requirements such as notifications of vessels, or the names of liaison officers, notification of start date or notice to mariners. Which require little or no consultation. While the more complex documents such as monitoring plans, installation methodologies and mitigation plans often took longer than the 4 months provided for within the licence. While the fact that simple documents were processed quickly and more complex documents took longer is hardly surprising, this does call into question the logic of a one size fits all approach of 4 months prior to construction. This is especially relevant to a project like Race Bank located within designated sites and with a significant monitoring effort aimed at validating decisions on the acceptability of impacts to those sites. For the post consent/construction variation requests the timeline agreed for consultation, required NE and MMO to consider

			documentation and provide responses within 2 weeks. For NE it also required review and response to an AA within 2 weeks as well, instead of the standard 4.
Q2.2.56	NE	Paragraph 5.4.11 of your representation [RR-097] stated that the benthic analyses were not appropriate for characterising the Markham's Triangle pMCZ. The Applicant concluded in [REP1-122] that only minor differences in the biotope classifications exist between those mapped in Sotheran et al. (2017) and the ES. Are you satisfied with the Applicant's response to this issue, as set out in [REP1-122], [REP1-131] and [REP3-023]? If not, why not? Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.	Our three main concerns were as follows: <u>The Cefas/Defra evidence for Markham's Triangle was not used in the characterisation of the Hornsea Project Three array area.</u> The Applicant refers us to their previous answers in REP1-122 and REP1-131. The reasons for including and excluding Cefas / Defra evidence within various infaunal analyses are discussed further in REP1-131. We remain unsure that Cefas / Defra data was used in the best manner to inform characterisation of the pMCZ. <u>The biotopes provided in JNCC Report 608 (Sotheran et al., 2017) were not used in the analyses, instead considering only suggested biotopes for the survey points within the pMCZ</u> We are content with the response in REP1-131. <u>Suggested biotopes for the Cefas / Defra data are quite dissimilar to the biotopes within JNCC Report 608</u> We continue to disagree that the biotopes interpreted by the Applicant are similar to those of previous surveys. For example, Sotheran et al (2017) notes that "Of the 50 samples analysed within this analysis, 38 (76%) were found to support the presence of 'Subtidal sand' in the area, having been allocated the biotope

			SS.SSa.CFiSa.EpusOborApri or habitat SS.SSa.CMuSa”. The Applicant recorded neither in their Markham’s Triangle dataset.
Q2.2.58	NE	You representation [RR-097] states that the features of the Markham’s Triangle pMCZ should be assessed separately rather than by using one feature as a proxy. The Applicant has since presented habitat loss numbers in tabular format, as set out in [REP2-004] and a supplementary assessment in [REP3-023]. Does this enable you to reach a conclusion on the assessment that has been undertaken? In your view, are there any outstanding matters regarding the Marine Conservation Zone Assessment [APP-104]?	We continue to believe that the Applicant has not undertaken their assessments in a way to allow best scientific understanding of impact. However, we do understand that much of the uncertainty over impact distribution between broad scale habitats will remain until firmer understanding of turbine placements in pre-construction.
Q2.2.59	NE	Paragraph 4.4.5 of your representation [RR-097] stated that the consideration	Natural England believes that there are two aspects to this question a) the combined repetitive impact to the same footprint

		of each phase in isolation failed to consider cumulative impacts over time. The Applicant has concluded in [REP2-005] that a phased build would not affect recoverability of the relevant features as it would not result in repeat physical disturbance of the same area of seabed across different phases, due to the risk this would pose to the integrity of installed export cables. It is said that the operation and maintenance activities would be highly localised and intermittent. Can you list which impacts are most likely to have a residual effect between each phase, the species and sites affected and your degree of certainty? Are you suggesting that the Applicant has failed to meet the requirements of paragraph 2.6.64 of National Policy Statement (NPS) EN-3?	area over different stages of installation and b) the combined impact to a feature in a phased build scenario. a)) the combined repetitive impact to the same footprint area over different stages of installation Often impacts from one phase of installation (i.e. preparation, installation and operation) persist into the next phase especially where recoverability is hindered by the different activities. By considering each of these phases in isolation, the applicant is making the underlying assumption that both impact and recovery occur at each phase, whereas in reality, the impacts of preparation activities may persist into the construction and operational phase and so on. This persistent impact over time could result in additional impacts which would prevent a feature meeting its conservation objectives. This approach therefore fails to capture the cumulative impact on a feature throughout the lifetime of a project. b) the combined repetitive impact to a feature over different phases While the proposed two phase build to this project is unlikely to directly have the same physical disturbance to a particular area; the impacts are still to the same feature of the site. Therefore the phase build will extend the timeframe of impacts on the feature and overall recoverability of said feature.

		Does this apply to any other cumulative effects? Are you satisfied that the information supplied by the Applicant at Deadline 2 is sufficient or do you still maintain your original position?	For example, the first phase of a project could result in impacts on a sandbank feature that result in that feature being in unfavourable recovering condition, with full recovery expected within 6 years. Within the second phase of the project, a different area of that same sandbank could be impacted, therefore resulting in that feature being in unfavourable condition for a further 6 years. Depending of the timing between phases this could mean that the feature is affected for 10+ years. Therefore the phased approach may mean impacts that are considered short term/temporary when considered in the context of a single phase, persist in the medium to longer term. This should be fully assessed including the implications for the site potentially being in unfavourable condition for 10+ years when considering impacts to sandbanks. Conclusion: As we have limited survey data from with the MPAs, the proposed techniques are fairly new for offshore windfarm developments and yet to be deployed on the scale proposed for this project there is uncertainty in relation to WCS because the actual scale of the works required is unknown and the likely level of success. Therefore the timeframes for any recovery are also uncertain. We therefore that believe that NP EN-3 to consider different stages of the lifespan have been met but not considered cumulatively *2.6.64*

			Assessment of offshore ecology and biodiversity should be undertaken by the applicant for all stages of the lifespan of the proposed offshore wind farm and in accordance with the appropriate policy for offshore wind farm EIAs. N.B. At Deadline 3 Natural England provided further comments on the information provided by the applicant at Deadline 2, in which we retain our original position
Q2.2.60	NE	Paragraph 2.12.2.3 of the ES [APP-062] identifies a number of impacts that have been scoped out of the cumulative impact assessment. You have stated in [REP1-212] that seabed disturbance from maintenance activities should not have been scoped out of the cumulative assessment as up to 25% of the cable corridor may need protective measures. How was this figure derived and what empirical evidence to you have to substantiate this point? Please provide copies of any publications you wish to rely	For clarification the 25% figure in relation to the Operation and Maintenance (O&M) deposition of cable protection was provided by the applicant in 5.2 Report to inform Appropriate Assessment (APP – 051 Page 30). The applicant considers this to be a 'replenishment' of rock armouring over the life time of the project that was deposited as part of the construction (i.e. 2.5% of the original cable length) However, it is not clear if the 2.5% relates length or volume. If it is volume then there is every probability the discrete areas may have a larger footprint than the original cable protection. In addition, the RIAA doesn't account for the spreading of cable protection over time, which could lead to the requirement for additional cable protection and therefore potentially greater impact footprint. In any case, this requirement would result in additional rock armouring within the marine environment, which is likely to have a persistent impact and could hinder the effectiveness of any removal activities.

		upon in evidence that have not already been provided.	For NE to consider supporting this being scoped out there would need to be a marine licence condition which stipulates the following:- a) No new locations can have rock armouring installed to those areas installed as part of the construction b) The dimensions of area of impact are limited to the footprint listed in the application document i.e. no wider c) The 25% is more clearly defined i.e. a quarter of the cable protection length/volume installed during construction only, not of the total amount permitted. Noting that if volume only is considered by the engineers /contractors then the area of impacts could be much greater and therefore should be avoided as per point (a) above d) If the replenishment is within designated sites then further sign off is required with the MMO and Natural England before works are undertaken
Q2.2.63	NE	Please provide the following publications that you have relied upon in evidence: Roberts et al (2016) [REP1-213] and/or Roberts et al (2014) [REP1-212]	There is an error in REP1-212 this should read 2016. Therefore the reference is for the same paper that has been included with our Deadline 4 response
Q2.2.64	NE	You have embedded a document in your Deadline 3 submission [REP3-076] that is not accessible to the ExA.	Natural England provided this by email on 19/12/2018.

		Please provide a copy of: Technical Guidance Note: Providing Management Advice on MPA Features – Guidance on Using Feature Data for the Purposes of Fisheries Management Including the Use of Buffers and Margins, 4 November 2016.	
		Marine Mammals	
Q2.2.69	WDC, NE	In [REP1-022] WDC highlighted a concern about the impact of increased vessel activity throughout the life of the development because increased vessel noise can interrupt harbour porpoise foraging behaviour and echolocation, which can lead to significantly fewer prey capture attempts. Please can WDC submit a copy of Wisniewska et al (2018). In [REP2-004] the Applicant has suggested a methodology for the assessment of vessel	The Wisniewska et al., 2018 paper shows that while there is an interruption of foraging (predominantly when vessels are passing close to animals), animals have recommenced foraging within approximately 10 mins (figure 3b). This is a similar result to the Pirotta et al., 2015 paper (in terms of a very short term disturbance), which is referenced in the marine mammal ES chapter. In addition, the applicant's ES also references a Heinanen and Skov (2015) report, stating – *"Heinanen and Skov (2015) identified that the occurrence of harbour porpoise declines significantly when the number of vessels in a 5 km2 area exceeds 80 in one day. With an average of 19.6 vessels per day as a baseline, with a maximum increase of 6 vessels per day, in an area considerably larger than 5 km2, vessel density will remain well below this threshold level for harbour porpoises."*

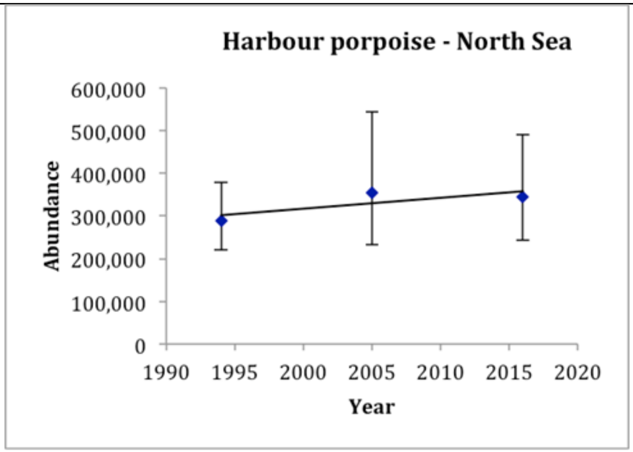
		movements and the associated ES conclusions have been agreed in the SoCG [REP1-218]. Does WDC concur with this view? Do the findings of Wisniewska et al (2018) change what NE has concluded in the SoCG?	Therefore given the assessment within the ES, and the short time foraging is affected, NE has not changed their opinion from that stated in the SoCG.
Q2.2.70	NE	The Applicant has stated [REP1-122] that it was not possible to quantitatively predict vessel impact exposure, in terms of the number of marine mammals affected, unlike piling noise disturbance. The Applicant went on to note that it has not been possible to provide any meaningful combined assessment of both activities and it has therefore relied upon a qualitative assessment. Are you satisfied with the qualitative in combination assessment that has been provided.	Natural England has discussed this matter with the applicant and agrees that a quantitative assessment cannot be undertaken given the mobile nature of both animals and vessels, and the short time the animals can be affected by vessel movements. Natural England is satisfied with the qualitative in combination assessment.

		If not, how could it be improved?	
Q2.2.72	TWT, NE	In [RR-047] TWT stated that fishing activity should be included in the in combination assessment rather than in the ES baseline. Paragraph 4.4.3 of EU guidance[1] suggests that completed plans or projects do not form part of the in combination assessment required by Article 6(3) but that their effect should still be considered if they have continuing effects on the site. Even if TWT considers fishing as a plan or project that has not been completed why would an in combination assessment not result in double counting if fishing has been included in the baseline? What legislative purpose does TWT think would be served by	Please see Natural England's response to Q2.2.73 below.

		assessing the effects of the continuing existing activity, i.e. fishing, a second time? Has a distinction been made between existing and future fishing activity in any of the Hornsea Project Three evidence? How can future fishing be taken into account before the outcome of any future licensing is known? What evidence does TWT have to suggest that the outcome of future licensing will intensify or extend fishing? [1] Managing Natura 2000 Sites. The provisions of Article 6 of the 'Habitats' Directive 92/43/EEC (2000)	
Q2.2.73	NE	You stated in [REP1-212] that where there is ongoing fishing activity on the site, it is appropriate to consider the effects of the plan or project	Where there is ongoing fishing activity in the site it is important that the impacts of the activity are captured within the assessment in the context of the conservation objectives of the affected designated site(s). This assessment will likely take place as part of the baseline characterisation of the development area, however, as fishing

		that is the subject of the assessment in the context of those prevailing conditions, of which fishing impact may be one. Does you consider that fishing should have been included in the ES as an in combination effect?	activity is mobile, variable and subject to change, there may be instances whereby fishing impacts are not adequately captured in the baseline characterisation and therefore may need to be considered as part of the in-combination assessment. This could be due to a change in effort; change in management; or a change in legislation amongst other things, and fishery managers (i.e. MMO and IFCAs) would be best placed to advise on this. There may also be occasions whereby there are plans for new fisheries, or changes to existing fisheries which could be captured in-combination. Again the fishery managers would be able to advise on this. In relation to the assessment of impacts on the SNS SCI, Natural England would consider that the impact of ongoing fishing activity in the context of the draft conservation objectives for the site, has been adequately captured for the purposes of the HRA. We are not currently aware of anything that would have significantly altered the levels of fishing activity within the site; any current plans for new fisheries, or changes to existing fisheries that have not been captured, but we would look to fisheries managers to advise more definitively on these points.
Q2.2.74	NE	In [RR-097] you stated that you did not agree with the approach of averaging the number of piling days per season when considering effects on the Southern North Sea candidate SAC (cSAC). You went on to suggest that	In [RR-097], Natural England was seeking to highlight the disparity between the scenario considered in the assessment and that which would potentially be permitted under the proposed DML condition i.e. the assessment assumes that piling is split equally across all months and does not assess the worst case scenario of more piling in the summer season (which would be permitted the proposed licence condition).

		work is more likely to occur during the summer months. The Applicant has since clarified in [REP1-131] that construction activity is likely to occur throughout the year and noted that the most weather sensitive component of the installation process is the blade lift with foundation installation commonly scheduled during the winter months to ensure that the installation of blades can occur during calmer, summer conditions. Please comment on the Applicant's response. Do you have any evidence to the contrary?	Natural England acknowledged the Applicant's clarification in [REP1-131], however, this does not necessarily rule out a worst case scenario of more piling taking place in the summer months. In Natural England's experience, construction activity doesn't always happen as scheduled, and this is precisely the reason that applicants seek flexibility within their DML conditions.
Q2.2.76	NE	In [REP1-023] TWT states that there is no understanding as to what the carrying capacity of harbour porpoise is in the Southern North Sea SCI. Therefore, in the opinion of TWT there is weak scientific	There are three estimates of abundance in the North Sea (SCANS surveys) and these were used to assess trend for the OSPAR Intermediate Assessment (OSPAR IA, 2017). The assessment concluded that there was no evidence of change in abundance over the period 1994-2016. We acknowledge that this is limited evidence, but it is suggesting that the population is stable.

		information underpinning any area-based approach to management and SNCB advice. Please respond to this point and provide any additional information that you wish to rely upon in evidence that has not already been provided.	 *Figure 1: Trend line fitted to harbour porpoise abundance estimates in the North Sea AU. Bars show 95% confidence intervals (Taken from OSPAR IA, 2017).* Therefore while there is no evidence of the carrying capacity of the Southern North Sea SCI, the available evidence is indicating that that the overall population of the North Sea Management Unit is stable. SNCB area based approach to management within the SCI is based on our best available evidence and was developed over several years and with considerable consultation. Whilst the limitations of the evidence base are acknowledged, no better way to assess the impact of noise within the SCI has been put forward. The SNCBs are clear that this is an initial approach, based on the available scientific information, and will be subject to change in the future if the evidence suggests that the thresholds or area based approach is not appropriate.

Q2.2.77	NE, MMO	WDC have stated in [REP1-022] that they wish to see temporary threshold shift as well as permanent threshold shift evaluated as an alone or in combination piling noise impact. The Applicant has indicated that in [REP1-218] you agreed that this is not required. Do you agree that an evaluation of temporary threshold shift is not required to inform the ES and HRA?	TTS (Temporary Threshold Shift) is a temporary, but recoverable, reduction in the sensitivity of a marine mammal's hearing, which they depend upon for foraging, navigation, predator avoidance and communication. Periods of TTS may therefore impair an animal's ability to forage, interact with others or avoid predators, i.e. cause cognitive impairments. However, despite extensive studies being undertaken to quantify TTS and define its onset, there is little scientific knowledge of where TTS becomes biologically significant, as there is a huge range of TTS impacts, from a small decrease in hearing sensitivity for a few minutes, up to near auditory injury (PTS), which may affect an animal for several days. The developer view is that *"Until such time that there is sufficient information to indicate a level and duration of TTS that may have a significant ecological effect on individuals, impact assessments should focus on the impacts that can be more justifiably assessed, namely PTS, to indicate a level of ecologically significant auditory injury, and on the potential behavioural effects of noise such as disturbance leading to disruption in natural behaviour (e.g. reduction in foraging efficiency)."* NE accepts this view, but would still like to see some assessment of TTS in the EIA to provide context to the overall assessment of noise impact. As such, NE believes that what is presented in the ES (the range (kms) of possible TTS, rather than the number of animals, which has no ecological meaning, given the range of TTS effects described above), is acceptable to provide this context.

			In terms of the HRA, the Conservation Objectives of the site are concerned with injury (i.e. PTS) and disturbance. Therefore the additional assessment of TTS is currently not required as TTS lies fully within the disturbance zone, and is therefore captured within the assessment. Whilst we agree that would be useful to assess the impacts of TTS, we accept that it is not currently possible to do this in a meaningful way (i.e. in terms of the level where it is significant for the animal). If future research provides the level where TTS becomes biologically significant, then we would expect this to fully be assessed within both the ES and HRA in future projects.
Q2.2.79	NE, MMO	WDC have pointed out [REP1-022] that an EPS license would be required for any pile-driving activity. With the Morge case in mind, is the project likely to infringe Article 12 of the Habitats Directive? If so, is it likely that a derogation, in the form of an EPS licence, would be granted?	As this question related to the granting of an EPS licence in relation to marine mammals, Natural England would respectfully defer to our colleagues the MMO to answer this point.

Q2.2.80	NE	In [REP1-212] you state that the JNCC piling mitigation protocol is out of date and that a range of other mitigation measures used in other European countries should have been detailed in the ES. You welcomed the DML conditions but needed further discussion of mitigation options. If revised piling mitigation protocol guidance is yet to be consulted upon what guidance should be used and given weight in this examination? The Applicant has made a commitment to a Marine Mammal Monitoring Plan and Site Integrity Plan. Why do you consider that these measures would be insufficient. The SoCG with WDC [REP1-219] establishes a 20% increase in piling duration, cost escalation and only	Whilst the piling protocol is useful it was published in 2010 and does not reflect the current scale and size of proposed new developments and the size of potential auditory and injury zones. There are now a range of other alternatives available and in use elsewhere in Europe that are not mentioned in the mitigation protocol. In Natural England's view these alternatives should be detailed within the ES (even if they are presented as a short list), and it should be clear that additional alternatives will be considered as they become available. With the development of a SIP, the potential in combination impacts of a project would be reassessed when there is more certainty of the other wind farm projects that might overlap with Hornsea 3 and mitigation can be detailed and agreed at this point as required. While NE does not have an issue with the SIP in principle, there is currently no process for how these plans will be monitored and managed over time. Our response to the BEIS RoC stated: ***Major comments concerning the SIP:*** • There is a need to put a timeframe on the SIP. At what stage will the developer be required to reassess whether the parameters that have been assessed within the HRA have been exceeded? We suggest less than 12 months before construction starts is too short a timeframe when finances are committed at the FiD stage.

		limited benefit. How effective would at-source mitigation be under these circumstances? Please provide copies of any publications you wish to rely upon in evidence that have not already been provided.	• More information is required from the MMO / BEIS on how spatio-temporal impacts will be managed to prevent exceedance of the thresholds. A process will need to be developed to ensure continuing adherence to the SCI thresholds as multiple SIPs are developed over time, especially when piling can take place over several years, and new projects can come online during this time. Should potential exceedance of the thresholds occur, a process for dealing with this issue needs to be in place – the affected developers / industries will need to work together with the regulator and SNCBs to prevent adverse effect on the SCI. However, this process needs to be developed and agreed before SIPs are placed onto DCOs. • While this list is not exhaustive, Natural England would expect the following to be included in the SIP: ○ Finalised design plan ○ Updated HRA ○ Updated mitigation measures (if required) – outlining potential mitigation that can and cannot be used and the reasoning. ○ Where modelling via the RoC has been updated (e.g. the Dogger projects), further mitigation may be required to ensure porpoises are out of an enlarged PTS zone than was predicted in the original EIA. For example, the presentation at the consultation meeting on the 29th November stated the Dogger Bank Creyke Beck PTS zone could be up to 2.4km, this is a larger zone than was originally predicted in the EIA. ○ Detail the requirement for EPS licences and Marine Licences for UXO detonation.

			- Provide a timetable for development of the plan. E.g. Post CfD, and again pre FID to ensure timely agreements and timeframes for finances to be agreed. - There should be an acknowledgement that the SNCB noise management threshold approach should be implemented in an adaptive management context with a programme of strategic research and monitoring to improve the evidence base. This should be a collaborative effort across all stakeholders. In terms of the comments made within the WDC SoCG, any cost benefit analysis for additional mitigation will need to be made within the SIP for the specifics of the Hornsea 3 site.

3. Marine Processes

Ref:	Question to	Questions
		The ExA does not have any questions under this heading.

4. Ecology – Onshore

Ref:	Question to	Questions

5. Navigation and other offshore impacts

Ref:	Question to	Questions

6. Commercial fishing		
Ref:	**Question to**	**Questions**

7. Landscape, seascape and visual impacts		
Ref:	**Question to:**	**Topic for question(s)**

8. Historic environment		
Ref:	**Question to**	**Questions**

9. Land use and recreation		
Ref:	**Question to**	**Questions**

10. Socio - economic		
Ref:	**Question to:**	**Questions**

11. Transport and highway safety		
Ref:	**Question to**	**Questions**

12.	**Living conditions for local residents**	
Ref:	**Question to**	**Questions**

13.	**Content of the DCO**		
Ref:	**Question to**	**Questions**	
		References to the dDCO in this section relate to the version submitted at Deadline 1 [REP1-127]	
		Schedule 11 – Deemed Marine Licence (generation assets)	
Q2.13.23	MMO, NE	The MMO has commented [REP3-092] that it has received reports on Offshore Wind Farms (OWF) under construction which have cast doubt over the efficacy of soft-start mitigation measures relating to piling. In Condition 18, the MMO (supported by NE) suggests an amendment to the effect that, if monitoring shows significantly different impacts to those assessed in the ES, piling activity should cease until an update to the marine mammal monitoring plan and further monitoring requirements have been agreed.	In the case the MMO are referring to in [REP3-092], the initial noise monitoring report submitted to the MMO by the developer indicated that the soft start mitigation was not effective. (At this stage it is important to note that after subsequent investigations it was concluded that the original results were likely due to defective monitoring equipment and that the soft start procedures were actually operating as they should have been so the efficacy of this procedure is no longer in doubt.) This case has cast doubt over the efficacy of this condition in such circumstances.

		Please provide evidence of the need for this approach.	In a scenario whereby the noise monitoring is indicating a difference to the modelled noise levels, this could indicate that the mitigation in place is not effective. This could in turn indicate that there is a risk of injury to Marine Mammals. In all offshore windfarm cases, we seek to mitigate for injury, so the efficacy of the mitigation is called into question, then there may be a need for the developer to seek an EPS licence to ensure that their construction activity remains lawful. Although the applicant has previously highlighted (DCO hearing) that the MMO have the ability stop construction work, upon submission of a monitoring report to the MMO it can take several weeks for the MMO to consult with their advisers, and to receive that feedback. This could therefore mean marine mammals are injured and the conservation objectives of the SAC are hindered in the intervening period. It could also mean that the developer is operating without the appropriate licensing and therefore committing an offence. (N.B In the case the MMO refer to, the developer was late submitting their report, so

			it was actually several months before the issue came to light). In light of this, Natural England strongly supports the MMOs proposed changes to the wording of Condition 18. The key point being that any discrepancies need to be reported directly to the MMO (not hidden in a report) and work should cease until it can be confirmed that sufficient mitigation is in place or an appropriate course of action is determined. This not only seeks to safeguard EPS, but also safeguards the applicant from committing an offence.
		Schedule 13 – Arbitration	
		The ExA notes that discussions are ongoing and has no questions at this stage.	

14.	**Compulsory acquisition**	
Ref:	**Question to**	**Questions**

15.	**General**	
Ref:	**Question to:**	**Questions**

End of questions

Annex 1: Sub features of the Wash and North Norfolk Coast SAC

The Wash and North Norfolk Coast SAC.	
Annex I Feature	Sub feature (if relevant)
Atlantic salt meadows (Glauco-Puccinellietalia maritimae)	N/A
Coastal lagoons	N/A
Harbour (common) seal (Phoca vitulina)	N/A
Large shallow inlets and bays	Complex subfeatures: Atlantic salt meadows (Glauco-Puccinellietalia maritimae); Circalittoral rock; Intertidal Biogenic Reef – Mussel Beds; Intertidal Biogenic Reef – Sabellaria spp; Intertidal coarse sediment; Intertidal mud; Intertidal rock; Intertidal sand and muddy sand; Mediterranean and thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi); Subtidal biogenic reefs – Mussel beds; Subtidal biogenic reefs – Sabellaria spp.; Subtidal coarse sediment; Subtidal mixed sediment; Subtidal sand; Subtidal stony reef.
Mediterranean and thermo-Atlantic halophilous scrubs (Sarcocornetea fruticosi)	N/A
Mudflats and sandflats not covered by seawater at low tide	Intertidal coarse sediment; Intertidal mixed sediment; Intertidal mud; Intertidal sand and muddy sand; Intertidal seagrass beds.
Otter (Lutra lutra)	N/A
Reefs	Circalittoral rock; Intertidal biogenic reef – Mussel Beds; Intertidal biogenic reef – Sabellaria spp.; Intertidal rock; Subtidal biogenic reef – mussel beds; Subtidal biogenic reed – sabellaria spp.; Subtidal stony reef.
Salicornia and other annuals colonising mud and sand	N/A
Sandbanks which are slightly covered by sea water all the time	Subtidal coarse sediment; Subtidal mixed sediment; Subtidal mud; Subtidal sand.

Annex 2: List of Ornithological Data Requirements.

Natural England has provided advice regarding the collection and analysis of data that we consider appropriate, and methods for use of these data for assessment of collision risk, displacement and population level impacts, in our submissions at Deadline 1, 2 and 3.

Additionally, in order to provide transparency and a clear audit trail for the competent authority to undertake a full assessment of the proposed project, Natural England requests that the Applicant provides the following information for the relevant species in the assessment:

Baseline characterisation data

- Raw digital aerial survey data giving the number of birds of each species recorded on each survey day and each transect, with birds in flight and birds on the water presented separately.
- Tables of raw numbers of birds recorded in each year and month of the baseline surveys – presented for Hornsea Three, Hornsea Three plus 2km buffer and Hornsea Three plus 4 km buffer. With numbers presented separately for birds in flight and birds on the water at each scale.
- Tables of population estimates for birds in each year and month of the baseline surveys – presented for Hornsea Three, Hornsea Three plus 2km buffer and Hornsea Three plus 4 km buffer. With numbers presented separately for birds in flight and birds on the water (availability bias corrected) and upper and lower 95% confidence intervals around each population estimate provided and the coefficient of variation presented for each estimate.
- Tables of population estimates with 95% confidence intervals, generated by bootstrapping all the transect data (i.e. from all four cameras used for the digital aerial surveys) for a given month and year (i.e. two separate monthly estimates where there are data from two years) calculated for birds on the water (with availability bias correction) and birds in flight combined. Presented for Hornsea Three, Hornsea Three plus 2km buffer and Hornsea Three plus 4 km buffer. Standard deviations and coefficients of variation should also be presented for each population estimate.

- Tables of density estimates for birds in flight for each year and month of the baseline surveys - presented for Hornsea Three, Hornsea Three plus 2km buffer and Hornsea Three plus 4 km buffer. With upper and lower 95% confidence intervals around each density estimate provided. Standard deviations and coefficients of variation should also be presented for each population estimate.
- Tables of density estimates for birds in flight for each year and month of the baseline surveys with 95% confidence intervals, generated by bootstrapping all the transect data (i.e. from all four cameras used for the digital aerial surveys) for a given month and year (i.e. two separate monthly estimates where there are data from two years). Presented for Hornsea Three, Hornsea Three plus 2km buffer and Hornsea Three plus 4 km buffer. Standard deviations and coefficients of variation should also be presented for each density estimate.

Collision Risk Modelling

- Band Model spread-sheets populated with all the project, turbine and bird parameters and data used for CRM for each species (gannet, kittiwake, lesser black-backed gull, great black-backed gull and Herring gull);

- Natural England advises that monthly density estimates and confidence intervals for the CRM assessments should be derived using only data collected from the digital aerial surveys of Hornsea Three, and that the precision of the density estimates is improved by analysing the data collected from all four cameras, rather than the data from just two cameras;

- Natural England requests that the Applicant presents collision outputs for each species that reflect the variability and uncertainty around densities, flight heights and avoidance rates as a minimum. This should include presentation of collisions calculated using the relevant mean avoidance rate and ± 2SD of the mean avoidance rate as given in JNCC et al. (2014); presentation of collisions using mean, upper and lower 95% confidence intervals around the mean flight density data by month; presentation of collisions using mean, upper and lower flight height distribution data from Johnston et al. (2014), and presentation of collisions that reflect variability in Nocturnal Activity Factors as set out in REP1-211 for species where relevant.

Data to inform displacement assessment

Natural England advises that monthly abundance estimates and confidence intervals for use in the displacement assessments (prior to calculating seasonal mean of peaks) should:

- be derived using only data collected from the digital aerial surveys of Hornsea Three;
- be presented as population estimates of the Hornsea Three footprint and a 2km buffer (total birds in flight and on the water, after correcting for survey effort and availability bias) on a month by month basis for all 20 months individually with associated upper and lower confidence intervals

Natural England advises that the precision of the population estimates is improved by analysing the data collected from all four cameras, rather than the data from just two cameras.

Age Class Data

Hornsea Three have presented an apportioning approach for gannet, kittiwake and puffin based on at-sea age class data. As previously requested a detailed breakdown of age class data from boat and digital aerial data sets should be provided in order for Natural England to assess suitability of the two data sets and help to establish suitable apportioning figures.

Natural England notes that the applicant has submitted some of these data in REP1-169 (Deadline 1 Appendix 3), and further data in REP3-026 (Deadline 3 Appendix 17).

REP1-169 presents monthly age class data collected during boat based surveys (conducted between March 2010 and February 2013) for the breeding season months alone for kittiwake, gannet and puffin. In the case of kittiwake and gannet the data was limited to data from transects that overlap with Hornsea Three (it is not specified whether this includes a buffer, or is for the footprint of the windfarm alone – could this be clarified?). Puffin data is derived from boat based data collected across the entire Hornsea Zone (due to low sample sizes).

REP3-026 presents monthly age class data collected during digital aerial surveys, for the breeding months alone (in the years 2016/2017) for gannet and kittiwake, for the Hornsea three site plus a 4km buffer.

However the applicant’s submissions do not completely fulfil our requirements and so we take this opportunity to re-iterate these:

- We request age class data for the following species:
  Puffin, Gannet, Kittiwake, Guillemot, Razorbill
- Digital aerial data:
  All age class data should be provided (including the un-aged class) for every survey month (2016 and 2017 should be presented separately), and at all survey scales (site footprint, site plus 2km buffer, site plus 4km buffer);
- Boat based data:
  All age class data should be provided (including the un-aged class) for every survey month (years presented separately) of the sub-set of data that applies to Hornsea Three. Or in the cases where there is insufficient data at the HOW3 level, (e.g. puffin) then from the entire Hornsea zone.

Furthermore we note that age class data from the same digital aerial surveys (but limited to the time period April 2016-February 2017) is presented in a report prepared and submitted to NE as part of the EWG process (Hi Def 23 May 2017, HC00002-002), On preliminary inspection the data presented in REP3-026 appears to differ substantially from the data presented in the Hi Def report, in particular the proportion of birds aged is considerably higher in the Hi Def report for both gannets and kittiwake. We query why these discrepancies exist and request the applicant submits the Hi Def report (23 May 2017) as part of the examination process.

We re-iterate that age class data from the digital aerial surveys should be obtained and presented from analysis of the full 4 camera survey strips within each transect, in order to increase sample size.

Population Modelling (PVA Outputs)

PVA output tables for gannet, kittiwake, guillemot, razorbill and puffin for FFC pSPA for each model (density dependent and density independent and each demographic rate set used for each species) that include:

- PVA Tables with median, 2.5% confidence interval (CI) and 97.5% CI of un-impacted population size in each year of the simulation n=0…35;
- PVA Tables with median, 2.5% confidence interval (CI) and 97.5% CI of impacted population size for each year of the simulation n=0…35, with impact levels equating to adult mortality at 5 bird increments e.g. 5, 10, 15, 20 …n. Where n is greater than the maximum published in-combination total for the species;

- PVA Tables with the median, 2.5% confidence interval (CI) and 97.5% CI of counterfactual of final population size calculated using matched pairs, for each year of the simulation n=0…35, with impact levels equating to adult mortality at 5 bird increments e.g. 5, 10, 15, 20 …n. Where n is greater than the maximum published in-combination total for the species.
- PVA Tables with median, 2.5% confidence interval (CI) and 97.5% CI of un-impacted population growth rate in each year of the simulation n=0…35
- PVA Tables with median, 2.5% confidence interval (CI) and 97.5% CI of impacted population growth rate for each year of the simulation n=0…35, and for impact levels equating to adult mortality at 5 bird increments e.g. 5, 10, 15, 20 …n. Where n is greater than the maximum published in-combination total for the species;
- PVA Tables with the median, 2.5% confidence interval (CI) and 97.5% CI of counterfactual of population growth rate, calculated using matched pairs, for each year of the simulation n=0…35, with impact levels equating to adult mortality at 5 bird increments e.g. 5, 10, 15, 20 …n. Where n is greater than the maximum published in-combination total for the species.
- Information on all the demographic parameters used in PVA models for each species;
- Information on the stable age structure of the PVA models for each species;
- Information on starting population sizes used in the models;
- Information on the growth rates of the projected populations in the different models for each species;

Cumulative and in-combination project figures

- Cumulative and in-combination project tables for the relevant species that contain all available data by month (if available) or season with no changes applied to the figures derived from the relevant project Environmental Statement (or whatever source the data were derived from). Data on bird numbers should be presented for all months irrespective of whether the Applicant has apportioned birds from a project to a SPA site or population during a particular month or season. The precise source of the data presented should also be clearly referenced.

References

Joint Nature Conservation Committee (JNCC), Natural England (NE), Natural Resource Wales (NRW), Northern Ireland Environment Agency (NIEA), Scottish Natural Heritage (SNH). (2014). Joint Response from the Statutory Nature Conservation Bodies to the Marine Scotland Science Avoidance Rate Review. 25th November 2014.

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MacArthur Green

EVIDENCE REVIEW TO SUPPORT THE IDENTIFICATION OF POTENTIAL CONSERVATION MEASURES FOR SELECTED SPECIES OF SEABIRDS

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CONTENTS

LIST OF TABLES & FIGURES

Tables:

Figures:

ACRONYMS

AEOI	Adverse effect on integrity (of an SPA)
AOBs	Apparently Occupied Burrows (census unit, roughly equivalent to 'breeding pairs')
AONs	Apparently Occupied Nests (census unit, roughly equivalent to 'breeding pairs')
AOSs	Apparently Occupied Sites (census unit, roughly equivalent to 'breeding pairs')
B_{lim}	Limit of stock biomass below which fishing should be closed to protect stock survival
DEFRA	Department for Environment, Food and Rural Affairs
DDT	Dichlorodiphenyltrichloroethane
EU	European Union
FAO	Food and Agriculture Organisation (of the United Nations)
HRA	Habitat Regulations Appraisal
ICES	International Council for the Exploration of the Seas
IROPI	Imperative reasons of Overriding Public Interest
JNCC	Joint Nature Conservation Committee
MANOVA	Multivariate Analysis of Variance
MIEU	Major Infrastructure and Environment Unit (of DEFRA)
MPA	Marine Protected Area
NAO	North Atlantic Oscillation
Natura 2000	Network of SPAs and SACs in EU member states
NGOs	Non-governmental organisations
NNR	National Nature Reserve
PCBs	Poly-chlorinated biphenyls
PFOW	Pentland Firth – Orkney Waters strategic area for development of wet renewables
POPs	Persistent Organic Pollutants
RSPB	Royal Society for the Protection of Birds
SCR	Seabird Colony Register
SNCBs	Statutory Nature Conservation Bodies
SPA	Special Protection Area
SPEC	Priority species (of birds) of European conservation concern, as defined by the EU
SST	Sea Surface Temperature
WECs	Wave Energy Converters
WWT	Wildfowl and Wetlands Trust

EXECUTIVE SUMMARY

MacArthur Green has been commissioned by CEFAS to produce an '*evidence review to support the identification of potential conservation measures for selected species of seabird* (from here on referred to as 'the Project'). The focus of the evidence review was on identifying measures that could be implemented either at protected sites or elsewhere with a view to informing considerations around the mitigation or compensation of predicted impacts from offshore marine developments. The review did not include mitigation measures that could be implemented at the site of offshore marine developments.

The evidence review identifies measures that could theoretically be applied to conserve seabirds where there are concerns about the potential impacts from marine developments and considers the evidence that is available to support the consideration of these measures. The measures identified raise a range of legal, administrative and policy issues. Their use in relation to any specific project or programme of marine development needs to be considered in a subsequent step based on the design of the project or programme and its specific environmental context.

The main factors affecting seabird population trends in the British Isles are food abundance, fisheries, predatory mammals at seabird colonies, and climate change. Both productivity and survival can be affected, with impacts on survival being especially influential but variation in productivity being high for all the species of concern except northern gannet (which has consistently high productivity).

Breeding populations of many seabird species increased during the 20th Century. However, in the last few years (particularly since 2000) populations of most seabirds in the UK have declined (including Manx shearwater, Arctic skua, lesser black-backed gull, herring gull, great black-backed gull, and kittiwake). Breeding populations of red-throated divers, Sandwich terns, common terns, common guillemots and razorbills have remained stable though have declined in Scotland and increased in England and Wales. Great skuas have declined at large colonies but increased at small colonies. Gannet numbers have continued to increase. Future trends are likely to be continued decreases in the numbers of most species. Given the EU objective to ban discarding by fishing vessels it is likely this will particularly affect those scavenging seabirds currently dependent on fishery discards.

Several management options to increase seabird productivity or survival, or both, have been identified in this evidence review. However, the scope for effective management varies among species. The most cost-effective, evidence-based management options for the species discussed in this report are:

- Provision of nest platforms for **red-throated divers**;
- Eradication of alien invasive mammal predators on islands with **Manx shearwater** colonies;
- Supplementary feeding of breeding pairs of **Arctic skuas**;
- Cessation of culling of breeding **lesser black-backed gulls**, **herring gulls**, and **great black-backed gulls**, and predator-proof fencing around mainland colonies of these gulls subject to fox predation;
- Closure of sandeel and sprat fisheries in UK waters to increase productivity and survival of **kittiwakes**, **common guillemots**, **razorbills, and Atlantic puffins**;
- Predator-proof fencing to exclude foxes from affected **Sandwich tern** colonies and engineering (every few years if necessary) to reduce risk of tidal flooding where this is a problem; and,
- Eradication of mink from islands with **common tern** colonies and deployment of predator-proof nesting rafts.

For gannets, scope to increase productivity or survival appears very limited, but includes a reduction in seabird by-catch by fisheries. The review also identified as a theoretical option that ending the legal harvest of chicks at Sula Sgeir would make a small contribution. For great skuas, any practical measures to increase productivity or survival appear very limited, although closure of sandeel and sprat fisheries may have a small beneficial effect, while supplementary feeding at colonies would be likely to be effective, but rather impractical. While presenting these single-species management options, we consider that interventions should aim to ensure functioning ecosystems as the highest priority. We therefore point out that some measures that could be implemented may benefit many seabird species rather than just a single species (e.g. closure of sandeel fishing), and we recommend consideration of the fact that great skua numbers (and those of some other scavenging seabird species such as herring gull, great black-backed gull and lesser black-backed gull) have been artificially elevated by human actions in the past (especially discarding by fisheries), and so numbers at SPAs at the time of designation represent elevated rather than sustainable population sizes, so desirable target population sizes may require further consideration by SNCBs.

SPECIES	**OPTION**
Red-throated Divers	Provision of nest platforms
Manx shearwater	Eradication of alien invasive mammal predators on islands with Manx shearwater colonies.
Arctic skuas	Supplementary feeding of breeding pairs.
Lesser black-backed gulls, herring gulls, and great black-backed gulls	Cease culling of breeding lesser black-backed gulls, herring gulls, and great black-backed gulls, and predator-proof fencing around mainland colonies of these gulls subject to fox predation.
kittiwakes, common guillemots and razorbills	Closure of sandeel and sprat fisheries in UK waters to increase productivity and survival of kittiwakes, common guillemots and razorbills.
Sandwich tern	Predator-proof fencing to exclude foxes from affected Sandwich tern colonies and engineering (every few years if necessary) to reduce risk of tidal flooding where this is a problem.
Common tern	Eradication of mink from islands with common tern colonies and deployment of predator-proof nesting rafts.
Gannets	Scope to increase productivity or survival appears very limited, but a reduction in seabird by-catch by fisheries would make a small contribution as might ending the legal harvest of chicks at Sula Sgeir.
Great skuas	Any practical measures to increase productivity or survival appear very limited, although closure of sandeel and sprat fisheries may have a small beneficial effect, while supplementary feeding at

	colonies would be likely to be effective, but rather impractical. Reduction in seabird by-catch by fisheries may also be useful for this species, but evidence base for this is very limited.

1. INTRODUCTION

MacArthur Green has been commissioned by CEFAS to produce an '*evidence review to support the identification of potential conservation measures for selected species of seabird'*(from here on referred to as 'the Project'). The focus of the evidence review was on identifying measures that could be implemented either at protected sites or elsewhere with a view to informing considerations around the mitigation or compensation of predicted impacts from offshore marine developments. The review did not include on-site mitigation measures that could be implemented at the site of developments. In the case of offshore wind these have been addressed by a recent and complementary Defra R&D report (Cook et al. 2011), but we do note in particular that recent studies show that most seabirds fly low over the sea so that raising turbine heights could potentially reduce collision risk for several species (Cook et al. 2012).

The assessment of adverse impacts on populations of certain bird species is potentially the most imminent barrier to the consenting of offshore wind energy development. The Habitats and Wild Birds Directives Marine Evidence Group and the Defra Major Infrastructure and Environment Unit (MIEU) have identified the need to collate information that would deepen technical and legal understanding of what might constitute compensation or mitigation (to reduce or avoid AEOI) measures for the impacts of birds from offshore wind. The evidence review identifies measures that could theoretically be applied to conserve seabirds where there are concerns about the potential impacts from marine developments and considers the evidence that is available to support the consideration of these measures. The project has dealt solely with technical aspects and makes no judgement about the legal position in respect of any of the measures identified.Their use in relation to any specific project or programme of marine development needs to be considered in a subsequent step based on the design of the project or programme and its specific environmental context.

The project has informed work by the MIEU to agree an outline definition of what constitutes mitigation and compensationby enabling a greater practical understanding of what measures are feasible and the effectiveness of each potential measure will feed into further consideration by the MIEU of the legal viability of particular options.

Given the potential for uncertainty in the interpretation of the terms 'mitigation' and 'compensation', we have minimised the the use of those terms in this report, and instead discuss management measures that aim to increase the survival rates or productivity of seabird populations.

2. AIMS AND OBJECTIVES

The CEFAS tender document (Ref: C5718I) for this project set out the following aim for this project : '*collate information to support the identification and development of potential measures that could be used to compensate or mitigate for the impacts of offshore developments on key bird species where there is an imminent consenting risk (i.e. where, on the basis of available evidence, assessments of proposed developments may predict an adverse effect on the integrity of sites designated for their protection or it may not be possible to ascertain that there will be no such effect)*'.

This was done by carrying out an evidence review that collates information to support the identification and development of measures to enhance survival and reproduction of key seabird species which may be affected by offshore wind farm developments where there is an imminent consenting risk (as defined above). The initial results of the evidence review (the draft report)were presented, by MacArthur Green, at a workshop involving regulators, advisers, industry and NGOs in May 2013. The main focus of the workshop was to test and understand the measures proposed. The findings were then revised (final report) taking into account the outcomes of the workshop where appropriate.

To inform the focus of the work, an initial brief review is presented based on up-to-date literature, on the most important factors that adversely affect seabird survival and productivity. This initial review considers all species of seabirds with a focus primarily on the populations in the British Isles, but considers literature from all parts of the world where that is informative. Following on from that initial review, the focus is on the key seabird species.

The key seabird species present on SPAs which are considered to be most at risk of adverse effects as a result of offshore wind farms developments are:

- Red throated diver *Gavia stellata*,
- Northern gannet *Morus bassanus*,
- Manx shearwater *Puffinus puffinus*,
- Arctic skua *Stercorarius parasiticus*,
- Great skua *Stercorarius skua*,
- Lesser black-backed gull *Larus fuscus graellsii*,
- Herring gull *Larus argentatus*,
- Great black-backed gull *Larus marinus*,
- Black-legged kittiwake *Rissa tridactyla*,
- Sandwich tern *Sterna sandvicensis*,
- Common tern *Sterna hirundo*,
- Common guillemot *Uria aalge*, and
- Razorbill *Alca torda*
- Atlantic puffin *Fratercula arctica*.

3. REPORT STRUCTURE

This report considers factors affecting seabird populations, especially within the British Isles (Section 4), introduces the use of population modelling to quantify the impact of changes in survival rates and productivity on seabird population trends (Section 5), and considers implications of life history ideas for management (Section 6).

For each of the key seabird species the following are considered in turn; the status and ecology of the species in the British Isles, recent and likely future population trends, factors affecting survival rates, breeding success of monitored populations in the British Isles, and the most promising management options that could increase survival rates or productivity of each species.

The report is structured to present evidence for each individual species separately in Sections 7 (red-throated diver) to 19 (razorbill). While there is considerable repetition across these species accounts sections, each one comprises a full account for that species in order to minimise the need for frequent cross-referencing within the report, and since it seems likely that only one or at most a few of the single species accounts will be of particular concern at some future date and should therefore each be complete in itself. Section 20 presents recommendations for management actions. We note that while the demography of seabirds means that measures to increase survival rates are likely to have a more powerful influence on population trend than measures to increase productivity, the evidence base to support decision making in relation to measures that affect survival is very weak, whereas there is a very strong evidence base to support decision making in relation to measures to increase productivity. Therefore, any evidence-based approach must focus on measures affecting seabird productivity even though these are likely to be less powerful than measures that increase survival rates.

4. GENERAL REVIEW OF FACTORS AFFECTING SEABIRD POPULATION SIZES WITH PARTICULAR REFERENCE TO THE BREEDING SEABIRDS OF THE BRITISH ISLES

4.1 Introduction

This review considers the consensus views in the scientific literature as to what factors primarily affect the trends in seabird population size, with particular reference to seabirds in the British Isles. This overview is intended as an introduction to the topic, before considering specific factors that might provide opportunities to carry out management actions that would recompense potential impacts of offshore wind farms on demographic trends in seabird populations.

Studies of seabirds have identified a large number of factors that affect seabird demography, and hence influence population trends, in the British Isles. These were listed by Mitchell et al. (2004) as:

- historic exploitation and persecution in the British Isles;
- current exploitation and persecution in the British Isles;
- exploitation and persecution of British and Irish seabirds while they are abroad;
- bycatch in nets;
- bycatch on longlines;
- collision with wind turbines;
- mammalian predation;
- avian predation;
- avian diseases and natural toxins (including botulism, puffinosis, ticks, red tides);
- food availability;
- effects of fisheries on food availability (through depletion of fish stocks, increases of non-target fish stocks and through provision of offal and discards);
- effects of offshore development on food availability;
- effects of refuse management on food availability;
- effects of farming on food availability;
- loss of nesting habitat;
- provision of new nesting habitat;
- oil pollution;
- persistent organic pollutants;
- heavy metal pollution;
- plastic pollution;
- global climate change;
- large-scale atmospheric and oceanographic events (e.g. El Niño, North Atlantic Oscillation); and,
- short-term weather events (e.g. storms).

Some of these factors affect only certain kinds of seabirds depending on their particular ecology, while some affect most or all species. Some have major impacts on demography of many species, while some have little or no detectable impact on most species.

Compared to most other kinds of birds, adult seabirds are exceptionally long-lived, with low reproductive output and late maturity (many species do not start to breed until three to eight years old, and many lay only a single egg which has a relatively low probability of survival to adult status). These demographic features of seabirds lead to low inherent population growth rates and generally poor abilities to recover from factors which reduce populations, particularly if these result from additional adult mortality. Consequently, any attempt to recompense for increased adult mortality by enhancing reproductive output will be constrained by both the limited extent to which low rates of reproductive output can be increased and its smaller relative contribution to population growth.

Thus, any management action that reduces mortality of adults is likely to have a greater influence on seabird population trajectory than action aimed at enhancing reproductive output, although the latter may still be effective even with long-lived seabirds with especially low reproductive rates (Finkelstein et al. 2010).

Compared to other types of birds, seabirds might be expected to have populations that remain relatively stable over periods of years or decades, because adult survival rates tend to be high, reproductive output tends to be low, and birds do not start to breed until several years old. So there is little scope for rapid increase in numbers, and unless mortality rates increase above 'normal' levels, numbers would not be expected to decrease rapidly. Nevertheless, many seabird breeding populations in Britain and Ireland have changed in numbers very considerably over the past 100 years, with a general trend for most species to have increased from 1900 to about 1990 to 2000, but in many cases to have declined since reaching a peak in breeding numbers late in the 20th Century; methods to census breeding seabirds are well developed, and counts of colony size are now fairly accurate for most species, though are less reliable for nocturnal and burrow-nesting seabirds (Mitchell et al. 2004, Forrester et al. 2007, Mitchell and Daunt 2010, Foster and Marrs 2012).

Identifying causes of population increase or decrease is difficult, as the response time of seabird populations tends to be slow. Any factor reducing breeding success, for example, will not be evident in terms of breeding numbers for several years, and may be delayed further by the buffering effect of a pool of non-breeders waiting to recruit into the breeding population (Klomp and Furness 1992). Furthermore, processes of emigration and immigration can also buffer local or even regional scale impacts on seabird demography. To compound this problem, counts of breeding numbers of seabirds at individual colonies do not necessarily reflect changes in numbers in the region as a whole as birds may move between colonies for various reasons such as local impacts of predators (Jennings et al. 2012), although such movements, which are typical of terns, may not occur in some other seabird species. Since national surveys of breeding seabird numbers tend to be made only every 10 to 15 years, and include a considerable inaccuracy in survey data, changes in breeding numbers may not become evident until several decades after the factor causing the change had its effect. Furthermore, in most cases, changes in population size are normally defined as changes in numbers of breeding pairs rather than from changes in total numbers in the population. These two metrics are not necessarily closely correlated. For example, breeding numbers could decline dramatically because increasing proportions of the population choose not to breed, while total numbers in the population may remain relatively stable. Such patterns appear to occur especially in terns where birds may opt not to attempt to breed when conditions are poor (Monaghan et al. 1989), but may apply to many other seabird species when conditions are extreme (Mavor et al. 2006).

Mitchell et al. (2004) suggested that the factors that most affected seabird populations in Britain and Ireland in the past were historic exploitation and persecution, exploitation of British and Irish seabirds abroad (i.e. outside the breeding season when the birds migrate), mammalian predation (especially involving introduced alien mammals), food availability, and fisheries. They also suggested that current population trends were most likely to be influenced by food availability, fisheries and climate change. Seabird researchers across nine nations identified the 20 highest priority global research questions regarding recent seabird declines and grouped them into six categories: population dynamics, spatial ecology, tropho-dynamics, fisheries interactions, response to global climate change, and management of anthropogenic impacts (Lewison et al. 2012). These six categories are all consistent with, or at least related to, the factors affecting seabirds covered in this review.

In this review, evidence is presented for each of the main factors affecting seabird populations, with particular reference to populations in the British Isles but taking global examples when these are informative and provide particularly clear examples. This initial introduction considers all species of seabirds, and not just the subset considered to be potential targets for management action to recompense impacts of offshore wind farms.

4.2 Exploitation and persecution

Mitchell et al. (2004) suggest that persecution and exploitation of seabirds during the 19th Century and the cessation of this during the 20th Century is '*likely to be at least partially responsible for the increases in most seabirds species observed between the 1930s and the mid-1980s in Britain*'. The same authors also suggest that current levels of persecution and exploitation in Britain can affect rates of population change of some species; culling of large gulls, shooting of great cormorants *Phalacrocorax carbo* under licence to protect freshwater fisheries (see also Smith et al. 2008), illegal shooting of skuas in some parts of Scotland, and the traditional harvest of northern gannet chicks on Sula Sgeir may all have reduced population growth rates or increased declines of particular populations. However, these effects appear to be less influential and also much more local than the effects of factors such as changes in food supply or climate change. Exploitation of British breeding seabirds in winter when they are in their wintering areas, or during migration, may influence breeding numbers of roseate terns *Sterna dougallii* in Britain (Mitchell et al. 2004). Although large numbers of auks have been shot on the coast of Norway and these include birds from British colonies, there is no evidence to indicate that this harvesting has affected numbers at British colonies (Mitchell et al. 2004). Overall, exploitation and persecution seem unlikely to be significant factors affecting current seabird numbers in the British Isles, except locally in a few cases.

4.3 Food supply

There is a broad consensus that seabird breeding numbers are particularly affected by food abundance, and that this factor (in some cases modulated by fisheries or by climate change) is the single most important influence on seabird population sizes at a regional level. Small, surface-feeding seabirds with short foraging ranges and a lack of alternative foods are especially vulnerable to such impacts (Furness and Tasker 2000). Changes in breeding numbers of seabirds resulting from changes in fish abundance can be dramatic. For example, common tern breeding numbers in the Firth of Forth were reduced to about half when sprat abundance fell (Jennings et al. 2012), Arctic tern and Arctic skua breeding numbers in Shetland fell by at least 50% after the decline of the Shetland sandeel stock (Forrester et al. 2007), common guillemot breeding numbers in the Barents Sea fell by over 90% when the capelin *Mallotus villosus* stock collapsed (Sakshaug et al. 2009). In contrast, swift tern and African penguin *Spheniscus demersus* numbers increased in part of the Benguela ecosystem when the sardine stock redistributed into the vicinity of these particular colonies, while breeding numbers in areas where fish abundance declined fell dramatically (Cury et al. 2011).

Mitchell et al. (2004) conclude that seabird demography in Britain and Ireland is '*strongly affected by the availability of food*'. Many breeding seabirds feed primarily on small schooling pelagic fish. These fish are important food because they tend to be abundant, available in the upper layers of the sea, have a high energy density, and are relatively small so are easy for seabirds to catch and swallow. In many different parts of the world and for many different kinds of seabirds, breeding success shows a strong sigmoidal correlation with the abundance of their preferred prey fish (Cury et al. 2011). Cury et al. (2011) identified a threshold of one third of the long-term maximum prey biomass of forage fish abundance. Below this stock biomass levelmany seabird species suffer from reduced and more variable productivity which is likely to lead to population decline if sustained. Food availability of preferred prey species varies across the oceans. For example, comparison of flight time data from geolocation loggers on wintering northern gannets suggested that food availability at this time is much more varied in the north of the range (Bay of Biscay, Celtic sea) but more consistent off Western Africa (Garthe et al. 2012). In waters around the British Isles, sandeels *Ammodytes marinus,* sprats *Sprattus sprattus,* and juvenile herring *Clupea harengus,* are preferred food for most seabirds in summer (Mitchell et al. 2004). In winter, a few deep-diving seabirds continue to feed on sandeels but because sandeels spend most of the winter buried in the sea bed, they are unavailable to most seabird species; seabird diets in winter tend to be more diverse than in summer, with less focus on small pelagic fish. In northern waters such as around Shetland, there are no sprats and no juvenile herring, so seabirds depend strongly on sandeels in summer and tend to move away from the area to spend the winter elsewhere. In coastal areas further south there may be sprats and young herring as well as sandeels, providing seabirds with a more stable prey base.

Several species' breeding success in Shetland, including that of the black-legged kittiwake (Votier et al. 2008), Arctic tern and Arctic skua*,* shows strong correlation with sandeel stock biomass (Furness, 2002). A potential minimum acceptable stock biomass (known as B_{lim}) has been put forward for seabirds (specifically kittiwakes and Arctic skuas) as a total stock biomass of 30,000 tonnes of sandeels in the Shetland stock (Furness, 2007). Below this, sandeel-dependent seabirds would be predicted to suffer from reduced breeding success, and hence potentially from population declines. One such example from the North Sea caused breeding failures for a variety of seabirds in 2004, after sandeel landings (reflecting stock biomass) decreased by over 50% between 2003 and 2006 (Frederiksen et al. 2006). In 2005 at North Sutor, North Scotland, black-legged kittiwakes suffered from complete breeding failure. Along with other species' low levels of breeding success, this decline was suspected to be linked to a shortage in sandeel abundance at the time (Mavor et al. 2005). On the Isle of May, there was a positive correlation between seabird breeding productivity and the size of sandeel prey (both in terms of individual fish size and stock biomass) (seen for Atlantic puffin, shag, common guillemot, razorbill and kittiwake; Frederiksen et al. 2006). Another study on the Isle of May kittiwake colony in relation to the opening and closure of the Danish sandeel fishery operating locally showed that breeding productivity was significantly reduced while the fishery was active, reducing the availability of the kittwake's preferred prey (Frederiksen et al. 2008).

Overall it is evident that particular species of seabirds in northern Scotland, namely black-legged kittiwakes, Arctic terns, Arctic skuas and Atlantic puffins have a strong prey preference for sandeels and hence are vulnerable to changes in their abundance. A few species of seabirds on the other hand appear unaffected by sandeel stock biomass: gannet breeding success in northern Scotland shows no correlation with the availability of sandeels. Although they will feed on sandeels when available, when sandeel abundance is low they switch to alternative prey such as adult herring or mackerel, fish that are too large for most other seabird species to swallow.

Arctic skua breeding success in Shetland has shown a strong correlation with the biomass of the Shetland sandeel stock, but in addition the breeding numbers in Shetland have declined substantially over the last 20 years. A study was carried out to find out the source of this population decline and it was found that breeding pairs supplemented with food had a higher nest attendance rate than those without supplement. Not only did food availability have a strong impact on breeding success, but it also affected adult survival. Birds given supplementary food were more likely to return to breed the next year than were unfed controls (Davis et al. 2005). This is a rare example of a measure that has been demonstrated to affect survival. It supports the frequent suggestion that improved food supply will increase overwinter survival or auunal survival of seabirds through improvement to body condition. However, demonstrating this effect is difficult and has been beyond the scope of most studies on seabird ecology.

The impact of food availability on adult survival is potentially more influential on population trend than the impact on breeding success. Similar relationships between food supply and seabird population size have been reported further afield. For example, off South African's Western Cape, swift tern *Sterna bergii* population numbers track the abundance of their prey; anchovy *Engraulis capensis* and sardines *Sardinops sagax* (Crawford 2009). In Japan, the availability of anchovy *Engraulis japonicus* and sandeel *Ammodytes personatus* prey was recorded to affect seabird chick diet, growth rate and breeding success of the rhinoceros auklet *Cerorhinca monocerata*, Japanese cormorant *Phalacrocorax filamentous* and black-tailed gull *Larus crassirostris* (Watanuki et al. 2012). In the Firth of Forth, common tern breeding numbers at individual colonies show rather different trends over recent decades, but the regional population size correlates with the abundance of sprats (their main breeding season food) in the area (Jennings et al. 2012). The variable dynamics of individual colonies seems to be driven by predation impacts and presence of gulls, with terns moving between colonies in response. Overall breeding numbers in the region varied much less than numbers at individual colonies. On the east coast of England and Scotland, large kittiwake colony "clusters" were associated with aggregations of sandeels whereas on the west coast, these predator-prey dynamics were not as apparent. The study showed that regional variation in prey abundance has a stronger impact on kittiwake populations than local prey depletion (Frederiksen et al. 2005, Fauchald et al. 2011). Harris et al. (2010) used geolocators to identify wintering areas of breeding adult puffins from the Isle of May. Mortality rates of puffins at this colony had increased and the authors concluded that increased mortality related to reduced food availability in the wintering areas used by these birds in the North Sea, and changes in their migration behaviour. This, and other studies, indicate a complex relationship between food, survival and productivity. Birds unable to find sufficient food during winter may not be in adequate body condition in spring to sustain successful breeding, so that food shortage may have complex 'carry-over' effects on productivity as well as migration behaviour and overwinter survival.

Not only are seabird populations affected by the amount of prey available, some can be vulnerable to changes in the prey age-class structure. This highlights how specific some seabird's reliance can be upon a relatively unpredictable food source. An unproductive breeding season for Atlantic puffins at St Kilda in 2006 was correlated with a low availability of the appropriate age class of sandeel (Mavor et al. 2006). The breeding success of common guillemots, razorbills and shags in the North Sea has shown a positive correlation to sandeel growth rates (Burthe et al. 2012). In Shetland and on the Isle of May, a positive correlation between 0-group sandeel abundance and adult survival of kittiwakes is apparent (Oro and Furness, 2002, Wanless et al. 2007). Food abundance can affect a wide range of demographic parameters and even such biometrics as egg size. Decreases in puffin egg size at colonies in Norway and Scotland have been related to effects of reduced food fish abundance (Barrett et al. 2012).

Food availability can affect the foraging ranges of seabirds, with decreased levels of food increasing the distance that birds will travel to feed. A study of breeding northern gannet colonies around the UK concluded that there was a positive correlation between population size and mean foraging trip duration (Lewis et al. 2001). Larger populations of seabirds increase competition for food and hence at larger colonies, birds will have to travel further to obtain food, depleting energy stores and potentially leaving nests unattended during the breeding season.

Great skua migratory routes were studied to identify changes over time and Scottish skuas were recorded to winter off northwest Africa, much further south than previously thought. It was suggested that this migration further south could be linked to an increase in fishery discards and increasing pelagic fish stocks in the area (Magnusdottir et al. 2012). The survival of Scottish adult common guillemots has been correlated for colonies which share wintering areas, suggesting that some environmental factor present at these shared wintering sites is affecting adult guillemot survival (Reynolds et al. 2011). Perhaps the most likely common factor affecting adult survival is food availability.

4.4 Fisheries

Fisheries can affect seabird populations in several ways, either directly, as for example through bycatch of seabirds in fishing gear and though provision of discards eaten by scavenging seabirds, or indirectly though influences on ecosystem functioning and fish community composition. Fisheries for the small pelagic fish that seabirds tend to target as preferred food can reduce food availability and so can cause breeding failures of seabirds (Wagner and Boersma 2011). However, fisheries for predatory fish may alter food web structure such that small pelagic fish stocks increase, and so trawl fisheries reducing predatory fish biomass can benefit seabird populations.

Fisheries can also alter food availability to seabirds through the provision of offal (fish guts) and discards (whole fish rejected as beyond quota or too small or not worth taking to market and thrown back at sea). Most discards tend to come from bottom trawl fisheries. This supply of food that would otherwise be inaccessible to seabirds (because the fish involved are generally much too big to be swallowed by the smaller species which are able to dive to the sea floor) can increase numbers of scavenging seabirds. However, reductions in the amounts of offal or discards can then lead to large scavenging seabirds such as great skuas, great black-backed gulls, herring gulls and lesser black-backed gulls to predation on smaller seabird species, adding a novel predatory impact onto small seabird populations in their vicinity (Votier et al. 2004).

4.4.1 Fishery bycatch

Seabird mortality through bycatch from fisheries can have a strong impact on seabird populations in certain parts of the world. In the UK, long-line fishing has been a cause of bycatch mortality in gannets, great skuas and northern fulmars (Dunn and Steel, 2001), while set gill nets have drowned large numbers of auks in the past. However, according to Mitchell et al. (2004) changes in food availability may be involved in declines in fulmar numbers in Shetland, and the role of longline mortality is unclear. Fisheries bycatch was also a named potential factor causing the decline in adult survival of Yelkouan shearwaters *Puffinus yelkouan* between 1969 and 1994 in Malta, (Oppel et al. 2011). The decline of common guillemots recorded during the 1980s at a Low Arctic colony in Newfoundland was considered to be associated with bycatch drowning from gillnets which overwhelmed any impact from climate change (Regular et al. 2010). The by-catch of albatrosses and petrels in the North Pacific and Southern Ocean is currently a major problem caused by long-line fisheries (Tasker et al. 2000). This is an example where major gains can be made by mitigation measures to reduce bycatch mortality. However, although bycatch mortality of seabirds due to fisheries can be an important issue in some parts of the world, it is apparently only a minor influence for seabird populations breeding in Britain. According to Mitchell et al. (2004) '*studies of bycatch mortality in Britain showed that large numbers of auks may be caught and drowned in these nets but the rates were insufficient to cause local population declines*' and '*mortality in nets outside British waters during winter was insufficient to cause population declines of auks in Britain and Ireland*'. There is a small amount of evidence to show that gannnets and great skuas occasionally get caught as fisheries bycatch. While not likely to contribute, in and of itself, to any population decline due to the (apparently) small numbers being caught, there is the potential to minimise this threat to adult survival within European waters through the EC-PoA for Seabirds, which is specifically addressing the issue of seabird bycatch in European waters and among European fleets. While such efforts are to be welcomed, as any reductions in adult mortality reduces pressure on declining populations, reductions in fisheries bycatch (for this species at least) could not be recommended as sufficient compensatory mitigation for the impacts of OWFs as the evidence base on the magnitude of this bycatch is deficient. However, future quantification of bycatch may indicate that there is potential to compensate by measures that reduce the bycatch. Recent research suggests that the bycatch of some UK seabirds may be much higher than previously thought. For example, it is estimated that the Gran Sol fishery accidentally caught 1,331 gannets per year during 2006/07 (BirdLife 2009), while recent unconfirmed reports from West Africa (a core winter ground for some UK seabirds including gannets and great skuas) indicate large numbers being caught by fisheries, with some of these being found in refrigerated containers destined for the Far East http://seabirds.net/posts/2013/02/13/evidence-for-massive-bycatch-in-chinese-fisheries/ .

4.4.2 Fishing of pelagic prey fish stocks

Although there are many examples of seabird breeding success relating to pelagic fish abundance, there is often dispute as to how much the abundance of pelagic fish is determined by fishing and how much variation is due to natural factors. The presence of sandeel fishing on the Wee Bankie, E Scotland, has been clearly correlated with low and variable kittiwake breeding success and reduced adult survival, whereas before the fishery opened (in 1990), breeding success at the Isle of May colony was much higher (Frederiksen et al. 2004, Scott et al. 2006). However, this relationship was complicated as there was also an influence of sea temperature (i.e. global climate change impact) in addition to the influence of presence or absence of a sandeel fishery. After closure of the sandeel fishery in this area in 1998, breeding success of kittiwakes within the fishery area improved

(Frederiksen et al. 2004) and matched that in a control (unfished) area (Frederiksen and Wanless 2006). This one case study therefore provides evidence that closing a fishery can improve food supply for seabirds. However, although there was also evidence of an improvement of breeding conditions for Sandwich terns, there was no significant change in breeding success of other species (such as auks) (Frederiksen and Wanless 2006).

In Shetland, although the relationships between sandeel stock biomass and seabird breeding success are clear, it is uncertain whether the decline in sandeel abundance in the late 1980s and since 2000 is due to fishing impacts or whether it is due to natural factors or to climate change. There is, for example, some evidence to suggest that the declines in sandeel abundance in the Shetland stock may have been influenced by top down predation impacts due to recovery of adult herring biomass in the area in the late 1980s and after 2000 (Frederiksen et al. 2007). Examples where impacts of the fishery on pelagic fish can be more clearly seen include the collapse of seabird populations in Namibia and parts of South Africa following depletion of sardine and anchovy stocks by overexploitation (Pichegru et al. 2010a, Cury et al. 2011) and the mass mortality of seabirds in Peru following depletion of the anchoveta *Engraulis ringens* stock by fishing (Wagner and Boersma 2011). The fact that closing fisheries around African penguin colonies can lead to rapid recovery of breeding success and numbers (Pichegru et al. 2010b) also provides clear evidence of a fishery impact on seabird prey abundance in that ecosystem. Changes in breeding numbers of seabirds resulting from changes in fish abundance driven by fisheries can be dramatic. For example, African penguin numbers fell by over 90% when sardines were overfished in Namibia (Cury et al. 2011), Atlantic puffin breeding numbers at Røst, Norway, fell by 50% after the herring stock was depleted by fishing (Gjøsæter et al. 2009, Cury et al. 2011). Richerson et al. (2010) showed by modelling seabird-fishery interactions that a 20% reduction in sandeel harvest from Shetland could potentially double breeding success of Arctic terns in Shetland, by allowing sandeel stock biomass to recover from the depletion caused by a fishery. However, in reality there has been no commercial sandeel fishery at Shetland since 1990, and stocks of this species have not recovered, indicating that such theoretical predictions do not necessarily match up with empirical observation. Other factors may subsequently have prevented the sandeel stock from recovering to levels that were present before the fishery of the 1970s and 1980s. Bertrand et al. (2012) showed that seabird foraging can be affected by fishery depletion of pelagic fish within a few days, with seabirds being forced by fishery depletion of local stocks to travel further to search for food.

4.4.3 Discards and offal

Although there are efforts currently in place to reduce the amount of fish being discarded from vessels at sea (Bicknell et al. 2013), scavenging seabirds still consume large quantities of discarded fish, and offal (60-80% of roundfish discards and 70-95% of offal discards) (Furness et al. 2007). Great skua colonies of the northern North Sea have seen rapid population growth from early in the 20th Century up to the end of the century, and this has been correlated to increases in fishery discards from the 1940s to the 1980s (Votier et al. 2004). From a study on great skua populations around Shetland it was clear that these birds relied heavily on fishery discards as a large component of their diet (Votier et al. 2008). Similar dependence on fishery discards is apparent in large colonies of great black-backed gulls and possibly in several other gull species (Mitchell et al. 2004). As reducing fishery discards remains a top priority for FAO's policy for Responsible Fisheries (Furness 2003, Bicknell et al. 2013), it seems reasonable to predict that unless these seabird species can switch to alternative diets, their current breeding success is unlikely to be sustained.

Wintering seabirds off the Western coast of Africa (Sahara and Mauritania) have also been reported to have a close association with fishing trawlers, feeding on the discards. However, hydrography (specifically cold water upwelling) there has perhaps an equally important role to play in seabird success (Camphuysen and Van der Meer, 2005). During a study to investigate the winter migration patterns of northern gannets, it was revealed that they often stay in areas of elevated discard availability. Off western Africa, 88.8% of wintering gannets were found to be associating with fishing vessels (Kubetzki et al. 2009). As gannets are adapted for diving and catching live fish prey, it is not surprising that they are switching to discards as a food source when availability of pelagic fish is low. It has been suggested, however, that fishery discards are a poor substitute for higher energy anchovy and sardine, which gannets would preferably feed on in favourable conditions. Between 1986 and 2006, anchovy and sardine availability off the west coast of South Africa decreased from ca.84% to ca.35%, correlated with a decline in growth rate of cape gannet *Morus capensis* chicks (Mullers et al*. 2009)*. This decline in anchovy and sardine availability can be linked to the activities of a purse-seine fishery, which in 2007 took 41% of the food needed for the Malgas Island cape gannet colony (72,000 birds) (Okes et al. 2009). Although northern gannets apparently switch from feeding mainly on discards in winter to feeding more on pelagic fish while breeding, even during the chick-rearing period almost all gannets seem to forage around trawlers at times (Votier et al. 2013).

A study of Balearic shearwater *Puffinus mauretanicus* diet pre-incubation and during the breeding period showed an interesting link to the energy differences in prey species. During the pre-incubation period, breeding adults were recorded to feed mainly on demersal (discarded) fish, whereas during incubation, had a tendency to switch their foraging effort to higher energy fish such as anchovies and pilchards *Sardina pilchardus*, (this was particularly prevalent in the female adults), (Navarro et al. 2009). This prey-switching behaviour in the shearwaters could be linked to natural (seasonal) or unnatural (caused by fisheries) variations in the availability of their preferred prey. However, the increased switching amongst females would suggest that this move away from discard feeding was linked to feeding their young since discards are lower in energy (Mullers et al. 2009). Österblom et al. (2008) put forward the “junk-food hypothesis” affecting marine top predators such as seabirds. This hypothesis suggests that it is also the quality, not just the quantity of food which affects seabird population trajectories. Given that the EU Common Fisheries Policy is currently being rewritten and that it is intended that the EU will introduce a ‘no discards’ policy, scavenging seabirds in the British Isles are very likely to face a drastic decrease in food availability (Votier et al. 2013). This could result in reduced breeding success and declines in breeding numbers of great skuas, great black-backed gulls, herring gulls, lesser black-backed gulls and possibly of northern fulmars and northern gannets (Garthe et al. 1996, Furness 2003, Bicknell et al. 2013).

4.5 Climate change

Although impacts on seabirds from climate change are mostly indirect, such as increased sea surface temperature (SST) altering the marine food web from plankton upwards, they can still have strong impacts on the breeding success of seabirds (Frederiksen et al. 2013). According to Sydeman et al. (2012), seabirds are responding to climate change across the globe. When predicting climate change impacts on seabirds it is apparent that no single factor acts alone on populations, all factors (such as food availability, warming oceans or nest desertion rates) are interconnected (Heath et al. 2009). Climate change models have predicted that by the end of this century, great skua and Arctic skua*,* two widespread seabirds in Northern Scotland will no longer breed in the UK (Mitchell and Daunt 2010). Measurements using the NAO index have estimated that 29 species of North Atlantic seabirds are significantly affected by climate (Sandvik 2012).

One such example is the kittiwake which has suffered breeding success declines with increasing Sea Surface Temperature (SST), as warming oceans are also correlated with a decrease in large *Calanus* copepod abundance (this being a key food for sandeels) (Frederiksen et al. 2007). Kittiwakes in the West Atlantic are also thought to be sensitive to deteriorating environmental conditions (changes in sea temperature affecting zooplankton affecting food fish abundance), leading to population declines, (Frederiksen et al. 2012); these deteriorating environmental conditions can only become more exaggerated with climate change given the adverse effect of warming sea temperature on copepods and hence on higher trophic levels. Further north, in a much colder climate, kittwakes of the Bering shelf region however may experience short-term demographic benefits from ocean warming effects (Satterthwaite, 2012). In Nunavut, Canada, although the reproductive success of thick-billed murres (Brunnich’s guillemots) *Uria lomvia* has not yet been reduced by an increasing SST and declining ice cover, a declining rate of energy supply to chicks has been observed and this suggests that reproductive success could be affected in the foreseeable future if this trend continues (Smith and Gaston, 2012).

Auks of the North Sea (common guillemots, razorbills and Atlantic puffins) have been recorded to be breeding later as a response to climate change (Wanless et al. 2009), probably because of an effect of reduced prey availability during the laying and incubation period. One of the more indirect effects of warming oceans due to climate change is the effect upon the marine food web and most importantly at the bottom of the web, plankton. With a decrease in large *Calanus* copepods correlated to increasing SST, many small fish (such as sandeel) will have a limited food source, which as discussed above, can have seriously detrimental impacts on seabird's breeding success. It is not only kittiwakes that are affected by climate change; there is evidence of effects on many other species, but not all effects are negative. In the Alaska Gyre, the abundance of 15 species of seabirds was tested against seasonal oceanic trends to predict the potential effects of climate change in the future. Overall, there was an increase in seabird abundance (9 out of 15 species), apparently a result of an increase in forage zooplankton and a lengthening of the growing season (Thompson et al. 2012).

The extent of climate change effects on seabirds will differ across the globe. Great skuas experience heat stress above certain temperatures causing increased rates of nest desertion. Above 16^{o}C, 10% of great skua territories were deserted by both parents leaving chicks vulnerable to predation and/or starvation (Oswald et al. 2008) at a study site in Foula, Shetland. At this same site, heat stress was recorded to have a "critical" effect on chick survival above 14^{o}C. During a seabird review throughout the UK climate change, along with prey availability, was found to correlate with the breeding success and survival rate of kittiwake, shag, Arctic skua, Arctic tern, common guillemot and Atlantic puffin*,* with infanticide being recorded in guillemot colonies as a critical impact *(*Mitchell and Daunt, 2010). As for climate change affecting migration routes and wintering areas of seabirds, Fort et al. (2012) found that winter hotspots for northern gannets could be dictated by the origin of the bird, suggesting genetic control of migration pathways, which may influence their susceptibility to climate change.

Little auks *Alle alle* of the North Atlantic have the ability to offset potential climate change impacts (i.e. ocean warming) through plasticity of their foraging behaviour (Grémillet et al. 2012). Kittiwakes of the North Sea on the other hand have been reported to correlate their laying dates with NAO and SST in the prebreeding period, whereas common guillemots made only minor adjustments to their breeding schedule even with large-scale climate changes (Frederiksen et al. 2004). Rising sea levels linked to climate change are another concern for seabird colonies on low-lying islands or exposed coastlines, with the potential for the nest habitats of ground-nesting birds to be washed away (Mitchell and Daunt, 2010).

Anticipated climate warming in the 'Green Belt' of the south-eastern Bering Sea is predicted to reduce availability of prey for planktivorous predators (in this case least auklets *Aethia pusilla* (Dorresteijn et al. 2012). Another predicted effect of climate change is an increased frequency of hurricanes and storms. Along the US Atlantic Ocean seaboard, modelling indicates that rises in numbers of hurricanes has the potential to increase the risk of extinction of black-capped petrel *Pterodroma hasitata* (Hass et al. 2012). The impacts observed from these studies are relatively strong, affecting seabirds at a population level and the effects are long-term with predictions for even stronger effects in the future. Climate change is therefore a relatively important factor affecting seabird populations (Frederiksen et al. 2013). In contrast to the effects of fisheries however, climate fluctuation mainly affected reproduction which is typically a "low elasticity" demographic trait and, as such, has lower impact on population growth rates compared with impacts on adult survival, a "high elasticity" trait (Barbraud et al. 2012).

4.6 Predation

Predation from 'natural predators' of seabirds (such as eagles, peregrines and other birds of prey) tends to have only a small impact on breeding success or survival rates of seabird populations, and tends to occur at sustainable levels over the long term. Human impacts can alter this balance where populations of predators increase. For example, great skuas and large gulls have increased in response to provision of large quantities of fishery waste, but may then switch to killing smaller seabirds when fishery management changes discard availability (Votier et al. 2004). Predation from introduced mammals, however, can have extremely serious unsustainable impacts on seabird populations, especially on smaller seabirds, as mammals tend to only attack seabirds that are smaller in size than they are (Towns et al. 2011). For example, the presence or absence of brown rats in Orkney and Shetland is the "*single most important influence*" on storm-petrel breeding distribution (de Leon et al. 2006). Evidence of this comes from a survey reporting that European storm petrels were restricted to rat-free and low-disturbance islands (de Leon et al. 2006).

It is estimated that 75% of threatened island birds are at risk from introduced species (Phillips 2010). For example, mammal predators were responsible for the extinction of three seabirds endemic to New Zealand: a species of penguin *Megadyptes waitaha*, Scarlett's shearwater *Puffinus spelaeus* and the southern merganser *Mergus australis* (Towns et al. 2011). Since European expansion in the 16th century, invasive mammalian predators have spread through UK Overseas Territories (e.g. Bermuda, Gibraltar, Falklands, Cyprus). These invasions have caused seabird extinctions and population declines which are ongoing across the islands (Hilton and Cuthbert 2010). Similar devastating impacts have frequently been seen throughout the world where alien mammals have been introduced onto islands with seabird colonies (Towns et al. 2011). For example, Pascal (1980) estimated that on Kerguelen Island sub-Antarctic Indian Ocean, cats killed 1.2 million seabirds each year during the 1970s.

Sooty terns nested in 'great abundance' on Raoul Island in the Kermadec group. Cats reached the island in the early 19th century, followed by brown rats after a shipwreck in 1921. In 1967, 80,000 adult terns were still present, but by 1994, these had been reduced to 1,300 and carcasses of terns killed by cats littered the beach. By 1997, the tern population had been extirpated (Peck et al. 2008, Towns et al. 2011). However, there are a few examples of cases where very large seabird populations have coexisted with rats for many decades without obvious impacts (Quillfeldt et al. 2008), possibly because rat numbers in these cases are simply too low relative to the huge numbers of seabirds in the colonies. This may explain why it took over 100 years for cats to eradicate sooty terns from Raoul Island. Similarly, cats on Ascension Island reduced sooty tern numbers from more than one million pairs in the 1940s to about 150,000 in the late 1980s (Nogales et al. 2004).

Brown rat and American mink are two introduced predatory species that have caused whole colony extinctions of terns, gulls, storm petrels, Manx shearwater and Atlantic puffin at many sites in the British Isles (Mitchell and Daunt 2010). The Shiant Islands of the Outer Hebrides experienced declines in nesting seabirds in the early 1990s which was correlated to the presence of ship (black) rats, (Key et al. 1998, Stapp 2002), although those rats and the seabirds appear to have co-existed on the Shiants since 1900 (Brooke 1972). Now that the predatory impacts of invasive mammals are well known, regulations are in place regarding the accidental transport and release of such animals, to reduce the chances of them reaching currently "safe" islands (Manchester and Bullock 2000).

In the Azores, the introduction of mammalian predators led to the disappearance of Procellariform seabirds from the main islands, all except the Cory's shearwater *Calonectris diomedea,* the largest abundant seabird of that archipelago (Fonataine et al. 2011). In combination with low natal philopatry, high predation pressure on the East Limestone Island colony of ancient murrelets *Synthliboramphus antiquus* in British Columbia has the potential to eradicate this seabird species from the island (Gaston and Descamps 2011). In the British Isles, foxes, badgers, stoats, American mink, otters, rats, and feral cats have all been identified as major influences on productivity of some seabirds. Not all impacts are from alien mammals. Otters, an endemic mammalian predator resident to Scottish coastlines, were the cause of a very low breeding success rate of Arctic terns in NE Scotland in 2005 (Mavor et al. 2006). Many areas that used to be predator-free nesting habitat have been invaded by mammal predators, in some cases naturally, but in most cases as a consequence of human influences that have increased populations of these mammals or have assisted their colonisation of seabird islands. Cats and foxes have been described as 'superpredators' because they kill adult seabirds as well as taking eggs and chicks. They may kill large numbers of seabirds that they then store for subsequent meals, and their populations on remote islands often require rodents to be present to sustain them through periods when seabirds are absent (Towns et al. 2011). American mink fit into this classification too. Rodents have been categorised as 'mesopredators' (Towns et al. 2011). Their impacts are predominantly on seabird breeding success rather than on adult survival, and can increase where superpredators such as cats have been eliminated, allowing mesopredators such as rats to increase (Rayner et al. 2007, Le Corre 2008).

There are numerous examples around the world of devastating impacts of mammal predators on seabirds, and more recently of the extirpation of mammals to restore seabird habitat and endangered populations. These examples include eradication of rats (Gaze 2000, Stapp 2002, Lock 2006, Towns et al. 2006, Appleton 2007, Bell 2007, Howald et al. 2007, Luxmoore 2007, Mitchell and Ratcliffe 2007, Patterson 2007, Stoneman and Zonfrillo 2007, Swann et al. 2007, Zonfrillo 2001, 2007, Jones et al. 2008, Mulder et al. 2009, Ratcliffe et al. 2009, Bellingham et al. 2010, Capizzi et al. 2010, Howald et al. 2010, Phillips 2010, Brown et al. 2011, Dunlevy et al. 2011, Kawakami and Aoyama 2011, Mulder et al. 2011, Oppel et al. 2011), mink (Craik 2007), feral cats (Veitch 2001, Keitt et al. 2002, Nogales et al. 2004, Dowding et al. 2009, Ratcliffe et al. 2009, Bellingham et al. 2010, Kawakami and Aoyama 2011, Oppel et al. 2011), hedgehogs (Jackson 2001), mice (Wanless et al. 2007, Angel et al. 2009), rabbits (which can be predators as well as keystone species affecting habitat) (Bried et al. 2009) and other mammals (Donlan et al. 2002, Carrion et al. 2011), and the use of predator exclusion from colonies such as by deployment of predator proof fencing (Young et al. 2012). Veitch and Clout (2002) review eradication of 138 populations of seabird predators in ten countries (many of these being eradication of Arctic fox populations in Alaskan islands and eradications of various species from small islands off New Zealand). The ability to eradicate invasive alien mammals has improved with experience and development of techniques (e.g. Eaton and Ogilvie 2009, Gsell et al. 2010). Eradications have been successfully completed on increasingly larger islands, with rat eradication on islands up to 11,300 ha (Campbell Island, sub-Antarctic New Zealand), cat eradication on islands up to 29,000 ha (Marion Island, sub-Antarctic South Africa), fox eradication on islands up to 90,574 ha (Attu, Aleutian Islands, Alaska United States) (Dunlevy et al. 2011).

In cases where seabirds have been extirpated by predators, seabird restoration programmes have been developed to restore populations after their habitat has been cleared of the invasive predator population (Rauzon et al. 2002, Parker et al. 2007, Rauzon 2007, Miskelly et al. 2009, Jones 2010a,b, Lavers et al. 2010, Jones et al. 2011, Jones and Kress 2012). Jones and Kress (2011) define seabird restoration as '*efforts to actively restore seabirds through direct management interventions rather than allowing seabirds to passively recover following the removal of disturbance factors such as invasive mammals*'. They recognise two forms of seabird restoration; chick translocation, and social attraction.

Chick translocation is labour-intensive, expensive, and is successful only for species with particular life-history traits (in particular chicks that fledge independently of parents and do not receive any parental protection or feeding after fledging). These include Manx shearwater and Atlantic puffin, for example. The idea is that chicks subsequently return several years later to breed where they fledged; this works because most seabirds show very strong fidelity to their natal colony. Chick translocation can be effective (see for example Miskelly and Taylor 2004, Bell et al. 2005, Miskelly et al. 2009); a meta-analysis by Jones and Kress (2011) identified 5 successful projects out of 5 using this method alone, and a success rate of 70% for 59 projects using a combination of 2 or 3 of chick translocation, acoustic attraction and decoys (the difference in success rates not being statistically significant due to the small sample size using chick translocations alone).

Social attraction aims to lure adult birds to restoration sites where there is no colony at the time. Models of adults, sound recordings, mirrors, artificial nests or burrows may be used in some combination to encourage birds to join what appears to be an active colony. Social attraction has been used particularly successfully with auks and terns (Jones and Kress 2011). An attempt to start a new Australasian gannet colony at Young Nick's Head, in New Zealand by social attraction was successful, but attempts using the same method with Australasian gannets at Mana Island, New Zealand, and with northern gannets in Nova Scotia and in Quebec failed (Jones and Kress 2011). Jones and Kress (2011) suggest that the key feature affecting the outcomes at these sites was that the Young Nick's Head site was in an area frequented by large numbers of potential colonists from a nearby colony, while the unsuccessful attempts were at sites far from source colonies. After rats had been eliminated from the island, an attempt was made using chick translocation and acoustic playback to attract Manx shearwaters to re-colonize Cardigan Island in Wales from which they had been extirpated by brown rats, but this failed, apparently due to high numbers of gulls deterring shearwaters from re-establishing (Gummer 2003).

However, costs of bringing seabirds back to an island from which they were extirpated by alien predators is usually much greater than the cost of removing the predators, and the probability of success is less than 1, so it would be very much better to remove predators before seabird colonies are completely abandoned (Jones et al. 2011). Kress and colleagues successfully used social attraction methods to encourage Atlantic puffins and terns to re-colonise islands in the Gulf of Maine (Kress 1983, 1992, Kress et al. 2008). The seabird restoration methods developed in Maine have since been used globally to restore at least 49 species of seabirds on 89 islands in 14 countries (Jones et al. 2011). Jones and Kress (2011) suggest that a typical restoration project for a seabird in a developed country may cost around £500,000 per annum over a project lasting at least 5, possibly 10 years on average. They point out that the successful project restoring Atlantic puffins to Eastern Egg Rock in Maine took 35 years of sustained effort to establish a population of 100 pairs of puffins.

Globally, there have been positive responses for at least 45 species of seabirds following removal of mammalian predators, so the effectiveness of predator removal in allowing seabird populations to recover is very clearly established (Towns et al. 2011). For example, eradication of cats on Ascension Island in 2004 resulted in an increase in breeding numbers of sooty terns by over 50,000 pairs in less than three seasons, and resulted in immediate natural recolonisation of the island by masked boobies and brown boobies (Hughes et al. 2008, Ratcliffe et al. 2009). Removal of mink from islands in the Baltic Sea led to increases in breeding numbers of Arctic skuas, Arctic terns and common gulls (Nordstrom et al. 2003). Numbers of breeding terns, black guillemots, eiders and gulls all increased on islands in Argyll where mink were trapped (Craik 1997, 2007). Evidence for the benefits of predator control can even be found in some cases where there has been no detectable impact on breeding seabird numbers. For example, controlling or eradicating rats resulted in white-chinned petrel breeding success at Crozet increasing from 16% to 50% (Jouventin et al. 2003). Control of rats on Lavezzi Island doubled breeding success of Cory's shearwaters (Pascal et al. 2008).

While invasive alien mammals have been eradicated from 12 relatively small islands around the British Isles, there are at least 80 islands or island groups around Britain where rats are present (Ratcliffe et al. 2009). Eradications that have been carried out including brown rats on Canna Scotland (Bell et al. 2011), brown rats on Ailsa Craig (100 ha, Scotland) (Zonfrillo 2001, 2002, 2007), brown rats on Handa (100 ha Scotland) (Stoneman and Zonfrillo 2005), brown and black rats on Lundy Island (500 ha England) (Bell 2004, Appleton et al. 2006), and brown rats on Ramsey Island (256 ha Wales) (Bell et al. 2000). ,

Numerous British islands of major importance for their seabird colonies have populations of invasive alien mammals, including several SPAs for seabirds:–

- Hermaness SPA has brown rats;
- Noss SPA has had feral cats, although it has been suggested that these have now died out and indeed storm petrels have returned to breed there in the last few years (M. Bolton pers. comm.);
- Shiant Islands SPA (one of the biggest puffin colonies in UK) has black rats;
- Foula SPA (the biggest great skua colony in the world) has feral cats and hedgehogs;
- Rum NNR and SPA (the biggest Manx shearwater colony in Scotland) has brown rats, red deer and possibly feral cats; and,
- Imperial Dock Lock SPA (the biggest common tern colony in Scotland) has brown rats and American mink.

In contrast, in Mexico, 12 species of invasive alien mammals have been eradicated from 31 islands (total of 51,000 ha) providing conservation gains for 227 colonies of seabirds (Aguirre-Munoz et al. 2008). Britain and Ireland lag far behind several other countries in terms of removing alien mammals, possibly because so many islands in Britain have been invaded for such a long time that these populations are now treated as part of the environment rather than a problem that can be solved; an issue also in some other countries (Ruffino et al. 2009). However, in many parts of the world, conservation managers recognise that eradication of alien mammals represents one of the most cost-effective approaches to protecting seabird populations (Dunlevy et al. 2011). Dunlevy et al. (2011) also identify a mindset among managers in those countries where eradication programmes are well developed, that while removal of invasive species is a necessity, monitoring the recovery of native species after eradications is an optional luxury. This can lead to reluctance to support new initiatives if the outcomes of previous financial outlays are not clear. This has encouraged a shift to evidence-based conservation, where the removal of introduced alien predators has been shown to have significant benefits (Dunlevy et al. 2011).

In the UK, DEFRA (2007) recognized the threat that invasive species pose to native biodiversity and drafted a strategy that requested prioritization of remedial management. Ratcliffe et al. (2009) provided exactly that required prioritization for eradications of rats on British islands that would maximize benefits for seabird conservation. They followed guidelines of Falklands Conservation (2008) in assuming a maximum swimming distance of 300 m for brown rats and black rats, and assessed 'eradication units' defined as islands at least 300 m from the mainland and at least 300 m from adjacent 'eradication units'. They also considered reintroduction risk, based on the premise that eradications would be less cost-effective if there was a high likelihood that rats would recolonize the island subsequently. Of the 12 eradications carried out in the UK, all were considered successful, but two of these sites were recolonized by rats from adjacent areas (Isles of Scilly and Looe Island). Recolonization risk increases with the size of the human population (de Leon et al. 2006, Ratcliffe et al. 2009). On this basis, they excluded from further consideration all islands with resident human populations exceeding 100 people. Ratcliffe et al. (2009) assumed a cost of eradication of £440 per hectare, based on the cost for eradication on Canna, probably the most expensive rat eradication so far carried out in the UK.

Mammals are obviously not the only predators of seabirds. Skuas feed on other seabirds; Arctic skuas mainly on young chicks but great skuas can take larger seabirds too. Their predominant prey is sandeels but in response to declines in sandeel availability and in the face of reduced rates of fishery discarding, great skuas have resorted to feeding more on other seabirds (Votier et al., 2007). To emphasise the extent of predation from skuas on other seabirds, it was recorded in 2005 that only 2 out of >300 pairs of Arctic terns fledged as a result of great skua predation and bad weather in North Ronaldsay (Mavor et al. 2006). Common terns are also strongly influenced by local predation, mainly by large gulls (Jennings et al. 2012). In North America, increasing populations of large gulls during the 20th century have been linked to declines of terns, with several tern colonies wiped out by gull depredations. Since 1997 almost all colonies of roseate terns in the United States have been managed to control predation by gulls (Whittam and Leonard 1999). Great skuas fed less on other seabirds at larger colonies than at small colonies (Votier et al. 2007). An interpretation of this finding was that at larger colonies, competition for other seabirds as prey was so intense that most of the skuas would feed on fish instead, even though foraging range would be wider.

As discussed above, following climate change, increasing temperatures (and hence a projected decline in sandeel stocks), seabird predation by great skuas (particularly affecting kittiwakes, Arctic skuas and Larus gulls in Orkney and Shetland) may well be subject to increase in the future. However, on a scale of factors affecting seabird populations, avian predators are a natural source of predation that tends to be sustainable, and alone (without the cumulative effect of climate change and fisheries management affecting skua and gull predation), have a relatively weak impact on seabird populations. Nevertheless, increased populations of gulls can have damaging effects on tern colonies in particular, and control of gull numbers near to tern colonies of conservation concern has been an effective method to increase tern breeding success (Whittam and Leonard 1999, Donehower et al. 2007).

4.7 Disturbance

Human disturbance of wild animals is often a concern. As for seabirds, ground-nesting species are obviously of greater disturbance potential than are cliff-nesters or burrow-nesters. At Alness Point, N. Scotland, a long term decline in common gull *Larus canus* numbers has been recorded, with human disturbance being the main factor reducing gull success; all nests failed in 2005 (Mavor et al. 2006). Kittiwakes and common guillemots at St. Abb's Head, East Scotland have shown reduced nesting success and even nest failure linked to human disturbance (Beale and Monaghan, 2004, 2005). In Orkney and Shetland, the presence of human visitors also appears to affect the distribution of storm petrels (de Leon et al. 2006). A study of Cassin's auklets (*Ptycoramphus aleuticus* - a ground-nesting seabird) off the coast of Mexico on West San Benito Island, has recorded human disturbance effects from a seabird colony less habituated to visitors. At this site, disturbed auklet chicks showed a lower mean peak mass than the control group of chicks left undisturbed. This showed that under strict experimental conditions, the growth rate (measured in mean peak mass) decreases with increasing disturbance. This considered, human disturbance is a relatively low impact factor on most seabird populations, being a significant hazard mainly for ground-nesting species frequently found on mainland sites (such as little tern). Although disturbance in studies cited above shows a negative correlation with both nesting success and chick growth rate, the effect of disturbance at a population level was weak compared to other factors affecting seabird success, although disturbance impacts should be considered at the site level for particularly vulnerable species such as terns and gulls at mainland colonies, red-throated divers nesting at lochans close to roads providing vehicle access for tourists. Disturbance is also a significant issue in the offshore environment, where boat traffic and offshore wind farms can disturb seabirds (which species such as divers and scoters being considered especially vulnerable).

4.8 Parasites and disease

Botulism is thought to have caused reductions in breeding numbers of large gulls where these birds regularly scavenge on refuse tips where botulism can develop (Mitchell et al. 2004). Puffinosis can cause death of large numbers of Manx shearwater chicks, with up to 4% dying in some years from this disease. However, the scale of mortality is thought not to affect breeding population trends (Mitchell et al. 2004). According to Mitchell and Daunt (2010), the detrimental impact of parasites on seabirds is increasing. However, a lack of research on parasitology of seabirds makes it difficult to gauge the importance of parasitic impact on seabird populations. The colonial nature of seabird breeding however represents an ideal niche for tick infestations. Tick infections have been recorded to reduce nestling condition, growth rates and survival of seabirds (Muzaffar and Jones 2004). Ticks as parasites will rarely kill their hosts, but can have a detrimental impact on their seabird hosts (especially when carrying viruses) which could impact the population dynamics long-term. Desertion of portions of kittiwake colonies during their breeding season has been associated with abnormally high levels of ectoparasite infestations (Boulinier and Danchin, 1996). However, these impacts are mostly not substantial enough to cause large-scale population changes among seabird colonies, although in some cases they might influence local population trends (Mitchell et al. 2004). Therefore parasites can be considered a relatively low-impact factor of seabird population change.

4.9 Oil Pollution

Whether from offshore oil platform leakages, oil spills at sea or general waste disposal, oil pollution remains a factor that affects seabirds and can kill large numbers of seabirds, especially auks. However, Mitchell et al. (2004) concluded that '*effects of large oil spills are relatively short-term and localised, with no evidence of wide-scale, long-term effects on auk population trends*'. Seabirds are described by Boulinier and Riffaut (2008) as the '*emblematic victims*' of oil pollution. It is thought that after an oil spill, when it affects adult seabird survival, population declines will occur the following year but that after the decline, populations generally recover again (Piatt and Roseneau, 1999).

Off the Dutch coast, quick post-spill recoveries have been recorded in gulls. All birds in the study managed to clean their plumage within a few weeks of the spill, in enough time to establish territories and breed that season (Camphuysen, 2011). However, a study on the effect of oil spills on adult common guillemots showed that major oil pollution incidences doubled their winter mortality rate (Votier at al. 2005).The Exxon Valdez oil spill of 1989 in Alaska caused population declines in affected common guillemot colonies and delayed breeding phenology leading to low reproductive success (Piatt and Roseneau, 1999). The Prestige oil spill, NW Spain, caused a delayed but sublethal impact on local seabirds, e.g. damage to kidneys and liver (Perez et al. 2010). In December of 1999, the Erika oil tanker spill in the Bay of Biscay resulted in the wreck of 80,000 seabirds being washed ashore, with 80% of these birds recorded as common guillemots (Riffaut et al. 2005).

These oil spills obviously had a large impact on local seabirds, particularly on common guillemots, but as the spills were rare events, alongside evidence for self-cleaning properties of some seabird species, populations had future opportunity to recover. Oil spills around the Scottish coastline are especially rare, however oil spills around the Northern Isles between December 1978 and March 1979 caused the death of 7,735 birds which were found oiled and dead on Orkney and Shetland islands (Richardson et al. 1982). Although the rarity of oil spills around Scotland presents a case for that factor to be of relatively low importance on seabird populations, it also means that the coastline is unprepared to deal with potential spills and clean up procedures, which could aggravate already serious impacts (Heubeck et al. 2003). Mortality peaks have also been recorded for foraging seabirds around offshore oil platforms as a result of ingested oil or damaged feathers (Wiese et al. 2001).

As for oil discharges from ships, along the coast of Newfoundland, Canada, illegal oil discharges were the cause of 315,000 ± 65,000 common guillemot deaths each year (Wiese et al. 2004). This emphasises the impact that oil can have on seabirds in areas where oil pollution is much more common. Although the impact upon seabird populations from various forms of oil pollution can be severe and often lethal, impacts vary greatly between species (Boulinier and Riffaut 2008) and they are usually centred around certain hotspots where commercial shipping is intense and frequent. In the British Isles therefore, the relative importance of oil pollution as a factor affecting seabird populations, is relatively low because pollution events are relatively infrequent. Worldwide, oil pollution represents a more important impact upon seabirds, but despite this is generally considered not to have a significant long term impact on breeding seabird numbers.

4.10 Persistent Organic Pollutants and heavy metals

Persistent Organic Pollutants (POPs) include pesticides such as DDT, and industrial chemicals such as PCBs and flame retardants. Some of these POPs have oestrogenic effects on birds, or are carcinogenic, and most are toxic although toxicity varies considerably among compounds (Knudsen et al. 2007). POPs tend to accumulate in body fat and increase in concentration up the food chain, so reach highest concentrations in top predators such as skuas and large gulls (Sagerup et al. 2009).

Increased POP levels can show a positive correlation with corticosterone levels in seabirds, as shown in black-legged kittiwakes in Svalbard.

This is an important environmental finding as corticosterone stress hormone levels predict the level of response from seabirds to changing environmental conditions (e.g. SST, food availability, parental effort) (Nordstad et al. 2012). Another POP study on seabirds in the Arctic researched the effects of POPs on the immune system of glaucous gull *Larus hyperboreus* chicks. The findings revealed that multiple POP exposures on the chicks had a negative effect on their immune system (Sagerup et al. 2009). Again, in the face of changing environmental conditions, a weakening of the immune system represents a negative impact on the survival rate of gull chicks. However, very few studies have found harmful effects of POPs on seabirds on a scale that would be likely to affect seabird population size, except in a very few cases such as the impact of a massive pesticide spillage in the southern North Sea in 1967 which killed large numbers of gulls, terns and cormorants (Mitchell et al. 2004). In the highly polluted Great Lakes, POPs caused embryo mortality and population decline in cormorants, gulls and terns, but levels of these compounds in British seabirds are far below the levels reported in seabirds in the Great Lakes.

Heavy metals, especially mercury and cadmium, are also considered to be a hazard for seabirds, but as with POPs, there is very little evidence that these metals influence seabird demography in any detectable way in the British Isles, and impacts elsewhere seem generally to be difficult to detect and probably only very local in exceptional circumstances (for example, lead pollution from paint peeling off military buildings affects albatross chick survival on Midway Island; Finkelstein et al. 2010).

4.11 Plastics

Plastics have been distributed around the oceans in increasing amounts over the past 40 years (Yamashita et al. 2011), acting as a source of pollution and a cause of potential harm to marine animals that may ingest plastic or become entangled. A group of short-tailed shearwaters *Puffinus tenuirostris* accidentally caught in the North Pacific Ocean in 2003 were studied for the potential effects of ingested plastics. Each bird's stomach contained a mean mass of 0.23g plastic. The mass of ingested plastic in the birds was found to correlate with concentrations of lower-chlorinated congeners found in the birds' tissues (Yamashita et al. 2011). Ingestion of plastics can be toxic to seabirds and could become more of an issue with an increase in the numbers of vessels using our oceans and amounts of plastic being used throughout the world. In a study along the coastline of Rio Grande do Sul, southern Brazil, stomach contents were studied from several species of dead seabirds, looking for plastics. Seabird species included shearwaters, albatrosses and petrels. Pieces of nylon line (from fishing boats) accounted for 17% of the plastics found in the seabird's stomach contents (Colabuono et al. 2010) while most was industrial raw plastic pellets or broken fragments of user plastic. Large accumulations of plastic fragments in the gizzard could reduce hunger or food assimilation. According to Colabuono et al. (2010), plastics could be an additional source (alongside transfer up the food chain) of POP exposure to seabirds, as well as a hazard blocking the intestine and filling the gizzard with indigestible material. However, attempts to measure harmful effects of ingested plastics in seabirds have been largely unsuccessful, suggesting that plastics are unlikely at present to have any detectable effect on seabird demography. Amount of plastic in stomachs of northern fulmars from the North Sea is used in the OSPAR Ecological Quality Objective (EcoQO) for marine litter. The preliminary EcoQO defines acceptable ecological quality as the situation where no more than 10% of fulmars exceed a critical level of 0.1 g of plastic in the stomach. During 2003-2007, 95% of 1295 fulmars sampled in the North Sea had plastic in the stomach and the critical level of 0.1 g of plastic was exceeded by 58% of birds. The EcoQO is now also used as an indicator for Good Environmental Status in the European Marine Strategy Framework Directive (van Franeker et al. 2011). However, there is no evidence to demonstrate that these amounts of plastic affect fulmar demography. During winter 2012-13 there has been a pollution incident in southern England with large numbers of seabirds, mostly guillemots and razorbills, killed by plumage contamination with a slick of polyisobutene. This is not the first such incident (see Camphuysen et al. 1999) so this problem may be more common than has been recognised. While such pollution incidents receive considerable media attention, impacts of these winter mortality events are extremely difficult to detect in terms of breeding numbers at colonies.

4.12 Offshore wind farms

The European Union has set an ambitious target of generating 20% of energy consumption from renewables by 2020 (European Commission 2007). Offshore wind farms avoid most of the public opposition, visual intrusion, noise, environmental and land use conflicts that are increasingly evident with terrestrial wind farm developments, and also capture up to 50% more energy because of higher wind speeds and less turbulence over the sea (Kikuchi 2010). By January 2013, 1,662 turbines had been installed and grid connected at 55 offshore wind farms in ten European countries, with 52% of these turbines in UK waters and 65% of the European total in the North Sea (European Wind Energy Association 2013). European offshore wind energy capacity represents 90% of the world total offshore capacity, and over 10% of this capacity was installed during 2012, with a similar amount anticipated in 2013 (European Wind Energy Association 2013).

There is concern that birds collide with offshore wind farm turbine blades (Desholm and Kahlert 2005; Hüppop et al. 2006; Everaert and Stienen 2007). Seabirds are especially vulnerable to collision mortality because they are long-lived animals with deferred maturity and low reproductive rates. As such, their populations are strongly affected by factors that increase adult mortality, whereas populations of most terrestrial birds produce large numbers of offspring that can buffer impacts on survival rates (Exo et al 2003; Desholm 2009). Assessment of the vulnerability of different species of seabirds suggests that northern gannets *Morus bassanus* and large gulls are among the species with highest vulnerability in Europe (Garthe and Hüppop 2004; Furness et al. 2013). This relates in part to gannets and large gulls often flying high enough above the sea to be at risk of colliding with turbine blades, whereas many species of seabirds habitually fly too low over the sea to be at risk (Furness et al. 2013). Gannets also appear to be at higher risk than many other seabirds because when breeding they have especially large foraging ranges from colonies (Thaxter et al. 2012) so may frequently commute past offshore wind farm sites, and after breeding they migrate past offshore wind farms that lie between their breeding and wintering ranges (Kubetzki et al. 2009; Fort et al. 2012). Empirical evidence supports this view. Surveys of seabird numbers at proposed offshore wind farm sites in the UK record high numbers of gannets (WWT Consulting 2012).

Nevertheless, offshore wind farm developments will be localized and therefore not as widespread as the impacts that climate change or fisheries and food availability would have on seabirds. However, seabirds are at risk of collision mortality at offshore wind farms, and may also be affected by displacement (habitat loss) and barrier effects (increasing flight times and so energy costs) (Furness and Wade 2012). Conversely, there are fewer records of offshore collisions than onshore collisions (Boehlert and Gill 2010) and seabirds, wildfowl, and waders have been noted to avoid collisions with offshore turbines by altering their flight paths up to a few hundred metres around wind farms to avoid collisions (Exo et al. 2003), even at night time. In fact, out of all the ducks and geese recorded off Nysted wind farm, less than 1% flew close enough to the turbines during migration to be at any risk of a collision (Desholm and Kahlert, 2005). While the impact of offshore wind farms on seabird populations remains to be seen, there are concerns that collision mortality rates for some seabirds, such as northern gannets and gulls, which tend to fly at heights that make them relatively vulnerable to collisions with turbines (Furness and Wade 2012; Furness et al. 2013), may have detectable effects on population trends in these groups (Busch et al. 2013). Projected effects of Round 3 developments in UK waters are of particular concern given the relative novelty of offshore wind farms, the large scale of Round 3 developments, and uncertainty about impacts on seabirds, and the potential for cumulative impacts on seabird populations. These would most likely be detected in areas where vulnerable seabird species breed close to concentrations of offshore wind farms (such as with gannets and large gulls in south-east Scotland and east England). On the other hand, if offshore wind farms are closed to fishing, they may represent areas where fish populations could increase; whether this would benefit seabirds through higher densities of small pelagic fish remains to be seen, but is a possibility, and some preliminary evidence indicates that some fish populations do increase within offshore wind farm areas (Danish Energy Agency 2013).

4.13 Wave and tidal arrays

Impacts of wave and tidal arrays on bird populations remain to be seen, but are considered likely to be small and confined to a relatively small number of seabird species and populations (Furness et al. 2012). It is thought that diving birds will encounter a risk of entanglement, collision or blade strike with subsurface components (Boehlert and Gill 2010); subsurface components specifically of tidal turbines, as wave energy device structures will be situated mostly above the sea surface. Seabirds such as auks, divers, shags and cormorants dive deep below the sea surface to catch their prey hence any novel construction underwater has the potential to act as a barrier to their movements and a collision hazard.

There is also concern for seabirds during the construction and maintenance of new devices at sea, that boat traffic and disturbance will increase. Increase in boat traffic during the construction, decommissioning and maintenance of devices could flush auk species from hundreds of metres away (Langton et al. 2011). Divers have been reported to be especially sensitive to boat movements and therefore could be negatively impacted by an increase in boat traffic in the PFOW area during construction and maintenance of tidal stream and wave devices. For seabirds along the Oregon coastline, it has been predicted that stormy conditions such as high winds or poor visibility could increase collision rate with wave energy converters (WECs) and that continuous lighting present on any WECs could increase collision risk at night when birds could be attracted to the lights.

Alongside these potential negative impacts of the pending wave and tidal arrays, they also carry potential positive impacts to the local seabird colonies: modifications to water movements and turbulence could alter vertical movements of marine organisms and result in prey and predator aggregations (Boehlert and Gill, 2010). Langton et al. (2011) have also reported that fish move closer to structures after disturbance events and suggest that once tidal stream and wave energy devices are installed, this could increase the success of seabirds foraging around the new device structures. There is speculation that with rotating blades under the sea surface, there is potential for seabirds to collide with rotating blades as with onshore wind turbines. However, Faber Maunsell and Metoc (2007) believe that underwater, birds' moderately fast burst speed would enable escape from the path of tidal turbine blades. There still remains the risk of collision for diving birds underwater with newly installed turbines though, especially for those that actively forage underwater. It is important to emphasise here the novelty of these marine energy devices and hence the scarcity of available literature assessing their potential impacts.

It will be impossible to know the full extent of these device instalments upon seabird populations until they have been installed and the local area and seabird colonies surveyed. However, the likely effects of wave energy and tidal stream turbine arrays have been assessed by McCluskie et al. (2012) and by Furness et al. (2012). These two reviews reach broadly similar conclusions. Impacts of wave energy devices are likely to be substantially less than impacts of tidal stream arrays, and both technologies are likely to have less impact on seabirds than development of offshore wind farms. For example, displacement of seabirds by tidal stream turbine arrays or wave energy devices is likely to be substantially less than from offshore wind farms because wet renewable developments occupy much smaller areas than taken up by offshore wind farms (McCluskie et al. 2012).

Seabirds most likely to be adversely affected can be identified based on knowledge of seabird ecology. For wave energy devices, the main hazards to seabirds are possible displacement of sensitive species from foraging habitat and possible injury through collision with structures either above or below water. While in the past there has been a tendency to assume that displacement equals death, this approach is no longer considered appropriate, and the effects of displacement are more appropriately assessed through a model linking behaviour to demography (McDonald et al. 2012). More speculative impacts include the possibility that such devices may provide 'stepping stones' permitting alien mammal predators such as mink to extend their range, and the possibility that pollutants may enter the marine environment by leakage from these devices (McCluskie et al. 2012). Seabirds most vulnerable to impacts of wave energy devices appear to be divers (all species), as these birds are particularly sensitive to disturbance.

For wave energy devices, divers are the species considered to be at risk of significant displacement, and both red-throated divers and black-throated divers breed in SPAs that have connectivity with parts of PFOW. For tidal stream arrays, of those species considered to be at high or moderate risk of impacts at the population level, several breed in SPAs that have connectivity with parts of PFOW. These are razorbill, shag, common guillemot, great cormorant, red-throated diver, Atlantic puffin and black-throated diver. Given the likely locations of MPAs with black guillemot as a feature, these may not have connectivity with PFOW sites, although the population of this species in PFOW represents a significant proportion of the total Scottish population of this species, as do the populations of all of the other species considered to be at high or moderate vulnerability.

4.14 Conclusions

To conclude, a range of factors in the marine environment impact upon seabird populations, and of these, the ones that have affected seabird populations in the British Isles most in the past are human exploitation and persecution, mammal predation, food abundance, and fisheries. Those most likely to affect seabird populations at present are food abundance, fisheries, mammal predation and climate change, and those most likely to affect seabird populations in the future are also likely to be food abundance, fisheries, mammal predation and climate change, with cumulative impacts likely where these pressures occur together. This would suggest that the most promising candidates for management action might be strategies that would enhance local food supplies around seabird colonies (such as closing these areas to fisheries for preferred food fish such as sandeels, sprats and young herring), or reduce predation impacts (such as eradication of alien mammal predators from seabird breeding habitats). Other possible strategies could involve reduction in levels of human persecution (for the few species that are still subject to legal culling or harvesting), or to the provision of safe nesting sites/colony sites for seabirds where habitat availability may limit breeding distribution.

5. A POPULATION MODELLING APPROACH TO ASSESSING THE RELATIVE EFFICACY OF MANAGEMENT ACTION AFFECTING SURVIVAL VERSUS REPRODUCTIVE OUTPUT

5.1 Introduction

Seabird population sizes are driven by rates of survival and reproduction. Influences on survival and reproduction can be inherent (e.g. competition for resources), external (e.g. predation, provision of fishery discards) and due to management interventions. The relative magnitude of effects on a population from changes in demographic rates will depend on which rates (e.g. adult survival, juvenile survival, fledging success) are affected and by how much. Population modelling provides a means to both quantify the magnitude of a potential impact and also predict the degree of benefit which may be achieved through a management intervention. As with any modelling, the quality and robustness of the results obtained are very dependent on the quality of the data used. Some seabird species have been comparatively well studied (e.g. gannet) with the consequence that demographic rates have been estimated with a reasonable level of confidence and model predictions can be considered reliable. Other species have been much less well studied, and thus greater assumptions about rates of survival and reproduction need to be made and the consequent model outputs are less certain. Nonetheless, certain features of seabird life histories are common, and thus commonalities amongst population models mean that certain general traits can be identified. In the following section the population modelling method is detailed and summary outputs for each species presented.

5.2 Methods

The same age-structued stochastic population model structure was used for all the species modelled here; red-throated diver, Manx shearwater, northern gannet, Arctic skua, great skua, lesser black-backed gull, herring gull, great black-backed gull, kittiwake, Sandwich tern, common tern, common guillemot and razorbill. The modelling followed best practice methods, as described in WWT Consulting (2012). Demographic data used in the models are provided in Table 5.2.1. For those species with only limited data on survival rates (red-throated diver, Manx shearwater, Arctic skua, great skua, Sandwich tern, commion tern) a standard deviation of 0.05 has been used. For the remaining species direct estimates both of mean rates and the standard deviations were available in the literature. Breeding success was taken either from published literature or the JNCC seabird breeding data base (see sections 7.1.4, 8.1.4, 9.1.4 etc and annual reports (e.g. Mavor et al. 2008)).

Table 5.2.1. Demographic rates used in the seabird populations models. Sources provided in table footnote. See also sections 7.1.3, 8.1.3. 9.1.3 etc.

Species (ref.)	**Age at first breeding**	**Mean survival rates (standard deviation)**					**Fledglings/pr.**	**Brood size range (min-max)**
		Adult	Year 1	Year 2	Year 3	Year 4		
Red-throated diver[1,2]	5	0.84 (0.05)					0.635 (0.41)	0 - 2
Manx shearwater[2,3]	5	0.9 (0.05)					0.591 (0.26)	0 - 1
Gannet[4,5,6]	5	0.919 (0.012)	0.42 (0.079)	0.829 (0.031)	0.891 (0.031)	0.895 (0.031)	0.697 (0.035)	0 - 1
Arctic skua[2,7]	4	0.84				NA	0.522 (0.37)	0 - 2
Great skua[2,8]	6	0.9 (0.05)					0.664 (0.1)	0 - 2
Lesser black-backed gull[2,9]	5	0.91 (0.025)	0.819 (0.027)				0.517 (0.37)	0 - 3
Herring gull[10,11,12]	4	0.898 (0.017)	0.82 (0.07)			NA	0.42 (0.28)	0 - 3
Great black-backed gull[9,10,11,12,13,14,15,16]	5	0.93 (0.025)	0.82 (0.03)				0.74 (0.297)	0 - 3
Kittiwake[11,17,18]	5	0.876 (0.035)	0.79 (0.1)	0.79 (0.05)	0.79 (0.05)	0.79 (0.05)	0.65 (0.098)	0 - 3
Sandwich tern[2,19]	3	0.9 (0.05)			NA		0.656 (0.136)	0 - 2
Common tern[2,20]	3	0.88 (0.05)					0.721 (0.49)	0 - 3
Guillemot[10,21,22]	5	0.965 (0.01)	0.56 (0.014)	0.792 (0.03)	0.917 (0.017)	0.938 (0.017)	0.335 (0.113)	0 - 1
Razorbill[10,11,22,23,24]	4	0.9 (0.028)	0.937 (0.028)			NA	0.38 (0.085)	0 - 1
Atlantic puffin[10,11,25]	5	0.924 (0.01)					0.345 (0.11)	0 - 1

1 – Hemmingsson and Eriksson 2002; 2 – JNCC reports on seabird numbers and breeding success (http://jncc.defra.gov.uk/page-2143); 3 – Perrins et al. 1973; 4 - Wanless et al. 2006; 5 – Nelson 2002; 6 – WWT Consulting 2012; 7 – O'Donald 1983, Furness 1987, Phillips et al. 1998; 8 – Furness 1987, Ratcliffe et al. 2002; 9 - Wanless et al. 1996; 10 - Mavor et al. 2008; 11 - Robinson 2005; 12 - Maclean et al. 2007; 13 - Garthe and Huppop 2004; 14 - Calladine and Harris 1996; 15 - Reeves and Furness 2002; 16 - Poot et al. 2011; 17 - Coulson and White 1959; 18 - Frederiksen et al. 2004; 19 – Robinson 2010; 20 – del Hoyo et al. 1992-2006, Becker and Ludwigs 2004; 21 - Harris et al. 2007; 22 - Birkhead and Hudson 1977; 23 - Lloyd and Perrins 1977; 24 - Chapdelaine 1997; Harris et al. 1997.

The model was matrix based, run for a simulated period of 25 years and for each modelled scenario 10,000 simulations were performed from which the median growth rate was calculated. The relative impact on the population growth rate of increases in mortality of all age classes and increases in the rate of reproduction were modelled to estimate the increase required in the latter to offset a given increase in the former. Four levels of additional mortality were trialled: 0%, 1%, 5% and 10%. At each of these additional mortality rates, productivity was modelled sequentially across the range 100% to 120% (at 1% increments). Thus, the increase in productivity required to offset additional mortality of 1%, 5% and 10% could be estimated.

The additional mortality was applied to each age class in proportion to their presence in the population. In some cases, this may not be an appropriate assumption, as immature birds may be more vulnerable to collisions, due to their inherent lack of experience (as represented in their generally lower survival rates compared with adults). Equally, in some instances they may be less vulnerable (through travelling to different areas from adults and so potentially avoiding threats in a specific area). However, such detail is beyond the scope of this modeling. Since additional mortality would be likely to operate as a per capita rate, rather than an absolute number, the number removed from the population at each time step was proportional to the population size. Thus, additional mortality remained at the same proportional level relative to the population size throughout the simulation, whether the population increased or decreased.

Closed populations were assumed since there is no information on rates of exchange (i.e. immigration and emigration) between the breeding colonies being assessed. Similarly there is no information on which to base density dependent population regulation, hence the models were density independent. While this is clearly unrealistic in the longer term, for the benefits of short term modelling of small populations, the risks from violating this assumption were considered to be small. The populations were modelled on an annual time step, with one year age classes up to adults, which is a multi-age class for all individuals of this age and older. Only the final age class breeds and the models were based on a post-breeding census structure (i.e. each census of the modelled population occurs immediately after the breeding season).

Environmental stochasticity was modelled using the mean rates and the standard deviations as listed in Table 5.2.1. Survival rates were drawn from beta distributions, and brood sizes from stretched beta distributions (Morris and Doak 2002). These distributions were used as they generate random numbers from probability distributions which have characteristics appropriate to the demographic rates (i.e. survival rates between 0 and 1, and brood sizes which lie between pre-defined limits).

Demographic stochasticity on survival was modelled using a binomial process, whereby the number of individuals which survive from one time step to the next was estimated using a binomial function (Akcakaya 1991). Thus, the number of individuals alive at time t+1 is generated by a 'coin-toss' process, using the number of individuals alive at time t and the randomly generated survival rate for that time step (as described in the preceding point). The difference between environmental and demographic stochasticity can be thought of as follows; environmental stochasticity generates random values for the probability of survival from one time step to the next; and, demographic stochasticity generates random numbers of individuals which survive from one time step to the next for any given survival probability. Thus, environmental stochasticity models variable environments (e.g. weather effects) while demographic stochasticity models the effects of chance, which are increasingly important as the population size falls.

For each species an initial population size of 10,000 was used. While this bears little resemblance to the real population of most of these species, this parameter has virtually no effect on the results obtained. The population models used for this assessment were stochastic and density independent. It may be argued that the models would be more realistic if they reflected current trends. However, such arguments are based on the premise that the baseline model for any given species should generate predictions which match the recent trend in the population of interest. While this is a reasonable request, it presupposes that the underlying reasons for such trends have been studied and are well understood. This is rarely the case. Most population models are, of necessity, based on demographic rates derived either from different populations or at some time in the past (or often both). Indeed in many instances there are few data on which to base the trends themselves. Population change occurs due to a wide range of factors, some intrinsic (i.e. population regulation through competition for resources, often referred to as density dependence), some extrinsic (e.g. weather conditions), and these two also interact so that intrinsic effects may be greater during periods of unfavourable weather. Without knowing the main drivers of such changes (which is typically the case), simply modifying the survival or reproductive rates in order that the population model generates a prediction in line with the estimated population trend (which may itself be poorly known) without understanding what has really caused observed changes has the potential to render the model very unreliable as a predictive tool. In addition, some population change may be due to movements of individuals between locations (i.e. immigration and emigration), unrelated to change in demographic rates.

In such circumstances the most robust approach for modelling is to avoid the temptation to include density dependence, since this is often based on the premise that ‘it must be operating, therefore it should be included’, even if the mechanism is unknown. Furthermore, this highlights that the most appropriate means for considering model outputs is in terms of the relative outcomes between alternative scenarios. For example, an approporiate measure to consider is the *change* in the population growth rate predicted to occur as a result of a given impact, not the absolute rate of growth itself, which has a high likelihood of being inaccurate. Thus, the onus on the absolute reliability of the model is eased and instead focus is directed towards assessment of the relative magnitudes of a range of predicted impacts.

5.3 Results

The percentage increase in reproductive rate (fledglings per pair) required to offset increases in mortality of 1%, 5% and 10% is shown in Table 5.3.1.

Table 5.3.1. Percentage increase in reproduction required to offset additional mortality of 1%, 5% and 10%.

	Percentage increase in reproduction required to balance increase in mortality		
Species	**1% increase in mortality rates**	**5% increase in mortality rates**	**10% increase in mortality rates**
Red-throated diver	4.47	21.98	43.98
Manx shearwater	4.87	23.41	47.65
Northern gannet	7.18	38.2	79.02
Arctic skua	3.91	21.1	44.1
Great skua	4.68	22.91	47.15
Lesser black-backed gull	6.58	31.8	67.2
Herring gull	5.85	30.64	63.87
Great black-backed gull	4.64	23.6	48.07
Kittiwake	4.6	24.94	51.95
Sandwich tern	2.93	14.3	29
Common tern	2.6	12.69	26.12
Common guillemot	13.76	69.38	146.1
Razorbill	5.14	27.6	58.35
Atlantic puffin	7.08	36.1	75.73

The increase in reproduction required to offset a 5% increase in mortality varied across the species modelled between 13% (common tern) and 70% (guillemot), while for a 10% increase in mortality the increase in reproduction required was in the range 26% to 146% (same species). At the lower end of the range are the shorter lived birds (e.g. terns), for which population growth rates are relatively sensitive to changes in reproduction, though still more strongly affected by survival. At the upper end are the longer lived birds (e.g. guillemot, puffin and gannet) for which population growth rates are much more sensitive to changes in adult survival. The terns, and to a lesser extent Arctic skua, breed earlier than most of the other species. This reduces the relative impact of additional mortality on these species as their populations are better able to replace themselves (through earlier reproduction) than ones which commence breeding at a later age. Conversely, a greater relative increase was required for guillemot. This species has the highest adult survival rate and the lowest reproductive rate, and hence its population can be considered as the most reliant on having long lived adults which although they breed at a low rate, do so for a comparatively longer life span.

It should be stressed that the numbers in table 5.3.1 need to be treated as a guide to the relative efficacy of alternative management options, not as an indication of how the population of any given species will definitely respond to any particular scenario. This is because the models are only as reliable as the data used to parameterise them (for some species very few demographic data are available) and also make several important assumptions, such as continuation of the conditions under which the demographic studies were conducted (i.e. the period over which the rates have been estimated will be representative of future conditions) and an absence of trade-offs between rates (e.g. survival costs of increased reproduction). Nonetheless, the broad message which emerges from this modelling is that for many seabird species, a given percentage of additional mortality can be offset by a rather larger percentage increase in reproduction, but the ratio varies considerably among species.

6. LIFE HISTORY CONSIDERATIONS AND THEIR IMPLICATIONS FOR MANAGEMENT

Life history theory envisages trade-offs between components of fitness, such as survival and reproductive investment (Stearns 1992). Long-lived birds, such as seabirds, may be expected to respond to adverse environmental conditions by reducing their investment in current reproduction in order to increase residual reproductive value and hence maximize lifetime reproductive success. Based on such considerations, Cairns (1987) proposed a model of seabird responses to food supply in which slight reductions in food supply affect breeding adult activity budgets and diet selection, but not breeding success or adult survival rate. Moderate reductions in food supply would affect breeding success, but only severe reductions in food supply would affect adult survival rates, by which time effects on breeding success would be catastrophic. Essentially the Cairns model proposes that because of the overwhelming importance of adult survival in seabird demography, the trade-off between investment in survival and breeding success is skewed strongly towards investment in survival. A similar trade-off would apply in relation to other environmental challenges, such as predation. According to the Cairns model, breeding adult seabirds should avoid risks of predation even at the expense of their own eggs or chicks.

Monaghan et al. (1992) proposed that an example of this trade-off would be that seabirds should abandon their breeding attempt if their body condition fell below a threshold value below which their survival might begin to be compromised. Wernham and Bryant (1998) showed that Atlantic puffins apparently do reduce the quality of their offspring rather than compromising their survival. However, in contrast to this theory and the examples mentioned above, several studies have shown that the trade-off between survival and productivity is less one-sided than this. In the kittiwake in Shetland, although breeding success correlated with sandeel stock biomass as predicted, contrary to the prediction above, adult survival rates of kittiwakes varied considerably from year to year (annual values from 0.98 down to 0.53, with standard error of around 0.02 so highly statistically significant variations). Contrary to the Cairns predictions, kittiwakes did not refrain from breeding in poor conditions, and the adult survival rate correlated with sandeel abundance. Experimental studies with breeding kittiwakes found similar effects, where the trade-off between survival and productivity led to considerable impacts on survival rates as well as on breeding success (Golet et al. 1998, Golet and Irons 1999). Similarly, Davis et al. (2005) showed that both breeding success and survival rates of Arctic skuas were affected by food supply and could be manipulated experimentally, with adults using some supplementary food to increase chick survival as well as using some to increase their own survival.

These results are important in relation to management to increase seabird populations. They imply that management measures that are primarily directed at influencing breeding success are likely to have consequences for survival rates too, and vice versa. This is particularly true where changes in food supply are concerned, but also applies for other challenges such as predation risk. In particular, the often quoted concept that seabird populations are primarily driven by variations in adult survival and not by variations in productivity (Stahl and Oli 2006, Finkelstein et al. 2010) appears to be a flawed concept for several reasons. Firstly, modelling seabird population dynamics (Section 5 above) indicates that changes in seabird population size are influenced by changes in productivity as well as by changes in survival rates for some UK seabirds (particularly those that start breeding at a relatively young age and rear several chicks per season) including terns and Arctic skua (Table 5.3.1). Secondly, modelling of the benefits to population conservation of increasing chick survival in the Laysan albatross by Finkelstein et al. (2010) showed that management to reduce lead poisoning of chicks significantly influenced the predicted population trajectory even in this extreme example of a seabird with especially high adult survival, delayed maturity and low productivity. Thirdly, empirical evidence from studies of seabird populations tends to indicate that both productivity and survival rates vary in relation to environmental factors, and therefore that there is scope for influencing seabird demography through manipulation of both or either of these parameters.

For many seabird species, although there has been some kind of estimate of the adult survival rate, either from studies of individually marked breeding adults at colonies or through analysis of ringing recovery data, there is a lack of evidence on the quantitative impact of environmental variables on survival rates. In contrast, the evidence regarding impacts of environmental factors on breeding success is very good for many seabird species. This leads to a better opportunity to demonstrate the evidence base for management actions to influence productivity, though paradoxically for most seabird species this may have a less powerful influence on population trend than actions to increase survival. Finkelstein et al. (2010) make the important conclusion '*overgeneralizations about demography can stifle useful conservation actions and highlights the need to consider the population-level benefits from multiple management strategies*'.

Another trade-off that should be considered is that between chick starvation and chick predation risk. In many studies it is evident that rates of predation of seabird chicks increase when there is a food shortage. For example, breeding adult Arctic skuas spend longer away from the nest searching for food when sandeel stock biomass is low, and as a consequence more chicks are killed by neighbouring great skuas (Phillips et al. 1998). Studies of common guillemots on the Isle of May showed that when sandeel abundance was severely reduced, adults spent less time at the nest site, and rates of predation of common guillemot chicks increased (Mavor et al. 2005). Productivity reductions ascribed to predation can be ultimately due to food shortage; management that increased food supply or that reduced predation threat could potentially increase productivity, but management action reducing predation when the ultimate cause of low productivity is lack of food might simply result in more chicks starving to death. Such interactions need to be given careful consideration for particular species.

7. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT OPTIONS FOR RED-THROATED DIVER

7.1 Red-throated diver ecology

Red-throated diver is a Schedule 1 and Annex 1 species, SPEC 3 (Unfavourable conservation status (Vulnerable) but not concentrated in Europe. The European breeding population is thought to be around 7,200 to 10,500 breeding pairs (Hagemeijer and Blair 1997).

7.1.1 The species in the British Isles

About 935-1,500 pairs of red-throated divers breed in the British Isles, with 100% of these in north and west Scotland (Forrester et al. 2007). Red-throated divers nest on the shores of freshwater lochs or pools, laying a clutch of one or two eggs (mean clutch size 1.8; Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). They commute from the nesting site to feed on small marine fish (especially sandeels) in shallow coastal waters (Forrester et al. 2007). About 395 pairs breed on 10 SPAs within the UK (Caithness and Sutherland Peatlands, Foula, Hermaness Saxa Vord and Valla Field, Hoy, Lewis Peatlands, Mointeach Scadabhaigh, Orkney Mainland Moors, Otterswick and Graveland, Ronas Hill – North Roe and Tingon, and Rum), representing an estimated 42% of the UK breeding population (JNCC web site).

Adult survival rate has been estimated to be around 0.84 (Hemmingsson and Eriksson 2002), but age of first breeding is thought to be at least 5 years old (Forrester et al. 2007), which would suggest that adult survival rate may well be higher than the one published estimate. UK breeding birds move to the coast after breeding, and overwinter at sea off sheltered, often estuarine, coasts. According to BirdLife International the maximum diving depth is 9 m and since this species often feeds on fish close to the seabed this may limit its foraging areas, although it regularly occurs in water up to 30 m deep in winter (http://seabird.wikispaces.com/Red-throated+Diver). Forrester et al. (2007) summarised the main threats to red-throated divers as disturbance making eggs and chicks vulnerable to predation by gulls and skuas, mink and otter predation of nests, lack of sandeels around Shetland, oil pollution, drowning in fishing nets, disturbance, displacement and collision mortality caused by offshore wind farms.

UK birds may winter from Orkney to the Atlantic coast of France (Forrester et al. 2007). Large numbers of red-throated divers arrive in British waters in autumn and remain until spring. These are thought to include birds from Greenland, Iceland and Scandinavia, and possibly from further east. A revised Great Britain wintering population estimate was compiled using data primarily from systematic line transect surveys by aircraft over marine nearshore areas conducted during 2001-2006. 17,116 (13,198-21,034, 95% confidence interval) red-throated divers were estimated to winter around Great Britain. The largest numbers were found off southeast and east Britain (59.3% of the total was between Flamborough Head, Yorkshire, and Dungeness, Kent), with large concentrations off the English south coast (10.9%), north Wales and Liverpool Bay (9.8%), and eastern Scotland (6.1%) (O'Brien et al. 2009). The Firth of Forth SPA holds 88 red-throated divers in winter. The Outer Thames holds over 6,000 red-throated divers in winter, the largest concentration in UK waters (Webb et al. 2009).

7.1.2 Present, and likely future trends

Breeding populations of red-throated divers in Shetland, Orkney and mainland Scotland probably increased in the second half of the 20th century, but surveys have not been considered highly accurate (Forrester et al. 2007). There was a decline of about 35% in breeding numbers in Shetland (the largest breeding population in the UK) between 1983 (when the population was thought to be at an 'all-time high' of around 700 breeding pairs) and 1994 (Pennington et al. 2004). Since then, breeding numbers appear to have been approximately stable. Wintering numbers are still not very well known, but may have declined since the 1990s in the Moray Firth and NE Scotland where there were previously large aggregations feeding on the local sprat stock. There is no obvious reason to expect major changes in breeding or wintering numbers in the UK in the immediate future (Forrester et al. 2007).

7.1.3 Factors affecting survival rates

Although there is one published estimate for the adult survival rate of red-throated divers, it is in a rather obscure 'grey' publication that was based on ringing relatively small numbers of breeding birds on lakes in part of Sweden (Hemmingsson and Eriksson 2002), so may not apply to populations elsewhere, and is of somewhat uncertain accuracy. There are no quantitative data on how survival rates are affected by particular factors such as food abundance, predators, drowning in fishing nets, oil pollution, or weather conditions. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

7.1.4 Breeding success in the British Isles

In 1988-90 red-throated diver productivity in Shetland was mainly reduced by low abundance of sandeels (Walsh et al. 1992), with minor additional impacts from predation, flooding of nests, and disturbance by people. Studies in Shetland indicated that human disturbance facilitates predation of red-throated diver nests. Adults which leave nesting lochs on the approach of humans are more likely to fail to rear young, especially when predation pressure by gulls and skuas is high (Walsh et al. 1992). Evidence of the effect of reduced sandeel abundance (which was measured around Shetland by fisheries surveys) includes a reduction in the proportion of broods of two chicks in years of low sandeel stock (Walsh et al. 1992). In addition, during 1971-80 when sandeel stock biomass was high, 97% of fish carried to feed chicks at Foula, Shetland, were sandeels but this fell to 17% in 1988 and was again low in 1989, coinciding with low breeding success (Walsh et al. 1992). According to Walsh et al. (1993), improved productivity of red-throated divers in 1992 compared to 1988-90 may be a reflection of increased availability of sandeels in 1992 compared to 1988-90. Marked variations in success between areas, was thought likely to reflect variations in predation and human disturbance (Walsh et al. 1993). In 1997, otters took eggs, a chick and an adult on Eigg, while great skuas took chicks on Foula (Thompson et al. 1998), but overall productivity in 1997 at 0.67 in Shetland, 0.47 in Orkney, and 0.33-1 elsewhere, was close to the average for 1986-1996. In 1999 in Shetland, some divers failed due to heavy rain in late May causing flooding of nests and otters killed young at five sites on Yell, but mean productivity at 0.59 chicks per pair was almost the same as the average for 1986-1998 (0.6) (Upton et al. 2000). High productivity in Orkney in 1999 (0.75 chicks per pair) was attributed to improved food supply (Upton et al. 2000). In 2000, a severe storm in June destroyed many nests of red-throated divers in Shetland, especially on larger lochs and easterly shorelines, although birds in Orkney and elsewhere were apparently unaffected (Mavor et al. 2001). However, despite the storm, productivity in Shetland in 2000 (0.53 chicks per pair) was only slightly below the mean for 1986-1999 (0.6 chicks per pair). An otter was believed to have taken both clutches of pairs on Eigg in 2000 (Mavor et al. 2001). In 2001, two pairs of divers nested for the first time on artificial sites provided in 2001. Both were successful, raising three chicks on reservoirs where fluctuating water levels had previously caused nesting failure (Mavor et al. 2002). In 2001 on Rum, atypically high rainfall in early June was thought to have reduced productivity there (0-0.25 chicks per pair), while apparently an otter on Eigg not only depredated two clutches and caused a third to be abandoned, but also killed an adult red-throated diver on the nest (Mavor et al. 2002). In 2003, red-throated diver breeding success was similar to the long-term average. Pairs at two lochs where breeding has tended to fail due to falling water levels in summer, were provided with an artificial island and a raft. Both pairs were successful at these artificial nest sites (Mavor et al. 2004), suggesting that artificial nest sites may improve productivity at lochs where nesting on the shore is often unsuccessful. In 2004, many red-throated diver chicks in Shetland died of starvation, and some of these were scavenged by great skuas, giving the false impression that low productivity was due to predation (Mavor et al. 2005). However, at many sites chick weights were low for their age and there was no sign of predation, indicating that food shortage was the main problem. In 2005, red-throated diver breeding success in Shetland was reduced primarily by food shortage, but also by flooding and predation (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for red-throated divers as follows. Success was especially low at Hermaness (0.38), where most pairs failed at an early stage, possibly due to food shortage, and on Foula (0.33) where many chicks disappeared during the second half of July possibly due to predation. On Fetlar, twelve out of 28 pairs had clutches depredated and four of seven chicks lost were known to have been taken by either great skuas or otters. In Shetland, due to a wet winter and above average rainfall between March and July, water levels at nesting lochs remained high throughout the breeding season, with little fluctuation in levels which can cause problems in some years. A small proportion of pairs did not attempt to breed. Few sandeels were seen to be delivered to chicks; nearly all food items identified were saithe, which have a lower calorific value than sandeels, but it is still possible for divers to successfully raise chicks on this lower quality fish. In Orkney, six out of 21 pairs monitored on Mainland failed due to low water levels in dry weather. On Eigg, three pairs all failed due to predation by otters, an annual problem on that island (Mavor et al. 2008).

Breeding success can be improved by provision of nesting platforms, especially on lochs where there is fluctuation in water levels (e.g. reservoirs) or high risk of human disturbance or predator activity (http://seabird.wikispaces.com/Red-throated+Diver). The benefits of nesting rafts for divers have also been established by trials in Argyll (Hancock 2000, ap Rheinallt et al. 2007).

Table 7.1.1 Meta-analysis of main factors contributing to reduced productivity of red-throated divers at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	12
Flooding of nests	8
Otter predation	6
Great skua predation	3
Gull predation	2
Drying out of lochs	2
Human disturbance	2

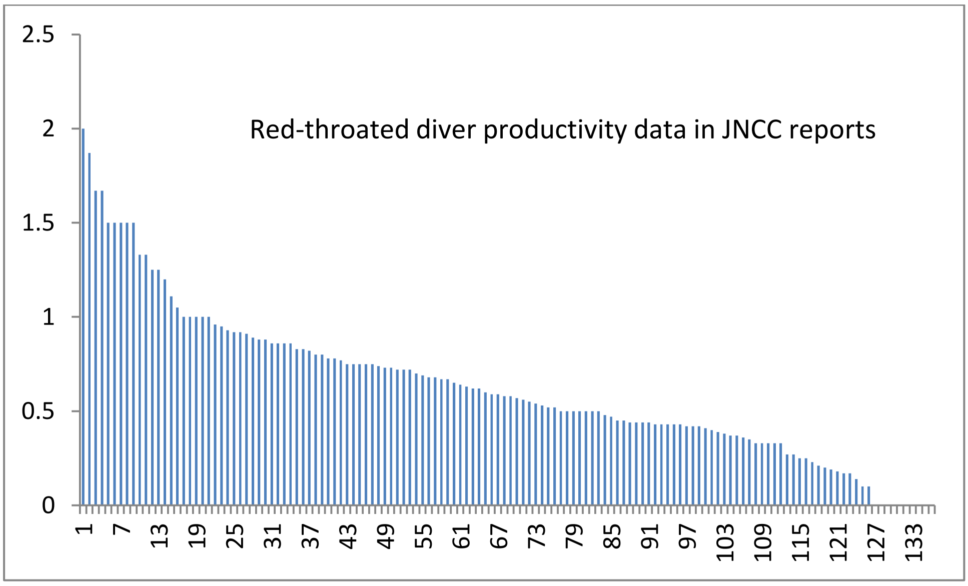

Figure 7.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of red-throated divers at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

7.2 Management options

	7.2.1 Nesting rafts	7.2.2 Closure of sandeel and sprat fisheries close to wintering areas	7.2.3 Closure of sandeel and sprat fisheries close to breeding areas	7.2.3 Closure of sandeel and sprat fisheries in all UK waters	7.2.4 Prevent oil spills
Evidence of success for this species	High C=High*	Low C=Low	Low C=Low	Low C=Low	Low C=Mod
Evidence of success for similar species	High C=High	Low C=Low	Low C=Low	Low C=Low	High C=Mod
Cost-effectiveness	High C=High	Uncertain C=Low	Uncertain C=Low	Uncertain C=Low	Uncertain C=Low
Feasibility	High C=High	Moderate C=Low	Moderate C=Low	Moderate C=Low	Low C=High
Practicality	High C=High	Moderate C=Low	Moderate C=Low	Moderate C=Low	Low C=High
Applies at SPA populations	Yes C=High	Uncertain C=High	Yes C=High	Yes C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

7.2.1 Provision of nest platforms on breeding lochs

There is strong evidence available that providing floating nesting platforms increases productivity considerably, especially on lochs where water levels fluctuate, predators are present, or human disturbance occurs. This also appears to be a management option that would be generally supported by the public, and where volunteer help and matched funding may be available.

A project was started in Argyll in 1976 to provide nesting rafts for breeding divers on two estates. This concept was then adopted by the RSPB for black-throated divers in Argyll and the Highlands, and by Forest Enterprise for red-throated divers in Argyll. Breeding success of red-throated divers in Argyll in 1973-78 (when there were no rafts) averaged 0.35 young reared per pair. Productivity of pairs provided with rafts increased to an average of 0.75 chicks reared per pair per year (ap Rheinallt et al. 2007). Rafts were found to eliminate the risks due to water level fluctuation, and reduce those due to human disturbance and natural predation from foxes and mink which were the main causes of failure in Argyll (Merrie 1996, ap Rheinallt et al. 2007). However, only 15% of red-throated diver pairs in Argyll were provided with rafts. Productivity of the closely related black-throated diver, which also nests in the same kind of nest sites as red-throated divers, was initially doubled on sites in Scotland where they were provided with nesting rafts (Hancock 2000). However, from 1985-2004, 56 breeding attempts at natural sites in Argyll fledged 19 young (0.34 per pair) and 60 breeding attempts at raft sites fledged 30 young (0.5 per pair) (ap Rheinallt et al. 2007) which represently slightly less than a doubling of productivity over a longer period, possibly because some rafts became less suitable over the years; maintenance of rafts or periodic replacement may be a way to maintain higher output over the long term.

In Shetland in 2001, two pairs of red-throated divers nested for the first time on artificial sites provided in 2001. Both were successful, raising three chicks on reservoirs where fluctuating water levels had previously caused nesting failure (Mavor et al. 2002). These pairs were deliberately selected for rafts because it seemed unlikely that they would ever nest successfully on lochs with large fluctuations in water height caused by human management.

Productivity of common loons (great northern divers) is also considerably increased when they are provided with suitable nesting rafts (Piper et al. 2002, de Sorbo et al. 2007).

In North America, the loon preservation committee puts out about 40 rafts each year for common loons (great northern divers) in New Hampshire, and strongly advocates the benefits of these rafts for increasing productivity in that state, protecting the birds from fluctuating water levels and reducing impacts of human recreational disturbance and nest predation by raccoons http://www.loon.org/nest-rafts.php. The Big Mantrap Lake Association also deploys rafts for nesting loons on Big Mantrap Lake Minnesota http://mantraplake.webs.com/loonnestingrafts.htm Their rafts are rather unattractive, with a metal mesh roof, whereas most diver rafts are designed to look like natural islets, but their design presumably helps to reduce predation risk, and they state that it achieves high productivity and is well used by their population of great northern divers. Because diver breeding lakes in North America freeze over in winter, they deploy rafts each spring when ice thaws, and remove them in the autumn. What appears to be an updated version of this raft is described in detail by De Sorbo et al. (2008) who also estimate the cost of materials to construct their design at around US$100 per raft for a raft that will require replacement about every 10 years. In Britain, diver rafts are left in place all year round, reducing the amount of volunteer effort required compared to the situation in North America. Maine Audubon Society provide guidance on construction and siting of diver nesting rafts http://maineaudubon.org/wp-content/uploads/2011/10/Loon-raft-plans.pdf as does Hancock (2000) for divers in Scotland.

Since there are about 395 pairs of red-throated divers breeding on ten SPAs in Britain, it would be necessary to deploy large numbers of nesting rafts in order to significantly increase productivity at the population level. This could best be achieved by selecting sites that have low breeding success because of known problems (fluctuating water levels, human disturbance, predation), since provision of nest rafts on sites where divers regularly nest successfully would not increase productivity. Normally, only one pair of divers nests on a particular loch, although there are a few exceptions to that generalisation. Lochs that have a small island and do not suffer from fluctuating water levels would not be improved by adding a raft; lochs without any islands or where water levels fluctuate strongly would be likely to show a large increase in diver productivity once a raft is available. This probably describes about 25% of red-throated diver nesting lochs, although that value will undoubtedly vary among regions depending on local conditions. The JNCC productivity monitoring data (Figure 7.1.1) shows a long ‘tail’ with poor productivity, suggesting that there is scope for many sites to be enhanced by provision of a nesting raft.

The cost of deploying 100 nesting rafts at suitably chosen red-throated diver lochs where productivity has tended to be low would depend on the logistics (such as distance of the loch from a road). However, typically it might be around £2,000 per site to install nesting rafts (David Okill Shetland Bird Club, Roger Broad RSPB, David Merrie Argyll Bird Club, pers. comms.). Rafts would require occasional maintenance checks, ideally once per year before the breeding season, and would probably require replacement every 10 to 20 years, depending on the exposure of the site (rafts on larger lochs tend to be more affected by wave action than on small lochs, and rafts on lochs sheltered from the wind last longer than those on exposed upland lochs (Roger Broad RSPB, David Merrie Argyll Bird Club, pers. comms.). Provision of nesting rafts at 100 of the least productive nesting lochs within the UK red-throated diver SPA suite could boost productivity at those 100 lochs from an average of around 0.3 to an average of around 0.7 chicks per pair (based on evidence outlined above in this section, and in section 7.1.4). This would be equivalent to boosting the productivity of the entire ca 400 pairs on SPAs by 0.1 chicks per pair (an additional 40 chicks per year). At many red-throated diver SPAs the sites which are least productive are well known as a result of monitoring. For example, Foula SPA has around 12 pairs of red-throated divers, but three of the sites are unsuccessful in most years; two of these are lochs used for the local pump-storage hydro-electric scheme which have severe fluctuations in water level and no islands, leaving incubating red-throated divers either stranded far from the water edge or flooded in most years, and one is the largest loch on the island, close to human disturbance, and which has no island so divers nest on the shore and are usually unsuccessful due to human disturbance. However, possibly a greater gain for the UK red-throated diver breeding population would be achived if nest platforms were distributed not just in SPA areas but across the entire UK population, targeting least productive sites which may often not be on SPAs.

7.2.2 Closure of sandeel and sprat fishing close to wintering area SPAs

Red-throated divers aggregate in sheltered bays and estuaries in winter, areas where they can feed in sheltered sea on small prey fish such as sprats and sandeels. Closure of fishing in areas close to SPAs for nonbreeding red-throated divers, or throughout UK coastal waters, should increase survival rates but evidence for this is lacking (because it would be difficult to gather). There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species. There are localised fisheries for sprats in UK waters, by Scottish fishermen off west Scotland (usually close to Mull), by English fishermen in the English Channel (Lyme Bay), and occasionally as a result of the activity of foreign industrial fishing fleets (ICES 2013). There have in the past been fisheries on sprats in the Moray Firth and in the Firth of Forth, fisheries which appear to have caused local depletion of those stocks and then been closed (Jennings et al. 2012). There has in the past been a fishery by Scottish fishermen on sandeels in Shetland which was closed in 1991 due to the depletion of that stock which has still not recovered, and the large industrial fishery for sandeels in the North Sea has moved around over the years to exploit different stocks within the North Sea, resulting in depletion of many of the distinct sandeel stocks in the northern North Sea, but continued exploitation by Danish fishermen of sandeel stocks in English waters (ICES 2010, ICES 2012). All of these sandeel and sprat stocks are likely to represent important preferred food of red-throated divers in UK waters in winter. Closure of sandeel fishing in UK waters would have low cost for UK fishermen, as almost the entire fishery is carried out by Danish fishermen. Closure of sprat fishing in UK waters would affect a small number of fishing vessels in west Scotland and in Lyme Bay, and limit potential development of sprat fisheries on stocks that are currently unfished (e.g. in Irish Sea, Clyde, Thames, Firth of Forth, Moray Firth).

7.2.3 Closure of sandeel and sprat fishing close to breeding area SPAs or all UK waters

There is strong evidence available that red-throated diver productivity is reduced when prey fish stocks are depleted. By analogy with evidence for some other species (e.g. Arctic skua, kittiwake) adult survival rates are also likely to vary with prey fish stock abundance. Red-throated divers feed in shallow sea within about 20 km of breeding areas. Closure of fishing for sandeels and sprats within 20 km of red-throated diver SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. Also, several of the red-throated diver SPAs are in Shetland and Orkney, regions where there is no fishing for sandeel or sprat (fishing for sandeels at Shetland was closed in 1991 due to depletion of the sandeel stock there and has never re-opened; the sandeel stock at Shetland remains depleted). Closure in all UK waters would include large areas of marine habitat not used by breeding red-throated divers, as their populations are in a limited range within north and west Scotland.

7.2.4 Reducing oil pollution

Red-throated divers are highly vulnerable to oil pollution when at sea, and during winter red-throated divers spend almost 100% of their time on the sea (small amounts of time in flight or underwater). So reducing risk of oil pollution would benefit diver over-winter survival as it would that of auks (see section 18.2.4 for example). However, the quantitative importance of oil mortality for red-throated diver populations is unknown, as is the quantitative effect of any reduction in amount of oil pollution in red-throated diver wintering areas.

8. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR MANX SHEARWATER

8.1 MANX SHEARWATER ECOLOGY

Manx shearwater is a SPEC 2 species (Unfavourable conservation status (localised) and concentrated in Europe. The world population is 340,000 to 410,000 breeding pairs (Mitchell et al. 2004), all located within Europe apart from a handful of pairs nesting in Canada.

8.1.1 The species in the British Isles

About 280,000 to 300,000 pairs breed in Great Britain, the Channel Islands and Isle of Man, and 27,000 to 61,000 pairs breed on Ireland. Manx shearwaters nest in burrows, predominantly in a small number of very large colonies, but with a number of small colonies scattered between the big ones. Virtually all colonies are in the Atlantic Ocean, Celtic Sea or Irish Sea, with the only colonies technically in the North Sea being very small colonies in Shetland. The adults only come ashore after sunset and departures occur before dawn. Adults at colonies during daylight are only those remaining within their burrow. The clutch size is a single egg. Manx shearwaters forage over considerable areas from their colonies, feeding predominantly on small shoaling pelagic fish.

According to JNCC, '*In the breeding season, the UK's SPA suite for Manx shearwater supports an average of 219,898 pairs. This amounts to effectively all of the British breeding population and most (nearly 83%) of the international population*' (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-7.pdf). These SPAs are Bardsey Island, Rum, Skomer Skokholm and Middleholm, and St Kilda. Copeland Island, Co. Down was added to the SPA suite for Northern Ireland in 2009.

Adult survival rate of Manx shearwater is around 0.9 (Perrins et al. 1973) but predation by great black-backed gulls at Skomer may add up to 2% to natural mortality rate of adults (Cramp and Simmons 1977-1994). Age of first breeding usually 5 or 6 years old (Cramp and Simmons 1977-1994). Manx shearwaters migrate from British colonies to winter over the continental shelf sea off South America. None remain in European waters in winter. According to Mitchell et al. (2004) the main threats to Manx shearwaters are introduced alien predators, especially brown rats and feral cats. Population declines and extinction have been recorded on islands that have been invaded by rats (through accidental introduction or otherwise). For example, the species became absent or extremely rare on the Calf of Man and on Lundy following the arrival of rats. Natural predators such as eagles, gulls and skuas have some impact, but generally not enough to cause population decline. Puffins may compete for burrows at some colonies, influencing local distribution. Climate change, especially heavy rainfall during incubation, represents a threat, as may fishery bycatch and impacts of fishing on pelagic fish abundance. Since there are relatively few Manx shearwaters breeding outside the British Isles, migration of birds from other populations through British waters is very limited, and almost all birds seen in British waters will be birds from British colonies.

8.1.2 Present, and likely future trends

Manx shearwaters were wiped out on the Isle of Man in the 1780s by brown rats that arrived there off a shipwreck, although a very few pairs recolonized the Calf of Man around 1999 (Mitchell et al. 2004). Most large colonies now remaining in the British Isles are on rat and cat-free islands, although there are several colonies where rats and other mammal predators are in the process of eradicating remaining populations. Over 90% of the British population of Manx shearwaters is on three islands: Rum in NW Scotland, Skomer and Skokholm (these two together with adjacent Middleholm representing a 'supercolony' in Pembrokeshire, Wales). Trends in these populations determine the overall population trend. However, Manx shearwaters breed at, at least 36 and possibly as many as 50, other colonies in Britain and Ireland (Mitchell et al. 2004). Trends in numbers breeding at the largest colonies are uncertain. There is some evidence suggesting that numbers may have increased on Skokholm (Mitchell et al. 2004). There is some slightly stronger evidence suggesting a decline in breeding numbers on Rum (Mitchell et al. 2004). At some smaller colonies, population change has been much clearer. Manx shearwaters were extirpated by brown rats on Canna in 2002 although the population had been between 1,000 and 15,000 pairs in the late 1970s (Mitchell et al. 2004). On the Isles of Scilly, there were 201 occupied burrows in 2000, whereas earlier counts were much larger (e.g. 900 pairs in 1974), so this population has probably been declining, possibly due to brown rats and feral cats (Walsh et al. 1995, Heaney et al. 2002, Mitchell and Ratcliffe 2007). On Lundy, only 154 responses to tape playback were obtained from over 7,000 potential burrows in May 2001, suggesting a major decline in breeding numbers there, possibly again relating to impacts of rats and feral cats (Mavor et al. 2002, Appleton 2007). In the Channel Islands, Manx shearwater colonies are thought to be on the verge of extinction due to brown rats and cats (Mitchell et al. 2004). On Rathlin Island County Antrim, Manx shearwaters have been extirpated from their main breeding areas and only a tiny number of pairs remain on 'inaccessible' ledges following introduction of ferrets to the island several years previously (Thompson et al. 1996, Mitchell et al. 2004). In Shetland, colonies on Foula, Horse of Burravoe and Yell were all thought to have been extirpated by mammal predators (especially feral cats), while the colony on Fetlar has been reduced almost to extinction, apparently also due to feral cats (Walsh et al. 1995, Pennington et al. 2004).

The trend for colonies to disappear due to mammal predators is likely to continue in the future, and the view that predation by brown rats on Rum may now be having a serious impact on productivity at that colony (Mavor et al. 2004, 2005, 2006) is of particular concern given that this island holds one of the two or three largest populations of this species in the world, is an SPA for this species, and is a National Nature Reserve.

8.1.3 Factors affecting survival rates

Although there are published estimates for the adult survival rate of Manx shearwaters which indicate an adult survival rate around 0.9 (Perrins et al. 1973), apart from the tentative suggestion that predation by great black-backed gulls may increase adult mortality by up to 2% (Cramp and Simmons 1977-1994), based on the estimate by Buxton and Lockley (2960) that great black-backed gulls on Skomer killed 2,500 Manx shearwaters a year on the island. Mylne (1960) also estimated that great black-backed gulls on Skomer were killing 5,000 to 10,000 Manx shearwaters a year on Skomer in the 1950s, which led to culling of gulls there, and reduction in numbers from over 300 pairs to about 40 pairs (Poole 1995). There are no quantitative data on how survival rates are affected by particular factors such as food abundance, other predators including alien mammals such as rats, drowning in fishing nets, oil pollution, weather conditions while breeding, or conditions on migration and in the wintering area off South America. However, there is evidence that mammal predators such as rats, mink and feral cats that become established on islands where there are Manx shearwater colonies can kill large numbers of adult shearwaters as well as taking eggs and chicks, and can in some cases cause breeding numbers to decline to extinction (Walsh et al. 1995, Mitchell et al. 2004, Lock 2006, Mavor et al. 2006, Luxmoore 2007, Patterson 2007, Swann et al. 2007). This implies the likelihood of a strong impact on adult survival rates. Estimation of the extent to which management might alter survival rates is therefore extremely difficult, but qualitative assessment is certainly possible.

8.1.4 Breeding success in the British Isles

In 1993 on Skomer, very wet weather in May was believed to have contributed to poor productivity as burrows were flooded (Walsh et al. 1994). In 1994, productivity on Canna was apparently around 0.75 chicks per occupied burrow, but fewer than 100 burrows there were occupied. Brown rats were known to be present on the island but were not thought to be affecting the shearwaters (Walsh et al. 1995). Colonies on Foula, Horse of Burravoe and Yell were all thought to have been extirpated by mammal predators (especially feral cats) (Walsh et al. 1995). In contrast, the small colony on Fetlar had recovered to 8 pairs in 1995 with no sign of cat predation that had affected that colony in previous years (Walsh et al. 1995). In 1995, occupancy of shearwater burrows on Canna was noted to be low (40%), and only 0.25 chicks fledged per occupied burrow, the lowest productivity recorded since 1982; there was evidence of predation by rats, with several partly-eaten chicks found (Thompson et al. 1996). On Rathlin Island in 1995, shearwaters have become confined to inaccessible grassy cliff ledges following introduction of ferrets to the island several years previously (Thompson et al. 1996). In 1997, a census at Canna indicated a decline in numbers from 1,000-1,500 pairs in the mid-1970s to 65 in 1997, attributed at least in part to depredations by brown rats, and possibly also feral cats; in addition, breeding success was only 0.07 chicks per pair for the few pairs attempting to breed there (Thompson et al. 1998). In Wales in 1997, many chicks on Skokholm drowned when their burrows flooded in exceptionally heavy rain, but breeding success was good on Skomer (0.64) and Bardsey (0.8) in the same season (Thompson et al. 1998). In 1998, flooding of burrows during heavy rain reduced productivity on Rum and on Lighthouse Island (Co. Down) (Thompson et al. 1999). In 1999, poor breeding success on Rum (0.56 chicks per occupied burrow) was attributed to predation by rats (Upton et al. 2000). On Lighthouse Island (Co. Down) wet weather resulted in chilling of many eggs and drowning of chicks (Upton et al. 2000). In 2000, all the shearwater burrows normally monitored on Canna had been abandoned as a result of the activities of brown rats (Mavor et al. 2001). There was a concern on Rum that numbers of fledglings were declining in the area where these are traditionally ringed on the surface of the colony, and that this decline may be due to activities of rats on Rum (Mavor et al. 2001). In 2001 on Skomer productivity was 0.43 chicks per nest, well below average apparently due to a thunderstorm on 6 July flooding many burrows (Mavor et al. 2002). On Lundy in 2001, signs of rat predation, including broken eggs and carcasses of adults, were apparent at several sub-colonies of the relatively small population on that island which is not regularly monitored (Mavor et al. 2002). On Bardsey in 2002, productivity was high (0.8) but carrion crows there took at least 80 eggs from exposed burrows (Mavor et al. 2003). In 2003 on Rum, breeding success in monitored burrows was higher than average, but there was evidence suggesting that the colony as a whole had reduced numbers of fledglings, and this was thought possibly due to predation by rats (Mavor et al. 2004). On Bardsey in 2003, a pair of carrion crows took eggs from 50 accessible shearwater nests (Mavor et al. 2004). In 2004, for the first time on Rum, eggs were eaten by rats in a small number of the monitored burrows (Mavor et al. 2005). This represents a change from the situation described by Thompson (1987) who found no evidence of rat predation on shearwater eggs on Rum in 1984 and 1985. On Canna, where rat predation led to the extirpation of the Manx shearwater colony in 2002, no shearwaters were found attending any burrows in 2004 and in 2005 one adult was found that had apparently been killed by rats (Mavor et al. 2006). On Bardsey, one pair of carrion crows took as many as 40 shearwater eggs from burrows (Mavor et al. 2005). In 2005, rats were again evident in the Manx shearwater colony on Rum, and took some eggs although the extent of depredation is unclear (Mavor et al. 2006). On Skomer, breeding success of shearwaters has been low since 2000; reasons for failures at 40 nests in 2005 were egg abandoned (13 nests), egg broken (8 nests), adults evicted (3 nests, two by puffins, one by another shearwater), burrow collapse (1 nest) and failed at egg or small chick stage (15 nests) (Mavor et al. 2006). These losses suggest food shortage as the likely main cause. Mavor et al. (2008) described the 2006 breeding season for Manx shearwaters as follows. Breeding success on Rum, at 0.59 chicks per egg laid, was well below the long-term average. The breeding season was exceptionally late and protracted, possibly in part due to very cold weather in April and May. On Bardsey, breeding success (0.74 chicks per occupied burrow) was lowest since monitoring began in 1996. Outwith the study plot, carrion crows targeted accessible nest chambers taking 29 eggs. Breeding success was also below average at Skomer, with many eggs abandoned during incubation, suggesting food shortage.

Table 8.1.1 Meta-analysis of main factors contributing to reduced productivity of Manx shearwaters at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Rat predation	12
Intense rainfall	6

Factor	Cases reported
Feral cat predation	4
Crow predation	4
Food shortage	2
Ferret predation	1

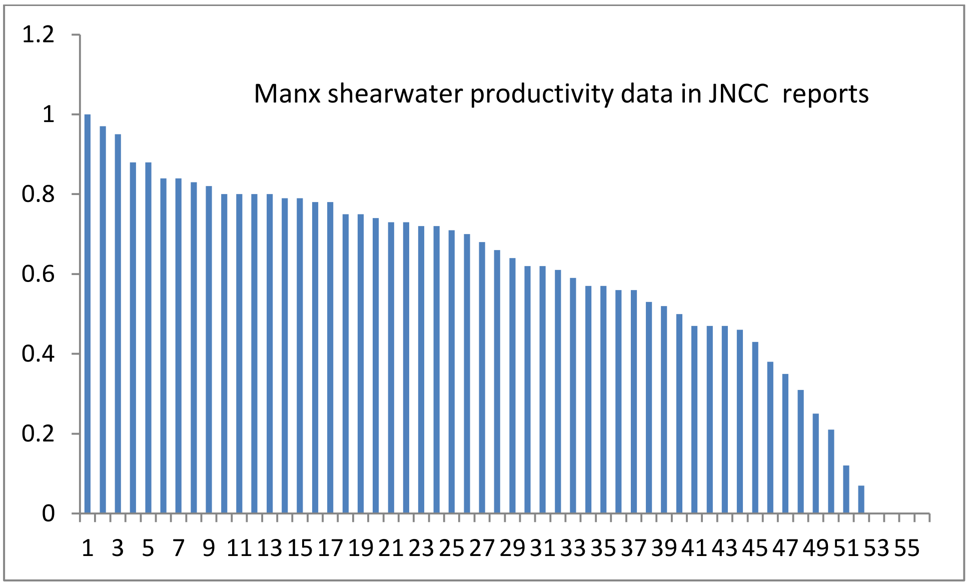

Figure 8.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Manx shearwaters at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

8.2 Management options

	8.2.1 Eradicate rats	8.2.1 Eradicate feral cats/ferrets	8.2.2 Exclude large gulls from around colony	8.2.3 Closure of sandeel and sprat fisheries close to breeding areas	8.2.3 Closure of sandeel and sprat fisheries in all UK waters
Evidence of success for this species	High C=High*	Low C=Low	Moderate C=High	Low C=Low	Low C=Low
Evidence of success for similar species	High C=High	High C=High	Moderate C=High	Low C=Low	Low C=Low
Cost-effectiveness	High C=High	High C=High	Moderate C=High	Uncertain C=Low	Uncertain C=Low
Feasibility	High C=High	Moderate C=High	Moderate C=High	Moderate C=Low	Moderate C=Low
Practicality	High C=High	Moderate C=High	Low C=High	Moderate C=Low	Moderate C=Low
Applies at SPA populations	Yes (Rum) C=High	No C=High	Yes C=High	Yes C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

8.2.1 Eradication of alien mammals from islands with Manx shearwater colonies

There is strong evidence that productivity is considerably reduced at some Manx shearwater colonies where alien mammals have been accidentally introduced and become established. Survival rates have almost certainly also been reduced at these colonies although evidence of this is lacking because survival rates at those colonies have not been measured, but Manx shearwaters have apparently been extirpated from several islands by alien predators (Ratcliffe et al. 2009) and reduced in numbers at several other islands (Table 8.2.1). Keitt et al. (2002) estimated that feral cats on Natividad Island were killing about 200 shearwaters (of a different species) per year per cat, showing how high cat predation impact can be for shearwaters due to their physical vulnerability while on land. Species of alien mammals currently reducing productivity at Manx shearwater colonies monitored in the UK include brown rats, feral cats, ferrets, red deer, and may include American mink (Table 8.2.1). Sites where alien mammals are reducing productivity at present include one SPA, the island of Rum, which may hold the largest colony of this species in the world. This is a particularly concerning development, as brown rats have been present on Rum for a long time, and were thought to be having no impact on shearwaters when this interaction was studied in the 1980s (Thompson 1987). However, more recent studies indicate an impact on productivity and possibly on survival rates, and suggest that this may now be starting to cause a decline in shearwater numbers (Mavor et al. 2006). This is consistent with many other examples, where rats and seabirds co-existed for some time with little evidence of impacts, but subsequently rat impact increased until the seabird population was eradicated. This is what was described, for example, for the island of Canna (Swann et al. 2007). Ratcliffe et al. (2009) identified Rum as one of the ten top sites in the UK for rat eradication to benefit seabirds. They stated '*The co-occurrence of brown rats with a large proportion of the national Manx shearwater population on Rum is cause for concern. Research during the 1980s showed that rats were relatively rare in this high-altitude colony during the breeding season, and predation was negligible (Thompson and Furness 1991), but recent monitoring has found that predation rates on viable eggs and chicks have increased (A.D. Ramsay unpubl. data). Experimental control of rats in the worst affected colony on the mountain of Hallival is being considered to quantify the impact of rat predation on Manx shearwater productivity (A. Douse pers. comm.)*'. Since Manx shearwaters have become extinct at some sites due to alien mammal predation, eradication of alien mammals at those sites may also require reintroduction programmes to encourage Manx shearwaters to re-colonize those sites.

Table 8.2.1. The main present and recently occupied Manx shearwater colonies in the UK, ranked by estimated colony size in 2000, and the status of invasive alien mammals at these sites.

Colony	**Pairs (AOS) in 2000**	**Alien mammals present**	**Impacts**	**Population trend**	**SPA status**
Rum, Lochaber	120,000*	Brown rats	Yes	Declining	SPA
Skomer, Dyfed	101,800**	No	No	Stable?	SPA
Skokholm, Dyfed	46,200	No	No	Stable?	SPA
Bardsey, Gwynedd	10,000	No	No	Increasing	SPA
St Kilda, Western Isles	4,800	No	No	Stable?	SPA
Copeland, Co. Down	4,600	No	No	Increasing?	SPA
Middleholm, Dyfed	3,000	No	No	Increasing	SPA
Treshnish Isles, Argyll	1,283	No	No	Unknown	
Ramsey, Dyfed	950	(brown rats eradicated 2000)	No	Now increasing	
Eigg, Lochaber	250	Cats	No?	Increasing	
Sanda, Argyll	200	Mink	?	Stable?	
Isles of Scilly	200	Brown rats, cats	Yes	Depleted	

Lundy, Devon	166	Black rats and brown rats eradicated 2004, cats	Yes	Depleted	
Calf of Man	34	Brown rats, cats	Yes	Depleted	
Channel Islands	10	Brown rats, cats	Yes?	Depleted?	
Fetlar, Shetland	7	Cats	Yes	Depleted	
Foula, Shetland	0	Cats	Yes	Extirpated?	
Canna, Lochaber	0	(brown rats eradicated 2005), cats	Yes	Depleted or Extirpated	
Rathlin, Co. Antrim	0-10	Ferrets	Yes	Depleted	
Cardigan Island, Dyfed	0	Brown rats eradicated in 1968 but no natural recolonization by Manx shearwaters there up to 2000.	Yes	Extirpated	

*The population on Rum is now thought to be closer to 60,000 to 70,000 pairs and possibly declining, although 120,000 pairs is the official census from Seabird 2000 (Dr Andy Douse, pers. comm.)

**Perrins et al. (2012) suggest that the true figure for this colony may be 316,000 breeding pairs in 2011, but the 2011 census by tape playback may either indicate a large increase in numbers or a difference due to survey methodology.

Eradication of brown rats at Rum, Isles of Scilly, and Calf of Man, eradication of ferrets at Rathlin Island, eradication of mink at Sanda, and eradication of feral cats at Calf of Man, Fetlar, Foula, and Canna, would all be management measures that could allow increases in Manx shearwater productivity and survival, and in the longer term increases in breeding numbers at these sites, almost all of which have seen depletion of shearwater numbers. A LIFE-funded project to eradicate rats in the Isles of Scilly is currently underway. The eradication is focussing on those linked islands to which re-invasion is unlikely. Due to the linked nature of the remaining islands (all lie within rat swimming distance of each other) any subsequent eradication would have to embrace the entire archipelago, and may be beyond current feasibility limits (RSPB in litt.). Since feral cats and ferrets act as 'superpredators' the removal of their populations from islands should be especially targeted as it is likely to have a greater benefit than removal of rats alone. However, the logistics of eradicating rats are simpler than those of eradicating feral cats and local public approval of rat eradication is likely to be high, whereas killing cats is likely to be less widely supported by the resident human population or stakeholders (Ratcliffe et al. 2009). In particular, on islands such as Foula and Fetlar in Shetland, some cats are 'semi-feral' and will come into houses in winter but live on the cliffs feeding on seabirds in summer. Some of these animals will be considered to be domestic pets by residents rather than feral cats having serious impacts on seabird populations. For cat eradication projects to be successful in the long term on islands with human resident populations, there would be a need to control fertility of domestic cats and to limit future importation of domestic cats to neutered animals (Ratcliffe et al. 2009).

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island (Morgan 2012), Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for other British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000).

There seems to be little or no literature on the eradication of alien populations of ferrets (such as the one on Rathlin Island), but this would appear to be equivalent to eradication of species such as American mink, which can be trapped out relatively easily from small areas at low cost (Craik 2007), although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. Rathlin Island lies somewhere between these two extremes.

8.2.2 Exclusion of large gulls from Manx shearwater colonies

Great black-backed gulls were identified as increasing adult mortality by up to 2% (Cramp and Simmons 1977-1994), based on the estimate by Buxton and Lockley (2960) that great black-backed gulls on Skomer killed 2,500 Manx shearwaters a year on the island. Mylne (1960) also estimated that great black-backed gulls on Skomer were killing 5,000 to 10,000 Manx shearwaters a year on Skomer in the 1950s, which led to culling of gulls there, and reduction in numbers from over 300 pairs to about 40 pairs (Poole 1995). There is no evidence of gulls killing Manx shearwaters on Rum (Furness 1988). Small numbers are killed by great black-backed gulls on St Kilda, and at that colony a few are also killed by great skuas (Furness, pers. obs.). There seem to be no major problems with shearwaters being killed by gulls at other colonies in the UK (Mitchell et al. 2004; data in 8.1.3). It would, therefore, appear that removal of gulls that had been a problem for Manx shearwaters has already been carried out at colonies where this was perceived to be a major issue. There might be limited scope for removal of large gulls that can be identified as killing Manx shearwaters at colonies, but the scope for this to significantly improve shearwater survival rates seems limited.

8.2.3 Closure of sandeel and sprat fishing close to breeding area SPAs or all UK waters

Evidence indicates that Manx shearwater productivity can be reduced by food shortage around breeding colonies, although this is less influential and less frequent than predation impacts. Given that Manx shearwaters forage at considerable distances from colonies (Langston 2010 indicates a foraging range averaging 172 km), effective closures would need to be over considerable distances from Manx shearwater SPAs. This would appear to be much less practical and less effective than eradication of alien mammal predators. However, if closures to sandeel and sprat fisheries in UK waters were to be implemented, this would be likely to have a small net benefit to Manx shearwater productivity, and possibly also to survival rates.

9. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR NORTHERN GANNET

9.1 Northern gannet ecology

Northern gannet is a SPEC 2 species (Unfavourable conservation status (localised) and concentrated in Europe. The world population is around 390,000 breeding pairs (Mitchell et al. 2004), with most of these breeding in Europe apart from 77,700 pairs in six colonies in Canada.

9.1.1 The species in the British Isles

About 230,000 pairs breed in Great Britain, the Isle of Man and Channel Islands, and 33,000 pairs on Ireland (Mitchell et al. 2004). Gannets tend to breed in a small number of large colonies, and show reluctance to establish new colonies, though when new colonies are founded they tend to grow rapidly in size through immigration in the early years (Mitchell et al. 2004). The clutch size is a single egg. Gannets feed predominantly on pelagic fish, but can take sandeels when these are abundant, or larger fish such as adult herring and mackerel. In winter gannets feed extensively on fishery discards. Discards appear to be less frequent in the breeding season diet, although tracking data indicate that breeding adults will often feed behind trawlers even when rearing chicks (Votier et al. 2013).

The UK's SPA suite holds about 197,000 breeding pairs of gannets, representing around 98% of the UK breeding population (http://jncc.defra.gov.uk/pdf/ukspa/ukspa-a6-10.pdf). These are on ten SPAs (Ailsa Craig, Fair Isle, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Grassholm, Hermaness Saxa Vord and Valla Field, North Rona and Sula Sgeir, Noss, St Kilda, and Sule Skerry and Sule Stack). Adult survival rate is 0.92 (Wanless et al. 1996).

Gannets are believed to start breeding from age 5, although data on this are limited and come only from the Bass Rock colony (Nelson 1978, 2002). Non-breeding by adults that have nested before is thought to be very infrequent (Nelson 1978, 2002, WWT Consulting 2012). Gannets from colonies in the British Isles tend to winter from the southern North Sea to the continental shelf off West Africa, with younger birds travelling further south on average. Tracking suggests that adult gannets may be wintering on average further south now than they did a few decades ago, possibly in response to changes in fisheries in the North Sea and off West Africa (Kubetzki et al. 2009, Garthe et al. 2012).

Few threats to gannets have been identified, partly because their numbers have continued to increase up to the present. Gannets accumulate relatively high levels of pollutants compared to other European seabirds, but there is no evidence of toxic impacts, some gannets are killed by oil pollution, by entanglement in fragments of fishing net or on fishing lines, but none of these are thought to represent major threats. Bycatch in fisheries may be a greater problem than is currently recognised (BirdLife 2009), but there is inadequate data to assess this fully. The ability of gannets to swallow large fish as well as small ones, their extremely long foraging range, and their aggressive nature and large size giving them a dominant position in mixed-species feeding groups, makes them relatively insensitive to fluctuations in abundance of any particular fish species. Gannets from some other populations may visit waters around the British Isles. Norwegian gannets appear to winter further north than British gannets (Fort et al. 2012) and many of them winter in the North Sea, but their population is very small (3,850 pairs) compared to that of the British Isles (less than 2%). Icelandic gannets may winter west of the British Isles (WWT Consulting 2012), but their population (25,400 pairs) is relatively small compared to that of the British Isles (about 10%).

9.1.2 Present, and likely future trends

The 2003/04 national census of gannets found nearly 261,000 pairs of gannets in 24 colonies in Britain and Ireland, with three new colonies founded since the previous national census in 1994/95 (Wanless et al. 2005). The rate of increase between 1994/95 and 2003/04 (1.2% per annum) was considerably less than in previous decades, suggesting that the population may be approaching carrying capacity after 100 years of sustained and well-documented increase (Wanless et al. 2005). However, there is no evidence yet of any density-dependent decrease in gannet productivity (Mavor et al. 2005, 2008, WWT Consulting 2012). Changes in the Common Fisheries Policy, and especially the likely phased introduction over the next few years or decades of a zero discarding policy is likely to reduce food supply for gannets (Votier et al. 2013), which may be especially important for gannets in winter when availability of pelagic fish in British waters tends to be lower and gannets then feed more extensively on discards (Garthe et al. 1996). Whether this change will affect gannet numbers is uncertain. Gannets are the dominant scavenger behind fishing vessels and can swallow larger fish than any other UK seabirds (Hudson and Furness 1989), and have exceptionally long foraging range, so they are likely to be less affected than other scavenging seabirds. Changes in discarding may simply encourage a higher proportion of gannets to overwinter off west Africa rather than off Britain (Garthe et al. 2012). Most probably, future increases in gannet breeding numbers are likely to occur at a slower rate, and numbers may possibly stop increasing or possibly even decline.

9.1.3 Factors affecting survival rates

Only one study has investigated survival rates of northern gannets, and that analysis was based on ring recovery data (Wanless et al. 1996). Rather surprisingly, no studies have colour ringed gannets for survival analysis. Wanless et al. (1996) estimated adult survival rate to be 0.919 (0.915 to 0.922), 4th year survival 0.895 (0.889 to 0.900), 3rd year survival 0.891 (0.886 to 0.896), 2nd year survival 829 (0.821 to 0.836) and 1st year survival 0.424 (0.410 to 0.439). The data suggested a decline in survival rate in 1990 to 2000 compared to 1960 to 1990, but there were too few data to determine if this suggestion was a real change or an artefact of small sample size in the recent time period. The study found no clear evidence for differences in survival rates between colonies and did not investigate whether annual variations were caused by specific environmental factors (partly because the data set is not large or robust enough for more detailed analysis than was carried out). Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

9.1.4 Breeding success in the British Isles

In Shetland in 1988-90 when sandeel abundance fell to low levels, gannets switched from feeding primarily on sandeels while breeding to feeding on herring and mackerel, and maintained high breeding success (Walsh et al. 1991). Productivity at Ailsa Craig in 1993 was only 0.53, an exceptionally low value for an established colony, and this appeared to be caused by extremely cold weather (snow and ice) in May causing many birds to fail at the egg stage (Walsh et al. 1994). In 1994, lower than normal productivity at Troup Head (0.5 chicks per pair) was attributed to disturbance or predation at that colony (Walsh et al. 1995). Despite severe weather in 1997, there was no evidence that gannet productivity was adversely affected (Thompson et al. 1998). In 1999, mean productivity at monitored colonies was 0.66 chicks per pair, very slightly below the average for 1986-1998 (0.67), with many nests at Fair Isle washed away by a severe storm in May, and breeding success at Ailsa Craig reduced by disturbance by helicopters flying close over the colony (Upton et al. 2000). In 2000, at Hermaness the productivity was 0.57, the lowest on record there, possibly due to the severe storm in June 2000 (Mavor et al. 2001). However, there was no impact of this storm at Noss (0.73 chicks per pair) and the mean for 2000 for all monitored colonies (0.66) was close to the long-term mean (1986-1999, 0.69). A landslide at Hermaness destroyed several hundred gannet nest sites between 1999 and 2003, causing a reduction in breeding numbers at that colony in contrast to increases continuing elsewhere (Mavor et al. 2004). A new colony on Sule Skerry, which held 15 nests in mid-July 2003 lost the contents of 5 nests due to depredation by great black-backed gulls (Mavor et al. 2004). Mavor et al. (2008) commented that breeding success at newly formed gannet colonies tends to be slightly lower than at established colonies, because colonizing birds tend to be less experienced breeders than birds at established sites. Otherwise, breeding success tends to be consistently high (around 0.68-0.71) at all established colonies in all years, with only small variations between years and between sites. Mavor et al. (2005) stated that the northern gannet’s ability to travel hundreds of miles when foraging, coupled with a diet less reliant on sandeels, ensured that productivity remained high compared with other Shetland seabirds. In 2004, gannets at Shetland colonies fed mainly on mackerel (Mavor et al. 2005).

A licenced harvest of chicks at Sula Sgeir has been carried out traditionally each summer, with around 2000 gannet chicks killed there each year by the men of Ness in Lewis (Beatty 1992, Murray 2008).

Table 9.1.1 Meta-analysis of main factors contributing to reduced productivity of Gannets at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Storms	3
Great black-backed gull predation	1
Helicopter disturbance	1
Cold weather	1
Landslides	1
Human disturbance	1

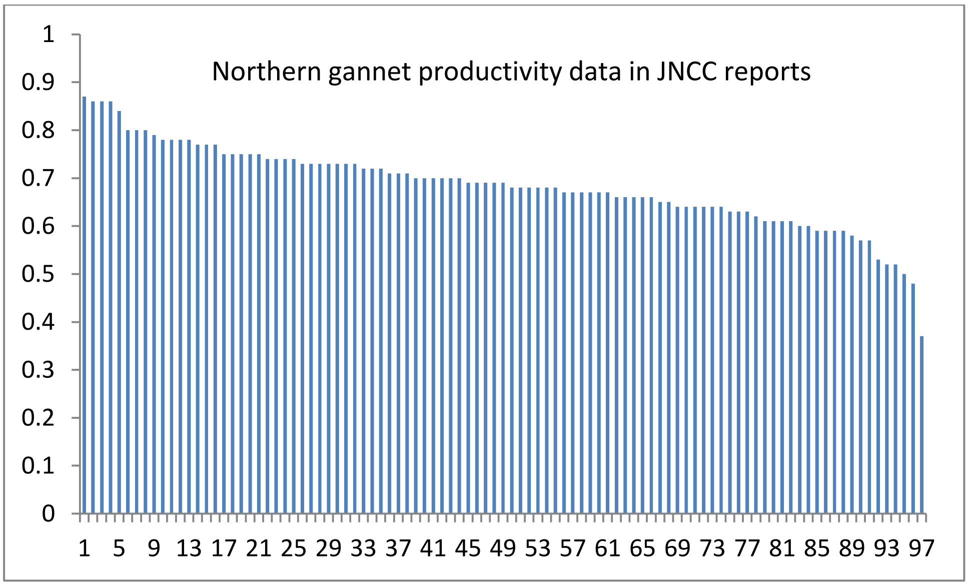

Figure 9.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of northern gannets at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

9.2 Management options

	9.2.2 End harvest of chicks	9.2.3 Encourage establishment of new colonies	9.2.4 Reduce bycatch in fisheries
Evidence of success for this species	High C=High*	Low C=Low	Low C=Low
Evidence of success for similar species	High C=High	Moderate C=Low	Low C=Low
Cost-effectiveness	High C=Low	Low C=Low	Low C=Low
Feasibility	Low C=High	Moderate C=Low	Low C=Low
Practicality	Low C=High	Low C=Low	Low C=Low
Applies at SPA populations	Yes C=High	No C=High	No C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

9.2.1 Management to increase survival rates

No management options have been identified that would have a strong probability of increasing adult and/or immature survival rates. One possibility might be reductions in fishing for herring and mackerel, which are important preferred food of gannets. Management measures that increased stock sizes of these fish species might increase overwinter survival and improve body condition of gannets, but there is no evidence available to test whether or not this would be the case, and it is possible that there would be no effect. The fact that rates of increase of gannet breeding numbers did not noticeably decline during the years when herring and mackerel stocks in UK waters were depleted by fishing in the 1960s-1980s, would tend to suggest that gannet survival rates are, at least at present and in past decades, not closely related to abundance of these fish stocks. There is, therefore, no clear evidence base to support management decisions to attempt to increase gannet survival rates.

9.2.2 End harvest of gannet chicks on Sula Sgeir

At almost all colonies, gannet productivity varies very little among colonies or among years, and is normally high. So there is almost no scope for increasing productivity of gannets. The one clear exception to this is Sula Sgeir, where there is a licenced harvest of around 2000 gannet chicks per year by ‘The men of Ness’. Gannet numbers on Sula Sgeir have increased less than at other colonies, indicating that this harvest has apparently affected the rate of colony growth. The harvest of 2000 chicks per year also probably affects productivity of unharvested nests through the human disturbance involved. Ending this harvest would increase productivity at that colony. However, it is unlikely that any other measures could significantly increase gannet productivity or survival at other sites. Figure 9.2.1 shows a plot of the increase in breeding numbers of gannets at different colonies in relation to the size of each colony at the 1969 census. It is clear from this graph that Sula Sgeir is an ‘outlier’. The rate of growth of that colony is much lower than the rate of growth predicted from the data from other gannet colonies. This reduced performance is most likely to be due to the lower productivity at Sula Sgeir and the associated disturbance of birds by the gannet harvest.

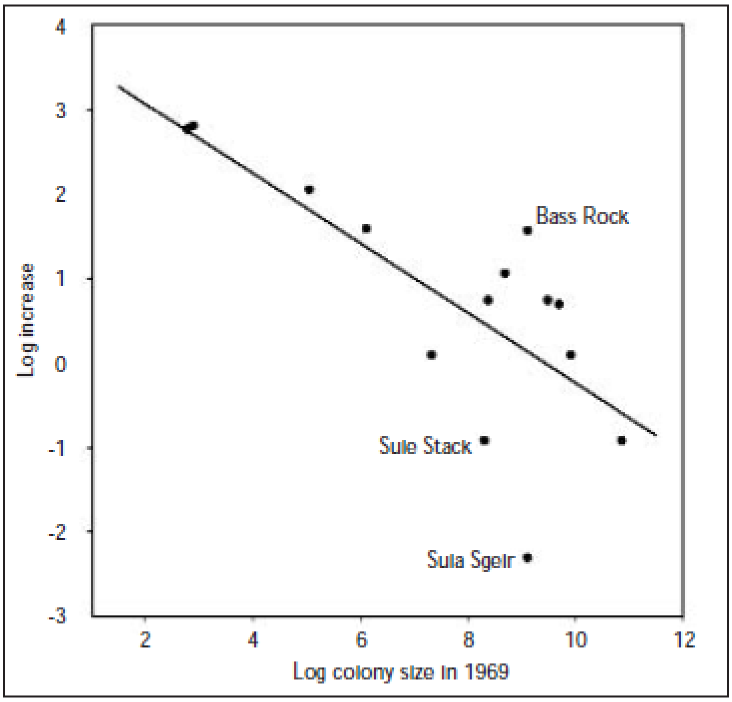

Figure 9.2.1. Rate of increase of gannet colony sizes in recent decades in relation to breeding numbers in 1969 (both axes plotted on log scales). The plot shows the colony at Sula Sgeir to be far below the expected rate of population growth compared to other colonies. From Wanless et al. 2005.

Ending the harvest of gannet chicks (gugas) at Sula Sgeir would increase productivity at that colony by at least 2000 chicks per year, and would be likely to result in more rapid growth of breeding numbers there. However, such a measure may not be acceptable for cultural reasons as this harvest is an important part of the local culture in north Lewis (Murray 2008).

9.2.3 Encourage establishment of new colonies

It might be possible to encourage gannets to form new colonies at locations where the species does not currently breed that are some distance from existing colonies. Birds would be likely to be able to exploit local fish resources more efficiently where they did not have to travel long distances from their colony to feeding areas, and where nesting numbers were smaller so reduced competition. Behavioural attraction methods developed in Maine have since been used globally to restore at least 49 species of seabirds on 89 islands in 14 countries (Jones et al. 2011). Jones and Kress (2011) suggest that a typical restoration project for a seabird in a developed country may cost around £500,000 per annum over a project lasting at least 5, possibly 10 years on average. They point out that the successful project restoring Atlantic puffins to Eastern Egg Rock in Maine took 35 years of sustained effort to establish a population of 100 pairs of puffins. An attempt to start a new Australasian gannet colony at Young Nick's Head, in New Zealand by social attraction was successful, but attempts using the same method with Australasian gannets at Mana Island, New Zealand, and with northern gannets in Nova Scotia and in Quebec failed (Jones and Kress 2011). So it is uncertain whether northern gannets could be encouraged to colonise new sites, and the cost of attempting to stimulate colonisation would be quite high.

9.2.4 Reduce bycatch in fisheries

There is too little data on bycatch rates to be able to assess whether reducing bycatch would significantly increase gannet survival rates, but it has been estimated, for example, that the Gran Sol fishery accidentally caught 1,331 gannets per year in 2006/07 (BirdLife 2009). If such large numbers are killed as bycatch in several fisheries in the wintering areas used by gannets, then reducing bycatch rates could represent a cost-effective compensation measure. There is also a lack of evidence regarding bycatch rates of gannets in EU fisheries, but the current belief is that numbers killed as fishery bycatch in EU waters are probably relatively small.

10. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR ARCTIC SKUA

10.1 Arctic skua ecology

The Arctic skua has a circumpolar breeding distribution, between 56°N and 82°N, on tundra and coastal moors. The world population is probably between 85,000 and 340,000 pairs, with 15,000 to 35,000 of these in the NE Atlantic (Mitchell et al. 2004).

10.1.1 The species in the British Isles

Seabird 2000 estimated 2,100 pairs breeding in Great Britain, with all of these in N and W Scotland, especially Orkney and Shetland (Mitchell et al. 2004). Arctic skuas nest on the ground in a shallow scrape, mostly on moorland close to colonies of terns, kittiwakes and/or auks, from which they steal fish. Most pairs lay two eggs, and the mean clutch size is 1.8 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). The UK SPA suite includes 780 pairs of Arctic skuas (24% of the British population) nesting on 7 sites; these are Fair Isle, Fetlar, Foula, Hoy, Papa Westray, Rousay, and West Westray (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-79.pdf). Adult survival rate (in the absence of illegal shooting) has been estimated at 0.84 (O'Donald 1983, del Hoyo et al. 1992-2006), 0.89 (Furness 1987), and 0.90 (Phillips and Furness 1998), and the mean age of first breeding is 4.5 years (O'Donald 1983). British Arctic skuas migrate to spend the winter in the South Atlantic; none remain in British waters overwinter (Wernham et al. 2002). Threats to British Arctic skuas include depletion of sandeel stocks, climate change (as this species is at its southerly breeding limit in Scotland), territorial conflicts with and depredation by great skuas (Forrester et al. 2007). Migrations of Arctic skuas from higher latitudes (Scandinavia, Iceland, Faroes, the Arctic tundra) brings birds past the British Isles in autumn (when moderate numbers linger in the North Sea and elsewhere during migration, stealing fish from terns and small gulls in particular) and in spring (the latter migration tending to be rapid and off the west of the British Isles) (Forrester et al. 2007).

10.1.2 Present, and likely future trends

Arctic skua breeding distribution in Britain, concentrated mainly in Shetland, Orkney and the Western Isles, has hardly changed over the last 150 years (Mitchell et al. 2004). Breeding numbers increased somewhat between 1969 and 1988, but the trend in recent years has been for large declines throughout the breeding range (Figure 10.1.1), but especially in Shetland and Argyll. The Arctic skua consequently moved from the 'Green' list directly to the 'Red' list, as one of the UK's most rapidly declining breeding birds. There is little suggestion of any population recovery, though since some birds in recent years have refrained from breeding, the decrease in total population size may be slightly less than the decrease in breeding numbers, and some nonbreeding birds might return if conditions improve in future. However, the prospects for this species do not appear to be good. While the declines in Shetland and Orkney are clearly related to reductions in sandeel abundance, declines in Argyll may possibly be due to climate change, as there is no evidence of decreases in small pelagic fish stocks in Argyll waters and breeding numbers of seabirds from which Arctic skuas steal food have not declined in Argyll (ap Rheinallt et al. 2007). However, Arctic skua has been identified as one of the seabird species most likely to be adversely affected in the British Isles by climate change (Oswald et al. 2011). Foster and Marrs (2012) estimated that Arctic skua breeding numbers in Scotland declined by 74% from 1986 to 2011. There is no reason to expect the breeding population of Arctic skuas in Britain to recover in the foreseeable future.

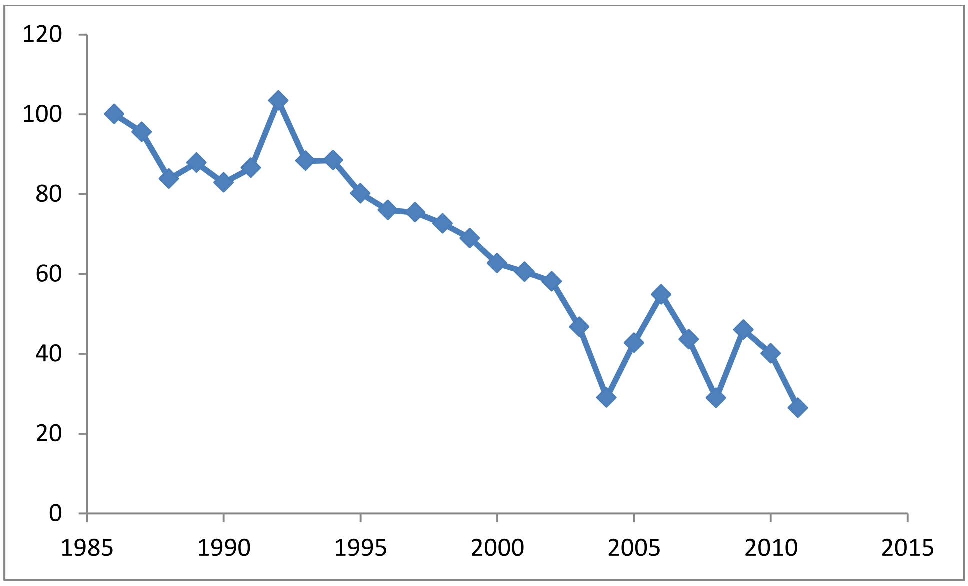

Figure 10.1.1. Arctic skua breeding population index for Scotland. Data from JNCC online database.

10.1.3 Factors affecting survival rates

On Fair Isle, annual survival rates of breeding adult Arctic skuas averaged 0.801 in 1948-62, 0.886 in 1973-75, and 0.747 from 1976-78 (O'Donald 1983). The high rate in 1973-75 was attributed to the colony being studied intensively by scientists so that probably no illegal shooting of adults took place during that period. In contrast, in 1976-78 scientists were only occasionally present and many adults were found shot dead on the island. During 1948-62 it is thought that there was some illegal shooting, but less than in 1976-78 (O'Donald 1983). Shooting clearly affected survival rates of this species on Fair Isle, but shooting of Arctic skuas appears to be very much less common at other colonies; Fair Isle is somewhat exceptional in this regard (Furness 1987). However, survival studies at other colonies have been much less complete than on Fair Isle so survival rates at less disturbed colonies are uncertain. On Foula, Shetland, survival rates of colour ringed adults were reported by Phillips et al. (1998) as 0.883 between 1992 and 1995. Survival rates of pale phase adults were 0.917 in 1993-94 and 0.902 in 1994-95, and of dark phase birds were 0.890 in 1993-94 and 0.906 in 1994-95 (Phillips and Furness 1998a). Davis et al. (2005) carried out an experiment to test the hypothesis that low breeding success of Arctic skuas at Foula, Shetland was due to food shortage (specifically to low abundance of the Shetland sandeel stock). They provided supplementary food to a sample of breeding Arctic skuas within the colony and used other pairs as controls. Supplementary fed pairs not only achieved higher productivity, but adults also spent significantly less time away from the territory searching for food, retained some of the supplementary food themselves rather than feeding it all to their chicks (demonstrated by measurement of body condition and of stable isotopes which differed between the natural and supplementary foods), and showed a significantly higher return rate to the colony the next season than did controls. The study concluded that poor food supply not only reduced productivity, but also reduced adult survival rate. This effect of food supply on adult survival is also consistent with analysis of kittiwake adult survival rates in relation to sandeel abundance (Oro and Furness 2002) or presence of a sandeel commercial fishery (Frederiksen et al. 2004).

10.1.4 Breeding success in the British Isles

In 1988-90 in Shetland, many Arctic skua colonies fledged no young, because they were unable to find adequate amounts of food (sandeels) (Walsh et al. 1991). All monitored colonies in Shetland and most in Orkney showed improved breeding success in 1991 compared to 1986-90, apparently related to high recruitment of sandeels in 1991 (Walsh et al. 1992). On Westray in 1991, many pairs failed at the egg stage apparently due to sheep or large gulls (Walsh et al. 1992). Most monitored colonies also showed higher productivity in 1992 than in 1988-90, apparently reflecting increased sandeel stock biomass since 1991 (Walsh et al. 1993). Slight reductions in productivity were caused by predation and disturbance: on Mousa predation was by otters, on Noss disturbance from tourists and neighbouring Great skuas reduced hatching success, Great skuas caused post-fledging mortality of Arctic skuas on Foula, Fair Isle and six colonies on mainland Shetland, Arctic skua chicks were killed pre-fledging on Hoy, and predation of eggs by sheep and common gulls occurred on Westray (Walsh et al. 1993). The main factor affecting Arctic skua productivity in 1993 (which averaged 1 chick per pair at Shetland colonies) was reported to be predation from great skuas and large gulls (Walsh et al. 1994). In 1995, productivity at Shetland colonies varied between 0.9 and 1.21 chicks per pair, the only factor noted to adversely affect productivity at any of these colonies being wet weather during incubation on Foula (Thompson et al. 1996). Low food availability in Shetland was considered to be the main cause of reduced productivity there in 1997, which also resulted in low attendance by adults (Thompson et al. 1998). Low food availability was considered to affect productivity in 1998 in Shetland, but many surviving chicks were killed either before or after fledging by Great skuas (Thompson et al. 1999). In 1999 in Shetland, productivity (0.46 chicks per pair) was apparently reduced by poor food availability, and chicks and fledglings were killed by Great skuas, whereas in Orkney a good food supply led to higher productivity (0.83 chicks per pair) (Upton et al. 2000). In 2000 in Shetland, productivity averaged 0.57 chicks per pair, this low value being attributed primarily to low abundance of sandeels but also some predation of chicks and adults by Great skuas, and some losses of clutches to the severe storm in mid-June (Mavor et al. 2001). In 2001, Arctic skua productivity was the lowest yet recorded during the monitoring programme, due to scarcity of sandeels in Shetland waters (Mavor et al. 2002). In 2002, lack of sandeels around Shetland greatly reduced productivity (0.18 chicks per pair), whereas food availability at Orkney was considered to be good, and productivity was moderately high (0.74 chicks per pair) (Mavor et al. 2003). In 2003, scarcity of sandeels around Shetland resulted in long-term decline in Arctic skua breeding numbers, extensive nonbreeding, late laying, and breeding success much below levels seen before the collapse of the Shetland sandeel stock (Mavor et al. 2004). The sandeel shortage that affected Arctic skua breeding success on Shetland in 2001-2003 recurred in 2004 and resulted in the lowest productivity since the monitoring programme began in 1986 (Mavor et al. 2005). In addition to adults abandoning eggs and chicks starving, predation by great skuas was also intense. On Handa in 2003, breeding success was high and food availability was apparently high throughout the season, but predation of fledglings by great skuas resulted in 50 to 60% of fledglings being killed (Mavor et al. 2004). Breeding success in Orkney in 2004 was also affected by food shortage, but this was not the case in NW Scotland (Mavor et al. 2005). Mavor et al. (2008) described the 2005 and 2006 breeding seasons for Arctic skuas as follows. The 2005 breeding season was poor, with few young fledged. Food appeared to be scarce. Only two chicks hatched on Foula. On Fair Isle, depredation by great skuas and low food availability resulted in only five young fledging from 71 territories. In Orkney, food availability appeared to be low, some pairs did not attempt to breed, and productivity averaged only 0.3 chicks per pair. On Handa in 2005, productivity was reduced below normal levels by unidentified predators taking eggs, and by great skuas taking Arctic skua chicks before fledging. Post-fledging mortality of Arctic skuas on Handa (killed by great skuas) was reported to be lower than normal in 2005, but still around 40%. In 2006, food was scarce around Foula. Productivity on Fair Isle was higher than in 2005, but post-fledging mortality was high as great skuas were seen to kill fledglings on several territories. In Orkney, breeding success was higher than in 2005, but food shortage in 2006 reduced chick survival. On Coll, heavy depredation by great skuas and great black-backed gulls reduced fledging success.

Davis et al. (2005) carried out an experiment to test the hypothesis that low breeding success of Arctic skuas at Foula, Shetland was due to food shortage (specifically to low abundance of the Shetland sandeel stock). They provided supplementary food to a sample of breeding Arctic skuas within the colony and used other pairs as controls. Supplementary fed pairs achieved significantly higher productivity, partly because the supplementary food allowed chicks to grow better, but also because the supplementary food allowed adults to spend more time guarding their chicks so reduced the numbers lost to predators. This work therefore demonstrates not only that productivity was limited by food supply but also that predation rates interact with food supply such that losses to predators increase when birds are struggling to find food.

Table 10.1.1 Meta-analysis of main factors contributing to reduced productivity of Arctic skuas at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	37
Great skua predation	19
Gull predation	4
Sheep	2
Wet weather	2
Human disturbance	1
Otter predation	1

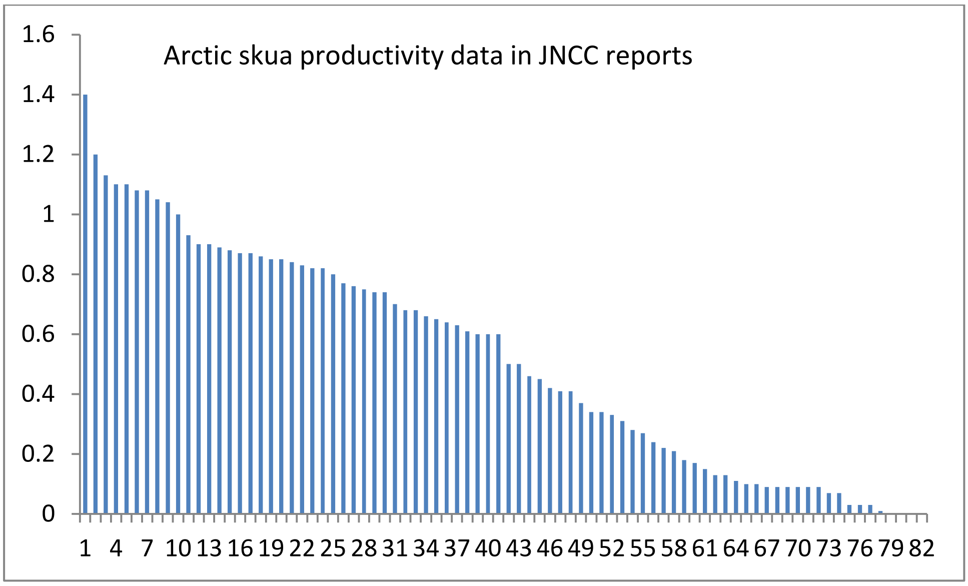

Figure 10.1.2. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Arctic skuas at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

10.2 Management options

	10.2.1 Closure of sandeel and sprat fisheries close to breeding area SPAs	10.2.2 Provision of supplementary food to breeding pairs	10.2.3 Exclusion of great skuas from buffer zone around colonies
Evidence of success for this species	Low C=High*	High C=High	Moderate C=Mod
Evidence of success for similar species	High C=Mod	High C=High	Moderate C=Mod
Cost-effectiveness	Uncertain C=Low	High C=High	High C=Mod
Feasibility	Moderate C=Low	High C=High	Low C=Low
Practicality	Moderate C=Low	Moderate C=High	Moderate C=Low
Applies at SPA populations	Yes C=High	Yes C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

10.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs

There is very strong evidence that Arctic skua productivity and survival are greatly affected by prey fish abundance around colonies. Arctic skua foraging mostly occurs within 28 km of colonies but maximum foraging range may be as much as 100 km. Closure of fishing for sandeels and sprats within 50 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

The SPAs for Arctic skua are in Shetland (Fair Isle, Fetlar, Foula) and Orkney (Hoy, Papa Westray, Rousay, and West Westray). These are also the regions around the British Isles where sandeel stock declines have been most pronounced in recent decades. Sandeel stock biomass in Shetland waters has never recovered fully since the local sandeel fishery at Shetland was closed in 1990 due to the depleted state of that sandeel stock. However, the extent to which declines in sandeel stocks at Orkney can be attributed to fishing pressures is unclear. There has been no fishery targeting sandeels in Orkney waters, but connectivity between sandeel populations in Orkney and elsewhere in the North Sea is uncertain. It is possible that changes in sandeel abundance have been influenced more by climate change effects or predation impacts on sandeel larvae from increased stocks of adult herring in the northern North Sea (Frederiksen et al. 2007).

10.2.2 Provision of supplementary food to breeding pairs

There is strong evidence that Arctic skua productivity and survival are greatly affected by food supply. Supplementary feeding (for example on cat food, hen's eggs or day-old chicks) increases productivity and survival. Given that Arctic skua colonies are relatively small it would be possible to provide supplementary food to boost productivity and survival of Arctic skuas, in a similar way to the established practice of supplementary feeding of hen harriers to reduce their predatory impact on red grouse.

Breeding adult Arctic skuas in the UK weigh around 430g on average (Phillips and Furness 1997). Based on the allometric equation relating field metabolic rate to body mass of Charadriiformes (Ellis and Gabrielsen 2002), an adult Arctic skua has a field metabolic rate of 894 kJ/day. Assuming a food utilisation efficiency of 80% (Hilton et al. 2000), this means a food intake of 1,118 kJ/day. Each pair of Arctic skuas therefore requires at least 2,236 kJ per day from its food, and slightly more than this if feeding chicks. Davis et al. (2005) provided 625 kJ per day to experimental pairs, and compared to control pairs given no extra food, this increased productivity from 0.52 ±0.1 to 0.89±0.11, and increased apparent adult survival rate (measured as return rate the next season) from 0.725 to 0.900. Supplementary food provided chicks with 20% (range 5 to 40%) of their dietary protein (Davis et al. 2005). This study indicates that providing pairs of Arctic skuas with supplementary food increases productivity and survival. Provisioning Arctic skuas from around the time they start to lay eggs (May) until chicks fledge (July) would be a simple mamagement measure, since most Arctic skua colonies are small (tens of breeding pairs, even in the SPA populations – see Table 10.2.1) and most are on fairly accessible moorland terrain. There would be a need, however, to establish the best food to provide. Davis et al. (2005) provided cat food and hens' eggs, but day-old chicks or fish might be suitable options. Most Arctic skua pairs do not readily adapt to taking supplementary food, but require a "training period" during which they need to be provided with the supplementary food (usually cat food) alongside a broken hen's egg (broken because they have difficulty in breaking the shell themselves). Once trained, individuals will then accept the supplementary food alone. Davis et al. (2005) mention a few other considerations, including the need to avoid attracting other scavengers to the food (for example by feeding skuas in the evening when their attendance on the territory is high), and placing food close to the centre of each territory to avoid disputes between neighbouring pairs. Such care is critically important since uneaten food could attract large scavengers such as great skuas, great black-backed gulls or ravens and these could have strong negative effects on Arctic skua productivity. All of the SPA populations (Table 10.2.1) are on inhabited islands. Fair Isle has a Bird Observatory employing seasonal staff, Fair Isle and Foula have employed rangers on the islands, Fetlar, Papa Westray and Hoy have RSPB reserves with staff. So it would probably be fairly easy to establish a supplementary feeding regime at any of these SPAs. Supplementary feeding would require about half a day of the time of one person each day from early May to mid-July (a minimum of about 80 days). For benefits of increased chick survival to be fully realised, it might also be necessary at some colonies to consider whether control of great skuas may be necessary to avoid fledgling Arctic skuas being killed by neighbouring great skuas (see section 10.2.3). Supplementary feeding of Arctic skuas would not raise such major issues of public disquiet as supplementary feeding of great skuas. Arctic skuas are generally liked and their numbers are relatively small so the cost of supplementary food for these birds would be very considerably less than for populations of great skuas. Nevertheless, supplementary feeding may not be popular with the general public so the issue would need to be treated sensitively. For that reason, and because care would need to be taken to avoid food attracting larger scavengers, practicality is classified as Moderate.

Table 10.2.1. Most recent published counts of Arctic skua numbers in SPA populations

Colony	Pairs	Year	Great skua pairs adjacent to Arctic skuas out of whole colony	Reference
Fetlar*	5	2011	Tens of pairs out of ca. 400 pairs	Shetland Bird Club (2012)
Foula*	41	2011	Tens of pairs out of ca. 1,800 pairs	Shetland Bird Club (2012)
Fair Isle*	70	2010	Tens of pairs out of 280 pairs	Shaw (2012)
Papa Westray	44	2010	About half of the 29 pairs	Meek et al. (2011)
Westray	27	2010	About half of the 19 pairs	Meek et al. (2011)
Rousay	37	2010	Tens of pairs out of 85 pairs	Meek et al. (2011)
Hoy*	16	2010	Tens of pairs out of 1,346 pairs	Meek et al. (2011)

*These islands are also designated as SPAs for the great skua

10.2.3 Exclude great skuas from buffer zone around Arctic skua colonies

The main predation factor reducing Arctic skua productivity (and reducing adult survival) is predation by neighbouring great skuas. Not only is this recognised in the JNCC data meta-analysis (Table 10.1.1), but it has been reported in several papers. Furness (1977) observed the spread of great skua colony edge into Arctic skua territory on Foula in the 1970s, when great skuas killed many adult Arctic skuas during conflicts over territory ownership. More recently, with reduced food availability, great skuas have killed not only a high proportion of Arctic skua chicks, but also a very high proportion of recently fledged Arctic skuas. Predation by great skuas tends to occur especially where great skuas hold territories on the edge of Arctic skua territory (Phillips et al. 1998), so that they can very easily move in to kill any unattended chicks, and can attack fledglings as they make their first and relatively inept flights (Phillips et al. 1998, Mavor et al. 2008). Estimates vary, but in many cases great skuas kill all surviving Arctic skua chicks either before or after fledging, and even when food supplies for both species are good, great skuas kill around 10 to 20% of Arctic skua fledglings (Furness 1987).

Establishing a buffer zone around Arctic skua colonies from which great skuas are prevented from establishing breeding territories, would reduce the predation impact of great skuas on Arctic skuas. All Arctic skua SPAs are on islands where there are also great skuas nesting, but only relatively small numbers of great skuas nest adjacent to Arctic skua territories (Table 10.2.1) because great skuas prefer areas with less human activity than Arctic skuas will tolerate, and prefer areas with taller vegetation than Arctic skuas prefer (Furness 1987). Once their territory is established, removal of great skuas could only be achieved by killing the adults or by trapping and removing the birds into permanent captivity, since released birds will return to their territory. Since Fair Isle, Fetlar, Foula, and Hoy are also SPAs for great skuas, it might be particularly difficult to manage their breeding distributions on those islands. Westray and Papa Westray are SPAs for Arctic skua but have only recently been colonised by great skuas and removal of great skuas from those islands might be considered as a more acceptable management strategy to protect Arctic skuas. Although Rousay is an SPA for Arctic skua but not for great skua, it has a colony of great skuas that has increased to 85 pairs (there were only 13 in 1982 when there were 96 pairs of Arctic skuas there, Meek et al. (2011)).

11. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR GREAT SKUA

11.1 Great skua ecology

Great skua is classified as a SPEC 4 species; Favourable conservation status (secure) but concentrated in Europe. The world population of the species totals around 16,000 breeding pairs, with all of these within Europe (Mitchell et al. 2004). Numbers have increased during the 20th century, and the species has colonised new regions, including Norway, north Russia, Svalbard, Bear Island and Jan Mayen (Mitchell et al. 2004).

11.1.1 The species in the British Isles

Seabird 2000 estimated that 9,600 pairs bred in Great Britain (all in N and W Scotland) (Mitchell et al. 2004). Great skuas nest on moorland and rough grassland mostly in upland areas away from human habitation. Nests are depressions in the vegetation with minimal lining material. Most pairs lay a clutch of two eggs, the mean clutch size being 1.8 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Great skuas catch sandeels when these come to the sea surface, steal some from other seabirds, scavenge discards from fishing boats, and attack and kill a wide range of seabirds, terrestrial birds and mammals up to the size of swans and lambs (Furness 1987). The UK SPA suite includes about 6,300 pairs of great skuas (74% of the British population) on 9 sites: Fair Isle, Fetlar, Foula, Handa, Hermaness Saxa Vord and Valla Field, Hoy, Noss, Ronas Hill – North Roe and Tingon, and St Kilda (http://jncc.defra.gov.uk/pdf/ukspa/ukspa-a6-80.pdf). The adult survival rate in Shetland has averaged 0.9 (Ratcliffe et al. 2002) but appears to have declined as sandeel abundance fell and birds worked harder to try to rear chicks. The mean age of first breeding is around 6 years old (Furness 1987). British great skuas migrate to spend the winter over the continental shelf seas of southern Europe or West Africa (Magnusdottir et al. 2012). The main threats to British great skuas appear to be lack of sandeels and reductions in fishery discarding in the seas around their main colonies (Forrester et al. 2007). Great skuas have high pollutant burdens compared to most European seabirds but toxic impacts are not detectable (Bourgeon et al. 2012). Great skuas from Iceland use two wintering areas, one off Canada and one in southern Europe. Birds migrating to the former do not come into British waters, while birds wintering in the latter area migrate through British waters and mix to some extent in winter with British great skuas although on average Icelandic birds appear to winter slightly further north than those from British colonies (Magnusdottir et al. 2012). Given the slightly smaller population size in Iceland, Faroes and Norway than in Britain, and the fact that about half of the Icelandic birds migrate to Canada, it is likely that ‘foreign’ birds represent less than 25% of those seen migrating through British waters in autumn or spring.

11.1.2 Present, and likely future trends

Breeding numbers of great skuas increased from 1900 up to about 2000 (Mitchell et al. 2004), but while numbers may have continued to increase at small colonies, in recent years the numbers at the largest colonies have declined. At Foula (Shetland), there were 2,495 pairs in 1985-86 but only 2,293 in 2000 (Mitchell et al. 2004), and numbers there have fallen further since then. On Unst (Shetland) there were 1,569 pairs in 1992 but 1,385 in 2000 (Mitchell et al. 2004). At Hoy (Orkney), there were 1,973 pairs in 2000 but 1,346 pairs in 2010 (Meek et al. 2011). These changes have been described as a density-dependent response to a combination of reduced food supply and increased predation (Meek et al. 2011). Continued scarcity of sandeels in the northern North Sea and likely further reductions in fishery discarding culminating in a probable discard ban in the future (Votier et al. 2013), is likely to result in some further decrease in great skua breeding numbers, especially at the larger colonies where competition for food is highest among great skuas (Votier et al. 2007, 2008), since great skuas feed extensively on discards not only as nonbreeders but also throughout the breeding season, and their breeding success is also correlated with sandeel stock biomass (Votier et al. 2004). These larger colonies also tend to be SPAs for the species, so breeding numbers are likely to decrease particularly strongly at colonies designated as SPAs for great skuas.

11.1.3 Factors affecting survival rates

Based on ring recovery data from British great skua colonies, Furness (1978) estimated adult great skua survival of 0.93 between 1938 and 1974, a period of continuous growth of great skua numbers and typically high breeding success (Furness 1987). Based on observations of individually colour ringed birds, annual survival rates of breeding adult great skuas at Foula, Shetland, varied between 0.93 and 0.82, averaging 0.89 over a period of years (1989-1999) when numbers in the colony were declining and breeding success was low (Ratcliffe et al. 2002). Annual survival rate correlated with Shetland sandeel stock biomass, indicating that this was a major environmental driver of survival in this population. However, since most mortality appeared to occur in winter, Ratcliffe et al. (2002) suggested that nutritional stress and reproductive effort breeding in years of poor food supply affect survivorship on migration or in wintering quarters. The results obtained by Ratcliffe et al. (2002) suggest that adult survival rate may typically be 0.93 when food is abundant, but that the survival rate may fall by 0.11 under conditions of food shortage during the breeding season. Great skuas can be killed as bycatch on long line fisheries and through entanglement in fishing nets, but the magnitude of fishery bycatch is unknown. There are suggestions that bycatch in fisheries in wintering areas used by great skuas (especially off West Africa) might be higher than has been recognised. However, there is a need to quantify this bycatch before it would be possible to assess whether reducing bycatch would represent a cost-effective compensation measure.

11.1.4 Breeding success in the British Isles

In 1991, Great skua productivity in Shetland (0.69 chicks per pair) on Fair Isle (0.7) and on Orkney (0.8) was higher than in 1988-90 apparently a consequence of high recruitment of sandeels in 1991, with this improved food supply leading to higher attendance of chicks by adults so higher chick survival (Walsh et al. 1992). In 1993, productivity was generally good in Shetland (around 0.8 chicks per pair), but reduced by ‘cannibalism’ in Orkney (0.4 chicks per pair) with some chicks being killed by neighbouring adults (Walsh et al. 1994). In 1997, adult attendance was unusually low (compared with previous years) suggesting a shortage of food, and there was considerable killing of chicks by neighbouring adults, resulting in low productivity at most Shetland colonies (Thompson et al. 1998). In 1998 in Shetland, productivity averaged 0.6 chicks per pair, with predation, poor weather and food shortage all contributing to reduction in productivity (Thompson et al. 1999). In 1999, productivity averaged 0.71 chicks per pair in Shetland and 0.51 in Orkney. Reasons for losses included poor weather, reduced food availability, and predation by conspecifics (Upton et al. 2000). In 2000, productivity in Shetland averaged 0.8 chicks per pair, with most losses attributed to low abundance of sandeels and some consequent ‘cannibalism’ of chicks. In 2001, productivity at monitored sites was the lowest since the programme began in 1986 with the exception of 1998; however, low availability of sandeels at Shetland affected great skuas less than Arctic skuas (Mavor et al. 2002). It is believed that low availability of fishery discards contributed to poor productivity in 2001, and also led to some chicks being killed and eaten by neighbouring adults (Mavor et al. 2002). Poor food supply depressed breeding success at the main Shetland colonies in 2003, whereas productivity was above average in Orkney (Mavor et al. 2004). Many great skua chicks at Shetland colonies were killed and eaten by neighbouring adult great skuas, although this predation was evidently a consequence of food scarcity (Mavor et al. 2004). The lack of sandeels in 2004 resulted in very low productivity in Shetland and Orkney, but there was no evidence of food shortage in NW Scotland (Mavor et al. 2005). In 2005, great skua productivity at Handa was lower than normal, and analysis of regurgitated pellets indicated less fish than normal (with over 40% of pellets consisting of bird remains) (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for great skuas as follows. Productivity was higher than in recent years, though still reduced as a consequence of food shortage. In addition, killing of great skua chicks by adult great skuas from neighbouring territories reduced breeding success further.

Table 11.1.1 Meta-analysis of main factors contributing to reduced productivity of Great skuas at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Food shortage	25
Great skua predation of chicks	9
Wet weather	2

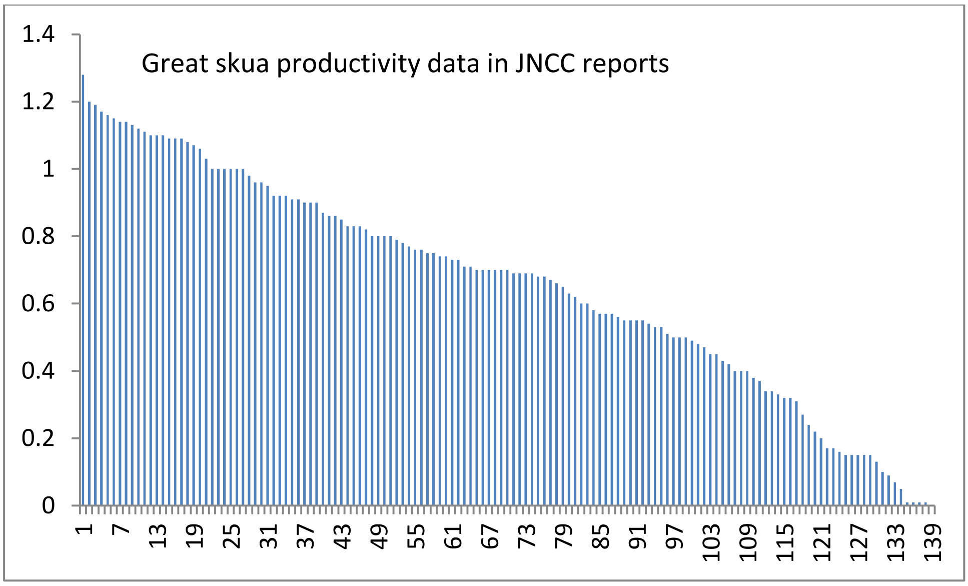

Figure 11.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of great skuas at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

11.2 Management options

	11.2.1 Closure of sandeel and sprat fisheries close to colonies	11.2.2 Supplementary feeding at colonies	11.2.3 Reduce fishery bycatch
Evidence of success for this species	Low C=Low*	High C=High	Low C=Low
Evidence of success for similar species	High C=Low	High C=High	Low C=Low
Cost-effectiveness	Uncertain C=Low	Low C=High	Low C=Low
Feasibility	Moderate C=Low	High C=High	Low C=Low
Practicality	Moderate C=Low	Low C=High	Low C=Low
Applies at SPA populations	Yes C=High	Yes C=High	No C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

11.2.1 *Closure of sandeel and sprat fishing close to breeding area SPAs*

There is strong evidence that great skua productivity and survival are greatly affected by prey fish abundance around colonies. Great skua foraging mostly occurs within 36 km of colonies but maximum foraging range may be as much as 100 km or more. Closure of fishing for sandeels and sprats within 50 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. Furthermore, great skua SPAs are almost all in areas where there are currently no sandeel or sprat fisheries (six in Shetland - Hermaness, Foula, Fetlar, Ronas Hill, Noss, Fair Isle; Hoy in Orkney, St Kilda in the Western Isles, and Handa in NW Scotland). Discards from trawl fisheries are also important food for great skuas, but it would be nonsense to encourage continued discarding to support populations of scavenging seabirds that are being sustained at artificially high levels by the subsidy of discards they have been receiving for many decades (Votier et al. 2004, 2007, 2008, 2013).

11.2.2 *Supplementary feeding of birds in SPAs*

Great skuas are scavengers, and will take a very wide variety of foods on an opportunistic basis. Supplementary feeding of breeding pairs of great skuas with cat food has been carried out on an experimental basis to test ideas about their ecology and life history (e.g. Ratcliffe unpubl. PhD. Thesis, University of Glasgow; Kalmbach unpubl. PhD. Thesis, University of Glasgow; Hammer pers. comm.). Great skuas are quick to learn of new feeding opportunities (Furness 1987). It would be relatively easy to feed great skuas in SPAs with supplementary food. However, the relatively large colony sizes (1,000 to 2,000 breeding pairs at Foula, Hoy, Hermaness for example) would require considerable quantities of food to be provided to increase breeding success and survival rates. Typically, a great skua has a field metabolic rate of around 2000 kJ/day (Ellis and Gabrielsen 2002). With a food utilisation efficiency of around 0.8 this represents about a pair of breeding great skuas will consume around 1,000 g of food per day. So a colony of 1,000 pairs needs around 1 tonne of food per day, from early May to late July (about 80 days). Providing even half of this requirement as supplementary food would be logistically challenging and expensive. There would be strong public opposition to feeding great skuas in many parts of the species' breeding range, as this species is highly unpopular in areas such as Orkney and Shetland. Furthermore, the general public throughout the UK would be likely to question whether spending money on food for scavenging seabirds would be acceptable. So the Practicality of this measure is scored Low despite the evidence for it being a measure likely to succeed in the objective of increasing survival and productivity.

11.2.3 Reduce bycatch in fisheries

There is too little data on bycatch rates to be able to assess whether reducing bycatch would significantly increase great skua survival rates. If large numbers are killed as bycatch in several fisheries in the wintering areas used by great skuas, then reducing bycatch rates could represent a cost-effective compensation measure. There is also a lack of evidence regarding bycatch rates of great skuas in EU fisheries, but the current belief is that numbers killed as fishery bycatch in EU waters are probably relatively small.

12. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR LESSER BLACK-BACKED GULL

12.1 Lesser black-backed gull ecology

The lesser black-backed gull is classified as SPEC 4 Favourable conservation status (secure) but concentrated in Europe. The total breeding population of the species is around 300,000 pairs, with 179,000 pairs in the subspecies '*graellsii*' which occurs from Greenland to Portugal (Mitchell et al. 2004).

12.1.1 The species in the British Isles

Around 117,000 breeding pairs of lesser black-backed gulls were recorded in the Seabird 2000 survey in Great Britain, the Isle of Man and Channel Islands, and 4,800 pairs in Ireland (Mitchell et al. 2004). Lesser black-backed gulls nest on the ground in colonies that tend to be of moderate or large numbers. Colonies are often on islands or in sand dunes, but can be on moorland some distance from the coast. Clutches of 2 or 3 eggs predominate, the mean clutch size being 2.6 eggs (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Lesser black-backed gulls tend to have a more marine diet than herring gulls, feeding to a greater extent on small fish, and less on terrestrial or intertidal foods (Kim and Monaghan 2006). There is an urban-nesting population of this species which has been growing faster than the population using natural breeding sites, but numbers of urban-nesting lesser black-backed gulls are much smaller than numbers of urban-nesting herring gulls and represent a very small fraction of the total population of the species (Raven and Coulson 1997).

The SPA suite in the UK supports bout 88,600 pairs, which is 'virtually the whole UK population' (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-84.pdf). These breed on ten SPAs: Ailsa Craig, Alde-Ore Estuary, Bowland Fells, Firth of Forth Islands, Isles of Scilly, Lough Neagh and Lough Beg, Morecambe Bay, Rathlin Island, Ribble and Alt Estuaries, and Skomer and Skokholm.

The adult survival rate has been estimated at 0.91 (Wanless et al. 1996) and 0.90 (Poole et al. 1998), and the mean age of first breeding is 4 years old (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). British lesser black-backed gulls mostly migrate to spend the winter in north Africa, but increasing numbers (though still a small minority) remain close to breeding areas overwinter. Few 'foreign' lesser black-backed gulls seem to migrate through British waters. Some Icelandic birds may do so, but there is very little evidence of that from the ringing that has been done in Iceland. Continental birds of the subspecies 'intermedius' (from southern Scandinavia and Netherlands) are very infrequently seen in the British Isles, while birds of the subspecies 'fuscus' which breed in northern Scandinavia migrate southeastwards to the Middle East and East Africa, and do not normally pass through British waters (Forrester et al. 2007).

12.1.2 Present, and likely future trends

Lesser black-backed gull breeding numbers in Britain and Ireland increased considerably from 1900 to 2000, reaching 116,684 pairs in the Seabird 2000 census with 57% in England the Isle of Man and Channel Islands, 21% in Scotland, 18% in Wales and 4% in Ireland (Mitchell et al. 2004). This increase was initially triggered by protective legislation and reduced exploitation, but subsequently also encouraged by increased feeding opportunities from fishery discards and edible waste at landfill sites (Mitchell et al. 2004). Since reaching a peak around the mid-1990s, breeding numbers have fallen slightly (Figure 12.1.1). Reasons for the recent decline are thought to include culling (between 1999 and 2002 some 29,000 gulls mostly lesser black-backed gulls were culled at Tarnbrook Fell alone (Mitchell et al. 2004)), reduced food availability from changes in refuse disposal and reduced discarding by fisheries, predation, competition from other large seabirds for food and nest sites, and habitat changes (Mitchell et al. 2004, Forrester et al. 2007). Figure 12.1.1 suggests that breeding numbers are currently (i.e. since about 2005) approximately stable in the UK as a whole. Given that this species appears generally to be more dependent on marine foods than are herring gulls, reductions in amounts of fishery discards may affect this species more, and some decrease in breeding numbers seems likely in response to reduced fishery discarding in coming years.

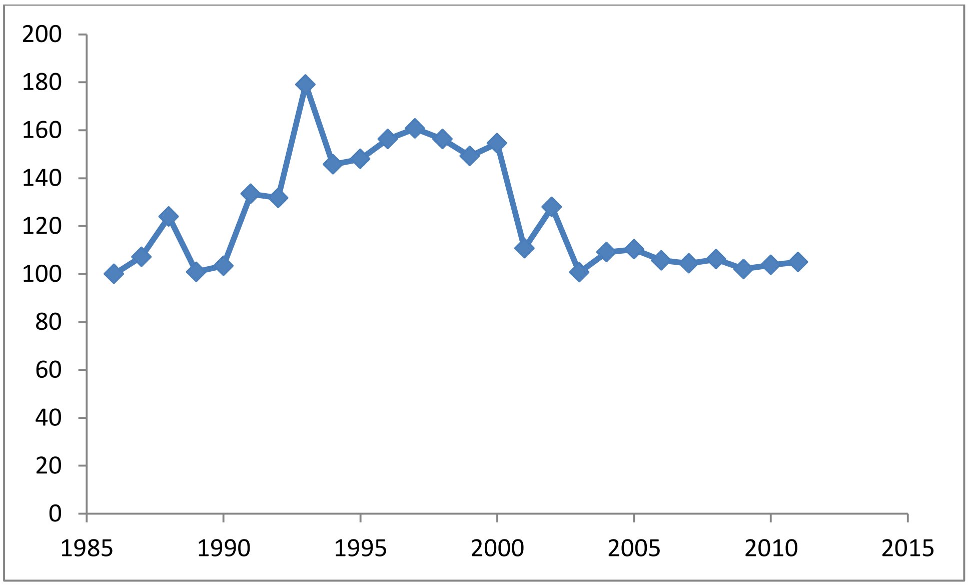

Figure 12.1.1. Lesser black-backed gull breeding population index for the UK. Data from JNCC online database.

12.1.3 Factors affecting survival rates

Poole et al. (1998) estimated adult survival rate at 0.9 on Skomer, but declining from 0.976 in 1978 to 0.806 in 1994 but for uncertain reasons, though possibly related to reduced food supply there in the 1990s (Thompson et al. 1996). Wanless et al. (1996) estimated adult survival rate at 0.91 (standard error 0.012) on the Isle of May for a sample of colour ringed breeding adults between 1989 and 1994. They found no significant annual variation in survival rates of this sample (although annual values varied between 0.89 and 0.94), so were unable to identify any environmental factors affecting survival rates. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

12.1.4 Breeding success in the British Isles

In 1995, productivity was low at Skokholm and Skomer, possibly related to low activity of fishing boats trawling in the general area (Thompson et al. 1996). In 1998, causes of low productivity were mainly unidentified, but there were several instances of possible botulism, at Eigg (significant mortality of juveniles), Strangford Lough (up to 400 dead adult and sub-adult gulls), and South Walney (two waves of mortality of adults and chicks) (Thompson et al. 1999). At Orford Ness in 1999, 16% of nests failed due to fox predation (Upton et al. 2000). A total of 50 adults (from a population of 250 pairs) died of botulism at Old Lighthouse Island (Co. Down) in 1999 (Upton et al. 2000). In 2000 an outbreak of botulism caused substantial adult mortality at South Walney for the third successive year (Mavor et al. 2001). In 2000, 11,946 birds were culled at Tarnbrook Fell as part of a long-term water quality management strategy. At Orford Ness, in 2000, 75% of nests (in a colony of 23,000 pairs), failed due to fox predation (Mavor et al. 2001). Breeding numbers at Orford Ness fell from 24,000 pairs in 2001 to 6,500 pairs in 2002 due to fox activity at the colony because fox control was not carried out there in 2002 (Mavor et al. 2003). Mink greatly reduced breeding success at Argyll colonies where no mink control was carried out (Mavor et al. 2004). At Rockliffe Marsh in 2004, more than half of the nests were abandoned in early June, apparently due to food shortage (Mavor et al. 2005). Several colonies in Argyll have been abandoned due to the presence of mink (Mavor et al. 2006). Colonies where mink were present produced 0.13 chicks per pair whereas a colony where mink were trapped out produced 0.5 to 1 chicks per pair (Mavor et al. 2006). At Tarnbrook Fell, the deployment of falcons and controlled disturbance reduced breeding success and reduced breeding numbers in the centre of the colony, but resulted in the formation of new satellite colonies developing some distance away from the original colony (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for lesser black-backed gulls as follows. Data indicated low breeding success at most colonies in 2006. Experimental removal of mink from around certain colonies in Argyll indicated that mink removal boosted breeding output in this species in 2006 in Argyll by 50% relative to control sites where mink were not trapped. Breeding success of urban-nesting lesser black-backed gulls tends to be high except where efforts are made to reduce numbers breeding in urban environments (Raven and Coulson 1997).

Table 12.1.1 Meta-analysis of main factors contributing to reduced productivity of Lesser black-backed gulls at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Botulism	7
Mink predation	5
Food shortage	3
Fox predation	3
Culling operations	1

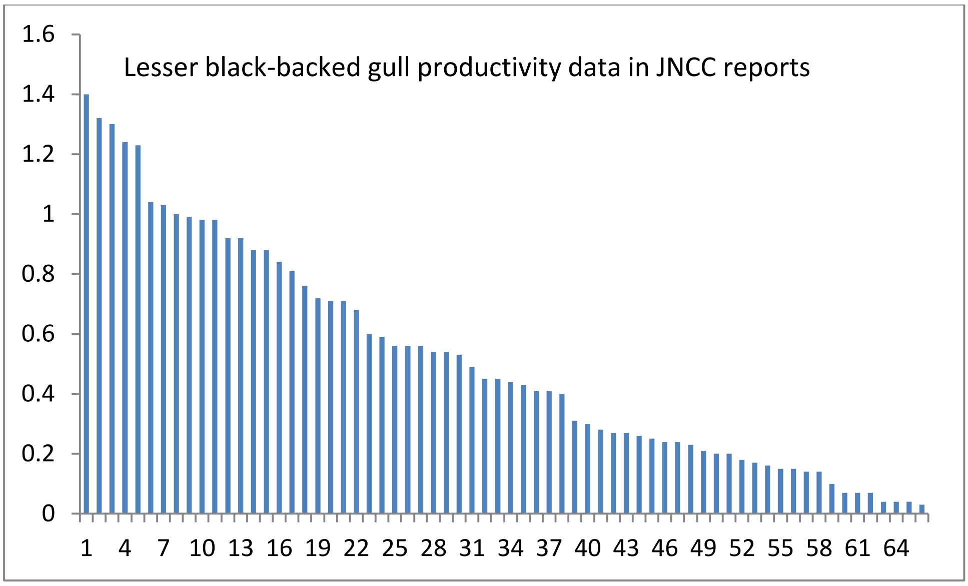

Figure 12.1.2. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of lesser black-backed gulls at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

12.2 Management options

	12.2.1 Mink eradication	12.2.2 Fencing out foxes	12.2.3 End culling	12.2.4 Closure of sandeel and sprat fisheries	12.2.5 Rat eradication
Evidence of success for this species	High C=High*	High C=High	High C=High	Low C=Low	High C=High
Evidence of success for similar species	High C=High	High C=High	High C=High	High C=Low	High C=High
Cost-effectiveness	High C=High	Moderate C=High	High C=High	Uncertain C=Low	High C=High
Feasibility	High C=High	Moderate C=High	Moderate C=High	Moderate C=Low	High C=High
Practicality	Moderate C=Low	Moderate C=High	High C=High	Moderate C=Low	High C=High
Applies at SPA populations	No C=High	Some C=High	Few C=High	Yes C=High	Few C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

12.2.1 Eradication of American mink

Mink are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow lesser black-backed gull productivity to increase at colonies where this predator is present.

Eradication of mink from small islands can be achieved relatively easily by trapping, although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a 'mink-free' status for the seabird colony through the summer, resulting in considerably improved productivity (from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012) (see 12.2.2 below). By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al.

2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
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There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

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12.2.2 Exclusion of foxes from colonies

Foxes are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow lesser black-backed gull productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect gull colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes (see section 12.2.1).

12.2.3 End culling

Culling of breeding adult lesser black-backed gulls has previously taken place at many colonies for a variety of reasons, including reduction of impacts on tern colonies and reduction in bacterial contamination of drinking water supplies. Refusal to permit large-scale culling could increase survival rates of adult lesser black-backed gulls. There are additional measures taken to control impacts of gulls, including removal of nests from urban locations, and disturbance of birds by trained falcons to reduce their use of urban refuse. However, such measures are unlikely to have a significant impact on gull demography at the level of the national population.

12.2.4 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Lesser black-backed gulls depend more on small pelagic fish than do herring gulls. Mean foraging range of lesser black-backed gull is 72 km and maximum foraging range 181 km (Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

12.2.5 Eradication of rats

Although relatively few lesser black-backed gull colonies appear to be subject to rat predation impacts according to the JNCC annual reports, there is evidence that eradication of rats can increase lesser black-backed gull breeding success at islands where rats are numerous. On Ailsa Craig, brown rats colonized in 1889, and there was still a colony of lesser black-backed gulls present in 1990, but their breeding success was about one-third that expected, and the reduction was attributed to rats killing gull chicks (Zonfrillo 2001). Rats on Ailsa were eradicated in 1991 and lesser black-backed gull breeding success improved immediately, to about three times the productivity experienced when rats were present (Zonfrillo 2001, and pers. comm.).

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

At colonies that are on the mainland, or are on islands very close to the mainland so that rats would easily be able to recolonize the island, predator-proof fencing might be an alternative option (see 12.2.2.).

13. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR HERRING GULL

13.1 Herring gull ecology

The herring gull subspecies breeding in Britain is *Larus argentatus argenteus*, which has a total population of about 180,000 to 200,000 pairs distributed across the British Isles, the Netherlands, Belgium, northern France, western Germany, the Faroe Islands and Iceland. The subspecies *L. a. argentatus* breeds in Scandinavia, the Baltic States, Poland, Russia, parts of Germany, southern and eastern France, and numbers around 500,000 to 600,000 pairs (Mitchell et al. 2004), and birds from especially the northern part of its range visit the British Isles in winter.

13.1.1 The species in the British Isles

Seabird 2000 found around 143,000 pairs of herring gulls breeding in Great Britain the Isle of Man and Channel Islands, and about 6,500 pairs in Ireland (Mitchell et al. 2004). Herring gulls breed in a range of colonies from small to large, and in a range of habitats from flat ground to cliffs, and will readily nest on buildings. The urban-nesting population of this species has been growing faster than the population using natural breeding sites, but numbers of urban-nesting herring gulls represent a very small fraction of the total population of the species (Raven and Coulson 1997). The clutch size is usually 2 or 3 eggs, with a mean clutch size of 2.6 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Herring gulls are generalists taking a wide diversity of foods, including intertidal prey and terrestrial foods as well as marine foods. The UK SPA suite for this species holds about 54,600 pairs of herring gulls (about 32% of the UK population of the species) across 12 sites: Ailsa Craig, Alde-Ore Estuary, Buchan Ness to Collieston Coast, Canna and Sanday, East Caithness Cliffs, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Fowlsheugh, Morecambe Bay, Rathlin Island, St Abb's Head to Fast Castle, and Troup Pennan and Lion's Heads (http://jncc.defra.gov.uk/pdf/ukspa/ukspa-a6-85.pdf). Herring gull adult survival rate has been estimated at 0.93 (Glutz von Blotzheim & Bauer 1982), 0.92 (Coulson and Butterfield 1986), 0.94 (Chabrzyk and Coulson 1976), 0.88 (Wanless et al. 1996), 0.88 (Pons and Migot 1995), and 0.81 (Poole et al. 1998), and age of first breeding is 3 to 7 years, averaging 4.5 (Cramp and Simmons 1977-1994, Glutz von Blotzheim & Bauer 1982). British herring gulls are not migratory, but rather disperse over short distances, mostly remaining within Britain all year round (Wernham et al. 2002). Large numbers of herring gulls of the subspecies *argentatus* visit Britain to spend the winter in the North Sea and eastern Britain, though very few of those birds reach the west coast of Britain or sea areas to the west of the country (Wernham et al. 2002).

13.1.2 Present, and likely future trends

Herring gull breeding numbers increased considerably from 1900 to 1969-70 when an estimated 344,000 pairs nested in Britain and Ireland. From 1969 to 2000 numbers declined by about 50-60%, with the population in 2000 estimated at 150,000 pairs, with 49% in Scotland, 38% in England the Isle of Man and Channel Islands, 9% in Wales, and 4% in Ireland (Mitchell et al. 2004). From 2000 to 2011 numbers have declined further (Figure 13.1.1). The recent decline seems to have been greater in Scotland and in Northern Ireland than in the UK as a whole (compare Figures 13.1.1, 13.1.2 and 13.1.3). Foster and Marrs (2012) estimated a 58% decline in the index of breeding numbers in Scotland between 1986 and 2011. Mitchell et al. (2004) stated '*Whilst the increase in the herring gull population through much of the 20th century is attributable, at least in part, to the availability of plentiful and easily accessible food supplies from artificial sources, especially refuse tips, fishery operations and sewage outlets, the decline in recent decades in the amount of food from these same sources is suspected to have contributed to the population declines witnessed since the SCR Census* [1985]'. The anticipated phased introduction of a total ban on fishery discarding in EU waters is likely to further reduce food supply for scavenging seabirds over coming years or decades, and that is likely to affect herring gulls as well as other scavenging seabirds.

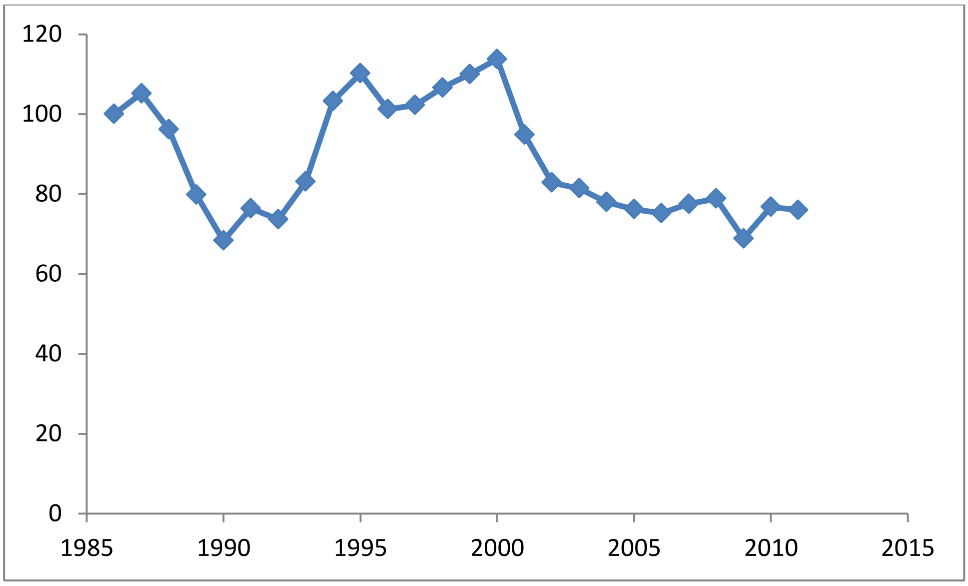

Figure 13.1.1. Herring gull breeding population index for the UK. Data from JNCC online database.

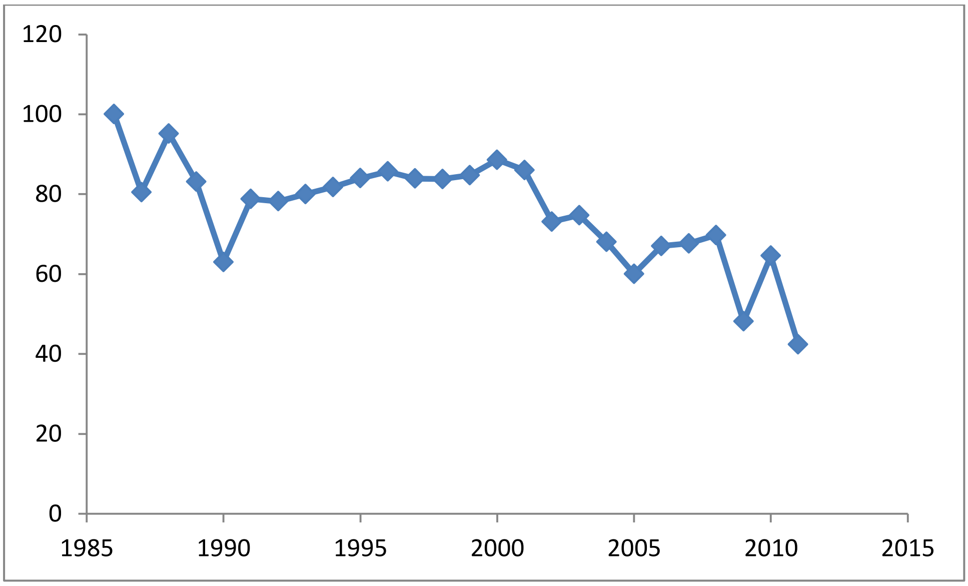

Figure 13.1.2. Herring gull breeding population index for Scotland. Data from JNCC online database.

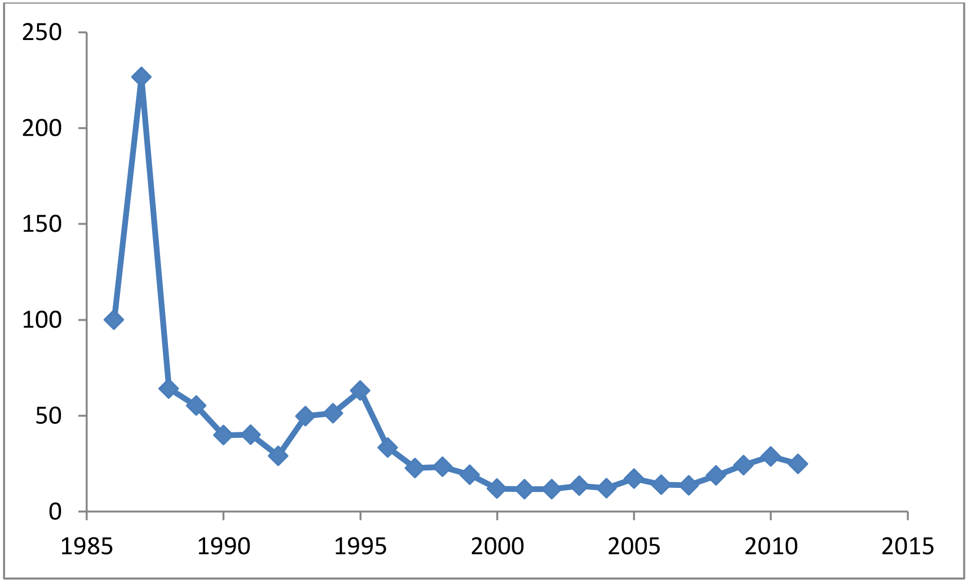

Figure 13.1.3. Herring gull breeding population index for Northern Ireland. Data from JNCC online database.

13.1.3 Factors affecting survival rates

Wanless et al. (1996) estimated adult survival rate at 0.88 (standard error 0.013) on the Isle of May for a sample of colour ringed breeding adults between 1989 and 1994, a period following a major cull that greatly reduced numbers in that colony. They found no significant annual variation in survival rates of this sample (although annual values varied between 0.84 and 0.92), so were unable to identify any environmental factors affecting survival rates. Survival of herring gulls on the Isle of May prior to culling was estimated at 0.935 (Chabrzyk and Coulson 1976), but with a large standard error (0.1) so that this value is not significantly different from that estimated by Wanless et al. (1996). Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

13.1.4 Breeding success in the British Isles

In 1992, flooding by high tides reduced productivity at one colony in East Anglia, and botulism affected productivity and adult survival at one colony in Northern Ireland (Walsh et al. 1993). In 1994 in Argyll, colonies affected by mink predation produced 0.16 chicks per pair whereas colonies where mink were absent produced 0.72 chicks per pair (Walsh et al. 1995). In 1995 in Argyll, colonies affected by mink predation produced 0.18 chicks per pair whereas colonies where mink were absent produced 0.77 chicks per pair (Thompson et al. 1996). In 1996 in Argyll, colonies affected by mink predation produced 0.16-0.23 chicks per pair whereas colonies where mink were absent produced 0.58-0.79 chicks per pair (Thompson et al. 1997). In 1997 in Argyll, at nine colonies where mink were absent or controlled productivity averaged 0.96 chicks per pair, whereas at ten colonies where there was evidence of mink activity in the area, productivity averaged 0.23 chicks per pair (Thompson et al. 1998). On Skokholm in 1997, only 0.44 chicks fledged per pair where some nests were washed away by heavy seas in mid-season (Thompson et al. 1998). In 1998 in Argyll, seven colonies where mink were controlled produced 1.07 fledglings per pair, while 25 colonies with no mink control produced 0.34 fledglings per pair (Thompson et al. 1999). In 1999 at Argyll colonies where mink were known to be active, only 0.11 chicks fledged per pair, whereas at colonies where mink were controlled, productivity averaged 0.81 chicks fledged per pair (Upton et al. 2000). In 2000 at Argyll colonies 0.33 chicks per pair were produced where mink was present, whereas 1.21 chicks per pair were reared at colonies where mink were controlled (Mavor et al. 2001). At Orford Ness in 2000, 75% of 6,750 pairs of Herring gulls suffered breeding failure due to foxes (Mavor et al. 2001). Outbreaks of botulism killed chicks and adults at Ynysoedd Gwylan and at South Walney (Mavor et al. 2001). In 2001, half of the herring gull colonies monitored in Argyll that were exposed to mink produced no young at all and averaged 0.13 chicks per pair, whereas colonies where mink were controlled produced 0.83 chicks per pair (Mavor et al. 2002). Low productivity at Canna in 2001 was attributed to a reduction in fishery discards in the area (Mavor et al. 2002). In 2002, mink were considered to be reducing productivity at unprotected colonies in Argyll by about 30% compared to colonies where mink were trapped (Mavor et al. 2004). Lack of predator control at Orford Ness in 2002 resulted in Herring gull breeding numbers falling there from 6,800 pairs in 2001 to 2,575 pairs in 2002 (Mavor et al. 2003). In 2003, mink were considered to be reducing productivity at unprotected colonies in Argyll by about 25% compared to colonies where mink were trapped (Mavor et al. 2004). Mink removal at some colonies in Argyll in 2004 increased productivity to 0.9 chicks per nest compared to 0.52 at colonies where mink were not controlled, suggesting that mink reduced productivity by 42% (Mavor et al. 2005). In 2005, mink reduced productivity by about 38% (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for herring gulls as follows. Breeding success varied considerably among colonies. Experimental removal of mink from around certain colonies in Argyll indicated that mink removal boosted breeding output in this species in 2006 in Argyll by 36% relative to control sites where mink were not trapped. Breeding success of urban-nesting herring gulls tends to be high except where efforts are made to reduce numbers breeding in urban environments (Raven and Coulson 1997).

Table 13.1.1 Meta-analysis of main factors contributing to reduced productivity of Herring gulls at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Mink predation	12
Flooding	2
Botulism	2
Fox predation	2
Food shortage	1

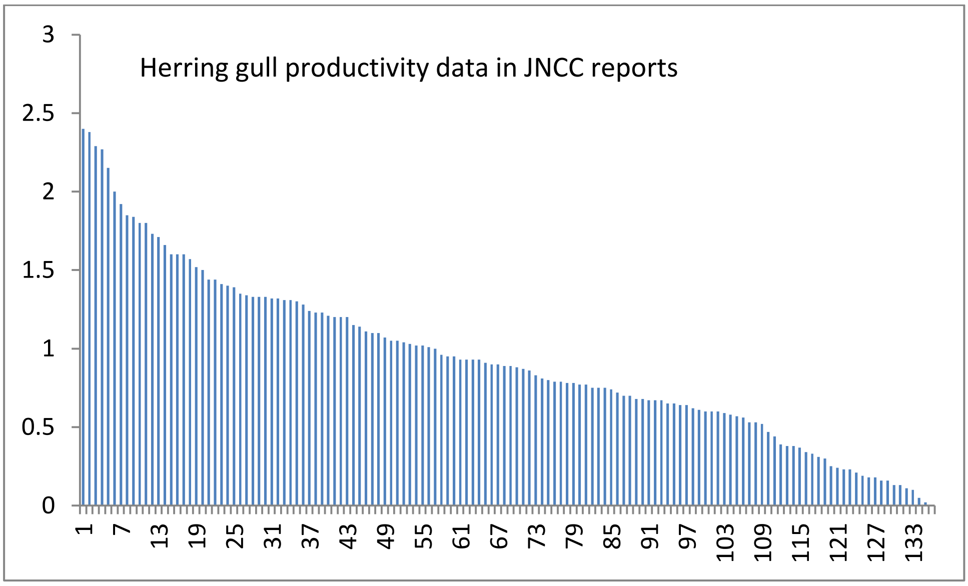

Figure 13.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of herring gulls at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

13.2 Management options

	13.2.1 Mink eradication	13.2.2 Exclusion of foxes	13.2.3 End culling	13.2.4 Rat eradication
Evidence of success for this species	High C=High*	High C=High	High C=High	High C=High
Evidence of success for similar species	High C=High	High C=High	High C=High	High C=High
Cost-effectiveness	High C=High	Moderate C=High	High C=High	High C=High
Feasibility	High C=High	Moderate C=High	Moderate C=High	High C=High
Practicality	Moderate C=High	Moderate C=High	High C=High	High C=High
Applies at SPA populations	No C=High	Some C=High	Few C=High	Few C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

13.2.1 Eradication of American mink

Mink are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow herring gull productivity to increase at colonies where this predator is present.

Eradication of mink from small islands can be achieved relatively easily by trapping, although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity (varying from year to year but on average from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 13.2.2 below).

13.2.2 Exclusion of foxes from colonies

Foxes are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow herring gull productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka’ ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect gull colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes.

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A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

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13.2.3 End culling

Culling of breeding adult herring gulls has previously taken place at many colonies for a variety of reasons, including reduction of impacts on tern colonies and reduction in bacterial contamination of drinking water supplies. Refusal to permit large-scale culling could increase survival rates of adult herring gulls. There are additional measures taken to control impacts of gulls, including removal of nests from urban locations, and disturbance of birds by trained falcons to reduce their use of urban refuse. However, such measures are unlikely to have a significant impact on gull demography at the level of the national population.

13.2.4 Eradication of rats

Although relatively few herring gull colonies appear to be subject to rat predation impacts according to the JNCC annual reports, there is evidence that eradication of rats can increase herring gull breeding success at islands where rats are numerous. On Ailsa Craig, brown rats colonized in 1889, and there was still a colony of herring gulls present in 1990, but their breeding success was about one-third that expected, and the reduction was attributed to rats killing gull chicks (Zonfrillo 2001). Rats on Ailsa were eradicated in 1991 and herring gull breeding success improved immediately, to about three to four times the productivity experienced when rats were present (Zonfrillo 2001, and pers. comm.).

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so

£150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

At colonies that are on the mainland, or are on islands very close to the mainland so that rats would easily be able to recolonize the island, predator-proof fencing might be an alternative option (see 13.2.2.).

14. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR GREAT BLACK-BACKED GULL

14.1 Great black-backed gull ecology

The great black-backed gull is a SPEC 4 species (Favourable conservation status (secure) but concentrated in Europe). The world population is around 170,000 to 180,000 pairs, with 100,000 to 110,000 of these in Europe (excluding Russia) and around 60,000 pairs in North America (Mitchell et al. 2004). Some 40,000 pairs nest in Norway, predominantly on the Arctic north coast; many of these birds visit Britain and the North Sea for the winter (Wernham et al. 2002).

14.1.1 The species in the British Isles

Seabird 2000 found about 17,000 pairs breeding in Britain and 2,300 in Ireland (Mitchell et al. 2004). Great black-backed gulls mostly breed in small colonies or scattered pairs along suitable coastlines, although there are some large colonies. Nests are on the ground, often close to colonies of seabirds on which these gulls may feed. The clutch is usually of 2 or 3 eggs, with a mean clutch size of 2.6 eggs (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Great black-backed gulls have a varied diet, more predatory than that of herring gulls. Birds nesting in large colonies mainly feed on fish, including fishery discards (which form a major part of their diet, especially in winter), whereas birds nesting as isolated pairs tend to feed on seabirds such as auks and kittiwakes. The UK SPA suite for this species holds about 4,400 pairs (23% of the UK total) across 6 sites: Calf of Eday, Copinsay, East Caithness Cliffs, Hoy, Isles of Scilly, and North Rona and Sula Sgeir (http://jncc.defra.gov.uk/pdf/UKSPA/UKSPA-A6-86.pdf). Adult survival rate is thought to be around 0.93 by analogy with herring and lesser black-backed gulls, and age of first breeding is around 4 or 5 years old (Cramp and Simmons 1977-1994, Glutz von Blotzheim & Bauer 1982). British great black-backed gulls rarely move more than a few tens of kilometres from their nesting sites, but large numbers of great black-backed gulls from north Norway visit the North Sea and eastern Britain during autumn and early winter. Those birds tend to return to north Norway by about February, and very rarely cross to western areas of the British Isles.

14.1.2 Present, and likely future trends

The breeding population increased during the 20th century, reaching 22,412 pairs of coastal-breeding great black-backed gulls in 1969-70 (a minimal number nesting inland were not surveyed). In 1985-88 there were 20,892 pairs, indicating a decline from what appears to have been an all-time peak in numbers around 1969. This declining trend was confirmed in Seabird 2000, when there were 19,691 coastal-breeding pairs plus 22 inland pairs (Mitchell et al. 2004). Of this total in 2000, 75% were in Scotland, 12% in Ireland, 11% in England the Isle of Man and Channel Islands, and 2% in Wales.

The increase in numbers through most of the 20th century has been attributed mainly to protection after long periods of persecution and exploitation, although this species also makes considerable use of trawl fishery discards, being able to swallow larger fish than any other UK seabird apart from gannet, and being able to rob discards from smaller scavenging seabirds (even great skuas) (Hudson and Furness 1988, 1989). Mitchell et al. (2004) suggest that fishery discards are almost certain to have influenced the continued growth in numbers of great black-backed gulls, although they point out that very little research into the demography or ecology of this species has been carried out. Hudson (1982) showed that birds at large great black-backed gull colonies in Ireland fed mainly on fishery discards while breeding, whereas pairs nesting in isolation or in small colonies fed mainly on smaller seabirds such as puffins. The same is true in Scotland, England and Wales (Poole 1995), with isolated breeding pairs also feeding on rabbits and hares as well as a wide range of seabirds (Mitchell et al. 2004).

Although in a few cases declines in breeding numbers since 1969 can be attributed to culling great black-backed gulls for the conservation of smaller seabirds, declines in breeding numbers since

1969 have occurred particularly at larger colonies, consistent with the idea that these represent density-dependent responses to reductions in fishery discard rates (which have already been occurring since the late 1960s (Votier et al. 2004)). In addition, in some northern colonies great black-backed gulls breed adjacent to great skua colonies. Increases in great skua numbers seem to have gone hand in hand with decreases in great black-backed gull numbers at such sites. A clear example of this is on Hoy, Orkney, where great black-backed gull breeding numbers fell from 3000 pairs in 1969 to 1,163 pairs in 1985 to 389 pairs in 2000, while great skua numbers increased from 72 pairs in 1969, to 1,563 in 1985 to 1,973 in 2000 (Mitchell et al. 2004).

The index of breeding numbers at UK monitored colonies (Figure 14.1.1) shows a decline from a peak reached around 2000 to 2011. However, the decline appears to be more severe in Scotland (Figure 14.1.2) than in the UK as a whole. Foster and Marrs (2012) reported a 53% decrease in an index of breeding numbers of great black-backed gulls in Scotland between 1986 and 2011. Phased elimination of discarding by EU fisheries is likely to reduce great black-backed gull numbers. This species, like the great skua, feeds extensively on discards while breeding as well as in winter. As Mitchell et al. (2004) stated '*it is probable that productivity during the breeding season and increased winter survival both increased as a consequence of feeding on discards*'. Removal of this food subsidy is likely to reverse that, leading to declines in great black-backed gull numbers, especially at the largest colonies, which are almost all SPAs for this species.

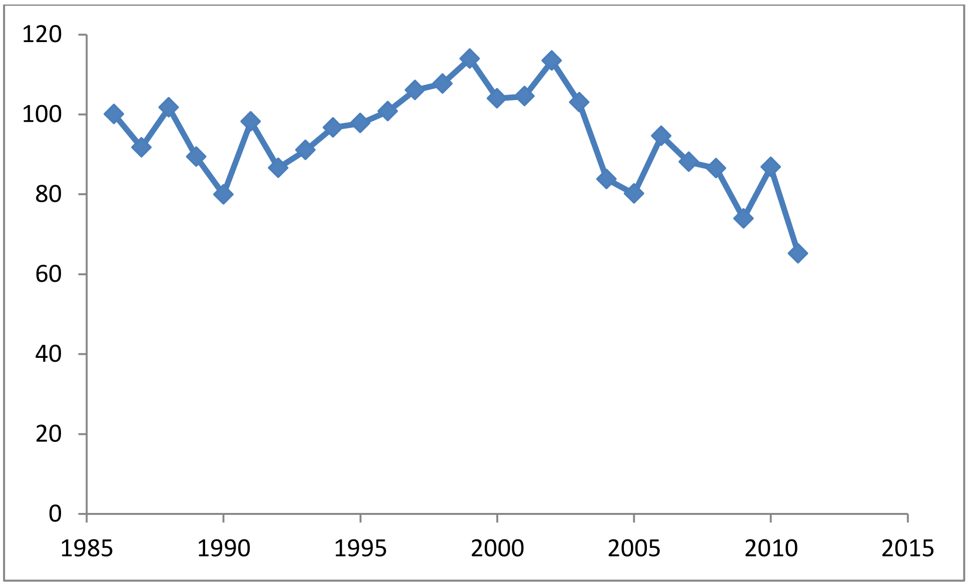

Figure 14.1.1. Great black-backed gull breeding population index for the UK. Data from JNCC online database.

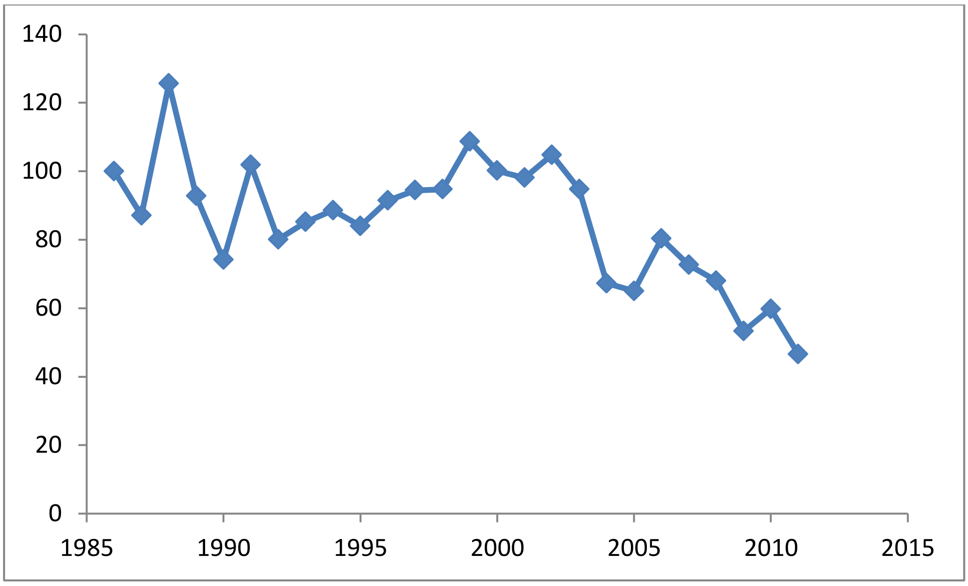

Figure 14.1.2. Great black-backed gull breeding population index for Scotland. Data from JNCC online database.

14.1.3 Factors affecting survival rates

There has been very little research into survival rates of great black-backed gulls. Adult survival rate is reported to be around 0.93 by analogy with related but slightly smaller gull species (Cramp and Simmons 1977-1994, Glutz von Blotzheim & Bauer 1982). However, factors affecting survival rates are uncertain, and have not been quantified. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

14.1.4 Breeding success in the British Isles

In relation to the 1995 breeding season, Thompson et al. (1996) commented that the two colonies with highest productivity (Isle of May and Nigg oil terminal) were those where there is protection of this species from disturbance and persecution. In 1998 in Argyll, colonies where mink were controlled produced 1.25 chicks per pair while those where there was no mink control produced 1.11 chicks per pair (Thompson et al. 1999). On Noss, a high proportion of fledglings were killed by Great skuas (Thompson et al. 1999). In 1999 in Argyll, there was complete breeding failure at 30 of 61 monitored sites, with mink apparently responsible for failure at 17 of these and possibly at another 6 (Upton et al. 2000). In 2000 in Argyll, Great black-backed gulls were monitored at 73 sites, mostly with small numbers of pairs. There was complete breeding failure at 18 sites, with mink considered responsible at 10 of these. However, breeding success overall averaged 1.17 chicks per pair (Mavor et al. 2001). In 2001, mink reduced productivity at 14 out of 55 Great black-backed gull breeding sites in Argyll with only 7 young fledging from 53 nests, whereas at all 58 sites productivity averaged 0.92 chicks per pair (Mavor et al. 2002). In 2001 at Nigg, there was evidence of mammal predation affecting productivity (Mavor et al. 2002). Mink in parts of Argyll were estimated to reduce productivity by 9% in 2004 (Mavor et al. 2005). In 2005, mink were estimated to reduce productivity by 41%, although other predators including brown rats may also have contributed to this reduction (Mavor et al. 2006). On Hoy in 2004 breeding success was very low, and chicks were underweight suggesting food shortage (Mavor et al. 2005). In 2005, complete breeding failure at the Nigg colony (for the third year in succession) was attributed to low food availability (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for great black-backed gulls as follows. Breeding success was high at some colonies but low in NW Scotland, NW England and NE Ireland. Mink control around some colonies in Argyll increased breeding success in those colonies to 0.81 chicks per nest compared to control colonies which produced 0.67 chicks per nest.

Table 14.1.1 Meta-analysis of main factors contributing to reduced productivity of Great black-backed gulls at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Mink predation	7
Food shortage	3
Great skua predation	2
Human disturbance	1
Rat predation	1

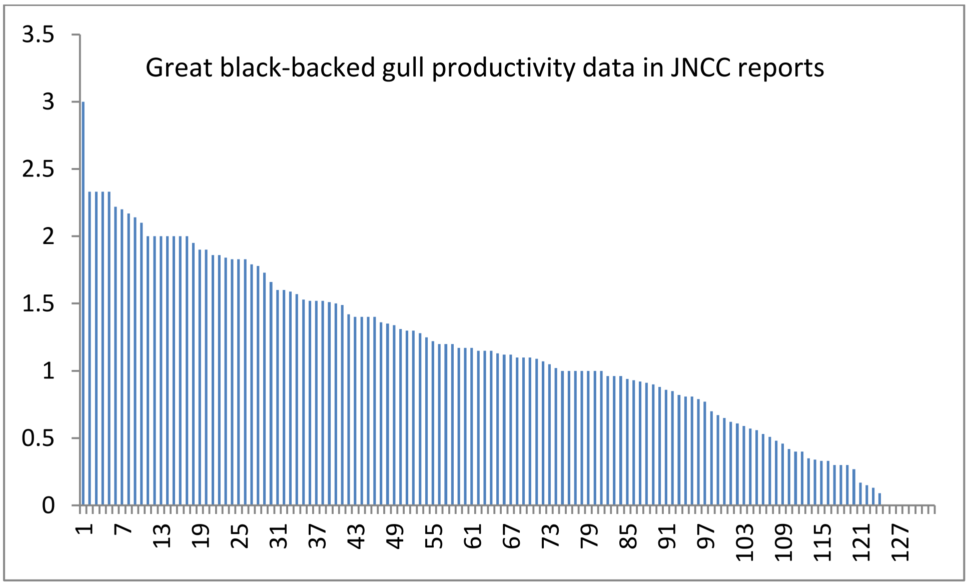

Figure 14.1.3. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of great black-backed gulls at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

14.2 Management options

	14.2.1 End culling	14.2.2 Mink eradication	14.2.3 Exclusion of foxes	14.2.4 Closure of sandeel and sprat fisheries close to colonies	14.2.5 Rat eradication
Evidence of success for this species	High C=High*	High C=High	High C=High	Low C=Low	High C=High
Evidence of success for similar species	High C=High	High C=High	High C=High	High C=Mod	High C=High
Cost-effectiveness	High C=High	High C=High	Moderate C=High	Uncertain C=Low	High C=High
Feasibility	Moderate C=High	High C=High	Moderate C=High	Moderate C=Low	High C=High
Practicality	High C=High	Moderate C=High	Moderate C=High	Moderate C=Low	High C=High
Applies at SPA populations	Few C=High	No C=High	Few C=High	Yes C=High	Few C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

14.2.1 End culling

Culling of breeding adult great black-backed gulls has previously taken place at many colonies for a variety of reasons, including reduction of impacts on tern and other seabird colonies. For example, breeding numbers on Skomer were reduced by culling from over 300 pairs in 1971 to below 40 pairs in 1990 to reduce their impact on Manx shearwaters (Poole 1995). Refusal to permit large-scale culling could increase survival rates of adult great black-backed gulls.

14.2.2 Eradication of American mink

Mink are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow great black-backed gull productivity to increase at colonies where this predator is present.

Eradication of mink from small islands can be achieved relatively easily by trapping, although eradicating mink from large areas is much more challenging. Efforts by Scottish Natural Heritage to eradicate mink from the Western Isles have been costly and protracted. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity (varying from year to year but on average from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 14.2.3 below).

14.2.3 Exclusion of foxes from colonies

Foxes are a major determinant of productivity at several colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Fencing of colonies to exclude foxes would allow great black-backed gull productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka’ ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect gull colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals

and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

14.2.4 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as cause of reduced productivity at some colonies in some years. Great black-backed gulls depend more on small pelagic fish than do herring gulls. Mean foraging range of great black-backed gull is around 40 km and maximum foraging range is likely to be at least twice this distance (Ratcliffe 2000 cited in Langston 2010), so closure of sandeel and sprat fishing within 60 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

14.2.5 Eradication of rats

Although relatively few great black-backed gull colonies appear to be subject to rat predation impacts according to the JNCC annual reports, there is evidence that eradication of rats can increase great black-backed gull breeding success at islands where rats are numerous. On Ailsa Craig, brown rats colonized in 1889, and there was still a colony of great black-backed gulls present in 1990, but their breeding success was about one-third that expected, and the reduction was attributed to rats killing gull chicks (Zonfrillo 2001). Rats on Ailsa were eradicated in 1991 and great black-backed gull breeding success improved immediately, to about three times the productivity experienced when rats were present (Zonfrillo 2001, and pers. comm.). Great black-backed gull breeding numbers increased, probably because rats had been eliminated from the competition for scavenging of bird corpses, so leaving more food for great black-backed gulls.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

At colonies that are on the mainland, or are on islands very close to the mainland so that rats would easily be able to recolonize the island, predator-proof fencing might be an alternative option (see 14.2.3.).

15. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR BLACK-LEGGED KITTIWAKE

15.1 Black-legged kittiwake ecology

The world population of the species is around 4.3 to 5.2 million breeding pairs (Mitchell et al. 2004), with around 2.5 to 3 million pairs breeding within the North Atlantic region (Stroud et al. 2001). The largest European populations are in Iceland (600,000 to 800,000 pairs), and Norway (770,000 pairs including Svalbard and Bear Island).

15.1.1 The species in the British Isles

Seabird 2000 found 370,000 pairs in Great Britain, the Isle of Man and Channel Islands and 49,000 pairs in Ireland (Mitchell et al. 2004). Kittiwakes mainly nest relatively low down on steep cliffs, though colonies can occur on waterside buildings and walls in places where natural cliff sites are unavailable or absent. Kittiwakes lay one to three eggs, most often two, with a mean clutch size of 2.01 in the British Isles (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). At North Sea colonies in Britain, kittiwakes feed their chicks primarily on sandeels (Furness and Tasker 2000), even at colonies where neighbouring common guillemots are simultaneously feeding chicks mainly on sprats or young herring (Lewis et al. 2001), but at colonies in the Irish Sea they may feed chicks mainly on sprats (Chivers et al. 2012). Euphausiids and other zooplankton can be important in their diet, and they will feed on small scraps of offal and discards at fishing vessels and on spilled stomach contents of larger fish, especially in winter (Garthe et al. 1996).

The UK SPA suite for this species holds around 384,000 pairs (78% of the UK total) across 33 sites: Ailsa Craig, Buchan Ness to Collieston Coast, Calf of Eday, Canna and Sanday, Cape Wrath, Copinsay, East Caithness Cliffs, Fair Isle, Farne Islands, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Fowlsheugh, Handa, Hermaness Saxa Vord and Valla Field, Hoy, Marwick Head, Mingulay and Berneray, North Caithness Cliffs, North Colonsay and Western Cliffs, North Rona and Sula Sgeir, Noss, Rathlin Island, Rousay, Rum, Shiant Isles, Skomer and Skokholm, St Abb's Head to Fast Castle, St Kilda, Sumburgh Head, Troup Pennan and Lion's Heads, West Westray. Adult survival rate averages 0.81 (del Hoyo et al. 1996) and age of first breeding is around 4 years old (Cramp and Simmons 1977-1994).

British kittiwakes mostly winter in the western North Atlantic, although a small proportion of the population may remain in British waters all year round (Frederiksen et al. 2012). Kittiwakes from other European and high latitude colonies in the North Atlantic and Barents Sea may pass through British waters on migration, but most of these birds also winter in the western North Atlantic in the same general area used by British birds (Frederiksen et al. 2012).

15.1.2 Present, and likely future trends

Kittiwake numbers increased throughout the British Isles from 1900 to around 1985. Operation Seafarer in 1969-70 found 448,000 pairs in Britain and Ireland. The SCR Census in 1985-88 found 540,000 pairs, while Seabird 2000 found 416,000 pairs (Mitchell et al. 2004). The long-term increase for most of the 20th century has been attributed in part to reduced persecution and exploitation, but changes in food supplies are also likely to have had an influence. Increases in sandeel abundance in the 1960s and 1970s followed depletion of large predatory fish such as cod and whiting, and predators on sandeel larvae and competitors for zooplankton (herring and mackerel). Kittiwakes feed primarily on sandeels while breeding, and their breeding success and survival rate are strongly influenced by sandeel stock size and by commercial fisheries on sandeels (Furness and Tasker 2000, Lewis et al. 2001a,b, Oro and Furness 2002, Mitchell et al. 2004, Frederiksen et al. 2004). Since the peak in numbers around 1985, declines in kittiwake numbers have been most severe in north Scotland (especially Shetland), associated with the collapse of sandeel stocks in the northern North Sea and consequent increases in predation impacts on kittiwakes, especially from great skuas in northern areas. JNCC monitoring data show a large decline in the index of kittiwake breeding numbers in the UK (Figure 15.1.1). However, this index includes varying trends in different national populations. In Scotland, (Figure 15.1.2) the decline is more extreme than in England (Figure 15.1.3), while in Welsh colonies the decline is smallest (Figure 15.1.4). Foster and Marrs (2012) reported a decline in the index of breeding numbers in Scotland of 66% between 1986 and 2011. The strong decline in Scotland emphasises the importance of the collapse of sandeel stocks in the northern North Sea and associated increases in predation by large gulls and great skuas.

Future prospects for kittiwakes look bleak in the north of Britain as the root of much of the problem seems to be climate change impacts on lower trophic levels affecting kittiwake food supply. In addition, a phased ban on fishery discards in EU waters will reduce food supply in the form of offal and fragments of discarded fish on which kittiwakes feed in winter though rather less in summer (Garthe et al. 1996). The discard ban will also reduce food supply to great skuas and great black-backed gulls, which is likely in the short term to cause those birds to increase depredation of kittiwakes, though in the long term may reduce numbers of these larger scavengers to more sustainable levels. There may be relatively few kittiwakes left in northern parts of the British Isles by then. In southern Britain, the situation may be less extreme, as sandeel stocks in the southern North Sea appear to be more sustainable than in the north, and possibly sprats may provide an alternative food in some areas such as the Irish Sea (possibly contributing to the lower decrease in kittiwake numbers at Welsh colonies).

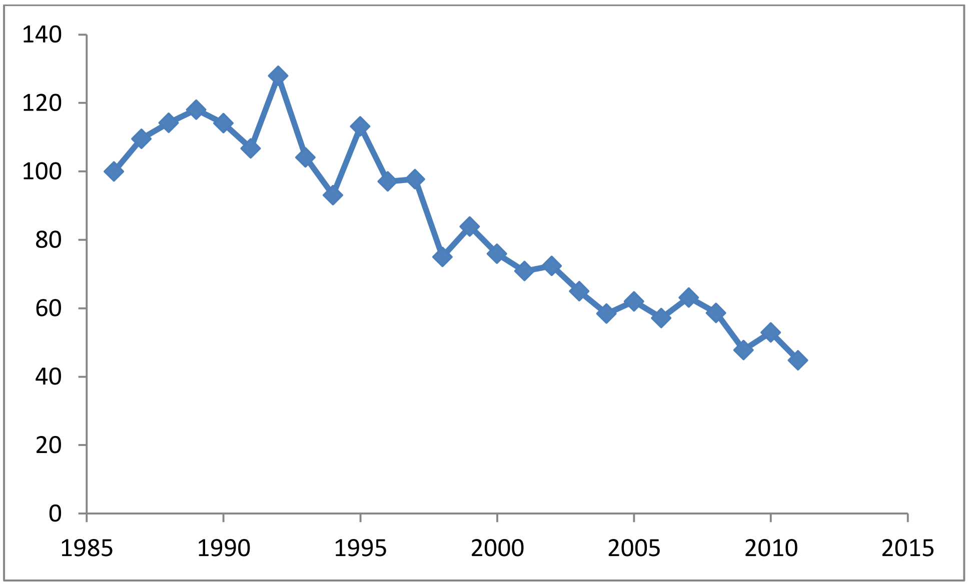

Figure 15.1.1. Kittiwake breeding population index for the UK. Data from JNCC online database.

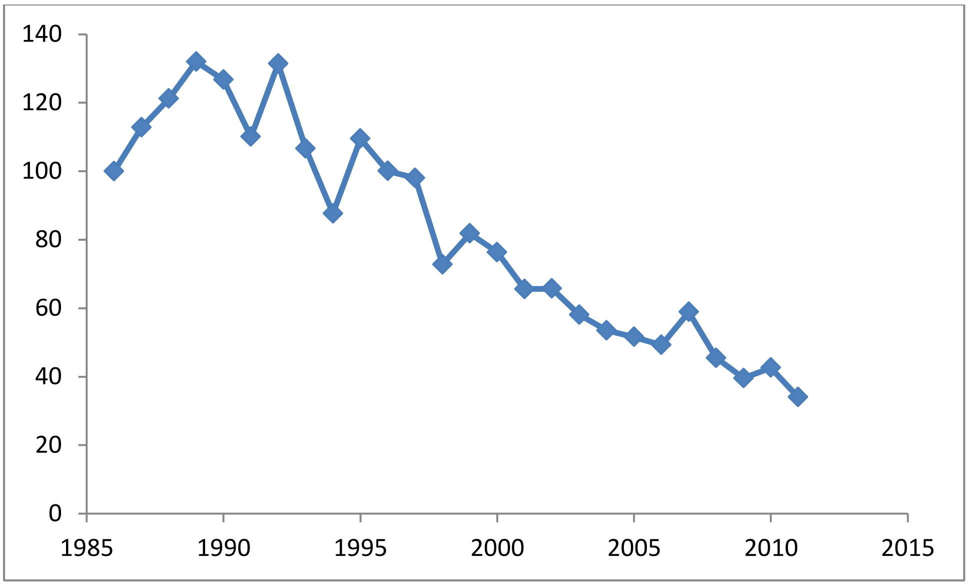

Figure 15.1.2. Kittiwake breeding population index for Scotland. Data from JNCC online database.

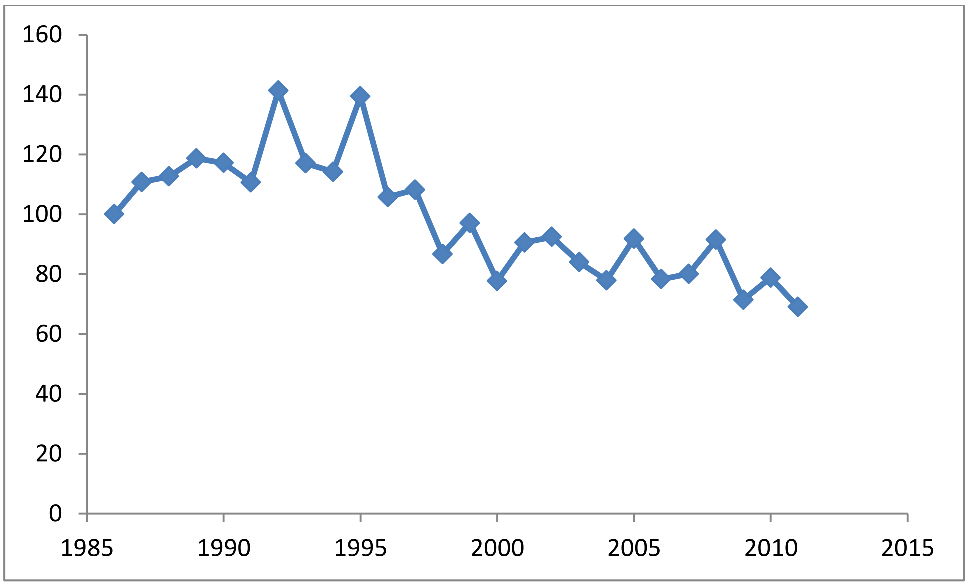

Figure 15.1.3. Kittiwake breeding population index for England. Data from JNCC online database.

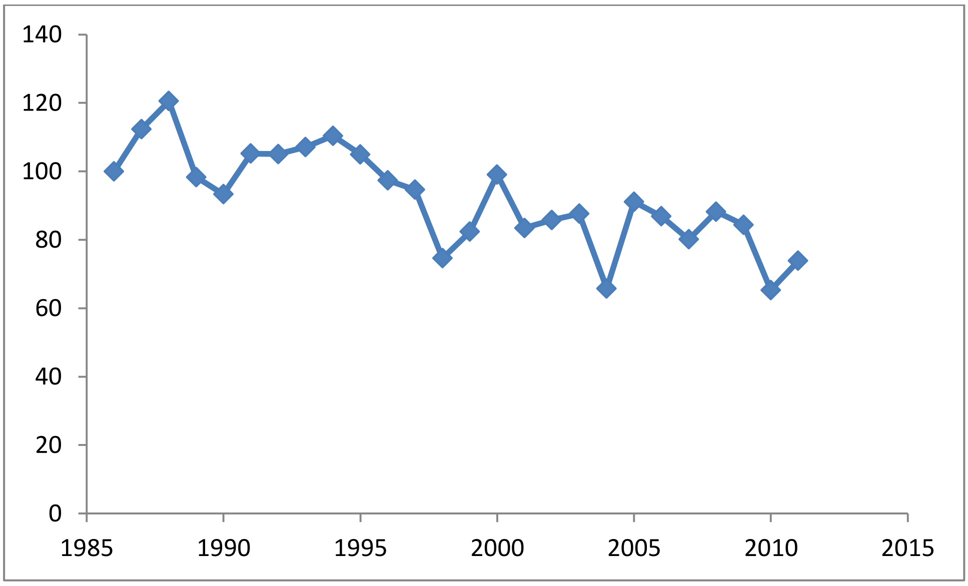

Figure 15.1.4. Kittiwake breeding population index for Wales. Data from JNCC online database.

15.1.3 *Factors affecting survival rates*

Several studies have reported kittiwake adult survival rates but without investigating influences on these rates of specific environmental factors (e.g. Danchin and Monnat 1992, Hatch et al. 1993 (mean 0.912 at a Pacific colony), Cam et al. 1998, Golet et al. 1998 (mean 0.922 at a Pacific colony), Coulson and Strowger 1999 (mean 0.79 at a colony in NE England), Harris et al. 2000 (0.882 at the Isle of May), Rothery et al. 2002 (mean 0.832 on Fair Isle), Lerche-Jørgensen et al. 2012) (mean 0.82 at a Danish colony)). Studying colour ringed adults in Brittany in 1980-1993, Cam et al. (1998) estimated annual survival at 0.79 for breeders and 0.65 for non-breeders; they suggested that lower survival of nonbreeders reflected their poorer individual quality or body condition. Aebischer and Coulson (1990) reported a mean survival rate of 0.8 but with variation from 0.85 in 1954-1965 to only 0.65 in 1982-85. They suggested that the decrease in survival in the 1980s might most likely be due to changes in abundance of small pelagic fish on which the kittiwakes depend. Sandvik et al. (2005) reported a mean survival rate of 0.88 for breeding adult kittiwakes colour ringed at Hornøya, north Norway, 1990-2002. In that analysis they showed that annual variation in kittiwake survival was strongly affected by pelagic fish stock biomass (herring and capelin both being important), and by sea surface temperature variation. Oro and Furness (2002) showed that kittiwake breeding adult annual survival rates at a colony in Shetland varied between 0.98 and 0.53 (with a mean of 0.8), with a strong effect of sandeel abundance and a weak influence of great skua breeding success. Survival of kittiwakes was higher when 0 group sandeel abundance was higher and was slightly reduced when great skua breeding success was higher (suggesting more predation by skuas when they had chicks to feed). Kittiwake survival increased by about 0.2 from lowest to highest observed sandeel abundance. Frederiksen et al. (2004) analysed environmental factors affecting survival rates of breeding adult kittiwakes at the Isle of May colony. They found that survival rate varied between 0.98 in 1986-87 and 0.82 in 1998-99, with 35 to 52% of the annual variation in survival rate being explained by the presence or absence of a commercial fishery for sandeels in the area and sea surface temperature (SST). Survival was lower when there was a sandeel fishery and when SST was higher. This is consistent with the fishery depleting the local sandeel stock, and with sandeel recruitment decreasing with higher SST (Arnott and Ruxton 2002). On average, kittiwake adult survival rate was reduced by about 0.05 during the period when a commercial fishery for sandeels was active in the area. The results presented by Frederiksen et al. (2004) are closely consistent with those of Oro and Furness (2002), but for kittiwakes breeding in different regions and associated with different stocks of sandeels and different commercial fisheries exploiting those stocks. These results are also consistent with changes in adult survival rates in relation to food supply reported for Arctic skua (Davis et al. 2005) and great skua (Ratcliffe et al. 2002).

15.1.4 *Breeding success in the British Isles*

Poor availability of sandeels in Shetland waters led to greatly reduced productivity in 1988-90 (zero at most Shetland colonies in 1988, 1989 and 1990) (Walsh et al. 1991). Kittiwake productivity in 1991 was much higher in Shetland (0.56 chicks per nest) than it had been there in 1988-90, apparently reflecting high recruitment of sandeels in 1991 (Walsh et al. 1992). In 1992, productivity averaged 0.73 chicks per nest, this improvement over previous years being attributed to increased abundance of sandeels in the northern North Sea and little impact of predation or weather (Walsh et al. 1993). In 1993, overall productivity was moderate, averaging 0.63 chicks fledged per nest. Reduction in productivity was broadly attributed to food shortage, but with some instances of predation; for example Kettla Ness colony failed completely due to predation by great skuas (Walsh et al. 1994). In 1994 productivity overall was moderately high averaging 0.72 chicks per nest, but was reduced at some Shetland colonies by Great skua predation, and at the Isles of Scilly by brown rat predation (Walsh et al. 1995). In 1994, poor weather was considered to have had a minor impact at some colonies, but food shortage was thought likely to be the most important factor determining Kittiwake productivity (Walsh et al. 1995). In 1995, poor productivity of colonies on southern Irish Sea coasts was thought to be related to low availability of food (Thompson et al. 1996). In 1996 in

Shetland, productivity was thought not to be affected by food shortage, but varied among colonies in relation to predator impacts, specifically the amount of predation by Great skuas (Thompson et al. 1997). The lowest breeding success in the west in 1996 was on St Kilda (0.62 chicks per nest) where a severe gale in late May washed nests off some study plots (Thompson et al. 1997). In 1997 in Shetland, productivity was reduced by lack of sandeels and by predation (mostly by Great skuas) (Thompson et al. 1998). Along the east coast of Britain, severe north-easterly gales with associated rain and heavy seas destroyed large numbers of Kittiwake nests at exposed colonies (Thompson et al. 1998). In 1998 in Shetland productivity (0.06 chicks per pair) was drastically reduced by scarcity of sandeels in Shetland waters leading to low colony attendance by adults, and starvation of chicks in nests in late June and early July (Thompson et al. 1999). Productivity at colonies in east Scotland and east England was very patchy, with some colonies affected by food shortage (e.g. Isle of May, where young were left unattended by adults and died) and some colonies affected by egg losses due to heavy rain (Thompson et al. 1999). On the Isles of Scilly, breeding success was only 0.14 chicks per pair, most losses being due to predation of chicks by cats (Thompson et al. 1999). In 1999, productivity averaged 0.81 chicks per nest across 42 colonies. In Shetland (average 0.74 chicks per nest) great skuas killed some chicks and fledglings (Upton et al. 2000). At Bullers of Buchan and Lowestoft, wet weather washed away some nests. At Canna, peregrines took several fledglings. At St Abbs Head, predation by mink was thought to be responsible for the loss of half of the chicks in one monitoring plot. On Tyneside, construction of an artificial tower next to a demolished mill colony attracted birds to nest on the new structure, with 65 pairs fledging an average of 1.17 chicks per nest (Upton et al. 2000). Breeding success in 2000 averaged 0.78 chicks per nest across 40 colonies, slightly above the long-term average 1986-1999 of 0.72 (Mavor et al. 2001). Productivity in 2001 was generally poor, with complete breeding failure at 8 colonies in Shetland '*undoubtedly caused by food (sandeel) shortage*' (Mavor et al. 2002). Mean body weights of adult birds caught at nests on Foula, Shetland, in 2001 were significantly lower than in previous years (Mavor et al. 2002). At St Abbs Head in 2001, mink were suspected of causing significant chick mortality (Mavor et al. 2002). Low productivity in Shetland in 2002 (0.2 chicks per nest) was attributed to scarcity of sandeels (Mavor et al. 2003). Low productivity in Shetland in 2003 was attributed to scarcity of sandeels; evidence for that included regurgitated food samples mainly comprising fishery discard fragments, and mean body weights of nesting adults being about 30g below normal (Mavor et al. 2004). In Orkney in 2003, many chicks died from mid-July onwards, but were intact in nests (i.e. not victims of predation), suggesting food shortage in Orkney waters but only late during the breeding season in contrast to the situation in Shetland (Mavor et al. 2004). In SE Scotland, kittiwakes achieved high breeding success and evidently found plenty of sandeels as these formed 91% of the diet by weight (Mavor et al.2004). In 2005, productivity was generally higher than in 2004, but in some areas food availability appeared to be low during chick rearing (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for black-legged kittiwakes as follows. Productivity at all monitored colonies in Britain and Ireland averaged 0.54 chicks per pair, below the long-term mean of 0.68 for 1986-2005. Low success in 2006 was primarily attributed to food shortage. For example, an apparent shortage of food, with resultant starvation of chicks, was noted on Noss. However, predation of chicks was considered to have reduced success at Hermaness and Foula. At North Sutor, low productivity was attributed in part to increased predation by great black-backed gulls due to low food availability. On the Isle of May, the body condition of chicks at fledging appeared to be very poor due to a rapid deterioration in feeding conditions late in the breeding season. Brood neglect was high at this time, leaving chicks exposed to weather and predators. On the Farne Islands, predation by large gulls was a problem at some kittiwake sub-colonies. At Lowestoft, many nests were destroyed early in the breeding season by foxes, although this was noted to be unusual. On Ailsa Craig, food became scarce during chick-rearing causing high mortality of chicks, with only 0.14 chicks fledged per nest, the lowest productivity at that colony since 1990. Productivity was also unusually low at Skomer in 2006, with predation by great black-backed gulls thought to be a major factor depressing productivity.

Frederiksen et al. (2004) showed that breeding success of kittiwakes on the Isle of May correlated strongly (and negatively) with sea surface temperature and was lower in years when a commercial sandeel fishery operated in the area. These two factors explained 81% of the variation in kittiwake breeding success.

Table 15.1.1 Meta-analysis of main factors contributing to reduced productivity of Kittiwakes at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Food shortage (often related to climate change)	43
Great skua predation	6
Extreme weather conditions	5
Gull predation	3
Mink predation	2
Fox predation	1
Feral cat predation	1
Rat predation	1
Peregrine predation and disturbance	1

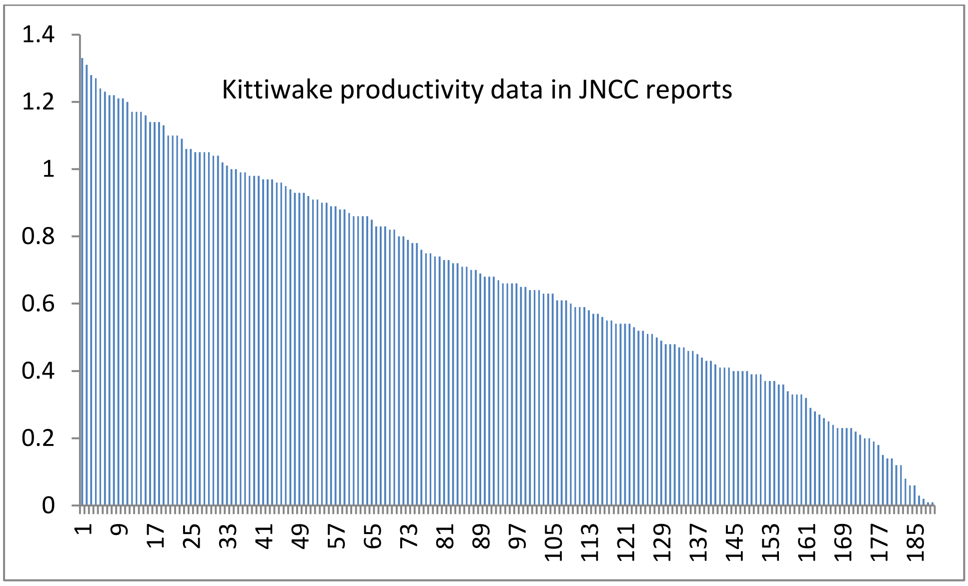

Figure 15.1.5. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of black-legged kittiwakes at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

15.2 Management options

	15.2.1 Closure of sandeel and sprat fisheries in UK waters	15.2.2 Eradicate Mink	15.2.3 Feral cat eradication	15.2.4 Rat eradication	15.2.5 Exclusion of foxes	15.2.6 Exclusion of great skuas	15.2.7 Artificial structures for colonies
Evidence of success for this species	High C=High*	Low C=Mod	Low C=Mod	Unknown C=Mod	Low C=Mod	Moderate C=Mod	High C=High
Evidence for similar species	High C=Mod	High C=High	High C=High	High C=High	Low C=High	Low C=High	High C=High
Cost-effectiveness	Uncertain C=Low	High C=High	High C=High	Low C=High	Low C=High	Moderate C=Low	Low C=High
Feasibility	Moderate C=Low	High C=High	Moderate C=High	Low C=High	Low C=High	Moderate C=High	Moderate C=High
Practicality	Moderate C=Low	Low C=High	Low C=High	Low C=High	Low C=High	Low C=High	Low C=High
Applies at SPA populations	Yes C=High	Few C=High	No C=High	Few C=High	Few C=High	Few C=High	No C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

15.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or throughout UK waters

The single most important factor that could be influenced by management, and which reduces productivity and survival of kittiwakes, appears to be food supply (identified 43 times out of a total of 63 in the meta-analysis of JNCC monitoring data), and especially abundance of sandeels which are the main diet of breeding kittiwakes at almost all UK colonies. Mean foraging range of breeding kittiwakes is around 26 km, but maximum range is at least 100 km, possibly 200 km. Closure of sandeel fishery within 200 km of SPAs would effectively be equivalent to closure of all sandeel fishing in UK waters given the widespread distribution of kittiwake SPAs around the British Isles. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes.

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

15.2.2 Eradication of American mink

Mink are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that relatively few kittiwake colonies could benefit from mink eradication.

Eradication of mink from islands can be achieved relatively easily by trapping. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present. Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a 'mink-free' status for the seabird colony through the summer, resulting in considerably improved productivity. However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012) (see 14.2.3).

15.2.3 Eradication of feral cats

Feral cats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of feral cats would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that relatively few kittiwake colonies could benefit from feral cat eradication.

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000).

15.2.4 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies (JNCC Annual Reports on Seabird Numbers and Breeding Success). Eradication of rats would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that relatively few kittiwake colonies could benefit from rat eradication.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

15.2.5 Exclusion of foxes from colonies

Foxes are a factor reducing productivity at a very few colonies (JNCC Annual Reports on Seabird Numbers and Breeding Success). Fencing exclosure of foxes would allow kittiwake productivity to increase at colonies where this predator is present, but evidence suggests that very few kittiwake colonies could benefit from fencing out foxes.

Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow kittiwake productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect kittiwake colonies from foxes, but might be appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

15.2.6 Exclude great skuas from buffer zone around kittiwake colonies

Several kittiwake colonies are affected by great skua depredations (Votier et al. 2004, 2007, 2008). Evidence indicates that the great skuas that kill kittiwakes tend to be birds nesting close to kittiwake colonies (Furness 1987, Votier et al. 2007). Removal of great skuas and prevention of great skuas establishing territories adjacent to kittiwake colonies could increase kittiwake productivity, but much of this predation stems from food shortage causing great skuas to increase depredations on other seabirds. Although this approach would probably be impossible at colonies where great skuas are an SPA feature, there are many kittiwake colonies where great skuas are not features of SPAs and are only present in relatively small numbers. Removal of great skuas from these sites could increase kittiwake survival and productivity.

15.2.7 Construction of artificial structures to support kittiwake colonies

Kittiwakes will breed on structures such as warehouses overhanging the sea, harbour walls, and even bridges over tidal rivers. So construction of artificial nesting sites for kittiwakes is possible, but in most areas of their breeding range there is no shortage of natural nesting habitat (cliffs), and not all of the potential nesting habitat is occupied, so provision of artificial cliffs would be unlikely to provide useful breeding habitat for this species and would be an expensive measure.

16. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR SANDWICH TERN

16.1 Sandwich tern ecology

Sandwich tern is an Annex 1, Schedule 1 species with SPEC 3 status (Unfavourable conservation status (declining) and concentrated in Europe). The world population is estimated at 160,000 to 170,000 breeding pairs, with three subspecies, one predominantly in Europe, one in North America, and one in South America (Mitchell et al. 2004). The nominate subspecies found in Europe and west Asia (*sandvicensis*) numbers around 90,000 to 100,000 breeding pairs (Mitchell et al. 2004).

16.1.1 The species in the British Isles

Seabird 2000 found 11,000 pairs in Great Britain the Isle of Man and Channel Islands, and 3,700 pairs in Ireland (Mitchell et al. 2004). Sandwich terns nest on the ground on low-lying offshore islets or in remote sand dunes. Most colonies are on North Sea or Irish Sea coasts, in areas with extensive sheltered shallow waters. Sandwich terns nest in a relatively small number of large and dense, highly synchronous, colonies on bare ground. Local colonies may be abandoned in response to predation pressures, competition with gulls, vegetation succession, and coastal erosion processes (Mitchell et al. 2004). The maximum clutch size is normally two eggs, and the mean clutch size is 1.6 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). While breeding they feed predominantly on small pelagic fish, in the British Isles on sandeels, sprats and young herring. Their fishing success is severely hampered by strong winds and rough seas (Dunn 1973, Taylor 1983, Stienen et al. 2000), so they tend to feed mainly in sheltered bays and estuaries (Mitchell et al. 2004).

The GB SPA suite holds around 10,000 pairs (72% of GB population) and there are 16 sites in the UK: Alde-Ore Estuary, Carlingford Lough, Chichester and Langstone Harbours, Coquet Island, Duddon Estuary, Farne Islands, Firth of Forth Islands, Foulness, Larne Lough, Loch of Strathbeg, Morecambe Bay, North Norfolk Coast, Solent and Southampton Water, Strangford Lough, Ynys Feurig Cemlyn Bay and The Skerries, Ythan Estuary Sands of Forvie and Meikle Loch.

Adult survival rate averages 0.87 to 0.94 (Robinson 2010), and the age of first breeding is usually 3 or 4 years (Cramp and Simmons 1977-1994). British Sandwich terns migrate along coasts to winter off west Africa. Other European populations also migrate coastally to broadly the same wintering area, and while some continental birds pass through British waters on migration, most probably remain on the continental side of the North Sea.

16.1.2 Present, and likely future trends

Seabird 2000 estimated the total British and Irish population of Sandwich terns at 14,252 pairs, with 63% in England, 26% in Ireland, 7% in Scotland, and 3% in Wales (Mitchell et al. 2004). This represents a slight decline from the total of 16,047 pairs in 1985-88, but a slightly higher number than the 12,073 pairs present in 1969-70 (Mitchell et al. 2004). Although the overall population in the British Isles has remained fairly stable over recent decades, numbers at individual colonies have often changed dramatically. Decreases at particular colonies have mainly been caused by predators causing breeding failure then abandonment of particular colonies (Mitchell et al. 2004). Mitchell et al. (2004) concluded '*only colonies on [predator-free] offshore islands are immune from attack, and these are scarce within the Sandwich tern's British and Irish range. Restoration of existing offshore islands or creation of new islands from dredge-spoil may be necessary to maintain Sandwich tern populations into the 21st century*'. Changes in numbers since Seabird 2000 are indicated by the JNCC index of breeding numbers from monitoring colonies; a high proportion of the main Sandwich tern colonies is included in this index so it is likely to be reliable. In the UK as a whole, the index of Sandwich tern breeding numbers has remained fairly stable from 1986 to 2011 (Figure 16.1.1). However, the index for England suggests a slight decline in numbers there (Figure 16.1.2), so there may have been some redistribution of birds into colonies in Ireland (where there are too few data for the index to be plotted with confidence). Foster and Marrs (2012) estimated a 48% decrease in the index of breeding numbers of Sandwich terns in Scotland from 1986 to 2011, but numbers breeding in Scotland represent only 5-7% of the total in the British Isles.

Future trends are difficult to predict for this species, although the suggestion of Mitchell et al. (2004) that the species may run out of safe breeding sites in the British Isles would indicate that a decline in numbers would be more likely than an increase, in the absence of sympathetic management.

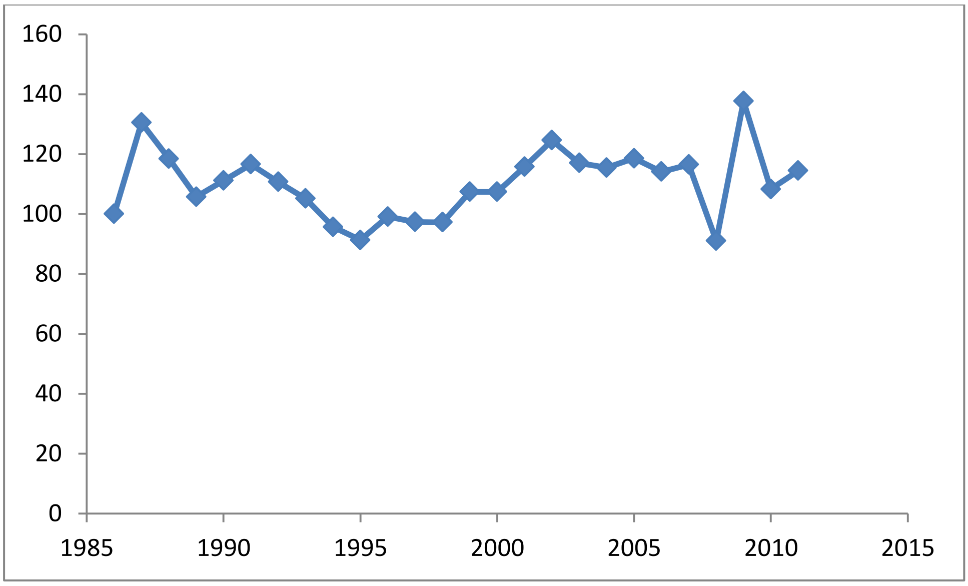

Figure 16.1.1. Sandwich tern breeding population index for the UK. Data from JNCC online database.

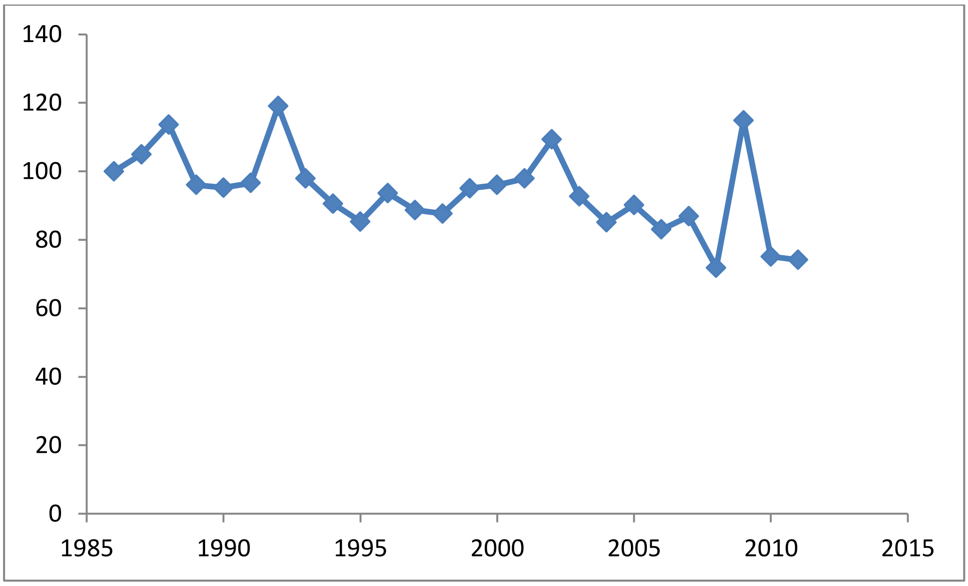

Figure 16.1.2. Sandwich tern breeding population index for England. Data from JNCC online database.

16.1.3 Factors affecting survival rates

Discounting earlier estimates of Sandwich tern survival rate which may have been biased by ring losses (Robinson 2010), the only published estimate of Sandwich tern survival rates that appears to be unbiased, estimated adult survival at 0.899 (standard error 0.029), 2nd/3rd year survival at 0.741 (s.e. 0.206) and 1st year survival at 0.313 (s.e. 0.191) (Robinson 2010). No environmental covariates were included in this analysis, so the influences of environmental factors on survival rates cannot be quantified. Estimation of the extent to which management might alter survival rates is therefore extremely difficult.

16.1.4 Breeding success in the British Isles

In 1986-1988, fox predation was identified as the main cause of reduced productivity of Sandwich terns at monitored colonies (Walsh et al. 1990). In 1989, fox predation at Scolt Head was very severe and few chicks survived (Walsh et al. 1991). In 1990, predation by foxes and flooding by high tides were identified as the two most important factors reducing Sandwich tern productivity at monitored colonies (Walsh et al. (1991). No Sandwich terns nested at Scolt Head in 1990, apparently in response to the breeding failure there in 1989 caused by fox predation (Walsh et al. 1991). In 1991, productivity averaged 0.45 chicks per pair across 16 colonies, the lowest value since monitoring started in 1986. The main impact on productivity identified was predation by foxes, which reduced productivity at Hodbarrow (520 pairs) to 0, at Foulney (332 pairs) to 0.15, at Sands of Forvie (1,115 pairs) to 0.3 (Walsh et al. 1992). In 1992, 360 pairs at Hodbarrow failed to rear any chicks as a result of depredation by foxes. Walsh et al. (1993) reported of Sandwich terns throughout the British Isles '*A major cause of breeding failure is predation by foxes, and where they are successfully excluded productivity can be greatly increased. The impact of foxes is illustrated by the situation at Hodbarrow where they have caused total breeding failure at the chick stage in the last two years. The use of electric fencing to deter foxes from entering colonies is encouraged where practicable*'. In 1993, predation reduced productivity at several colonies: foxes accounted for many of the failures at Strathbeg, and at Hodbarrow all young being raised by 100 pairs were taken by stoats. On Lindisfarne foxes depredated all 40 nests. In contrast, productivity was particularly high at Scolt Head, which was attributed to a good food supply and reduced predation by foxes. High productivity at Blakeney was attributed to a plentiful food supply. Severe weather was thought to have reduced productivity at Strathbeg and in NE Ireland (Walsh et al. 1994). In 1994, productivity was high at 0.84 across 13 colonies. Low productivity at Rye Harbour was attributed to chilling of chicks among tall vegetation during wet weather. Mink were recorded near colonies in NW Ireland, but had little impact on productivity. Foxes were noted as a likely predator in several regions (Walsh et al. 1995). In 1995, productivity averaged 0.66. Predation reduced productivity at several colonies: Havergate suffered complete failure, attributed to foxes, and foxes were also thought to be responsible for complete failure at Foulney. In contrast, at Cemlyn, stoat and crow control was carried out, and productivity was 0.78 chicks per pair (at a colony of 650 pairs). At Hodbarrow, heat stress may have contributed to reducing productivity (Thompson et al. 1996). In 1996, productivity averaged 0.68, and no specific impacts at individual colonies were identified (Thompson et al. 1997). In 1997, productivity averaged 0.55. Flooding by high tides washed out all nests at Scolt Head (productivity 0.0), and some at Blakeney (productivity 0.5). Productivity was reduced by mink and badgers at Dungeness (productivity 0.0), and by badgers at Rye (productivity 0.0). In 1998, productivity averaged 0.49, well below average. A few colonies were affected by bad weather: Scolt Head failed completely due to flooding by high tides, while wet and windy weather at Loch Swilly reduced foraging ability of adults but only reduced productivity slightly (0.88). Several colonies were affected by predators: herring gulls took chicks at Scolt Head, peregrines and large gulls affected productivity slightly at Anglesey (0.98 compared to 1.14 and 1.33 in the two previous years), and rat predation affected the colony at Lady's Island Lake (Thompson et al. 1999). In 1999, mean productivity was 0.53 chicks per pair, lower than in most years. At the Farnes, food shortage resulted in the colony of 1,946 pairs rearing only 292 chicks (0.15). Gull predation reduced productivity on the Isle of May, cold weather was blamed for low productivity at Loch Ryan, and grazing deer were thought to be responsible for extensive damage to eggs on Brownsea Island where productivity was only 0.01 chicks per pair (Mavor et al. 2000). In 2000, productivity was generally good (1 chick per pair). At Loch Ryan, despite some losses to flooding, productivity (0.92) was close to average. There was no sign of food shortage at the Farnes where productivity was 0.56, higher than in the previous year when food had been scarce. Herring gull predation at Blakeney resulted in productivity of 0.0 there (Mavor et al. 2001). In 2001, productivity was close to average in most colonies, but an estimated 50% of chicks at the Farnes died during poor weather in mid-June, and many of the survivors were depredated by gulls (Mavor et al. 2002). In 2002, overall productivity (0.69) was marginally below the long-term average for 1986-2001 (0.71). The colony at Loch Ryan failed (0.0) due to flooding. Lindisfarne failed (0.0) due to oystercatcher predation of eggs. Brownsea failed due to predation (but predator species was not reported) (Mavor et al. 2003). In 2003, mean productivity of 0.8 chicks per pair across 15 colonies was above average. However, productivity at Sands of Forvie (0.71) was reduced by stoat and gull predation, while it was reduced at Blakeney (0.79) by rat predation, and at Langstone Harbour (0.39) by food shortage, while flooding depressed productivity at Lough Swilly and mink reduced breeding success to only 0.12 chicks per pair at Mulroy Bay (Mavor et al. 2004). In contrast, productivity was at least 1 chick per pair at Loch Ryan, Holkham, Rye Harbour, Anglesey, and Hodbarrow, sites where no problems with weather, food supply or predation were reported (Mavor et al. 2004). In 2004, mean productivity across monitored colonies was 0.52 chicks per pair, the lowest for 12 years (Mavor et al. 2005). Breeding success in NE England was depressed by storms in June, while chicks that survived were severely affected by shortage of food (Mavor et al. 2005). In E England colonies were also affected by storms and by food shortage. Predation by foxes resulted in breeding failure at North Solent. In SW England food was plentiful and productivity was high although bad weather and predation accounted for some losses (Mavor et al. 2005). In 2005, productivity was reduced at Sands of Forvie by a presumed shortage of sandeels resulting in only 22 fledged young from 570 pairs, and presumed food shortages were blamed for low productivity at the two main colonies in NE England (Mavor et al. 2006). Predation depressed productivity at Blakeney, bad weather and food shortage reduced productivity at Langstone Harbour, while mink predation reduced productivity at Mulroy Bay to half the normal level (Mavor et al. 2006). In 2006, Sandwich tern productivity averaged 0.77 chicks per pair across twelve colonies, above the long term average, but predators reduced breeding success at Langstone Harbour to the lowest level for four years, and lesser black-backed gulls reduced productivity at Hodbarrow to well below average by eating tern chicks (Mavor et al. 2008). Human recreation activities (and dogs) can cause inadvertent disturbance to tern colonies accessible to walkers, although such disturbance was not identified at any of the colonies monitored in the JNCC annual monitoring of seabird numbers and breeding success. Human disturbance can act in combination with other factors, for example pushing colonies down the beach towards high tide line and therefore increasing risk of flooding or exposure to predation.

Table 16.1.1 Meta-analysis of main factors contributing to reduced productivity of Sandwich terns at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Fox predation	23
Tidal flooding	11
Food shortage	10
Gull predation	9
Extreme weather conditions	9
Mink predation	5
Rat predation	3
Stoat predation	3
Badger predation	2
Crow predation	2
Oystercatcher predation	1
Peregrine predation and disturbance	1
Heat stress	1
Deer	1

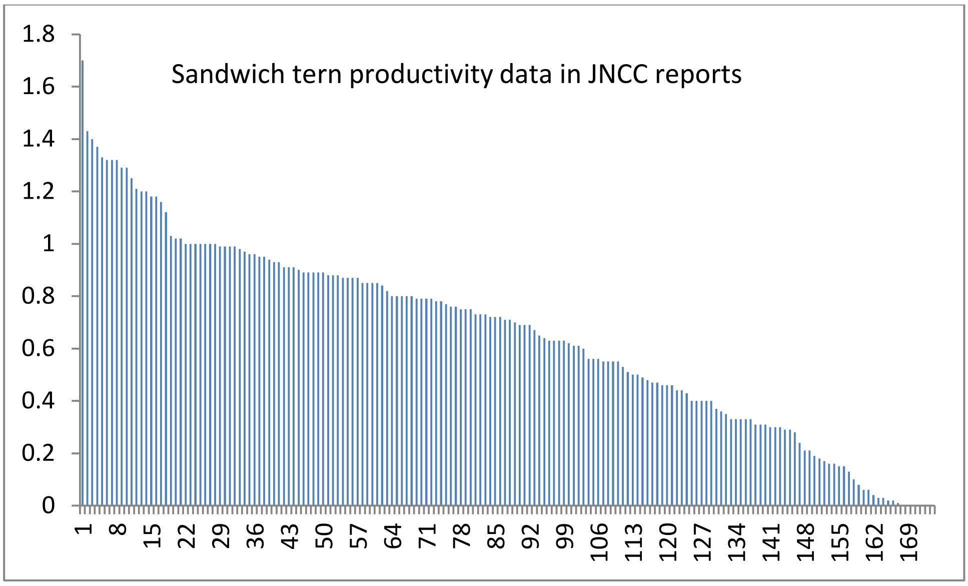

Figure 16.1.3. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Sandwich terns at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

16.2 Management options

	16.2.1 Closure of sandeel and sprat fisheries close to colonies	16.2.2 Eradicate Mink	16.2.3 Eradicate Feral cats	16.2.4 Eradicate Rats	16.2.5 Exclude foxes	16.2.6 Control stoats	16.2.7 Flood control	16.2.8 Exclude large gulls
Evidence of success for this species	Moderate C=Low	High C=High	Low C=Low	Low C=Mod	High C=High	High C=High	High C=High	Mod C=Low
Evidence of success for similar species	High C=Mod	High C=High	High C=High	High C=High	High C=High	High C=High	High C=High	Mod C=Low
Cost-effectiveness	Uncertain C=Low	High C=High	High C=High	High C=High	Moderate C=High	Moderate C=High	High C=Mod	Mod C=Low
Feasibility	Moderat C=Low	High C=High	Moderate C=High	High C=High	Moderate C=High	High C=High	High C=High	Mod C=Low
Practicality	Moderat C=Low	Moderat C=High	Moderate C=High	Low C=High	Moderate C=High	High C=High	Moderate C=High	High C=Mod
Applies at SPA populations	Yes C=High	No C=High	Few C=High	Few C=High	Some C=High	Few C=High	Some C=High	Yes C=High

*Confidence in each score is listed as ‘High’ ‘Moderate’ or ‘Low’ based on amount and consistency of evidence

16.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Mean foraging range of Sandwich terns is around 15 km and maximum foraging range is up to 70 km (Langston 2010), so closure of sandeel and sprat fishing within 60 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider.

However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes.

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental. No Sandwich terns breed in Shetland, so this case study does not provide evidence specifically for Sandwich terns, but the principles are thought to apply.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The body of strongly suggests that Sandwich tern productivity and adult survival would benefit from higher average abundances of these small prey fish. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to the isle of Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

16.2.2 Eradication of American mink

Mink are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien mink would allow Sandwich tern productivity to increase at colonies where this predator is present.

Eradication of mink from islands can be achieved relatively easily by trapping. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present (and most Sandwich tern colonies are either on islands very close to the mainland or are on the mainland). Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between.

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity of gulls (from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

16.2.3 Eradication of feral cats

Feral cats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of feral cats would allow Sandwich tern productivity to increase at colonies where this predator is present.

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000). An alternative to eradicating feral cats may be cat-proof fencing to protect colonies. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

16.2.4 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of invasive alien rats would allow Sandwich tern productivity to increase at colonies where this predator is present.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

An alternative to eradicating rats that would probably be more appropriate where colonies are on mainland sites or on islands close to the mainland coast, may be rat-proof fencing to protect colonies. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

16.2.5 Exclusion of foxes from colonies

Foxes are a factor reducing productivity at several colonies including important SPA populations, and their depredations appear to reduce adult survival although that impact has not been quantified. Exclosure fencing to keep foxes out of colonies would allow Sandwich tern productivity to increase at colonies where this predator is present.

Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland. Fencing of colonies to exclude foxes would allow Sandwich tern productivity to increase at colonies where this predator is present.

In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect colonies just from foxes, but might be especially appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is remarkably expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013 http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

16.2.6 Control of stoats close to colonies

Stoats are a factor reducing productivity at several colonies including important SPA populations, and their depredations appear to reduce adult survival although that impact has not been quantified. It may be necessary to deploy trapping to reduce stoat numbers at colonies or to eradicate them from islands. However, a more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Deploying such predator-proof fencing around colonies affected by stoats would allow Sandwich tern productivity to increase at colonies where this predator is currently present and reducing productivity.

16.2.7 Protection of colonies from flooding or engineering of new nesting habitat in safer locations

Flooding (by unusually high tides and storm-driven waves) has been identified as a factor reducing Sandwich tern productivity in some years at some colonies (the JNCC meta-analysis indicated this to be the main impact in at least 11 cases out of about 90 and to be the second most frequently identified impact). Engineering work to improve sea defences at colonies and to increase the amount of suitable nesting habitat in areas less at risk from flooding (because Sandwich terns need bare ground for nesting and are displaced from some nesting areas by excessive growth of vegetation) could greatly increase productivity by providing safer nesting sites. Costs would be extremely site-specific, but this seems a likely approach to identify since Sandwich terns select areas of bare ground for nesting and such sites tend to degrade due to natural processes of erosion or colonisation by vegetation. Creation of bare ground in well protected areas away from flooding risk would therefore seem to be a cost effective approach to increasing Sandwich tern productivity at some colonies. Such engineering work may need to be carried out every few years (outside the tern breeding season), depending on the mobility of coastal habitat and the rate of vegetation development on specific sites. Sites where such engineering work might be most cost-effective should be selected on the recent history of flooding and the ease with which engineering might be carried out. It would be important to monitor the effectiveness of such management, in relation to the limited plasticity of colony site characteristics and population distribution.

16.2.8 Exclude large gulls from nesting close to colonies

Predation by large gulls affects productivity at some colonies in some years (this was identified by the meta-analysis of JNCC data to be the third most frequently identified cause of reduced productivity, reported 9 times). Since there are relatively few important Sandwich tern colonies, removal of gull territories adjacent to these key colonies could improve productivity of Sandwich terns. It is likely that most of the Sandwich tern colonies affected by gull predation are affected by small numbers of gulls that have developed specialised habits of feeding on tern eggs and chicks, and so the numbers of gulls that would need to be removed to increase tern productivity are likely to be small.

17. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR COMMON TERN

17.1 Common tern ecology

Common tern is listed on Annex 1 and Schedule 1. The world population is around 460,000 to 620,000 pairs, in three subspecies, *hirundo*, *longipennis*, and *tibetana*. The nominate subspecies, *hirundo*, is widely distributed in Europe, the Middle East, west Asia, and North America. Within Europe the population is around 220,000 to 340,000 pairs (Stroud et al. 2001).

17.1.1 The species in the British Isles

Seabird 2000 found 10,000 pairs of common terns in Great Britain the Isle of Man and the Channel Islands, and 4,200 pairs in Ireland (Mitchell et al. 2004). Common terns nest on the ground in colonies that vary in size from a few pairs up to many hundreds of pairs, but very rarely reaching 1,000 pairs as birds feed on fish within a few km of the colony so large colonies become limited by local food supply. Colonies are usually on islets close to the shore, but can be on man-made structures including flat rooftops, floating platforms, derelict jetties, abandoned lock gates. Common terns nest on man-made structures more than any other seabird in the British Isles. Breeding sites are often abandoned if mammal predators can gain access, or if breeding numbers of gulls increase. Most pairs lay two or three eggs, the mean clutch size being 2.6 (Cramp and Simmons 1977-1994, del Hoyo et al. 1992-2006). Common terns feed on small fish, but may take a wider range of fish species than caught by Arctic terns or Sandwich terns. Nevertheless, most colonies depend on local stocks of either sandeels, or sprats or young herring. Common terns tend to favour more sheltered feeding areas than Arctic terns, and their fishing success is reduced by strong wind or rough sea (Cramp and Simmons 1977-1994).

The GB SPA suite holds around 5,700 pairs of common terns (46% of the population). The UK SPA suite holds 6,993 pairs, across 22 sites: Breydon Water, Carlingford Lough, Coquet Island, Cromarty Firth, Dungeness to Pett Level, Farne Islands, Firth of Forth Islands, Foulness, Glas Eileanan, Inner Moray Firth, Larne Lough, Lough Neagh and Lough Beg, Monach Isles, North Norfolk Coast, Poole Harbour, Ribble and Alt Estuaries, Solent and Southampton Water, Strangford Lough, The Dee Estuary, The Wash, Ynys Fenrig Cemlyn Bay and The Skerries, Ythan Estuary Sands of Forvie and Meikle Loch.

Adult survival rate averages 0.88 (del Hoyo et al. 1992-2006), and age of first breeding is usually 3 or 4 years old (Cramp and Simmons 1977-1994). British common terns migrate along the coast to winter off west Africa. Continental populations of common terns winter in a broadly similar area, and while some continental birds may migrate through British waters, most probably pass southwards along the eastern side of the North Sea.

17.1.2 Present, and likely future trends

Seabird 2000 estimated the population of common terns in Britain and Ireland at about 14,500 pairs, with 33% in England and the Channel Islands, 33% in Scotland, 29% in Ireland, and 5% in Wales (Mitchell et al. 2004). This total is only 2% lower that the numbers found in the SCR Census in 1985-88 and in Operation Seafarer in 1969-70. However, the apparent stability hides major changes in distribution.

Although relatively stable in the UK as a whole (Figure 17.1.1), numbers have decreased in Scotland (Figure 17.1.2), but increased since 1986 in England (Figure 17.1.3) and probably also in Ireland (where survey data are less complete). Foster and Marrs (2012) estimated a 43% decrease in the index of breeding numbers of common terns in Scotland from 1986 to 2011. The future trends in breeding numbers are difficult to predict, but the common tern is vulnerable to impacts of predators at colonies and the decline in Scotland in recent years probably reflects at least in part the lack of sandeels in Shetland and other northern areas in recent years.

Common tern colonies can be displaced by large gulls moving in to breed. There is clear evidence that such displacement reduces tern numbers locally, but terns may be able to re-locate to areas without nesting gulls, and at least in some cases the regional population may be unaffected even when large tern colonies have been displaced by gulls (Jennings et al. 2012). But in some places, alternative nesting habitat for terns may not be available; management options may therefore include either provision of artificial nesting sites for terns, or removal of gulls from existing tern colony areas.

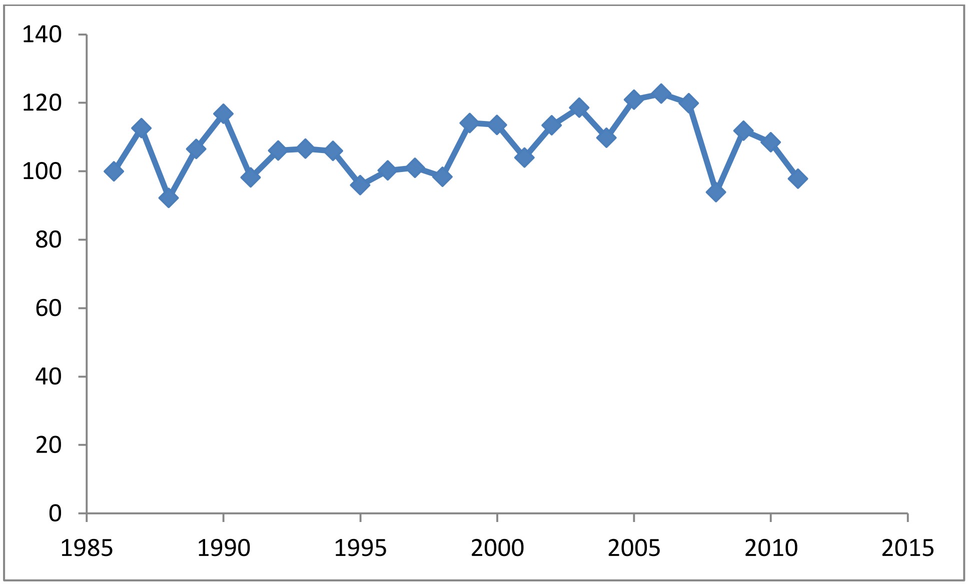

Figure 17.1.1 Common tern breeding population index for the UK. Data from JNCC online database.

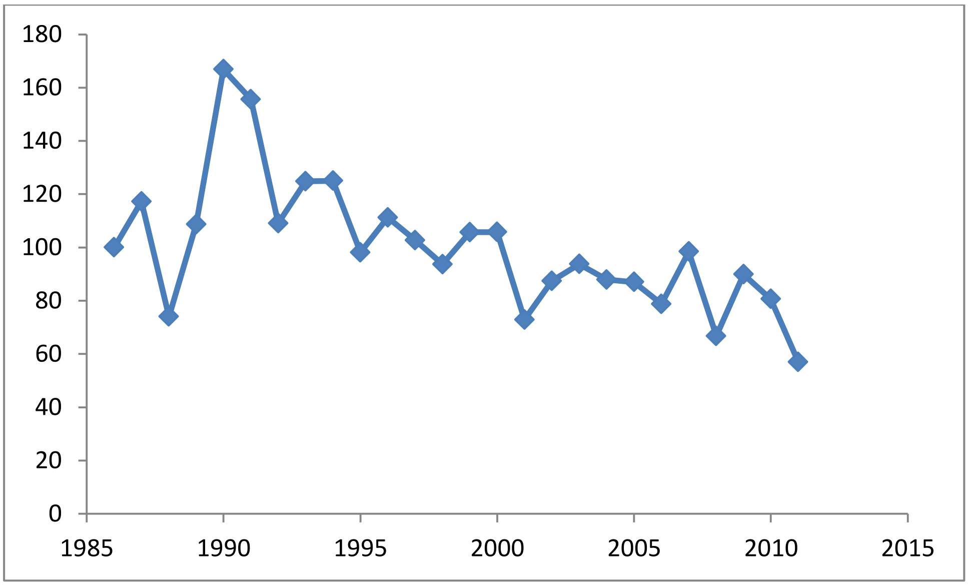

Figure 17.1.2. Common tern breeding population index for Scotland. Data from JNCC online database.

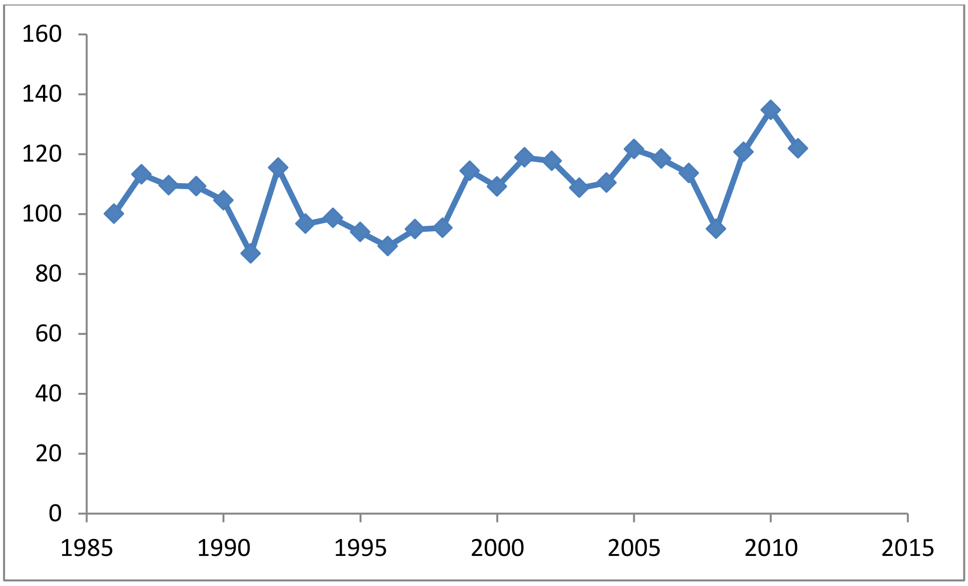

Figure 17.1.3. Common tern breeding population index for England. Data from JNCC online database.

17.1.3 *Factors affecting survival rates*

Del Hoyo et al. (1992-2006) indicated a survival rate of 0.88 for adult common terns. Nisbet and Cam (2002) estimated a survival rate of 0.88 (s.e. 0.04) for adult common terns at a North American colony. Becker and Ludwigs (2004), working at a colony in Germany, estimated adult survival at 0.9 and survival from fledging to age 2 at 0.47. Ezard et al. (2006) presented an estimate of adult common tern survival of around 0.9 for this same common tern colony, but also pointed out that population trend in numbers related more to the influence of environmental variation, especially small pelagic fish abundance, on demography than to the distance from environmental carrying capacity. However, none of these analyses quantified effects of environmental factors on survival rates. Craik (1995, 1997 and 2007) showed that invasive alien predators such as mink can strongly affect adult common tern survival as well as colony productivity, so predation impacts are likely to alter survival rates, as may pelagic fish abundance as suggested by Ezard et al. (2006). However, lacking quantitative assessments of the impacts of these factors, it is only possible to make qualitative or semi-quantitative estimates of consequences of management on survival rates of common terns.

17.1.4 *Breeding success in the British Isles*

In 1986-1989, predation by mink in the west of Scotland and by gulls and skuas in Shetland (combined with a shortage of sandeels around Shetland) were identified as factors reducing common tern productivity in those areas; by far the lowest productivity occurred in Shetland (0.035 chicks per pair in 1986-1989 compared with levels between 0.3 and 1 chick per pair elsewhere) (Walsh et al. 1990). In 1990, Walsh et al. (1991) concluded '*Reduced foraging success for [common] terns and direct losses of chicks through exposure may have contributed to poor success rates*' but indicated that predation was the most obvious cause of reduced productivity in this species. At least 12 out of 29 Scottish colonies monitored suffered total breeding failure, with mammalian predators (mink, otters, foxes) responsible in most cases. Poor weather and predation by gulls were also considered to be contributing factors. In 1991, productivity of common terns was low (averaging 0.4 chicks per pair in Scotland and 0.5 chicks per pair in England). In parts of Scotland food shortage was an issue, with abnormally small clutch sizes and adults deserting colonies. Predation affected many colonies: foxes, gulls, otters and mink were identified as predators at many sites in Scotland. In England, flooding, predation and human disturbance were listed as the main problems at most colonies. However, it was noted that success was highest at sites where artificial nesting rafts were provided for common terns (Walsh et al. 1992). In 1992, food shortage was reported at several colonies, including Lady's Island Lake (0.2 chicks per pair) and Ribble Marshes (almost total failure of >500 pairs). Predation was also reported: mink affected productivity at many colonies in west Scotland, and fox predation affected productivity at Hodbarrow (Walsh et al. 1993). Walsh et al. (1993) also noted that productivity was much higher at artificial sites than at natural colonies (on average about 30% higher) because predation and disturbance levels were much lower at artificial sites. In 1993, food shortages were quoted as a factor in reduced productivity at some colonies. Predation reduced productivity at several colonies: mink were the main problem in west Scotland, rats were responsible for taking many chicks at Shotton (Wales), although productivity there was still high at 1.22, but lower than the 1.45 achieved there in 1992. Predation by stoats, gulls and oystercatchers continued to be a problem at several colonies in NW England. At artificial sites throughout Britain success was again relatively high, as these sites provide improved protection against disturbance and predation compared to natural sites (Walsh et al. 1994). In 1994, mink caused some whole-colony failures, but mink control at several colonies in Argyll allowed productivity of 1 to 1.6 chicks fledged per pair (Walsh et al. 1995). In 1995, the main factor reducing productivity at many colonies was predation: mink at several colonies in Argyll, feral cats and mustelids at Nigg, mink and gulls at Dungeness, kestrels at North Solent NNR and peregrines at Cemlyn (Thompson 1996). In 1996, five colonies in Argyll where mink were trapped out had productivity of 1.3 fledglings per pair whereas 14 colonies where there was no mink control had a mean productivity of only 0.1 fledglings per pair). Mink removal boosted productivity by a factor of 13. However, several colonies were also affected by birds of prey, fox and otter predation. Predators affected productivity at several other colonies: stoats and polecat ferrets were trapped at the colony at St Fergus, the Isle of May colony suffered heavily from gull predation, while low water levels due to dry weather allowed mammal predators access to several colonies in SE England. In contrast, flooding reduced productivity at Rockliffe Marsh (Cumbria). At St Fergus, productivity was reduced by combined impacts of shortage of food, poor weather and human disturbance (Thompson et al. 1997). In 1997, five colonies in Argyll where mink were trapped out produced 722-942 fledglings from 873 pairs (productivity 0.83 to 1.08) whereas six colonies where there was no mink control fledged only 13-23 young from 156 pairs (productivity 0.08 to 0.15). Mink removal boosted productivity by a factor of 5.5 to 13.5. However, several of these eleven colonies were also affected by peregrine and otter predation. At St Fergus, productivity was only 0.1, as a consequence of lack of food, bad weather and disturbance. High tides washed out the majority of nests at six colonies in East England (productivity 0.3), while predation by badgers and foxes affected Dungeness and Rye Harbour while mink affected the colony at Dungeness (Thompson et al. 1998). In 1998, mink control at several colonies in Argyll increased productivity (814 pairs in six colonies fledged 469 chicks) by a factor of 2.5 to 3.4 above levels in colonies where mink were not trapped (139 pairs at ten colonies fledged 28 to 34 chicks) (Thompson et al. 1999). Elsewhere, weather affected several colonies: high tides and waves affected 15 pairs at Newton which only fledged two chicks, high tides and strong winds on the Isles of Scilly resulted in 130 pairs fledging between 20 and 30 chicks (in colonies possibly also affected by presence of brown rats), and bad weather at Rockabill was blamed for relatively poor productivity there (though the 0.96 chicks fledged per pair was well above the average for this species) (Thompson et al. 1999). In 1999, productivity averaged 0.82 chicks per pair. Productivity at Argyll colonies where mink were controlled was 6 times higher (0.66) than at colonies without mink control (0.11). At McDermott's Yard, feral cats caused total breeding failure. The Farnes colony suffered from food shortage and produced only 0.07 chicks per pair. Fox and badger predation reduced productivity at Dungeness (0.03) and Rye Harbour (0.0). Poor weather washed away nests at Lough Swilly but no productivity estimate was provided (Mavor et al. 2000). In 2000, productivity was generally good, averaging 1.09 chicks per pair. Otters greatly reduced productivity at Glas Eileanan (0.2) and Torinturk (0.16). Flooding reduced productivity at two sites in SW Scotland. Human disturbance at Alness Point reduced productivity there to 0.18, while a combination of predation, bad weather and food shortage resulted in productivity of only 0.14 at St Fergus, and predation by great black-backed gulls reduced productivity at Brownsea to 0.51 chicks per pair (Mavor et al. 2001). In 2001, overall productivity was 1 chick per pair, but lower in Scotland

(0.58 chicks per pair), apparently to a large extent due to sandeel scarcity (Mavor et al. 2002). In addition, several colonies were affected by predation and a few by weather. In Argyll, colonies were affected by mink, otter and peregrine predation. However, in north Scotland, productivity (0.71) was close to average despite some predation and disturbance (Mavor et al. 2002). Predation by black-headed gulls reduced productivity at Loch of Strathbeg to 0.16, poor weather and increased gull predation reduced productivity on the Farnes to 0.3, predation by great black-backed gulls reduced productivity at Newton, Isle of Wight, and predation (predator not identified) caused big losses at Hodbarrow where productivity was 0.0 and at Rockliffe Marsh (0.33). High winds reduced hatching success at Seaforth (Mavor et al. 2002). In 2002, overall productivity was similar to the average for earlier years. Reductions in productivity were attributed to predation, weather, and food shortage. Predation affected Glas Eileanan (otters reduced productivity to 0.0), Loch Melfort (peregrines reduced productivity to 0.71), St Fergus (carrion crows reduced productivity to 0.07), Blakeney (gulls reduced productivity to 0.27), Brownsea (predation and flooding reduced productivity to 0.06), and colonies in north Scotland (otters). Weather reduced productivity at colonies in north Scotland, Langston Harbour (0.08), Hayling Island (0.29), Brownsea (0.06), and Shotton (1.05). Food shortage affected colonies in north Scotland (Mavor et al. 2003). In 2003, productivity was above average. In Argyll, the highest productivity was achieved by a colony nesting on a converted mink-free mussel raft, the only monitored site to achieve over 2.1 chicks per pair (Mavor et al. 2004). Food shortage reduced productivity at Nigg (0.0) and at Langstone Harbour (0.23). Predators reduced productivity at many colonies: peregrines caused breeding failure at Sgeir nan Caillich (0.0), and at Dubh Sgeir (0.0), and reduced productivity at Rockabill to 1.17 which is lower than normal at that site (Mavor et al. 2004). Foxes and cats affected Alness and McDermotts, stoats and gulls affected Sands of Forvie (0.09), gulls affected Boultham Mere (0.12), Holkham (0.0), Blakeney (0.27), and Isles of Scilly. Mink killed over 200 chicks at Shotton to reduced productivity to 0.99 at this normally productive colony (Mavor et al. 2004). The colony at Breydon Water was affected by a helicopter, but birds relaid and productivity was close to normal (Mavor et al. 2004). Weather conditions had little impact, although they contributed to reduction in productivity on the Isles of Scilly and flooded a few nests at Chichester Harbour (Mavor et al. 2004). In 2004, poor weather combined with food shortages affected productivity in most regions (Mavor et al. 2005). At ten sites in Argyll where mink were successfully removed, productivity was 1.42 chicks per pair, compared to only 0.11 at five sites with no mink control (Mavor et al. 2005). At Sands of Forvie, avian predation and food shortage resulted in only one chick fledging from 19 pairs (Mavor et al. 2005). On the Isle of May, productivity of only 0.1 chicks per pair was attributed to scarcity of the preferred prey. Storms and food shortage depressed productivity at Coquet Island, and at all monitored colonies on the north Norfolk coast. Mink affected productivity at Alton Water with only four young fledged from 35 pairs on natural sites, but 17 pairs nesting on a raft fledged at least 30 young (Mavor et al. 2005). Storms, tidal inundation and fox predation affected productivity at colonies in SE England, while colonies in SW England were affected by storms, and predation by rats (Mavor et al. 2005). Storms caused egg losses at Shotton, Wales, but birds relaid and overall productivity was good. Low productivity in NW England was attributed to food shortage (Mavor et al. 2005). In 2005, presumed food shortage depressed productivity in west and north Scotland, while predation and food shortage reduced productivity at colonies in England (Mavor et al. 2006). Depredation by otters, herring gulls and mink, and starvation of chicks caused productivity in Argyll of less than half the long-term average (Mavor et al. 2006). Tidal inundation depressed productivity at Blakeney Point, Holkham and Scolt Head colonies and at the last of these sites food shortage was also evident (Mavor et al. 2006). On the Isles of Scilly, productivity was reduced by human disturbance as well as by predation by gulls and oystercatchers (Mavor et al. 2006). At Seaforth, depredations by moorhens and bad weather affected productivity. At Big Copeland, disturbance by black-headed gulls reduced productivity (Mavor et al. 2006). In 2006, particularly high productivity figures were recorded at common tern colonies in N Scotland, SE Ireland and Wales, but predation depressed productivity at several sites across Britain (Mavor et al. 2008). In SW Scotland, otters ate chicks at Sgeir na Caillich, the largest colony in the region, and greatly reduced productivity at Eilean nan Gabhar. Controlling mink at two colonies resulted in productivity above 1 chick per pair, whereas at colonies where mink were not trapped, productivity averaged below 0.5 chicks per pair (Mavor et al. 2008). At several colonies, including the Isle of May SE Scotland, Eilean an Ruisg, SW Scotland, and Havergate Island, E England, depredations by herring gulls depressed productivity. At Langstone Harbour, productivity was low due to predators, while at the Isles of Scilly productivity was low due to tides flooding the colony. Food shortage affected productivity at Seaforth, NW England, and at Coquet Island, NE England (Mavor et al. 2008). Human recreation activities (and dogs) can cause inadvertent disturbance to tern colonies accessible to walkers, and such disturbance was identified at several of the colonies monitored in the JNCC annual monitoring of seabird numbers and breeding success (Table 17.1.1). Human disturbance can act in combination with other factors, for example pushing colonies down the beach towards high tide line and therefore increasing risk of flooding or exposure to predation.

Ratcliffe et al. (2006) showed that eradiation of mink in the Western Isles improvced hatching success of terns, but in their study the fledging success was no higher than in control colonies where mink were not removed, owing to overwhelming impacts of food shortage and poor weather on chick survival. Craik (2008) suggested that gull productivity provides a better measure of the impact of mink because gull productivity is less sensitive to weather and food supply than that of terns, a point accepted by Ratcliffe (2008).

Table 17.1.1 Meta-analysis of main factors contributing to reduced productivity of Common terns at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	**Cases reported**
Food shortage	47
Mink predation	42
Gull predation	24
Extreme weather conditions	22
Tidal flooding	19
Otter predation	15
Fox predation	14
Peregrine predation and disturbance	8
Human disturbance	8
Feral cat predation	5
Rat predation	5
Stoat predation	5
Badger predation	4
Great skua predation	3
Oystercatcher predation	3
Ferret predation	2
Buzzard predation	2
Crow predation	2
Kestrel predation	1
Moorhen predation	1
Helicopter disturbance	1

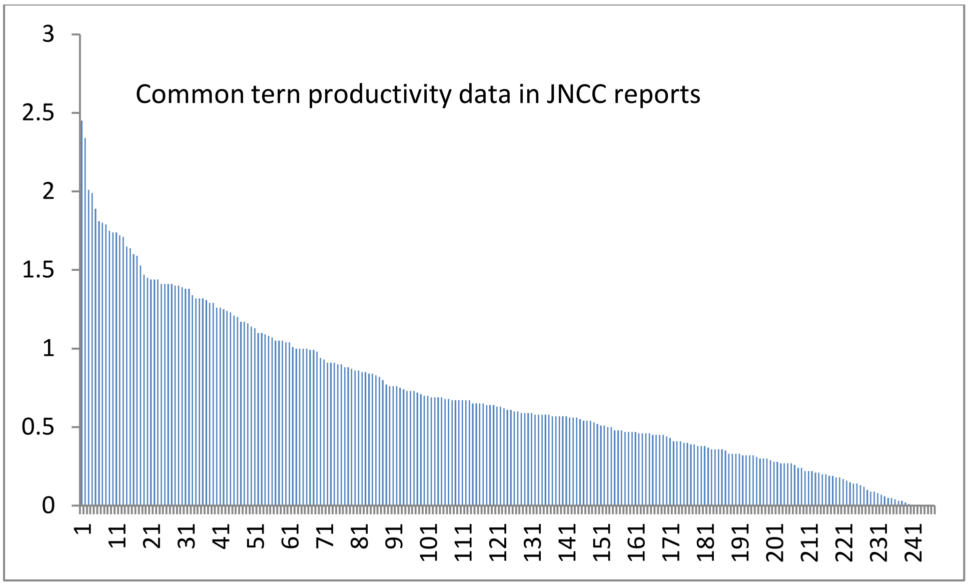

Figure 17.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of common terns at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

17.2 Management options

	17.2.1 Closure of sandeel and sprat fisheries close to colonies	17.2.2 Eradicate mink	17.2.3 Eradicate feral cats	17.2.4 Eradicate rats	17.2.5 Exclude foxes	17.2.6 Control stoats	17.2.7 Nest platforms	17.2.8 Exclude large gulls
Evidence of success for this species	Low C=Low	High C=High	Low C=Low	High C=High	High C=High	High C=High	High C=High	High C=High
Evidence of success for similar species	High C=Mod	High C=High	High C=High	High C=High	High C=High	High C=High	High C=High	High C=High
Cost-effectiveness	Moderate C=Low	High C=High	High C=High	High C=High	Mod C=High	Mod C=High	High C=High	High C=High
Feasibility	Moderate C=Low	High C=High	Mod C=High	High C=High	Mod C=High	High C=High	High C=High	Mod C=High
Practicality	Moderate C=Low	Moderate C=High	Mod C=High	Mod C=High	Mod C=High	High C=High	High C=High	Mod C=High
Applies at SPA populations	Yes C=High	No C=High	Few C=High	Few C=High	Few C=High	Few C=High	Yes C=High	Some C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

17.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. In the meta-analysis of JNCC data this factor came out top of the list, being identified as the cause of reduced productivity of common terns on 47 occasions. Mean foraging range of common terns is around 9 km and maximum foraging range is up to 37 km (Langston 2010), so closure of sandeel and sprat fishing within 30 km of SPAs should increase productivity and adult survival. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds (but not including common tern). They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant (p<0.0001). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes.

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The body of evidence strongly suggests that Sandwich tern productivity and adult survival would benefit from higher average abundances of these small prey fish, but there is no suitable evidence base to test this for common tern, so the assumed benefit is based on the similar ecology of these two species. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

Closure of sandeel and sprat fisheries close to common tern colonies (or to common tern SPAs) would close a smaller total area, but since there are 22 sites across the UK designated as SPAs for common terns, closures within 30 km of all of these 22 sites would be a complex management arrangement and it might be simpler to close all UK waters to these fisheries, with consequent benefits for other wildlife and predatory fish.

17.2.2 Eradication of American mink

Mink are a factor reducing productivity at several colonies, and their depredations reduce adult survival although that impact has not been quantified (Ratcliffe et al. 2008). Eradication of invasive alien mink would allow common tern productivity to increase at colonies where this predator is present.

Eradication of mink from islands can be achieved relatively easily by trapping. However, mink swim well, and are likely to re-colonise islands that are within 2 km of habitat with mink present (and most common tern colonies are on islands very close to the mainland or are on the mainland). Eradication of mink from mainland areas is more difficult, although there is evidence from mink eradication projects in the highlands of Scotland and in Iceland that mink can be eradicated from river catchments and in such cases re-colonisation by mink from adjacent river catchments may be inhibited by upland areas in between. The Hebridean Mink Project run by Scottish Natural Heritage provides a very relevant example of mink eradication benefitting terns (Ratcliffe et al. 2006).

Where eradication of mink is difficult to achieve due to re-colonisation from adjacent areas, Craik (2007) showed that trapping mink in the immediate vicinity of seabird colonies in late winter and early spring normally leads to a ‘mink-free’ status for the seabird colony through the summer, resulting in considerably improved productivity of gulls (from around 0.2 chicks per pair up to around 0.8 chicks per pair). However, trapping mink every year represents a long-term commitment that will be more expensive in the long term than eradication if that can be achieved. An alternative to annual control of mink may be mink-proof fencing to protect colonies, such that the need for annual trapping is removed once the area has been fenced and mink within the fenced area have been removed. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 17.2.5 below).

17.2.3 Eradication of feral cats

Feral cats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Eradication of feral cats would allow common tern productivity to increase at colonies where this predator is present.

Eradication of feral cats can be more complex than eradication of rats, especially where there is a resident human population with pet cats, as experienced on Ascension Island (Ratcliffe et al. 2009). In that example, large numbers of pet cats were killed by the poison deployment (38% of 168 domestic cats were accidentally killed) despite efforts to keep domestic pet cats safe (which had all been neutered as part of the project). Nevertheless, that eradication successfully removed all of the feral cats and has resulted in re-establishment of seabirds that had been extirpated from the main island (Ratcliffe et al. 2009). This project cost £650,000 over three years, with 24,200 person-hrs of cat control work and 14,480 person-hrs of monitoring work on an island of 97 km^2 (9,700 ha so a cost of £67 per ha which is not far from the costs per unit area of rat eradication programmes on UK islands) but where fieldwork is made relatively easy by the climate, terrain and existence of roads (Ratcliffe et al. 2009). Cats have been successfully eradicated from many islands, the largest being

Marion Island (290 km^2) in the sub-Antarctic Indian Ocean, where cats were eradicated in 2000, but this required a 15-year programme including shooting and poisoning and hunting with dogs and introduction of disease (feline panleucopaenia virus) to eradicate a population of around 3,400 cats at the start of the programme that were estimated to be killing over 450,000 seabirds per year (van Aarde 1980, Bloomer and Bester 1992, Bester et al. 2000). The second-largest island from which cats have been eradicated is Macquarie Island (120 km^2) in the sub-Antarctic which had a population of around 2,400 cats, and where the programme of trapping, hunting and deployment of the poison 1080, took 25 years before success was achieved around 2000 (Copson and Whinan 2001). Obviously these very long programmes are much more expensive, and the long timescale required can be attributed to relatively limited expertise in this type of work when those two projects started, and major logistical difficulties of working on uninhabited sub-Antarctic islands far from the mainland. Nogales et al. (2004) provide a useful tabulation of data about 48 islands where eradication of cats has been carried out and summary information on these eradication programmes, indicating that most programmes are now completed within 1-3 years, although none of the projects reviewed in that paper were in the British Isles. Although trapping, shooting, hunting with dogs, poisoning with baits and introducing disease have been the main methods used to eradicate cats, virus-vectored immune-contraception may hold promise in the near future (Courchamp and Cornell 2000). An alternative to eradicating feral cats may be cat-proof fencing to protect colonies. Predator-proof fencing to protect vulnerable populations of wildlife has been deployed very effectively in Hawaii at Ka’ ena Point Natural Area Reserve (Young et al. 2012) (see 16.2.5 below).

17.2.4 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies. Eradication of invasive alien rats would allow common tern productivity to increase at colonies where this predator is present.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a professional New Zealand company with considerable experience, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

17.2.5 Exclusion of foxes from colonies

Foxes are a factor reducing productivity at several colonies, particularly in England, and their depredations appear to reduce adult survival although that impact has not been quantified. Exclosure fencing to keep foxes out of colonies would allow common tern productivity to increase at colonies where this predator is present.

Foxes can be controlled on small islands by shooting, although recolonisation from the mainland may be an issue for islands situated within 1 or 2 km of the mainland.

Fencing of colonies to exclude foxes would allow common tern productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require some maintenance. A more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Such completely predator-proof fencing may be unnecessary to protect colonies just from foxes, but might be especially appropriate for colonies subject to predation by rats or mink as well as by foxes.

By 2006, in total, around 109 km of predator-proof fencing had been erected in various areas of mainland New Zealand to exclude predators from sites with important populations of native animals and birds (Scofield et al. 2011, Innes et al. 2012, Scofield and Cullen 2012). This predator-proof fencing is expensive, costing around £100 per m to construct, and around £1 per m per year to maintain, with a life-span in New Zealand of around 25 years, so a considerable rate of depreciation (Scofield et al. 2011). However, maintenance costs and life span will depend very much on the environment where the fencing is set up. In New Zealand, where much of the fencing is in forested habitat, trees falling onto the fence can cause expensive damage, as can cyclones (Scofield et al. 2011). In the predominantly open habitat of UK seabird colonies such fencing would be under less risk of damage, although corrosion from salt spray would be a consideration. There are several companies in New Zealand providing predator-proof fencing, for example see:
http://www.xcluder.co.nz/
http://www.meshindustries.com/Home
http://www.pestprooffences.co.nz/

There are several examples of the use of predator-proof fences to protect seabirds from mammals. A predator-proof fence completed in 2007 stretches 10.6 km across the neck of the peninsula from coast to coast at Cape Kidnappers Peninsula, North Island, New Zealand. This fence protects a privately owned and financed seabird restoration project where grey-faced petrels and Cook's petrels are being re-introduced (Cooper 2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels).

A good example of successful deployment of a predator-proof fence to protect a seabird colony is one erected in 2001 to protect 36-ha on Pitt Island (Chatham Islands, New Zealand) from feral cats and pigs. Between 2002 and 2005, 200 endangered Chatham petrel chicks from the only known breeding site on South East Island (Chatham Islands) were moved into the fenced reserve. In 2012, 17 pairs from these translocated birds returned to breed, producing 15 chicks http://www.chathams.co.nz/index.php/naturalheritage/116-chatham-petrel-recovery

Cooper (2013) http://www.acap.aq/index.php/en/news/latest-news/1359-predator-proof-fences-are-helping-to-protect-procellariiform-seabirds-including-acap-listed-albatrosses-and-petrels lists a further ten examples of deployment of predator-proof fencing around seabird colonies in New Zealand, Hawaii (USA) and Azores (Portugal).

17.2.6 Control of stoats close to colonies

Stoats are a factor reducing productivity at a few colonies, and their depredations appear to reduce adult survival although that impact has not been quantified. Stoat removal or exclusion would allow common tern productivity to increase at colonies where this predator is currently present and reducing productivity. It may be necessary to deploy trapping to reduce stoat numbers at colonies or to eradicate them from islands. However, a more expensive but more effective alternative is the use of predator-proof fences, as deployed in Hawaii at Ka' ena Point Natural Area Reserve (Young et al. 2012). These 2 m tall fences were set up in November 2010 to February 2011 around 20 ha of coastal habitat within Ka' ena Point to prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012). Deploying such predator-proof fencing around colonies affected by stoats would allow common tern productivity to increase at colonies where this predator is currently present and reducing productivity.

17.2.7 Construction and deployment of predator-proof nesting rafts

Evidence indicates that common terns readily take to nesting on purpose-built rafts, and that productivity on these artificial colonies is considerably higher than at natural colonies. Provision of predator-proof nesting rafts for common terns appears to be the most effective way to increase productivity in this species in the UK, and is almost certainly the most cost-effective approach. Such rafts are not inaccessible to avian predators of course, but evidence from their deployment is that avian predators rarely have a significant impact on productivity of colonies on rafts (Clive Craik pers. comm.; see also text below reviewing achieved productivity on rafts).

The outstanding case study of provision of nesting rafts for common terns is work being done by Dr Clive Craik in Argyll. Initially, he developed a derelict mussel farming raft on which a few pairs of common terns had started nesting. This raft was a wooden rectangular structure moored in a sea loch in Argyll which had been used for many years to grown mussels on ropes hanging from the raft. Clive enhanced the raft for terns, adding gravel substrate on which they could nest, and fences around the perimeter to deter mammal predators. Numbers of common terns nesting on the raft increased over the years, and with protection from mammal predators this colony has been highly productive. In 2003, this raft produced 2.1 chicks per pair, making it the most productive common tern colony out of the dozens monitored throughout the UK (Mavor et al. 2004), and this pattern has been replicated in most years since then (Clive Craik pers. comm.). In 2010 a decision was made to build more rafts at this site to provide a larger surface area for nesting terns. Construction of new rafts (developed to be predator-proof), and their licenced Crown Estate moorings, cost about £20,000. Half of this was provided by Argyll Bird Club, and half by Clive Craik himself. In 2011, the year these new rafts were constructed, over 300 pairs of common terns nested, and fledged over 300 chicks. In 2012, 600 pairs of common terns nested on these rafts making this the 2nd largest common tern colony in Scotland in 2012, and fledged more common tern chicks from this one site (over 600) than were produced by all of the natural common tern colonies in the whole of the west of Scotland. Common terns appear to be very quick to adopt artificial nesting platforms and experience high success rates on them. Similar success, though on a smaller scale, has been achieved by rafts placed for common terns at other locations. Herts and Middlesex Wildlife Trust placed rafts on Wilstone Reservoir in the 1990s, a site where common terns had not previously nested but occurred on migration in small numbers. Small numbers of common terns colonised these rafts, breeding successfully in most years. In 2008, twelve chicks fledged from the rafts http://www.tringreservoirs.org.uk/cmntern.html. Lincolnshire Wildlife Trust constructed a fibreglass raft to deploy on Grebe Lake, Whisby Nature Park, near Lincoln, to replace an older raft that was decomposing http://lincstrust.org.uk/conservation/article.php?id=21 and offer these rafts on a commercial basis. Nesting rafts for common terns are also promoted by the RSPB http://www.rspb.org.uk/ourwork/conservation/advice/rafts/.

A small, replicated, controlled study from May-August in 1982 on a concrete breakwater in Port Colborne, Canada (Richards and Morris 1984), found that common terns nested at higher densities on two plots enhanced with clumps of mossy stonecrop and driftwood (62% of 166 clutches in these plots), compared to plots layered with gravel (29% of clutches) or control plots of bare concrete (9% of clutches). Enhanced plots were also colonised earlier. Average clutch size and hatching rates were similar between plots (2.4-2.5 eggs/clutch and 76-86% hatching success), but the average number of chicks fledged per pair was significantly higher in enhanced (1.6) and control (1.3) plots than in gravel-layered plots (0.6).

A 1992 review of the use of artificial islands and floating platforms in 17 wetland nature reserves across the UK (Burgess and Hirons 1992) found that all seven species of gull and tern investigated used sparsely-vegetated islands and platforms at southern, coastal sites, but that nesting sites elsewhere were not used by four of the species. Sandwich terns used vegetated nesting sites at southern coastal sites, whilst common terns nested at all sites. At one site in Kent, the provision of 20 shingle islands attracted 350 pairs of Sandwich and common terns (Burgess and Hirons 1992).

A replicated study in 1987-1990 of a managed wetland in Macedonia, Greece (Pyrovetsi 1997) found that the target species, Dalmatian pelicans, did not benefit consistently from artificial habitats although other waterbirds did. Two constructed rafts and one artificial island were used extensively by a variety of waterbirds as resting and foraging sites. Common terns colonised the rafts in both years (average 12 nests and 14 fledglings per raft).

Dunlop et al. (1991) reported that rafts they built in spring 1990 in Totonto Outer Harbour were used by about 130 pairs of common terns, which achieved productivity of 1.3 chicks per pair, whereas this colony had been failing and declining in previous years due to predation, competition with gulls and human disturbance.

Although not based on deployment of rafts, a before-and-after study on Praia Islet (12 ha), off Graciosa in the Azores (Bried et al. 2009), found that the breeding population of common terns increased dramatically (from no pairs, to over 1,000 pairs) following the installation of nest boxes in 1996, combined with the eradication of rabbits and habitat restoration. It is unclear how much of this increase can be attributed to eradication of rabbits or to habitat restoration (which mainly involved removal of alien vegetation), or to provision of nestboxes. Although terns occupied many of the nestboxes, the breeding numbers increased far more than the numbers of nestboxes that were provided, suggesting that creation of more open ground without invasive vegetation may have been mostly responsible. However, provision of nestboxes on rafts may further aid increases in common tern numbers by providing shelter from the weather and some protection from avian predators.

17.2.8 Exclude large gulls from nesting close to colonies

Predation by large gulls affects productivity at some colonies in some years (this was identified by the meta-analysis of JNCC data to be the third most frequently identified cause of reduced productivity, reported 24 times). Removal of gull territories adjacent to these key colonies could improve productivity of common terns. It is likely that most of the common tern colonies affected by gull predation are affected by small numbers of gulls that have developed specialised habits of feeding on tern eggs and chicks, and so the numbers of gulls that would need to be removed to increase tern productivity are likely to be small. However, this measure might not be necessary if other measures can be put in place (such as providing nesting platforms which although they would not be immune to impacts from gulls would be likely to increase productivity sufficiently to make removal of gulls unnecessary.

18. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR COMMON GUILLEMOT

18.1 Common guillemot ecology

A widespread and numerous species, with a high-medium latitude breeding distribution around the northern hemisphere. The world population is around 7.3 to 7.4 million pairs, with around 2.8 to 2.9 million of these in the North Atlantic (Mitchell et al. 2004).

18.1.1 The species in the British Isles

Seabird 2000 found 890,000 pairs in Great Britain the Isle of Man and Channel Islands, and 160,000 pairs in Ireland (Mitchell et al. 2004). Common guillemots breed in dense colonies on cliff ledges, on sea stacks, or under boulders below cliffs. They lay a single egg onto bare rock, but will replace lost eggs around 14 days later. While breeding, common guillemots forage by diving to catch fish by underwater pursuit, feeding chicks especially on sandeels and sprats.

The UK SPA suite for this species holds around 693,000 pairs (95% of the UK total) across 34 sites: Ailsa Craig, Buchan Ness to Collieston Coast, Calf of Eday, Canna and Sanday, Cape Wrath, Copinsay, East Caithness Cliffs, Fair Isle, Farne Islands, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Fowlsheugh, Handa, Hermaness Saxa Vord and Valla Field, Hoy, Marwick Head, Mingulay and Berneray, North Caithness Cliffs, North Colonsay and Western Cliffs, North Rona and Sula Sgeir, Noss, Rathlin Island, Rousay, Rum, Shiant Isles, Skomer and Skokholm, St Abb's Head to Fast Castle, St Kilda, Sule Skerry and Sule Stack, Sumburgh Head, Troup Pennan and Lion's Heads, West Westray.

Adult survival rate averages 0.885 (del Hoyo et al. 1996) and mostly breed for the first time when 5 years old (Cramp and Simmons 1977-1994). British common guillemots disperse away from colonies in late summer, but mostly overwinter in British waters. Small numbers may overwinter further afield, from southern Norway to Iberia. Few birds from overseas populations migrate through, or overwinter in, British waters, although small numbers from Scandinavian, Faroese and Icelandic colonies reach northern Britain in winter (Wernham et al. 2002).

18.1.2 Present, and likely future trends

Seabird 2000 recorded about 1,600,000 individual common guillemots at colonies in Britain and Ireland, with 75% in Scotland, 15% in Ireland, 6% in England the Isle of Man and Channel Islands, and 4% in Wales (Mitchell et al. 2004). This was 32% higher than found by the SCR Census in 1985-88, which itself was 81% higher than the number recorded in Operation Seafarer in 1969-70 (Mitchell et al. 2004). The JNCC index of breeding numbers of common guillemots at UK colonies (Figure 18.1.1) indicates a further slight increase in numbers from 2000 to 2011, but the trend differs dramatically between northern and southern colonies. In Scotland, the index has declined (Figure 18.1.2), whereas in Wales it has continued to increase strongly (Figure 18.1.3). Foster and Marrs (2012) estimated a 24% decrease in the index of breeding numbers of common guillemots in Scotland from 1986 to 2011, but Figure 18.1.2 shows that the decrease has been even stronger than this if measured from the peak numbers recorded in 2001. The index shows a 40% drop in numbers from 2001 to 2011 at monitored Scottish colonies. This decrease within Scotland also varies regionally, being much greater at Shetland colonies than in SE or SW Scotland. It is difficult to predict future changes in common guillemot numbers, but the pronounced current trends of decreases in the north and increases in the south show no evidence at present of changing.

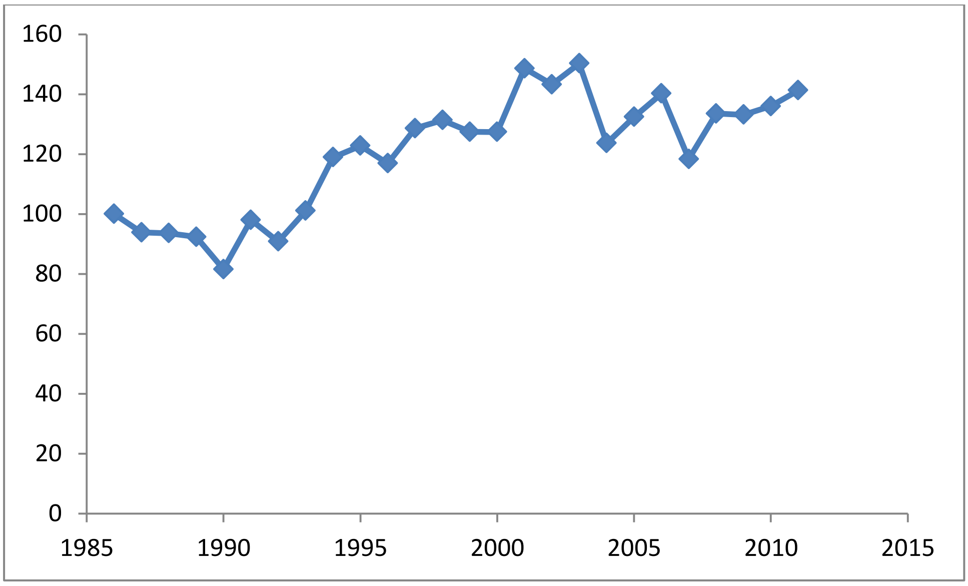

Figure 18.1.1. Common guillemot breeding population index for the UK. Data from JNCC online database.

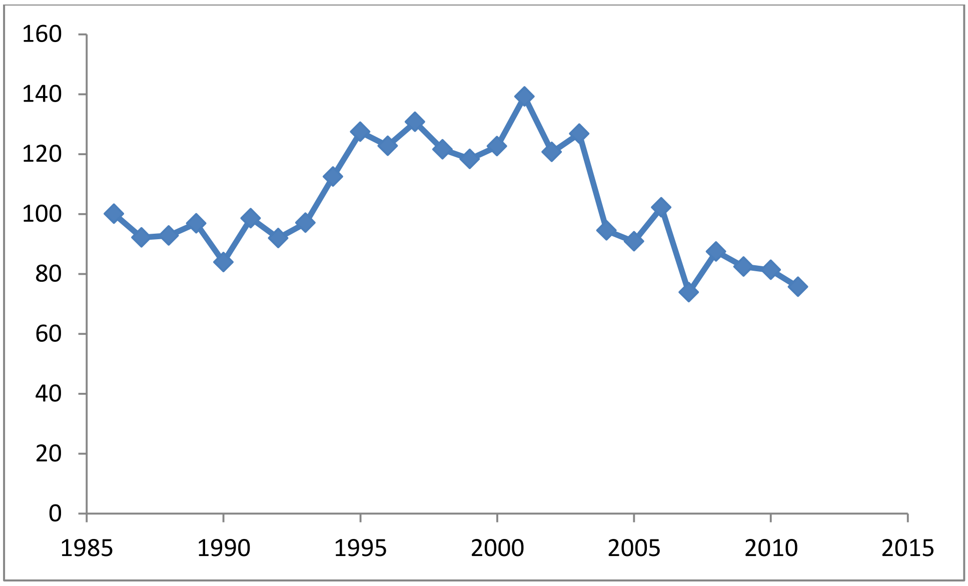

Figure 18.1.2. Common guillemot breeding population index for Scotland. Data from JNCC online database.

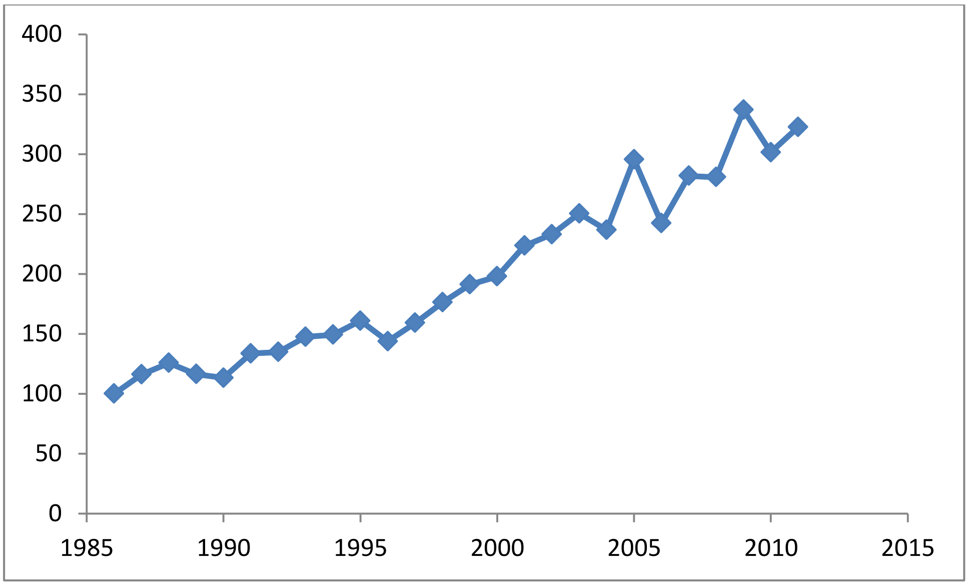

Figure 18.1.3. Common guillemot breeding population index for Wales. Data from JNCC online database.

18.1.3 *Factors affecting survival rates*

There have been numerous studies of adult survival rates of common guillemots, in the British Isles (Southern et al. 1975 0.87 NE Scotland, Birkhead 1974 British ringing recoveries 0.879, Mead 1974 British ringing recoveries 0.937, Birkhead and Hudson 1977 Skomer 0.904, Harris and Bailey 1992 Isle of May 0.94, Harris et al. 1992, Harris and Wanless 1995, 1996, Wernham et al. 1997, Poole et al. 1998, Harris et al. 2000 0.952 Isle of May 0.964 Colonsay 0.939 Canna, 2007, Votier et al. 2005, Reynolds et al. 2008, Votier et al. 2008), elsewhere in Europe (Olsson et al. 2000 Sweden 0.859, Sandvik et al. 2005 North Norway 0.961) and in North America (Sydeman 1993 California 0.939). Adult survival rate can vary between years and colonies. Most of these studies report survival rate estimates without including environmental factors as covariates. However Sandvik et al. (2005) showed a strong negative correlation between common guillemot survival at a colony in north Norway and SST, and a suggestion of a weak relationship with herring abundance. Votier et al. (2005) analysed the influences of oil spill incidence, North Atlantic Oscillation (NAO), sea surface temperature (SST) and indices of forage fish abundance on survival rates of adult common guillemots from Skomer, Wales. They found that survival rates varied between years, from 0.975 to 0.879, with the four lowest survival rates all occurring in years with major oil spills in the region. As a result, oil spill incidence significantly affected survival rate, with mortality doubling from 4.43% in years with no oil spill to 8.96% in years with a major oil spill. The NAO also influenced survival rate. Forage fish indices had a weak influence, possibly because the data on forage fish in the area are poor (there are no fisheries for sprats or sandeels in the region), and SST had a small influence but much less than the NAO. Harris and Bailey (1992) reported a strong correlation (0.8) between sprat abundance in the North Sea and the survival of first year guillemots from the Isle of May, but no detectable relationship for adult common guillemot survival rate across the same period (which varied only from 0.97 to 0.93), suggesting that inexperienced guillemots may be much more susceptible to reduced food abundance than are experienced adults.

18.1.4 *Breeding success in the British Isles*

In 1991, mean productivity at 11 monitored colonies was 0.73 chicks per pair, with colonies showing lower productivity mainly ones affected by predation (Herring gulls took guillemot eggs at Sumburgh Head and productivity there averaged 0.62) (Walsh et al. 1992). In 1992, productivity averaged 0.77 chicks per pair, with little geographical variation (Walsh et al. 1992). In 1993, productivity averaged 0.72 chicks per pair. Increased adult attendance at colonies suggested improved food supply compared to 1988-90 (Walsh et al. 1994). In 1994, overall productivity was high, averaging 0.76

chicks per pair, but poor weather was thought to have been the reason for a reduction in success at some Orkney colonies (e.g. from 0.77 chicks per pair at some Orkney colonies to 0.65 at Mull Head) (Walsh et al. 1995). In 1997, productivity was reduced at many colonies as a consequence of storms; mean productivity was 0.62 chicks per pair, suggesting that the exceptional weather had reduced productivity by about 10% from 'normal' levels (Thompson et al. 1998). In 1998, productivity averaged 0.73 chicks per pair, but there was evidence of food shortage at some colonies. At the Isle of May, chicks fledged 25% lighter than normal and productivity there was below the mean for the previous 17 seasons (Thompson et al. 1999). In 1999, breeding success at the Isle of May was the lowest recorded (0.66 chicks per pair) with evidence of food shortage (lowest recorded weights of chicks) (Upton et al. 2000). At North Sutor, predation by Great black-backed gulls contributed to low productivity (Upton et al. 2000). In Shetland in 1999, thousands of eggs were washed into the sea on west-facing coasts in a gale in May, and although a proportion of birds relaid, survival of late-hatched chicks was low due to predation by Great black-backed gulls (Upton et al. 2000). In 2000, productivity across 10 colonies averaged 0.74 chicks per pair, marginally above the average of 0.73 for 1986-1999 (Mavor et al. 2001). However, at Sumburgh some 8,000-9,000 chicks were washed into the sea by a severe storm on 13 June 2000 (Mavor et al. 2001). It was a relatively poor season in 2001, with 8 out of 10 monitored colonies recording below average productivity, this low productivity being attributed to adults experiencing difficulties finding food (Mavor et al. 2002). That supposition was supported by the lowest chick provisioning rates yet recorded on Fair Isle, and a high level of chick neglect on the Isle of May and at Sumburgh (Mavor et al. 2002). In 2002, productivity averaged 0.75 chicks per pair; lowest productivity was in Shetland (0.64) where attendance of chicks by adults was noted to be very low, suggesting food shortage. Further evidence for food shortage at Shetland in 2002 was lower chick weights that year compared to a sample taken in 1999 (Mavor et al. 2003). In Shetland, breeding success in 2003 was lower than in any previous year, with low growth rates of chicks and low attendance by adults indicating food shortage. Low attendance of adults also allowed increased rates of predation of eggs and chicks by large gulls (Mavor et al. 2004). On Canna in 2003, eggs depredated by brown rats were found and it appeared that rats had been responsible for redistribution of breeding common guillemots into areas inaccessible to rats (Mavor et al. 2004). The breeding success of common guillemots was lower in 2004 than in any previous year of the monitoring programme started in 1986. Many chicks were left unattended suggesting that adults had difficulty finding food (Mavor et al. 2005). Mavor et al. (2005) specifically stated '*Low availability of high quality food (e.g. sandeels) was likely the cause of the much reduced breeding success at all the aforementioned colonies, whether directly – due to chick starvation – or indirectly – due to hypothermia, predation, or killing of unattended young by neighbouring adults' 'Wing/weight ratios of chicks were nearly identical to that recorded in 2003, also a year of low success, but chicks of all wing lengths were, on average, 50g lighter than in 1999, a year of relatively high success*'. Productivity was only marginally better in 2005, but still very poor and apparently reflecting low availability of food fish. Chicks on Fair Isle and at Compass Head were around 50g lighter than normal for their ages, strongly implicating food shortage (Mavor et al. 2006). Although food was considered to be the main factor affecting productivity, predation of eggs and chicks by gulls was observed at North Sutor in 2005, especially in smaller sub-colonies (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for common guillemots as follows. Productivity was generally low in 2006 in all regions of Britain and Ireland. Mean success was 0.48 chicks per pair, well below the long-term average of 0.69 chicks per pair for 1986-2005. Many losses occurred at the egg stage, but in addition, chick weights were lighter than normal, suggesting that food shortage was the main factor reducing productivity. On the Isle of May, chick survival was estimated at 49%, whereas 90% had previously been typical. Many chicks were left unattended, indicating that adults were finding it hard to find food. Chick losses were due mainly to starvation or attacks from neighbouring adults, with gull predation a minor factor. In contrast, on Skomer, predation by great black-backed gulls was thought to be the main cause of chick loss in 2006 (Mavor et al. 2008). This was also the case in 2004 at this colony, where productivity was relatively high (0.66 chicks per pair) and not apparently affected by food shortage (Mavor et al. 2005).

Table 18.1.1 Meta-analysis of main factors contributing to reduced productivity of Common guillemots at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	22
Gull predation	7
Extreme weather conditions	5
Rat predation	1

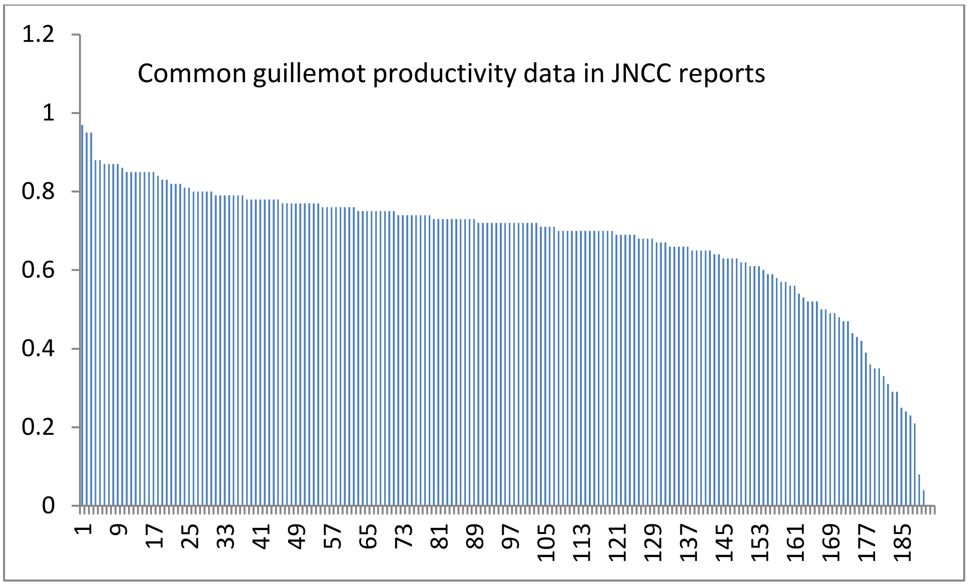

Figure 18.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of common guillemots at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

18.2 Management options

	18.2.1 Closure of sandeel and sprat fisheries in all UK waters	18.2.2 Closure of sandeel and sprat fisheries in wintering areas	18.2.3 Eradicate rats	18.2.4 Prevent oil spills
Evidence of success for this species	Low C=Low	Low C=Low	Low C=Mod	High C=Mod
Evidence of success for similar species	High C=Mod	Low C=Low	High C=High	Low C=Low
Cost-effectiveness	Uncertain C=Low	Uncertain C=Low	High C=Low	Uncertain C=Low
Feasibility	Moderate C=Low	Moderate C=Low	High C=High	Low C=High
Practicality	Moderate C=low	Moderate C=Low	Moderate C=High	Low C=High
Applies at SPA populations	Yes C=High	Uncertain C=High	Few C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

18.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or in all UK waters

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Indeed, food shortage was identified in the meta-analysis of JNCC monitoring as the most important factor reducing common guillemot productivity at colonies in Britain and Ireland (22 cases out of 35). Mean foraging range of common guillemots is around 38 km and maximum foraging range is up to 200 km (Langston 2010, Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival, but that would amount to closure of sandeel and sprat fishing in British waters given the wide distribution and large number of SPAs for these species. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes. So, despite the clear evidence from the JNCC monitoring that food shortage has a strong impact on common guillemot productivity, analysis of the data from the Isle of May and east Scottish sites from the 1990s did not show this effect. Possibly food fish abundance needs to fall to lower levels to affect common guillemots than it does to affect kittiwakes and terns (Furness and Tasker 2000).

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles, but evidence that this would affect common guillemot productivity is less strong. Closure of sandeel fisheries in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

18.2.2 Closure of sandeel and sprat fishing in areas where these species are aggregated in winter

Crashes of common guillemot populations have occurred in winter as a result of food shortage, and ‘wrecks’ of common guillemots occur in autumn/winter in some years, suggesting that winter may be a critical season determining survival rates. If so, protection of prey fish stocks that are important as winter food may be of greatest importance. Common guillemots feed extensively on sprats during winter, and also take some sandeels (even though sandeels are mostly buried in the sand in winter, guillemots will dig them out of the sand; M.L. Tasker pers. comm.) and sandeels were well represented in guillemot stomachs from birds killed in the Braer oil spill in Shetland which occurred in mid-winter. British common guillemots mainly overwinter in British waters, though often south of their breeding areas, so closure of sprat and sandeel fisheries in southern Britain may be most effective in protecting their food supply. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider.

There are localised fisheries for sprats in UK waters, by Scottish fishermen off west Scotland (usually close to Mull), by English fishermen in the English Channel (Lyme Bay), and occasionally as a result of the activity of foreign industrial fishing fleets (ICES 2013). There have in the past been fisheries on sprats in the Moray Firth and in the Firth of Forth, fisheries which appear to have caused local depletion of those stocks and then been closed (Jennings et al. 2012). There has in the past been a fishery by Scottish fishermen on sandeels in Shetland which was closed in 1991 due to the depletion of that stock which has still not recovered, and the large industrial fishery for sandeels in the North Sea has moved around over the years to exploit different stocks within the North Sea, resulting in depletion of many of the distinct sandeel stocks in the northern North Sea, but continued exploitation by Danish fishermen of sandeel stocks in English waters which appear to have been more resilient than stocks further north in the North Sea (ICES 2010, ICES 2012). All of these sandeel and sprat stocks are likely to represent important preferred food of common guillemots in UK waters in winter. Closure of sandeel fishing in UK waters would have low cost for UK fishermen, as almost the entire fishery is carried out by Danish fishermen. Closure of sprat fishing in UK waters would affect a small number of fishing vessels in west Scotland and in Lyme Bay, and limit potential development of sprat fisheries on stocks that are currently unfished (e.g. in Irish Sea, Clyde, Thames, Firth of Forth, Moray Firth).

18.2.3 Eradication of rats

Brown rats are a factor reducing productivity at a few colonies. Eradication of invasive alien rats would allow common guillemot and razorbill productivity to increase at colonies where this predator is present. However, this applies at a very small proportion of colonies and it is very unlikely that rat eradication would be an effective conservation measure for common guillemots.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

18.2.4 Prevent risk of major oil spills near to SPAs

The effect of oil pollution on seabird populations is surprisingly uncertain. Votier et al. (2005) showed that major oil spill incidents doubled mortality of common guillemots from a Welsh colony in the years in which these occurred. Major incidents such as the oil spill accidents considered by Votier et al. (2005) probably kill fewer seabirds than the chronic discharges of small quantities of oil from illegal discharges (Dunnet 1982). So the high influence of these major oil spills on common guillemot survival rates probably understates the influence of oil pollution. Nevertheless, previous attempts to relate changes in numbers of common guillemots attending colonies to oil spill mortality have often failed to show any detectable impact (see for example the lack of impact of the Braer oil spill in Shetland on seabird breeding numbers or breeding success http://www.nature-shetland.co.uk/birdclub/braer/Part10.html), suggesting that the consequence of increased mortality is not seen at the level of breeding numbers at colonies, so relationships between survival rates and breeding numbers may be quite complex. According to JNCC '*In all of the oil spill disasters that have occurred during the SMP, guillemots and razorbills have predominated in the seabirds recovered. But despite large numbers of birds being killed, there does not appear to have been any substantial lasting effect on UK seabird populations*' http://jncc.defra.gov.uk/pdf/SMP%20Review%20Report%20Appendix%202%20-%20Impacts%20of%20pressures%20on%20Seabirds.pdf . However, this statement, based on changes in numbers, is not entirely consistent with evidence that recruitment rates of immature birds increased in years following oil-related mortality of adults (Votier et al. 2005, 2008). That observation indicates that although breeding numbers did not change, there is a reduction in the size of the nonbreeding pool that may play an important role in buffering effects of environmental change.

Management to reduce amounts of oil pollution in UK waters would contribute to increasing survival rates of common guillemots. While it is reasonable to assume that strenuous efforts are taken to minimize risk of serious oil spills, survival rates of common guillemots (and presumably of razorbills) could be increased if incidence of oil spills could be reduced in the wintering areas used by common guillemots and razorbills. Preventing major accidents may be very difficult since there is already a very considerable effort to avoid such disasters. Preventing illegal discharges of oil that contribute to chronic oil pollution might be improved. Enforcement of laws prohibiting discharge of oil into the sea, with increased detection of culprits and increased effort into successful prosecution of offenders could reduce the amount of oil illegally entering UK waters.

19. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR RAZORBILL

19.1 Razorbill ecology

SPEC 4 Favourable conservation status (secure) but concentrated in Europe. The world population is around 610,000 to 630,000 pairs, with two subspecies, *torda* and *islandica*. The nominate subspecies breeds in Denmark, Norway, Russia, Finland, Sweden, Greenland, Canada and USA. The subspecies *islandica* breeds in the British Isles, Faroes, Iceland, France and Germany. The largest population is in Iceland (380,000 pairs) (Mitchell et al. 2004).

19.1.1 The species in the British Isles

Seabird 2000 found 110,000 pairs in Great Britain the Isle of Man and Channel Islands, and 35,000 pairs in Ireland (Mitchell et al. 2004). Razorbills breed in colonies on cliff ledges, on sea stacks, or under boulders below cliffs. Nest sites differ from those of common guillemots, with razorbill nests more often among boulders and more often scattered rather than in high-density groups. They lay a single egg onto bare rock, but will replace lost eggs around 14 days later. While breeding, razorbills forage by diving to catch fish by underwater pursuit, feeding chicks especially on sandeels and sprats, and generally taking smaller fish than caught by common guillemots.

The UK SPA suite for this species holds around 81,000 pairs (76% of the British total and 26% of the all-Ireland total) across 19 sites: Cape Wrath, East Caithness Cliffs, Fair Isle, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Fowlsheugh, Handa, Mingulay and Berneray, North Caithness Cliffs, North Rona and Sula Sgeir, Rathlin Island, Shiant Isles, Skomer and Skokholm, St Abb's Head to Fast Castle, St Kilda, Troup Pennan and Lion's Heads, West Westray.

Adult survival rate averages 0.905 (del Hoyo et al. 1996) and most breed for the first time when 4 or 5 years old (Cramp and Simmons 1977-1994). British razorbills generally move south after breeding, wintering from British waters to Iberia including the western Mediterranean (Wernham et al. 2002). However, many British razorbills, especially adults, remain in British waters all year round. Rather few foreign-ringed razorbills have been recovered in Britain, but these together with biometrics of tideline corpses indicate that some birds from Scandinavia (from the subspecies *torda*) winter in British waters and some birds from Iceland do too (Wernham et al. 2002).

19.1.2 Present, and likely future trends

Seabird 2000 recorded about 216,000 individual razorbills at colonies in Britain and Ireland, with 64% in Scotland, 24% in Ireland, 6% in England the Isle of Man and Channel Islands, and 6% in Wales (Mitchell et al. 2004). This was 23% higher than found by the SCR Census in 1985-88, which itself was 5% higher than the number recorded in Operation Seafarer in 1969-70 (Mitchell et al. 2004). The JNCC index of breeding numbers of razorbills at UK colonies (Figure 19.1.1) indicates a further increase in numbers of about 15% from 2000 to 2011, but the trend differs between northern and southern colonies. The index for Scotland indicates a small decrease from 2000 to 2011 (Figure 19.1.2), whereas in Wales numbers have continued to increase (Figure 19.1.3). Predicting future changes in numbers is very difficult, but the conditions appear to be more favourable in the south and less so in the north.

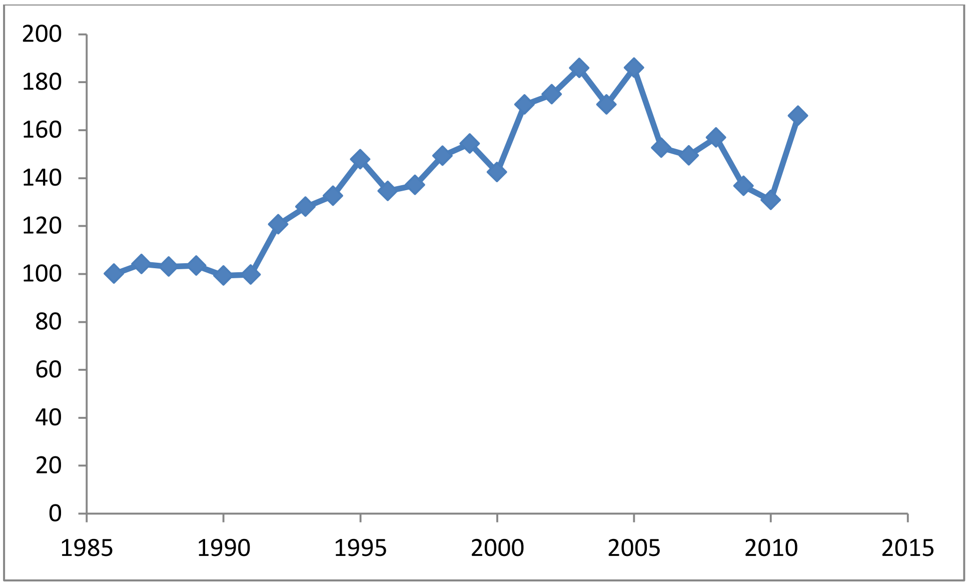

Figure 19.1.1. Razorbill breeding population index for the UK. Data from JNCC online database.

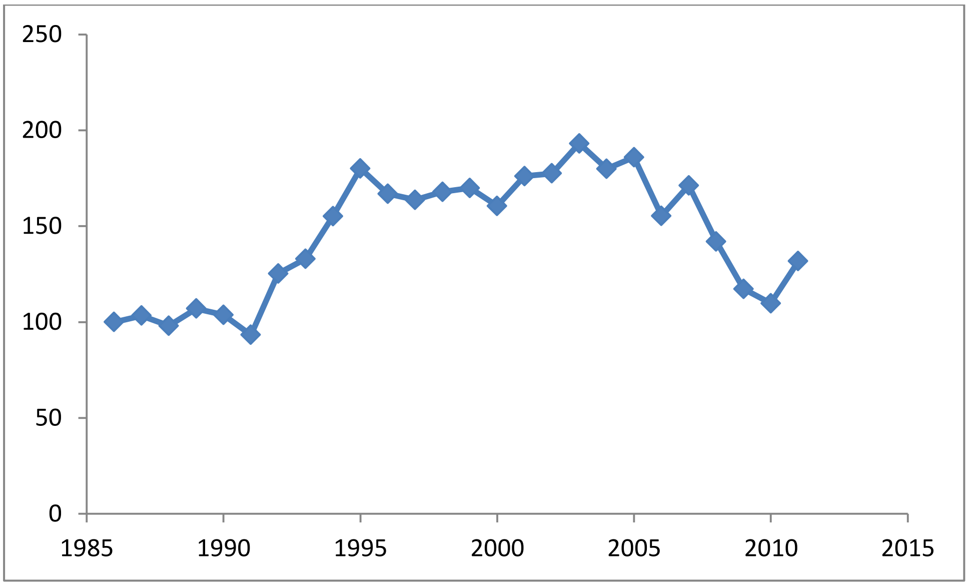

Figure 19.1.2. Razorbill breeding population index for Scotland. Data from JNCC online database.

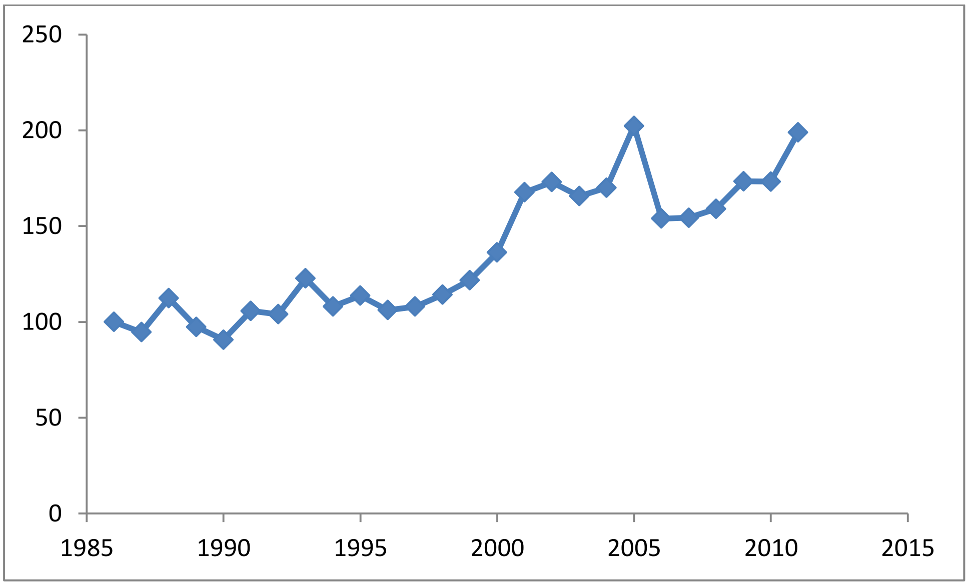

Figure 19.1.3. Razorbill breeding population index for Wales. Data from JNCC online database.

19.1.3 *Factors affecting survival rates*

Mean survival rates of adult razorbills have been reported as 0.919 at Hornoya north Norway (Sandvik et al. 2005), 0.905 at the Isle of May (Harris et al. 2000), 0.898 at Skomer (Hudson 1979), 0.901 at Skomer (Poole et al. 1998), 0.895 in Canada (Chapdelaine 1997), 0.92 at the Shiants (Steventon 1979), 0.96 at Clo Mor (Lloyd and Perrins 1977), 0.81 to 0.89 at Skokholm (Lloyd and Perrins 1977), 0.89 from British ring recovery data (Lloyd 1974) and 0.914 from British ring recovery data (Mead 1974). Sandvik et al. (2005) showed that razorbill survival rate decreased with increasing SST at Hornoya, but none of the other studies included environmental covariates, so the influences of environmental factors on survival rates of British razorbills remain uncertain. Estimation of the extent to which management might alter survival rates is therefore extremely difficult. However, it is recognised that razorbills can be subject to mass mortality incidents ('wrecks') in autumn and winter which seem to be mainly due to food shortage (scarcity of small pelagic fish). For example, a wreck in autumn 2007 involved thousands of dead razorbills, especially in the Skagerrak and Kattegat, was attributed to food shortage (Heubeck et al. 2011). A similar wreck in 1983 was attributed to low abundance of sprats in the north-western North Sea (Blake 1984).

19.1.4 *Breeding success in the British Isles*

Although productivity in 1988-90 was poor in Shetland, it improved in 1991, and growth rates of chicks at Hermaness were higher in 1991, indicating a better food supply. It is known that there was high recruitment of sandeels around Shetland in 1991(Walsh et al. 1992). In 1992, productivity was high at all study colonies, and chick weights at Hermaness were above those in 1988-90 which suggests that this improved productivity was due to improved food supply (Walsh et al. 1993). Walsh et al. (1995) noted that Razorbill nest sites on open ledges on Skomer were noticeably less successful than enclosed sites, probably reflecting exposure to predation. In 2001, Razorbill productivity was 0.7 chicks per pair across the 6 monitored colonies, similar to the average for 1986-2000 (Mavor et al. 2002). On Canna in 2003, eggs depredated by brown rats were found and it appeared that rats had been responsible for redistribution of breeding Razorbills into areas inaccessible to rats (Mavor et al. 2004). In 2004, there was low productivity of Razorbills in the Northern Isles, but higher productivity further south. Food samples indicated a scarcity of large sandeels in the north, but no evidence of food shortage in the south, strongly suggesting that reduced productivity was caused by lack of suitable food. In 2005, chicks at Fair Isle were noted to be underweight for their ages and some lost weight rather than growing, providing strong evidence for food shortage being the cause of low productivity (Mavor et al. 2006). Mavor et al. (2008) described the 2006 breeding season for razorbills as follows. Breeding success was generally poor, averaging 0.48 chicks per pair. In Shetland, chick survival was very poor, attributed to a lack of food (sandeels). Chicks showed low rates of weight gain, and some cases of weight loss, consistent with severe food shortage. On the Isle of May, productivity at 0.64 chicks per pair was close to the long-term average, and adults appeared to be having little difficulty finding food around that colony.

Table 19.1.1 Meta-analysis of main factors contributing to reduced productivity of Razorbills at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	8
Gull predation	1
Rat predation	1

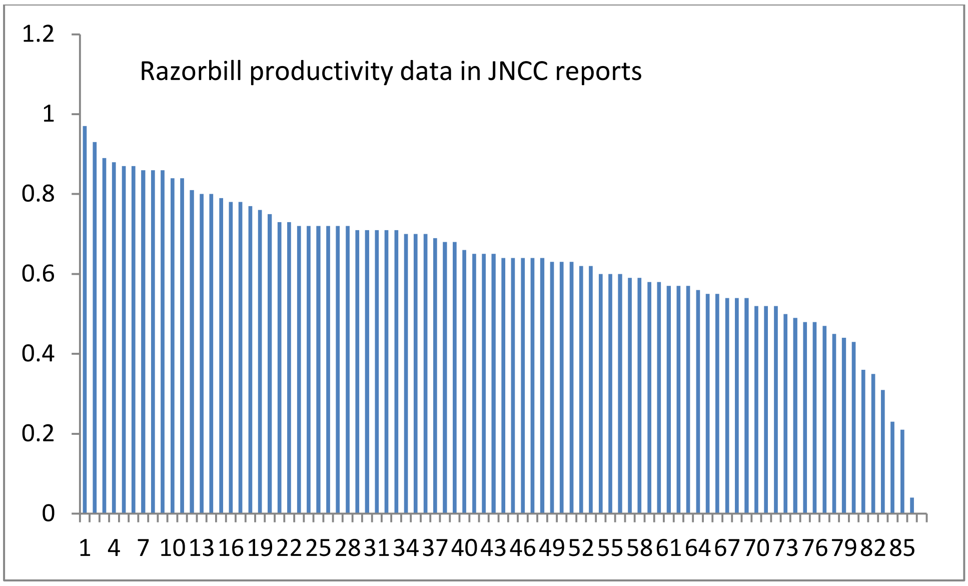

Figure 19.1.4. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of razorbills at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

19.2 Management options

	19.2.1 Closure of sandeel and sprat fisheries in all UK waters	19.2.2 Closure of sandeel and sprat fisheries in wintering areas	19.2.3 Eradicate rats	19.2.4 Prevent oil spills
Evidence of success for this species	Low C=Low	Low C=Low	Low C=Low	Low C=Low
Evidence of success for similar species	High C=Mod	Low C=Low	High C=High	High C=Mod
Cost-effectiveness	Uncertain C=Low	Uncertain C=Low	High C=Mod	Uncertain C=High
Feasibility	Moderate C=Low	Moderate C=Low	High C=High	Low C=High
Practicality	Moderate C=Low	Moderate C=Low	Moderate C=High	Low C=High
Applies at SPA populations	Yes C=High	Uncertain C=High	Few C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

19.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or in all UK waters

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Mean foraging range of razorbills is around 24 km and maximum foraging range is up to 95 km (Langston 2010, Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival, but that would amount to closure of sandeel and sprat fishing in British waters given the wide distribution and large number of SPAs for these species. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes. So, despite the clear evidence from the JNCC monitoring that food shortage has a strong impact on razorbill productivity, analysis of the data from the Isle of May and east Scottish sites from the 1990s did not show this effect. Possibly food fish abundance needs to fall to lower levels to affect razorbills than it does to affect kittiwakes and terns (Furness and Tasker 2000).

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Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles, but evidence that this would affect razorbill productivity is less strong. Closure of sandeel fisheres in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

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Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

19.2.4 Prevent risk of major oil spills near to SPAs

The effect of oil pollution on seabird populations is surprisingly uncertain. Votier et al. (2005) showed that major oil spill incidents doubled mortality of common guillemots from a Welsh colony in the years in which these occurred. The same is likely to be true of razorbills, given the similarity in their at sea behaviour. Major incidents such as the oil spill accidents considered by Votier et al. (2005) probably kill fewer seabirds than the chronic discharges of small quantities of oil from illegal discharges (Dunnet 1982). So the high influence of these major oil spills on common guillemot survival rates probably understates the influence of oil pollution. Nevertheless, previous attempts to relate changes in numbers of common guillemots or razorbills attending colonies to oil spill mortality have often failed to show any detectable impact (see for example the lack of impact of the Braer oil spill in Shetland on seabird breeding numbers or breeding success http://www.nature-shetland.co.uk/birdclub/braer/Part10.html), suggesting that the consequence of increased mortality is not seen at the level of breeding numbers at colonies, so relationships between survival rates and breeding numbers may be quite complex. According to DEFRA '*In all of the oil spill disasters that have occurred during the SMP, guillemots and razorbills have predominated in the seabirds recovered. But despite large numbers of birds being killed, there does not appear to have been any substantial lasting effect on UK seabird populations*' http://jncc.defra.gov.uk/pdf/SMP%20Review%20Report%20Appendix%202%20-%20Impacts%20of%20pressures%20on%20Seabirds.pdf.

Nevertheless, management to reduce amounts of oil pollution in UK waters would contribute to increasing survival rates of razorbills. While it is reasonable to assume that strenuous efforts are taken to minimize risk of serious oil spills, survival rates of razorbills could presumably be increased if incidence of oil spills could be reduced in their wintering areas. Preventing major accidents may be very difficult since there is already a very considerable effort to avoid such disasters. Preventing illegal discharges of oil that contribute to chronic oil pollution might be improved. Enforcement of laws prohibiting discharge of oil into the sea, with increased detection of culprits and increased effort into successful prosecution of offenders could reduce the amount of oil illegally entering UK waters.

20. EVIDENCE REVIEW OF POTENTIAL MANAGEMENT FOR ATLANTIC PUFFIN

20.1 Atlantic puffin ecology

SPEC 2 (Unfavourable conservation status (vulnerable) and concentrated in Europe). The Atlantic puffin is found around much of the North Atlantic and adjacent seas, with three sub-species. The nominate race *F. a. arctica* breeds at mid-latitudes, in eastern North America, Iceland, north Norway to southern Novaya Zemyla. *F. a. naumanni* breeds further north in high Arctic waters in Greenland, Svalbard and northern Novaya Zemyla. *F. a. grabae* breeds in southern areas of the species' range; in southern Norway, the Faeroes, the British Isles, and France. The total population of *F. a. grabae* (the biogeographical population) is estimated at 901,000 pairs (Cramp 1977-1994; Lloyd et al. 1991). The combined British and Irish population of 469,500 pairs in 1985–1987 made up 52% of this biogeographical population (Stone et al. 1997).

20.1.1 The species in the British Isles

Puffins are crevice and burrow-nesting seabirds, with colonies often found in highly inaccessible locations, such as on grassy slopes half way down cliffs. They can be very difficult to census, and their activity at colonies fluctuates dramatically, often showing large variations in colony attendance over periods of a few days throughout the breeding season. As a consequence trends in puffin colony sizes are often uncertain. The majority of the British and Irish population breed in Scotland with the St. Kilda archipelago holding the largest colony (Lloyd et al. 1991). Other principal breeding areas include Shetland (particularly Fair Isle and Foula), Orkney, Sule Skerry, the Isle of May and the Western Isles including the Shiant Islands. The main breeding areas in England are the Farne Islands, Coquet Island and the Bempton-Flamborough cliffs. Major colonies in Wales include Skomer, Skokholm and Ynys Gwylans. Rathlin Island (Antrim) is the main breeding location in Northern Ireland (Lloyd et al. 1991).Puffins feed on small pelagic fish (Hislop and Harris 1985), mostly caught in the upper 10 m of the sea by shallow dives from the surface. They carry multiple fish to the burrow to feed their chick (unlike common guillemots which carry a single fish), Breeding numbers at some colonies apparently declined considerably in early to mid 20th century. The very large colony on Ailsa Craig was apparently extirpated by brown rats (Zonfrillo 2001). Tens of thousands of pairs apparently disappeared from Skomer and St Kilda (Cramp et al. 1974). More recent studies have suggested roughly stable numbers at many colonies since the 1970s, although large increases occurred at the Isle of May, Farne Islands and Coquet Island. Food availability appears to be a major factor influencing population change. The cessation of increase in the Isle of May colony coincided with a reduction in the numbers of sprats in the North Sea and a doubling of the annual mortality rate of breeding adults (Harris and Wanless 1991). The large decrease in the northern Norway population in the 1970s and 1980s was linked to a crash in Atlantic/Scandinavian herring stocks (Anker-Nilssen and Barrett 1991). Declines have often been attributed to local factors such as oiling or mammalian predation (Harris et al. 1997). Conditions in the wintering areas also appear to be critical for puffins (Harris et al. 2005, 2010). Studies of puffin egg size over the last 30 years also indicate that climate change, through its effect on puffin prey stocks, has been causing a long-term reduction in egg size (Barrett et al. 2012). Finney et al. (2003) showed that puffin recruitment into a colony is influenced by the density of breeding gulls around the puffin colony. Puffin recruitment was reduced in areas with larger numbers of gulls. Furthermore, puffins breeding in areas of the colony that were gull free achieved higher rates of chick-feeding (because fewer fish were stolen by gulls), although there was no overall impact on productivity in the year of the study (Finney et al. 2001).

The UK's SPA suite for puffin supports around 470,000 pairs. This amounts to the vast majority of the British breeding population, and about 12% of the all-Ireland population. The SPA suite contains 21 sites where puffin has been listed as a qualifying species; Canna and Sanday, Cape Wrath, Coquet

Island, East Caithness Cliffs, Fair Isle, Farne Islands, Firth of Forth Islands, Flamborough Head and Bempton Cliffs, Flannan Isles, Foula, Hermaness Saxa Vord and Valla Field, Hoy, Mingulay and Berneray, North Caithness Cliffs, North Rona and Sula Sgeir, Noss, Rathlin Island, Shiant Isles, Skomer and Skokholm, St Kilda, and Sule Skerry and Sule Stack.

Adult survival rate averages 0.95 (del Hoyo et al. 1996) and most breed for the first time when 4 or 5 years old (Cramp and Simmons 1977-1994). After breeding, puffins disperse away from colonies. Birds breeding on North Sea coasts are thought to remain predominantly within the North Sea during winter, while birds breeding to the north and west of Britain disperse mainly into the North Atlantic, apparently ranging widely across the ocean at low densities (Harris 2011).

20.1.2 Present, and likely future trends

Seabird 2000 recorded about 600,000 apparently occupied burrows (approximately equivalent to pairs) in Britain and Ireland, of which 82% were in Scotland, 13% in England, 4% in Ireland and 2% in Wales (Mitchell et al. 2004). This represented a substantial (104%) increase in England and a small (13%) increase in Scotland since 1985. The greatest increase has occurred in SE Scotland and NE England (Isle of May, Inchkeith, Farne Islands and Coquet Island), Despite sandeel shortages in Shetland and puffin breeding failures at many Shetland colonies, numbers in Shetland have not shown clearly detectable decreases, except at the formerly very large colony at Foula (Mitchell et al. 2004). The JNCC index of breeding numbers of puffins at UK colonies does not provide a reliable population trend because the number of colonies where puffins are monitored is small, and count accuracy is considered to be relatively low (JNCC database). Foster and Marrs (2012) were unable to estimate the current puffin population trend for Scotland for the same reasons. Puffin populations appear to be rather robust to environmental pressures such as food shortage (Mitchell et al. 2004) and while impacts of climate change and shortage of sandeels are likely, these may be difficult to detect in view of the large uncertainty and variability in counts of puffin numbers at colonies. Predicting future trends is difficult, but it is also likely that trends will be quite difficult to detect from empirical data except at the very few intensively monitored colonies such as the Isle of May.

20.1.3 Factors affecting survival rates

Mean survival rates of breeding adult puffins averaged 0.93 at five European colonies where long time series of data are available (Skomer, Isle of May, Fair Isle, Røst, Hørnoya), and did not differ significantly among colonies (Harris et al. 2005), though the rate is slightly lower than the previous estimate of 0.95 for Skomer and 0.975 for Isle of May (Harris et al. 1997). The survival rates were considered by Harris et al. (2005) to be 'virtually identical' among the five colonies, despite differences in rates of growth or decline between colonies, indicating that local demography was not determined primarily by adult survival rate. Annual survival rates at four of the five colonies showed a significant negative correlation with sea surface temperature (SST) two years earlier. The exception (Hørnoya) was the northernmost colony (in Arctic Norway), where survival did not relate to local SST. Harris et al. (2005) point out that most mortality of adult puffins appears to occur during winter, so may be driven by environmental conditions in the wintering range of the birds. However, puffins from the Isle of May wintered predominantly within the North Sea, puffins from Skomer wintered predominantly west and south of the British Isles, while puffins from Røst wintered predominantly north of the British Isles, so the similarity of adult survival rates is despite these populations showing largely non-overlapping winter distributions. The influence of SST on adult survival rates was interpreted by Harris et al. (2005) as indicating 'bottom-up' effects of sea temperature on food abundance (such as sandeel stocks, and herring recruitment) affecting puffin survival.

20.1.4 Breeding success in the British Isles

Breeding success of Atlantic puffins is monitored at a rather small number of colonies around the British Isles; factors affecting productivity may differ between monitored sites and those that are not monitored (which tend to be less accessible locations). However, the JNCC monitoring reports identify food shortage as the most frequently reported factor causing breeding failure of puffins at monitored colonies. This has been especially frequent at Shetland, but has also affected colonies further south in the UK in some years. Flooding by intense rainfall events during May or June was also recorded several times (Table 20.1.1). Several research studies have reported on the influence of food availability on puffin productivity (reviewed in Harris 2011), reaching a clear conclusion from the evidence that scarcity of small pelagic fish (such as sandeels, juvenile herring, sprats or capelin) tends to result in breeding failure of puffins, with chicks coming out of burrows and then being taken by predators or dying of starvation or exposure. Productivity may be slightly influenced also by the presence of large numbers of kleptoparasites (great skuas, Arctic skuas, gulls, jackdaws), and in some colonies may be influenced by predators (including rats, feral cats, ferrets, great black-backed gulls and great skuas).

Table 20.1.1 Meta-analysis of main factors contributing to reduced productivity of Atlantic puffins at monitored colonies in Britain and Ireland 1986-2006. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

Factor	Cases reported
Food shortage	16
Flooding of burrows	8
Feral cat predation	2
Ferret predation	1

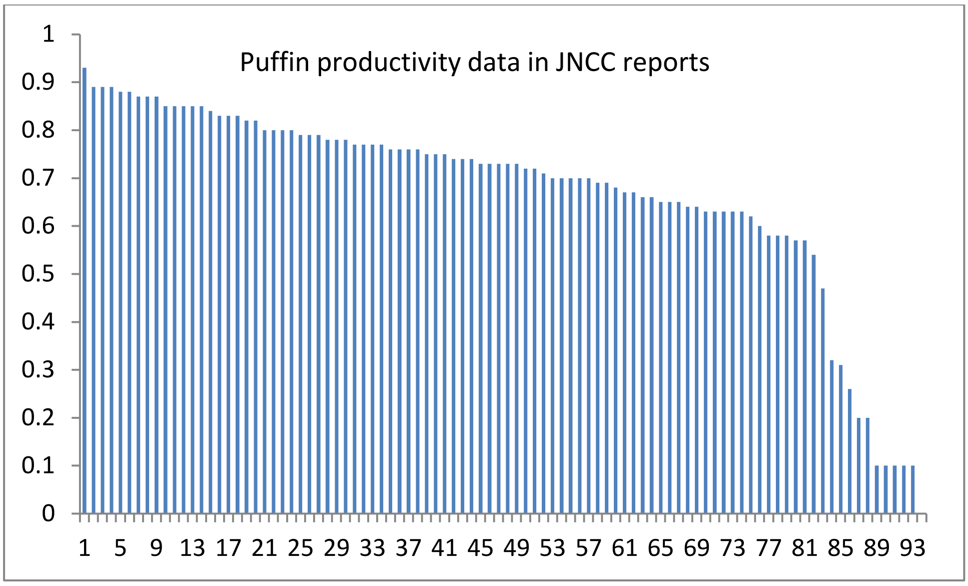

Figure 20.1.1. Ranked values (highest to lowest) of annual breeding productivity (chicks per pair) of Atlantic puffins at monitoring colonies. Data from JNCC annual reports on Seabird numbers and breeding success in Britain and Ireland, 1986 to 2006.

20.2 Management options

	20.2.1 Closure of sandeel and sprat fisheries in UK waters	20.2.2 Eradicate rats	20.2.3 Reduce oil spills
Evidence of success for this species	Low C=Low	Low C=Low	Low C=Low
Evidence of success for similar species	High C=Mod	High C=High	High C=Mod
Cost-effectiveness	Uncertain C=Low	High C=High	Uncertain C=Mod
Feasibility	Moderate C=Low	High C=High	Low C=High
Practicality	Moderate C=Low	Moderate C=High	Low C=High
Applies at SPA populations	Yes C=High	Some C=High	Yes C=High

*Confidence in each score is listed as 'High' 'Moderate' or 'Low' based on amount and consistency of evidence

20.2.1 Closure of sandeel and sprat fishing close to breeding area SPAs or in all UK waters

Food shortage is implicated as a cause of reduced productivity at some colonies in some years. Mean foraging range of Atlantic puffins was only 4 km at the one colony where this has been measured, but the maximum foraging range averaged about 100 km across several studies (Thaxter et al. 2012), so closure of sandeel and sprat fishing within 100 km of SPAs should increase productivity and adult survival, but that would amount to closure of sandeel and sprat fishing in British waters given the wide distribution and large number of SPAs for these species. There are clear difficulties with a management option involving changes to the EU Common Fisheries Policy, and commercial interests of fishermen to consider. However, of all the management options presented in this report, closure of selected fisheries is the option which appears to offer the greatest benefit, to the greatest number of seabird species.

An area off the east coast of Scotland was closed to sandeel fishing from 2000 because kittiwake breeding success in the area had fallen to very low levels after intensive fishing for sandeels for several years by the Danish industrial fleet on the sand banks close to the east coast of Scotland. The aim of this fishery closure was to allow sandeel stocks in the area to recover (which sampling of sandeels in the area confirmed to be the consequence; Greenstreet et al. 2006), and kittiwake productivity was used as the proxy for sandeel abundance in the absence of data on sandeel stock biomass (Camphuysen 2005). Frederiksen and Wanless (2006) reviewed the evidence that this closure increased productivity of kittiwakes and other seabirds. They found clear evidence that the closure resulted in increased productivity of kittiwakes within the study area compared with a control area outside the closure zone, results which agreed with the earlier findings of Frederiksen et al. 2004) that the sandeel fishery in this area reduced productivity of kittiwakes on the Isle of May during the years of the fishery. Productivity of kittiwakes did not differ between fishery and non-fishery years outside the closure zone, but inside the zone breeding productivity was considerably lower during fishery years (the difference was 0.28 chicks per nest and statistically highly significant ($p<0.0001$). Analysis of productivity data for monitored colonies of other seabirds was based on much smaller sample sizes, and showed less clear results due to chance variations. However, for Sandwich tern the pattern was similar, within the study zone there was higher productivity in no fishery years (0.62 chicks per pair) than in fishery years (0.38 chicks per pair) whereas productivity in the control area was consistently high. However, high variation and small sample size (there were only two colonies within the closure zone) resulted in this difference not being statistically significant. Data on Sandwich terns in the closure zone showed that the colony at Sands of Forvie was more or less completely deserted from 1992-1998 and that this species did not breed on the Isle of May until 1999. Combined with the observed (non-significant) reduction in productivity in the closure zone during the fishery years, Frederiksen and Wanless (2006) concluded that '*this suggests that Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes*'. For Arctic tern, European shag, common guillemot, razorbill and Atlantic puffin, the closure could not be shown to have any strong effect on productivity by analysis of data for each individual species in turn, but a one-way MANOVA with data for all species showed a significant effect of the fishery closure on productivity of the seabird community, but this was predominantly driven by the strong and clear effect on kittiwakes. So, despite the clear evidence from the JNCC monitoring that food shortage has a strong impact on puffin productivity, analysis of the data from the Isle of May and east Scottish sites from the 1990s did not show this effect. Possibly food fish abundance needs to fall to lower levels to affect puffins than it does to affect kittiwakes and terns (Furness and Tasker 2000).

In Shetland, sandeel stock biomass fell rapidly in the late 1980s during the period of a local industrial fishery for sandeels, and that fishery was closed in 1990. Breeding success of kittiwakes in Shetland in 1990 was 0.12 chicks per pair. In 1991, the first year of the sandeel fishery closure, sandeel recruitment was higher than it had been in any of the previous seven years during heavy fishing, and kittiwake productivity in 1991 at 0.57 chicks per pair was the highest at monitored Shetland colonies since 1986 (JNCC data and ICES sandeel stock data). However, it is unclear whether the high recruitment in 1991 was caused by the fishery closure or whether the timing was coincidental. This fishery re-opened in 1994 based on high recruitment in the early 1990s, but local fishermen did not resume sandeel fishing to a significant extent, and the fishery effectively became extinct when sandeel abundance fell again in the early 2000s.

Closure of all sandeel and sprat fisheries in UK waters would bring the UK into the same management position as exists in the USA, where fishing on the small pelagic fish such as sandeels that are keystone species for marine food webs (including large predatory fish of high commercial importance) is prohibited. The considerable body of evidence shows that kittiwake productivity and adult survival would benefit from higher average abundances of these small prey fish, and the JNCC data together with the detailed research at the Isle of May (Frederiksen et al. 2004, Frederiksen and

Wanless 2006) show that the depletion of sandeel stocks has been the single most influential factor reducing productivity of kittiwakes at colonies in the British Isles, but evidence that this would affect puffin productivity is less strong. Closure of sandeel fisheries in UK waters would have little direct cost for British fishermen, since the fishery is almost entirely carried out by the Danish fishing fleet. Closure of sprat fisheries in UK waters would affect small numbers of fishermen in NW Scotland (there is a small fishery for sprats carried out by a few local fishing vessels near to Mull in Argyll) and there is a small fishery for sprats carried out by a few local fishing vessels in the English Channel (ICES 2013). Closure of these fisheries would be likely to have some beneficial effects for commercially important fish stocks such as cod and whiting (Greenstreet et al. 2006).

20.2.2 Eradication of rats

Rats are a factor which may be reducing productivity at a few colonies. Eradication of invasive alien rats could allow puffin productivity to increase at colonies where this predator is present. This applies at a very small proportion of colonies, but these include the Shiants where black rats have been present for around 100 years, co-existing with one of the largest puffin colonies in the British Isles. Whether black rats affect puffin productivity on the Shiants is unclear, as breeding success has not been monitored there. RSPB are currently seeking funding to eradicate black rats from the Shiants, in order to reduce the risk to the puffin colony on the Shiants and to increase prospects for storm petrels and Manx shearwaters to colonise.

Costs of rat eradication depend on the local logistics and especially on the size of the island, and whether or not there are important populations of native species that might be affected by an eradication programme deploying poison baits. For example, on Canna, special consideration had to be given to avoid eradication of wood mice and white-tailed eagles on the island (Bell et al. 2011). On Ailsa Craig, rat eradication had to avoid impacts on native slow worms, common lizards, pygmy shrew, gulls and ravens (Zonfrillo 2001). Programme adaptation can increase costs if such species need to be trapped and held in safe areas while poisoning is carried out. Nevertheless, the cost of projects on Ailsa Craig (a total of around £15,000 in present-day terms on an island of 100 ha so £150/ha, but including free helicopter transportation, Zonfrillo pers. comm.), and at Handa and Lundy, averaged £164 per hectare, with a range of £14 to £191/ha (Appleton et al. 2006). However, rat eradication on Canna, a larger island with a resident human population and several important native animal populations, and carried out by a very professional New Zealand company, cost £440/ha (Ratcliffe et al. 2009). Programmes to eradicate brown rats on Ramsey Island, Ailsa Craig, Handa, Canna and to eradicate both brown and black rats on Lundy, have shown a 100% success rate, indicating the potential of this approach for British islands. Furthermore, these successful projects provide a detailed literature on how to carry out successful eradication projects even on islands (such as Lundy) where there is a resident human population with a range of commensal animals (Appleton 2007).

20.2.3 Prevent risk of major oil spills near to SPAs

The effect of oil pollution on seabird populations is surprisingly uncertain. Votier et al. (2005) showed that major oil spill incidents doubled mortality of common guillemots from a Welsh colony in the years in which these occurred. The same is likely to be true of razorbills, given the similarity in their at sea behaviour. Major incidents such as the oil spill accidents considered by Votier et al. (2005) probably kill fewer seabirds than the chronic discharges of small quantities of oil from illegal discharges (Dunnet 1982). So the high influence of these major oil spills on common guillemot survival rates probably understates the influence of oil pollution. Nevertheless, previous attempts to relate changes in numbers of alcids attending colonies to oil spill mortality have often failed to show any detectable impact (see for example the lack of impact of the Braer oil spill in Shetland on seabird breeding numbers or breeding success http://www.nature-shetland.co.uk/birdclub/braer/Part10.html), suggesting that the consequence of increased mortality is not seen at the level of breeding numbers at colonies, so relationships between survival rates and breeding numbers may be quite complex. According to DEFRA '*In all of the oil spill disasters that* *have occurred during the SMP, guillemots and razorbills have predominated in the seabirds recovered. But despite large numbers of birds being killed, there does not appear to have been any substantial lasting effect on UK seabird populations'* http://jncc.defra.gov.uk/pdf/SMP%20Review%20Report%20Appendix%202%20-%20Impacts%20of%20pressures%20on%20Seabirds.pdf.

There is no evidence to suggest that oil spills contribute extensively to puffin mortality. Nevertheless, management to reduce amounts of oil pollution in UK waters would contribute, if only to a small extent, to increasing survival rates of puffins. While it is reasonable to assume that strenuous efforts are taken to minimize risk of serious oil spills, survival rates of puffins could presumably be increased slightly if incidence of oil spills could be reduced in their wintering areas. Preventing major accidents may be very difficult since there is already a very considerable effort to avoid such disasters. Preventing illegal discharges of oil that contribute to chronic oil pollution might be improved. Enforcement of laws prohibiting discharge of oil into the sea, with increased detection of culprits and increased effort into successful prosecution of offenders could reduce the amount of oil illegally entering UK waters.

21. SPECIES SPECIFIC RECOMMENDATIONS

Many of the seabird populations around the British Isles are currently declining in numbers after reaching long-term peak numbers late in the 20th century. Current declines are thought to be due to a wide range of factors, but primarily to climate change impacts on food abundance (especially sandeels), effects of changes in fisheries management such as reducing volumes of discards, and presence of alien mammal predators. Breeding numbers of seabirds on many SPAs are now below levels present at site designation; site condition monitoring is likely to lead to a conclusion of unsatisfactory condition for these populations. It is recommended that consideration is given to the fact that great skua, herring gull, great black-backed gull and lesser black-backed gull numbers in the UK may be elevated above naturally sustainable levels by historical provision of fishery discards (and for the gulls also urban refuse), and therefore that reduced numbers of these species at some UK colonies might be a desirable conservation objective.

If management action to increase survival or productivity of red-throated divers was considered to be desirable, the provision of nest platforms on selected breeding lochs (7.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of Manx shearwaters was considered to be desirable, the eradication of brown rats on Rum SPA, feral cats on Eigg, brown rats and feral cats on the Calf of Man, brown rats and feral cats on the Isles of Scilly, ferrets on Rathlin Island, feral cats on Fetlar, and feral cats on Foula (8.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of northern gannets was considered to be desirable, thetermination of the licenced harvest of young gannets from Sula Sgeir (9.2.2) is the most cost-effective approach. Such action would have only a very limited buffering capacity, but no other feasible management options that would have a greater effect on gannet demography have been identified in this evidence review. Quantification of fishery bycatch of gannets would be useful to assess if reducing bycatch rate could provide a potential compensation.

If management action to increase survival or productivity of Arctic skuas was considered to be desirable, the provision of supplementary food to breeding pairs (10.2.2) is recommended as the most cost-effective approach, providing carried out with necessary care to avoid attracting larger scavengers, and may need associated control of nearby great skuas'.

It is recommended that consideration is given to the fact that great skua numbers in the UK are elevated above naturally sustainable levels by historical provision of fishery discards, and therefore that reduced numbers of this species at UK colonies might be a desirable conservation objective. If management action to increase survival or productivity of great skuas was considered to be desirable, we suggest that there is no truly cost-effective approach to achieve this. Two options that could be considered further are closure of areas within 50 km of great skua SPAs to sandeel and sprat fishing (11.2.1) and supplementary feeding of great skuas at SPA colonies (11.2.2). Quantification of fishery bycatch of great skuas would be useful to assess if reducing bycatch rate could provide a potential compensation.

If management action to increase survival or productivity of lesser black-backed gulls was considered to be desirable, the end of culling of breeding adults (12.2.3) is recommended as the most cost-effective approach, with mink eradication from islands with colonies (12.2.1) and establishment of predator-proof fencing around mainland colonies subject to mammal predation, especially foxes (12.2.2) as potential further measures.

If management action to increase survival or productivity of herring gulls was considered to be desirable, the end of culling of breeding adults (13.2.3) is recommended as the most cost-effective approach, with mink eradication from islands with colonies (13.2.1) and establishment of predator-proof fencing around mainland colonies subject to mammal predation, especially foxes (13.2.2) as potential further measures.

If management action to increase survival or productivity of great black-backed gulls was considered to be desirable, the end of culling of breeding adults (14.2.1) is recommended as the most cost-effective approach, with mink eradication from islands with colonies (14.2.2)

If management action to increase survival or productivity of kittiwakes was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (15.2.1) is recommended as the most cost-effective approach. A further action could be exclusion of great skuas from holding breeding territories in buffer zones around kittiwake colonies (15.2.6).

If management action to increase survival or productivity of Sandwich terns was considered to be desirable, theexclusion of foxes from colonies by predator-proof fencing (16.2.5) and protection of colonies from flooding (16.2.7) are recommended as the most cost-effective approaches.

If management action to increase survival or productivity of common terns was considered to be desirable, the eradication of mink from islands with natural colonies of common terns (17.2.2) and deployment of predator-proof nesting rafts (17.2.7) are recommended as the most cost-effective approaches, and possibly the control of local gulls that specialise in eating tern chicks (17.2.8).

If management action to increase survival or productivity of common guillemots was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (18.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of razorbills was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (19.2.1) is recommended as the most cost-effective approach.

If management action to increase survival or productivity of Atlantic puffins was considered to be desirable, the closure of all UK waters to fishing directed at sandeels and directed at sprats (20.2.1) is recommended as the most cost-effective approach, and possibly eradication of rats from islands with large colonies of puffins (20.2.2).

Notwithstanding specific requirements of the Birds Directive, we believe that we should be seeking to intervene to ensure functioning ecosystems rather than attempting to manage individual populations as if these do not interact. If management actions are required for several species, some of these recommended approaches may be effective across several species (for example closure of sandeel and sprat fisheries could benefit many seabird species) while other measures could potentially have negative interactions (for example measures to increase productivity or survival of some large gull populations may have negative consequences for some tern populations). In such cases there may be a need to consider spatially resolved management to reduce undesirable interactions.

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