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**Developing a Habitat Loss Method for Assessing Displacement Impacts from  
Offshore Wind Farms**

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**Evidence Quality Assurance**

The contractor (APEM) has followed the JNCC Evidence Quality Assurance Policy in compiling this report <http://jncc.defra.gov.uk/default.aspx?page=6675>.

As part of the JNCC Evidence Quality Assurance (EQA) process, the tender specification for this contract was reviewed internally within JNCC and externally by at least three SNCB colleagues from amongst the other agencies. The final draft report was circulated for peer-review and received input from two SNCB colleagues (Natural England and Scottish Natural Heritage) and an independent, external peer-reviewer (RSPB).

## Summary

The impacts of offshore wind farms (OWFs) on seabirds are typically divided into three forms: collision mortality, barrier impacts and displacement impacts. This report investigates the validity of assumptions underlying a recently suggested approach, defined as the 'Displacement as Habitat Loss' approach. This approach interprets displacement impacts as a time restricted, quasi-permanent habitat loss rather than a year-on-year effect across the entire operational lifetime of an OWF. It proposes that after displacement impacts have been exerted on a population, a new, stable population equilibrium is reached at some level below the initial starting population. By examining the assumptions underlying this approach and outlining ways of translating it into a staged assessment framework, this report provides an analysis of the pros and cons of the 'Displacement as Habitat Loss' approach.

An assessment framework has been developed that is presented, discussed and applied in case studies for two species. Additionally, alternative ways of developing a staged framework to the 'Displacement as Habitat Loss' approach are considered. It has been found that, independent of the focus of any assessment framework, challenges relating to the understanding and assessment of ecological processes governing the strength of any potential displacement effect, will affect the delivery of any potential assessment framework. Estimating (by season) the size of the original habitat, the habitat remaining after displacement impacts have occurred, and the likely number of birds that may need to relocate to alternative (remaining) habitat, appears possible but is challenging in its own right. Available evidence on percentage displacement suggests strong site-specific variations. For assessments the use of generic displacement categories for species (such as high, medium or low sensitivity) may be best, unless variations can be explained by, for example, habitat quality, which could represent a factor governing the strength of any site-specific displacement effect.

Considerable evidence gaps as well as difficulties with quantitatively assessing those ecological processes, which are subject to considerable natural variability, impede the assessment of the biological consequences of displacement. Quantification of the likely fitness implications of displacement requires a detailed understanding of the degree to which local populations are below, at, or above carrying capacity. This appears to be a precondition of any attempt to estimate what proportion of displaced birds are forced to relocate to alternative habitat and the proportion of those that might be accommodated within remaining habitat without considerable energetic costs. Moreover, if population equilibrium exists in seabird populations, it may not be possible to reliably characterise the point at which this equilibrium has been reached, due to the strong fluctuations that are inherent in marine ecosystems. Population equilibrium is unlikely to be constant and thus separating displacement effects from natural population fluctuations, as well as other pressures on the population being assessed, may be impossible.

A clear benefit of a staged assessment framework is that it provides transparency on where assumptions are made in the process and at which stage they are introduced. This is an advantage over the existing 'Displacement Matrix' approach as it helps breakdown the particular areas of uncertainty needing further research. The 'Displacement as Habitat Loss' framework, particularly in later stages, requires empirical evidence on seabird species that are absent from the literature. Each stage has particular uncertainties associated with it, meaning that cumulatively an unacceptable level of uncertainty can be the result. The 'Displacement as Habitat Loss' approach provides a thorough theoretical framework to assess displacement impacts from OWFs. However, the absence of empirical evidence to support/refute various the assumptions required throughout mean that in practice the framework is currently unlikely to be applied in a 'real-world' scenario, until some of the uncertainties have been further explored.

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# 1 Introduction

## 1.1 Background

The impacts of the development of offshore wind farms (OWFs) on seabirds are typically divided into three forms: collision mortality, barrier impacts and displacement impacts. This report focuses on the assessment of displacement impacts in the context of ornithological impact assessments for OWFs and investigates the validity of assumptions underlying a recently suggested approach, interpreting displacement impacts as a time restricted, one-off habitat loss effect rather than a year-on-year effect across the entire operational phase of an OWF.

Displacement assessment represents one of the main current challenges in environmental impact assessment (EIA) and habitat regulation assessment (HRA) studies addressing the impacts of OWFs on seabirds and their habitats. Several seabird species show a behavioural response to specific stimuli from operational OWFs such as rotating blades or other activities related to the OWF operation and maintenance (Fox and Petersen 2006). Several species permanently avoid entering or closely approaching OWFs and accordingly are displaced from the OWF footprint and a buffer zone around the development with the consequence that these areas are lost (or partially lost) as foraging habitat. Accordingly longer term displacement can be described as a physical habitat loss. While some of the seabird species most vulnerable to displacement effects have been identified and the need for detailed consideration of potential impacts is commonly agreed, considerable evidence gaps can impede current assessments.

Evidence is sparse, for example, in relation to the ecological consequences of seabirds experiencing displacement. The underlying ecological processes governing the strength of any potential impact are often understood and assessments attempt to acknowledge those. Nonetheless, it often remains challenging to breakdown those large scale processes that are subject to strong variability due to the spatial scale and the number of individuals relevant to an impact assessment.

### 1.1.1 Current interim advice

Interim advice from the statutory nature conservation bodies (SNCBs) in the UK currently recommends a broad level assessment of potential displacement impacts based on consideration of the full range of potential displacement of birds within defined project areas (0-100%) and mortality rates of the displaced birds (0-100%) with the help of a displacement matrix approach (Natural England and JNCC 2012). To narrow down the assessment range, JNCC have previously recommended, for certain specific project applications, that the disturbance sensitivity scores of Furness *et al* (2013), translated into percentage displacement categories (see Table 1), be used to frame the 'Displacement Matrix' approach (in the absence of species-specific, robust empirical data). However, this represents a theoretical framework, informed by relative disturbance rankings, rather than being based on empirically derived data, which are largely absent. Alternatively, using the highest reported species-specific displacement rates from post-construction monitoring studies has been suggested. Given the very great uncertainty concerning realistic levels of mortality of displaced birds, JNCC have previously advised, for certain specific project applications, that a 10% adult mortality rate could be used for some key species (e.g. guillemot, razorbill, kittiwake, gannet and Manx shearwater) (JNCC 2014a), while some developers argue that those values are overly precautionary.

**Table 1.** JNCC have advised the following displacement categories, based on Furness *et al* (2013) disturbance sensitivity score.

Disturbance Score	% Displacement	Example Species
1	0-20	Fulmar, Manx shearwater
2	20-40	Sandwich tern, lesser black-backed gull
3	40-60	Guillemot, razorbill, shag
4	60-80	Common goldeneye, greater scaup
5	80-100	Red-throated diver, common scoter

It should be noted that other (though similar) categories have been provided in advice from other SNCBs in the recent past. For example, Natural England advised for auks in relation to the Hornsea Project One a displacement range of 30-70% to be combined with an adult mortality of between 1% and 10% (Natural England 2014).

In this context it is important to note that there are indications that reduced breeding success may represent the main ecological consequence of displacement rather than adult mortality, at least during the breeding season (e.g. Langton 2013, Furness 2013). However, methods to appropriately integrate predictive productivity and adult mortality impacts into a single displacement assessment framework have yet to be developed.

### 1.1.2 Focus of this report

Key for the questions addressed in this report is the fact that in recent impact assessments displacement impacts have been considered as additive impacts that occur repeatedly each year across the entire operational phase of an OWF. It is assumed that a given percentage of birds normally present within a site are displaced and that a certain percentage, of those that are displaced, suffers from mortality as a result. This impact should then be added annually to baseline natural mortality rates, in addition to collision mortality predictions, and examined using a population model considering the lifetime of the OWF.

However, this approach suffers from a lack of evidence on various aspects of displacement, including:

- the spatial extent and degree of displacement;
- whether the ecological consequences are more likely to be adult mortality and / or reduced reproduction;
- temporal consistency of displacement across different seasons (breeding / non-breeding season); and
- the length of time over which impacts would occur at a particular strength.

This has led to a variety of different assumptions and approaches being taken for assessing displacement impacts, and different frameworks being suggested to broadly account for these impacts.

This report focuses on a proposal by an OWF developer to treat displacement impacts as a time restricted or 'one-off' habitat loss event which would result in a short-term reduction in the size of the impacted population as it reaches equilibrium with the reduced habitat and thus resources available (Furness 2013, Forewind 2014). Once this new population equilibrium is achieved, no additional annual mortality or reduced reproduction might reasonably be predicted occur as a result of the initial habitat loss. It is also suggested that displacement impacts could be assessed by the current displacement matrix approach

(Natural England and JNCC 2012), but that the derived displacement-mortality figure would only need to be accounted for once, i.e. as a one-off event.

In that approach, displacement is considered to represent effective habitat loss (of foraging resource) to birds, the loss of which may lead to mortality. However, mortality would not be considered to represent an on-going annual event, but would involve adjustment of the population to the revised foraging resource area available once an OWF became operational. Following this perspective displacement impact can be interpreted as a **one-off event** that may have an **initial population impact**. Subsequently the population would then be subject to the 'normal' drivers of change.

This interpretation of displacement impacts itself is based on a number of assumptions which have not been verified so far and will be investigated in this report. Furthermore, no assessment framework implementing the suggested approach in a way applicable to EIA studies has been drawn up. Within this report we suggest a potential framework to implement the approach and assess whether the assumptions underlying each stage are defensible, in an early attempt to determine whether the suggested approach has value.

While the approach is sometimes referred to as the 'One-off Habitat Loss' approach, this denomination appears misleading because 'one-off' suggests an almost immediate realisation of all impacts; thereafter returning to the status quo.

To better reflect the assumptions underlying the approach developed in this report, we suggest it is referred to as the '**Displacement as Habitat Loss**' approach and that phrase is used for the purpose of this report.

This report principally focuses on evidence of displacement as affecting seabird populations using United Kingdom waters.

## 1.2 Objectives

The following objectives have been considered in this report:

1. To identify, describe and consider the validity of assumptions underlying the treatment of displacement impacts as a time restricted habitat loss (principally reduction in population size to equilibrium after the initial period).
2. To review the empirical / theoretical evidence underpinning each assumption identified in Objective 1, allowing the assessment of which method (compared to the displacement matrix approach) has more empirical support.
3. To provide a methodological framework for implementing a displacement impact assessment process using the suggested approach, clearly sign-posting where the various assumptions fit into the framework.
4. To provide final recommendations on the overall suitability of categorising displacement impacts using the 'Displacement as Habitat Loss' concept.

5. Identify what additional monitoring or research could be done to shed light on some of the most critical (or least evidence based) assumptions, that could act to improve the validity and wider application of any 'Displacement as Habitat Loss' displacement assessment tool.

### **1.3 Structural approach**

Based on discussion among the project team and JNCC it was agreed to start the work by developing an assessment framework implementing the 'Displacement as Habitat Loss' approach to displacement (Section 3) and to then identify assumptions underlying each stage of that assessment framework. In addition further assumptions underlying the approach but not covered within the stages of the assessment framework were identified.

After this process had been finalised a literature review (Section 2) was carried out to identify available evidence and, based on this, formulate evidence summaries (Section 4) investigating the validity of each assumption listed.

Based on the evidence summaries it then was possible to broadly assess the suitability of the 'Displacement as Habitat Loss' approach and to identify the stages of the derived assessment framework implementing the approach in context of EIA studies that are backed up with evidence and whether the information needed for a successful implementation would be available (Section 3).

To illustrate the applicability of the assessment framework, red-throated diver, a species thought to be vulnerable to displacement from OWFs in a UK context, has been taken through the assessment process (Section 5). Lesser black-backed gull, a species that does not appear to be vulnerable to displacement has also been taken through the assessment process to provide context.

In addition, this assessment of a potential framework to implement the approach has been able to identify key evidence gaps requiring research and targeted monitoring in the context of displacement assessment (Section 7).

## 2 Methods

### 2.1 Introduction

The key methodological tool applied to assess the validity of the 'Displacement as Habitat Loss' approach was a directed literature review investigating the appropriateness of the assumptions underlying the approach and any assessment framework implementing it.

The literature review was structured to ensure a comprehensive and unbiased assessment of the assumptions identified, based on the most relevant literature on the various ecological concepts underlying the approach and comprised two steps. The extraction of the relevant literature (1.) and the evaluation of the relevant literature (2.) aiming at the compilation of evidence summaries providing a condensed overview on the validity, respectively state of knowledge, of assumptions to be tested.

### 2.2 Extraction of relevant literature

The search for relevant literature considered peer-reviewed publications as well as grey literature on the subject.

#### 2.2.1 Peer-reviewed literature

Peer-reviewed publications were extracted from the comprehensive bibliographic database 'Scopus' that provides access to more than 55 million articles from about 20,000 peer-reviewed scientific journals. Our search results were reported as an 'EndNote' library.

A list of key terms to be searched for within the database was derived from identified key assumptions underlying the 'Displacement as Habitat Loss' approach (see Table 2). These terms were searched for in combination with the term 'bird' (e.g. displacement + bird) in 'Scopus' to acquire broad information on the topic that is relevant to birds.

All search results in 'Scopus' were exported to an 'EndNote' library and literature folders (so called 'smart groups') were set up for each key term. Moreover, search results were ordered by the number of citations in 'Scopus' and the top 10 most cited and relevant papers were identified. A relevant paper in this context is one that relates to our offshore biodiversity interest; thus, if e.g. during a search for the term 'carrying capacity' papers on economy rather than population ecology occurred among the top 10 most cited, the economy papers would be excluded and only the top 10 cited papers interpreting carrying capacity as an ecological term identified and listed. The literature review focused on the top 10 most cited relevant papers, but in situations where these did not provide sufficient information, further search results were considered.

In addition, each key search term was then combined with each species / species group identified to be of relevance, e.g. 'carrying capacity + auks' etc (see Table 2) to identify seabird-specific evidence on the respective key search terms, respectively ecological concepts. UK seabird groups with a medium to high disturbance sensitivity score in Furness *et al* (2013) were used as a starting point. However, not all such groups were considered as some are considered of minor OWF relevance and little is published about them (e.g. grebes), and for others such as great cormorant there is evidence that the species is attracted to OWFs (Leopold *et al* 2011). In addition, well researched seabird species / species groups were considered, independent of their vulnerability to displacement, to make sure relevant, potentially transferable evidence on underlying ecological processes was not missed by the literature review. As an initial search indicated that including 'diver' as a search term could result in the identification of many irrelevant papers (e.g. relating to scuba

diving etc), 'Gavia' was allowed as an alternative search term for this species group if no suitable literature could be found using the original search term. Again all search results of the search strings were exported from 'Scopus' and the top 25 most cited papers were identified to form the core of the review, with further search results being available within the 'Endnote' library if required in search string specific literature folders.

The initial evaluation of relevant literature was done by reviewing abstracts and whenever possible full versions of key papers were accessed and evaluated.

A few further terms of relevance were identified (see Table 2, right hand column) that were used in a second stage filtering of the search results. These search terms were not used as search terms to extract literature from 'Scopus' (a series of trial searches identified that too many irrelevant search results were returned) but were used for searches within the 'EndNote' library that had been constructed from the initial literature search.

**Table 2.** Search terms for literature extraction.

	<b>Key search terms from underlying assumptions</b>	<b>Species groups</b>	<b>Further terms used for search within body of extracted literature</b>
1.	displacement	seabirds	impact
2.	survey methodology	divers (alternative: <i>Gavia</i> )	assessment
3.	foraging	seaducks	population
4.	habitat quality	auks	modelling
5.	carrying capacity	gannet	variability
6.	density dependence	kittiwake	
7.	mortality	fulmar	
8.	reduced reproduction	shearwater	
9.	equilibrium	shag	
10.	immigration	waders	
11.	habituation		
12.	habitat extent		

### 2.2.2 Grey literature

A second source of information was also compiled from the grey literature. This was compiled jointly from a 'Google' and 'Google Scholar' search and the library of industry and SNCB publications that had already been brought together by APEM's technical staff and held on its own servers. The Google searches and the search of APEM's library focused on seabird, waterbird and wader displacement resulting from industrial development (including OWFs). This identified in particular the information contained in OWF post-construction monitoring reports (including initial results on displacement rates) and the research reports on displacement as a result of development commissioned by UK Governmental bodies.

## 2.3 Literature analysis

The literature analysis focused on the assumptions identified to underlie the 'Displacement as Habitat Loss' approach and investigated their validity. This was done by assessing the

literature within the specific 'EndNote' library folders and searching the grey literature for the respective key term. For each assumption an evidence summary was compiled.

Identified evidence (especially from grey literature, as this has not been subject to peer-review) was critically assessed. To give an indication of whether the specific references used within the evidence summaries (Section 4) have undergone a quality assurance process, a categorisation was used. The categorisation was based on whether the publication had been subject to independent and anonymous (blind) peer-review; to have been reviewed by some kind of external steering group; or to have only been subject to internal review. The category is identified through each reference in the evidence summaries (Section 4), being followed by a superscript abbreviation indicating the quality assurance process that the reference has undergone.

To record the categorisation described above, the references cited in the evidence summaries are coded with the following superscripts:

- **Peer-Reviewed** evidence (PR) = peer-reviewed journal papers.
- **Externally quality assured Publication** (EP) = books and evidence based grey literature such as reports to government departments or agencies, e.g. the CEH report to Marine Scotland (Searle *et al* 2014) and comprehensive reports for long term monitoring studies that have been quality assured by a project steering group drawn from a wide range of stakeholders.
- **Internally quality assured Publication** (IP) = reports to private clients that, as far as can be determined, have not been reviewed other than internally (i.e. by the consultant and the client), such as environmental statements, site monitoring reports and topical reviews.

### **3 Displacement assessment framework**

One key objective of the project is to deliver a methodological framework for a displacement impact assessment process that implements the 'Displacement as Habitat Loss' approach. The framework that is sought has to be both detailed enough to cover and consider the complex ecological processes underlying displacement impacts and also pragmatic enough to provide a non-specialist with a quick overview of the approach and relevant information for the impact assessment decision making process.

The framework that has been developed to implement the 'Displacement as Habitat Loss' approach is illustrated in Figure 1. It proceeds through 10 stages and accompanying each stage is one or a series of assumptions. Those assumptions that underlie each stage of the assessment process are tested for their validity, based on the evidence available, in Section 4.

During the course of the project various potential assessment frameworks have been considered. While there are different ways of approaching a staged displacement assessment, key challenges - relating to the understanding and assessment of ecological processes governing the strength of any potential displacement effect - are an integral part of all the frameworks trialled.

Nonetheless, in addition to an individual-centred assessment framework representing the core assessment framework suggested by this project and discussed in detail in sections 3.1.1 to 3.1.10 (Figure 1), an alternative, more habitat-centred assessment framework is presented too (Figure 2) and shortly discussed in Section 3.2.

#### **3.1 Stages of the assessment framework**

The sections below describe each stage of the displacement assessment framework (as illustrated in Figure 1) that implements the 'Displacement as Habitat Loss' approach. The stage-by-stage descriptions identify the information and measures needed to move through each stage of the process.

While Stages 1 to 4 slightly differ according to season, assessment for breeding and non-breeding seasons progresses in the same manner from Stage 5 onwards. To estimate year-round displacement impacts, assessment results from breeding and non-breeding seasons would need to be added together after apportioning impacts to colonies / regional populations interacting with the OWF assessed during both seasons. It has to be noted that relating non-breeding season impacts back to specific colonies may not always be possible. In those situations non-breeding season impacts may only be assessed against a spatially distinct Biologically Defined Minimum Population Scale (BDMPS) (Furness 2015) in combination with displacement impacts from other OWFs affecting the respective non-breeding season population scale.

Section 4 that follows considers the assumptions underlying each stage of the assessment framework (as identified on Figure 1), providing an evidence summary for each.

In order to establish the framework and its stages, a pragmatic approach has been taken in this section, moving sequentially through each stage and introducing generic assumptions or rules in order to ensure that there is a progression through the stages and the entire description of the assessment is completed. The analysis of the adequacy of evidence behind those generic assumptions or rules is set out in Section 4.

### 3.1.1 Stage1: Define size of impacted area

The definition of the size of the impacted area will consist of two steps:

- i. The first step is a generic one. Independent of the species for which displacement impacts should be assessed, a definition of the size of the development footprint will form the basis for defining the size of the area where a displacement affect would occur. It also contributes to the calculation of the remaining habitat extent applied in *Stage 4: Define geographic extent of remaining habitat*.
- ii. In the second step, a species-specific buffer zone will need to be added to the development footprint to derive the species-specific spatial extent of the area / habitat that would be affected by a displacement impact. It also contributes to the calculation of the remaining habitat extent applied in *Stage 4: Define geographic extent of remaining habitat*.

Whilst information on species-specific buffers is sparse, there are studies for some key species that identify the geographic extent of the effect, as described in the evidence summary relating to Stage 1 (Section 4.1). In situations where such studies are missing generic buffer zones would need to be applied. This could be in a form of ranked scale (e.g. adding a 1km buffer for relatively insensitive species, 2km buffer for moderately sensitive species and 4km buffer for highly sensitive species). The decision on how precautionary an approach is taken to placing species on such a scale, in the absence of suitable evidence, should be made with the advice of the SNCB responsible for the relevant area of sea.

Stage	Underlying assumptions
1.	Sufficient knowledge on appropriate, species-specific buffers to define geographic extent of impacted area
2.	Survey methodology and assessment methods suitable to report densities at a precision allowing assigning changes to OWF operation
3.	Sufficient knowledge on % displacement
4.	Ability to define the extent of specific seasonal habitat available to a local population
5.	Sufficient knowledge about carrying capacity of respective population
6.	
7.	Ability to define proportion of birds that could still be supported by remaining habitat/available resources considering habitat quality & density dependence
8.	Immediate impact
9.	Ability to define if affected pop. suffers from mortality or reduced reproduction; Pop. will adjust to new equilibrium; length of such adjustment process; no immigration into affected population causing ongoing effect;
10.	Ability to reliably predict time scales over which new equilibrium is achieved; consider potential habituation impacts

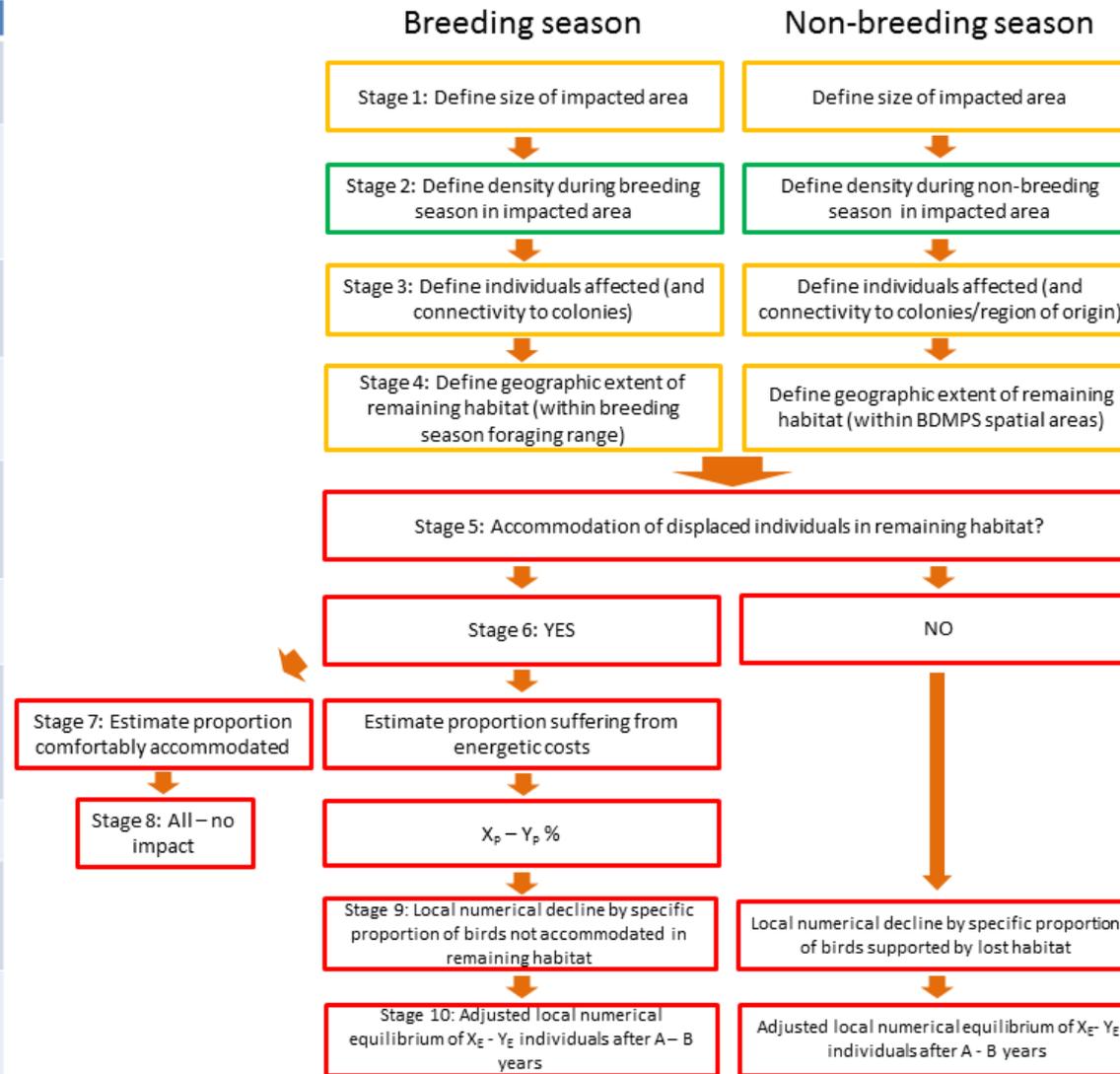


Figure 1. Displacement assessment framework.

### **3.1.2 Stage 2: Define density in impacted area (season specific)**

The use of surveys and appropriate analytical tools to determine seabird densities within marine areas is a standard procedure for establishing baseline data for impact assessments of OWFs. Information about the species-specific (natural) densities before any impact occurs will be defined by the baseline surveys (normally derived from two years of survey). Densities are normally expressed as individuals per unit area (e.g. ind/km<sup>2</sup>) and may be presented for different sub-areas of the survey area. Accordingly information on densities within the OWF footprint and relevant buffer zones (generic or derived from a species specific evidence base) will be available to inform this stage of the assessment process. If species-specific studies on the extent of displacement within buffer zones are available, densities for these respective zones around the footprint should be included. Densities are likely to fluctuate between seasons and years. Accordingly, seasonal densities (as a minimum breeding and non-breeding seasons and where appropriate the spring and autumn migration periods) should be considered independently when assessing displacement impacts (current SNCB guidance on species specific seasons is contained in Furness 2015). To at least partly account for the strong inter-annual variability within particular seasons the use of mean densities derived across the years for which data are available should be considered. Where available, densities should be related to hydrodynamic data in distribution models to explain some of the variation in bird numbers. To balance precaution within any assessment it may be sensible to take mean maximum seasonal densities forward to assessment.

### **3.1.3 Stage 3: Define individuals affected (and connectivity to colonies / region of origin)**

Stage 3 combines the information gathered by Stages 1 and 2 of the displacement assessment framework by calculating the maximum number of individuals of each relevant species that could be affected by a potential displacement impact. This is done by multiplying the extent (km<sup>2</sup>) of the species-specific impacted area (Stage 1) with the species-specific density (ind/km<sup>2</sup>) (Stage 2). The next step is that the strength of any species-specific displacement impact, i.e. what percentage of displacement is likely to occur, needs to be considered. While it is apparent from recent studies (Section 4.3.1) that displacement in most cases does not affect 100% of individuals of even sensitive species, there is still considerable uncertainty about the extent of displacement (in percentage terms) likely for a given species. Moreover, there are indications that the strength of displacement impacts for a species may strongly differ from site to site, probably according to the habitat quality of the impacted area and other factors (Furness 2013). While for some key species, like red-throated diver or guillemot, species-specific estimates of percentage displacement are available from post-construction monitoring, for many species this information is lacking. Where empirical data are lacking, one currently advised approach is to use the disturbance sensitivity score in Furness *et al* (2013) as a rough proxy of percentage displacement (see Table 1). Once individuals potentially affected by an OWF development have been identified, connectivity to specific colonies (during the breeding season) or a spatially distinct biologically defined minimum population scale (Furness 2015) (during the non-breeding season) has to be assessed. For the breeding season, there are established apportioning processes which may be suitable for application. These consider colonies within foraging range of a project and then apportion potentially displaced individuals according to the distance and size of their respective colonies. If available, tracking data should also be considered. During the non-breeding season, when individuals are much more dispersed, affected individuals could be allocated to a spatially referenced biologically defined minimum population scale (Furness 2015). Turnover of individuals within an OWF site may influence the number of birds potentially displaced (potentially increasing it) or the severity of any displacement effect (potentially reducing it). Due to a lack of information on turnover rates

for specific sites (which to remedy would require information acquired from, for example, extensive vantage point surveys from anchored vessels or individually marked birds) it appears unlikely that the potential implications of the turnover of individuals could be considered in most displacement assessments.

### **3.1.4 Stage 4: Define geographic extent of remaining habitat (season specific)**

The definition of extent of remaining habitat, after the realisation of a displacement effect, requires the extent to which the originally available habitat is reduced by the development footprint and buffer (defined in Stage 1). Representative breeding season foraging ranges can give an indication of the approximate extent of the foraging habitat of breeding seabirds. Nonetheless, if available, additional layers of information, like colony-specific tracking data, at-sea surveys and data on environmental variables should be modelled together to identify the likely areas of usage for particular colonies (see [Thaxter \*et al\* 2012a](#), for methods on characterising areas used by colonies). It is worth noting that difficulties accessing all of the necessary information and allowing for the fact that seabirds can increase their foraging ranges (e.g. in years of low prey abundance) may make this difficult. The loss of potentially suitable foraging habitat due to displacement may lead to extended/deviated foraging flights and associated effects on energy budgets and attendance by adults of the egg(s) and chick(s) at the colony. Available evidence summarised in Section 4.4 indicates that while it appears possible to roughly estimate the extent of colony-specific foraging habitat, its extent is likely to vary in accordance with prey availability. Accordingly the extent of the remaining habitat after realisation of a displacement impact is likely to fluctuate. Simulation models such as those of Langton (2013), predicting the maximum range adults can forage while still achieving a specified fledging success, allow the identification of areas at sea where successful individuals from specific colonies could be foraging, may present a potential way addressing this stage of the assessment process.

For the non-breeding season it appears unrealistic, for most species, to define the extent of colony-specific foraging habitats (see Section 4.4). During this season impact assessment could focus on spatially distinct biologically defined minimum population scales (BDMPS) (Furness 2015). Defined spatial units, such as the NW North Sea, SW North Sea and Channel etc, identified for all relevant UK seabird species could be used as the non-breeding season assessment scale, providing a rough estimate of the non-breeding season foraging habitat for birds originating from a range of colonies. The spatial extent of OWF developments and buffers within defined sea areas could be combined and used to calculate the extent of the remaining non-breeding habitat available within those defined sea areas. This does though assume that there are no other human impacts in those defined sea areas that could also be displacing birds (for example ship traffic can displace certain seabirds: Schwemmer *et al* 2011). For certain species, such as those that are highly aggregated and / or bound to certain water depths (e.g. seaducks, divers), there may be potential to refine further the extent of remaining habitat within BDMPS spatial areas by considering factors such as water depth. This aspect of different areas of habitat quality is considered as part of the alternative, habitat based assessment framework discussed in Section 3.2.

### **3.1.5 Stage 5: Accommodation of displaced individuals in remaining habitat?**

To implement Stage 5 of the assessment framework, it is key to be able to determine whether the remaining habitat has the potential to accommodate additional individuals, or whether the locally affected population within its defined habitat is at carrying capacity.

A key challenge in this context, again, is the natural variability in prey resources. Furness (2013) argues that seabirds are pre-adapted to cope with variable prey densities and that even in years with low prey stock there is little adult seabird mortality, just a decline in

reproductive output. Depending on year and season-specific prey abundance, local populations and breeding colonies and larger non-breeding season populations at BDMPS spatial areas (Furness 2015) may be at or approaching carrying capacity during one year and accordingly unable to accommodate displaced individuals without fitness consequences, while in another year displaced individuals from elsewhere may be accommodated without any noticeable effect on the fitness of the individuals already present. In years of low prey abundance information on reproduction rates could indicate that a local population is either close to carrying capacity if there is a marked reduction in reproduction, or not if reproduction remains stable / high. Where a local population's productivity yields a demographic excess, while showing a stable or decreasing population status, this could be indicative of the population been above carrying capacity during the non-breeding season.

**Stage 5 has two possible outcomes** (defined in stage 6):

**NO** – the remaining habitat cannot accommodate additional / displaced individuals because the local population is at carrying capacity. This could be indicated, for example, by a long-term decline in the local breeding and / or non-breeding population or low breeding success in the local colony before any displacement impact. Accordingly displaced birds would have to leave their traditional habitat in search of alternative prey resources. [Burton \*et al\* \(2006\)](#) demonstrate a 7% reduction in the annual survival of wintering adult redshanks displaced from Cardiff Bay following its flooding due to a significant decline in winter survival for (at least) three years after displacement.

**YES** – the remaining habitat can accommodate additional individuals because the local population is not at carrying capacity. Whether local populations are likely to be at carrying capacity or not in any one year can only realistically be assessed from long-term demographic data, ideally in combination with long-term information on prey abundance. Relatively stable populations could indicate a population close to carrying capacity during years of average prey abundance and a long-term increase in the population could indicate that the population is not yet at carrying capacity. These scenarios would indicate the absence / presence of the capacity of the local population to accommodate additional individuals that might be displaced from elsewhere.

### **3.1.6 Stage 6: YES / NO (intermediate stage)**

As indicated above, stage 6 represents an intermediate stage.

**YES** - If Stage 5 identifies that the remaining habitat has the potential to accommodate displaced individuals, the assessment moves on to stage 7, which looks in more detail at the potential costs for individuals accommodated within the remaining habitat.

**NO** - If Stage 5 identified that the relevant population is at carrying capacity, and thus unlikely to be able to accommodate displaced individuals, the assessment moves on to Stage 9.

### **3.1.7 Stage 7: Estimate proportion comfortably accommodated / Estimate proportion suffering from energetic costs**

Implementation of Stage 7 requires the ability to determine the extent to which a local population is at carrying capacity. Furness (2013) indicates that the biomass of key seabird prey species (sandeel, sprat and capelin) varies considerably. In the Doggerbank region, for example, sandeel recruitment varied by a factor of up to 190 between 1983 and 2011 (ICES 2010). Therefore estimating the proportion of displaced individuals that may be comfortably accommodated in the remaining habitat, as well as those affected by energetic costs in

specific years, is very difficult. In addition to variability in prey abundance, species-specific information on density-dependence and information on habitat quality have to be considered when estimating the proportion of displaced individuals that may be accommodated.

### **Density-dependence**

Problematic in this context may be that, once a local population is at carrying capacity, the energetic cost of additional displaced individuals attempting to find accommodation within the remaining habitat, may be spread across the entire population rather than affecting only those relatively few additional displaced individuals. This complexity may need to be ignored (in a quantitative sense) to be able to proceed with the assessment, by assuming that those additional individuals would be excluded from the local population and would need to cope with the ecological consequences of displacement. Accordingly, for simplicity, one may have to assume that displacement consequences are only exerted on those birds that have been displaced and are not exerted on the 'receiving' population (that is the population of birds in to which the displaced individuals attempt to mix).

Evidence summaries on these aspects (see 3.5, 3.7, and 3.8) indicate that while the relevance of, for example, density-dependence to regulate population growth rates in seabirds has been demonstrated; it may be very difficult and probably often unrealistic to be able to tell, at what number of individuals per km<sup>2</sup> (Ind./km<sup>2</sup>) density-dependent processes would start influencing population growth rates. Nonetheless, if that information is available for specific species, the extent of the remaining habitat could be multiplied by the number of individuals / km<sup>2</sup> (density) that could be supported before density-dependent effects occur. If the result indicates that all displaced individuals could be accommodated, this may speak against a displacement impact.

### **Habitat quality**

Another important factor influencing the capacity of the remaining habitat to accommodate displaced individuals, and an alternative way to approach this assessment Stage, will be the quality of the lost habitat in the context of the quality of the remaining habitat.

Habitat quality in the marine environment is very difficult to define as it strongly fluctuates with abundance and distribution of prey resources. This is the case for mobile prey resources, like small pelagic fish, that are the key prey resource for the majority of seabird species (seaducks feeding on molluscs represent an exception). Long term data series on prey abundance or seabird densities, acting as a proxy for prey abundance, should be considered to identify patches of different habitat quality within the overall area of usage during a particular season. One source of information could be the Seabird Mapping and Sensitivity Tool (SeaMaST) commissioned by Natural England and the Marine Management Organisation (Bradbury *et al* 2014). The tool is based on the European Seabird-At-Sea database (ESAS) and provides access to long time series of seabird density at particular locations. The ESAS data has been used in the past to assess at a relatively coarse scale the habitat quality in different OWF project areas to inform advice on potential mortality levels resulting from displacement (e.g. Hornsea Project One and Dogger Bank Creyke Beck projects).

Good quality habitat could be indicated by relatively small variations in high average seabird densities across years, medium habitat quality by average densities with higher variability across years, and low quality habitat by low average densities and probably low variation across years. An understanding of the relative quality of lost and remaining foraging habitat may make it possible to estimate the capacity of the remaining habitat to accommodate displaced individuals. One possible way forward could be to define broad categories relating the quality of the lost / impacted habitat to the proportion of displaced individuals likely to find

accommodation within the remaining habitat. For example, if based on Stages 5 and 6, the assessment were to conclude that the local population was not at carrying capacity and that the remaining habitat had the capacity to accommodate displaced individuals, it could be presumed that higher quality habitat would be able to accommodate larger numbers of birds than an equal area of lower quality habitat (although evidence has not been found to support this). However, there is currently no information available from which to derive any quantitative measures or ranges for such an effect.

### **3.1.8 Stage 8: All – no impact / range of impacts**

**All – no impact** – If Stage 7 concludes that all displaced individuals are comfortably accommodated within the remaining habitat, the assessment concludes no displacement impact and the assessment process can be stopped at this point.

**Range of impacts** – If Stage 7 concludes that a certain proportion or range of displaced individuals is likely to suffer from energetic costs as a consequence of displacement, then the assessment of the likely consequences of displacement for these individuals continues under Stage 9. In Figure 1 and Figure 2 this range in the proportion of the population that are displaced and are subject to energetic costs is indicated with the symbol “ $X_p - Y_p$  %”.

### **3.1.9 Stage 9: Local numerical decline by specific proportion of birds not accommodated in remaining habitat / supported by lost habitat**

At this point of the assessment process the number of individuals likely to suffer from displacement impacts has been identified. Any ecological consequences of displacement to these individuals must be defined. Independently of whether the displacement impact results in reduced reproduction or adult mortality, only a proportion of the birds attempting to find accommodation in the fully occupied remaining habitat, and therefore bearing the energetic costs of displacement, can be expected to breed successfully and/or survive.

As indicated in Section 3.1.7, for simplicity, and because conclusive evidence is missing, one may have to assume that, at least numerically, the displacement consequences are only exerted on those birds that are not accommodated in the remaining habitat once carrying capacity has been reached. Those birds suffering from energetic consequences (e.g. reduced survival) do not necessarily need to represent the same individuals displaced from the OWF footprint and buffer. It is likely that the birds which do suffer from energetic consequences are adults in poor body condition (e.g. due to parasites etc) and/or less experienced juvenile and immature birds drawn from the enlarged population of birds (i.e. the incoming birds and the receiving population). Exactly which individuals will depend on the relative fitness of the individuals among the incoming birds and the receiving population. In other words, the energetic costs of accommodating additional birds, which push a local population over carrying capacity, would be spread across the entire local, enlarged population. This could potentially mean that a larger number of individual birds could be affected by displacement impacts than the actual number displaced under certain environmental and population conditions. Such an assessment would represent an alternative and even more precautionary way of assessing displacement impacts and it would be even more challenging to address than the current worst case assumption that the upper limit is that all birds that are displaced suffer mortality (the 100% mortality column in the matrix approach).

According to the 'Displacement as Habitat Loss' approach, the local population would then be reduced by that proportion of individuals and achieve a new population equilibrium at some point in the future. In terms of quantifying this in the assessment process, and

depending on the degree of precaution one aims to integrate into the assessment, the local population would need to be reduced by:

- a) the number of displaced individuals that cannot be accommodated within the remaining habitat;
- b) the number of individuals that cannot be accommodated multiplied by a percentage mortality level that is considered appropriate; or
- c) the number of (juvenile) individuals that would not be recruited into the local population if the number of individuals that were accommodated, but suffered an energetic cost, were less productive in the breeding season.

For breeding season assessments that figure would need to be apportioned between potentially impacted breeding colonies. For the non-breeding season assessments that figure would be compared to the biologically defined minimum population estimated for the respective BDMPs spatial area.

Based on modelling or theoretical considerations, as reported in the evidence summaries 4.9 and 4.10, the ecological consequences of displacement are likely to be reduced productivity and adult mortality. There are indications that displacement is more likely to result in reduced productivity during the breeding season and increased adult mortality during the non-breeding season (Langton 2013, Furness 2013). Depending on the apportionment of the impacts into reduced reproduction and mortality, the time needed to achieve new population equilibrium may differ strongly.

The review of the evidence (Section 4) has not identified empirical evidence on the likely percentage mortality or reduced reproductive output caused to seabirds that are displaced and are not accommodated within remaining habitat. The most recent study modelling the demographic impacts for seabirds at Scottish SPAs of consented OWFs (Searle *et al* 2014), estimated cumulative worst case decreases in colony-specific adult mortality of -1.97% for kittiwake, -3.32% for puffin, and -0.82% for razorbill, though for most species / SPA combination decreases of less than 0.5% were predicted. The estimated adult mortality impacts for a single OWFs were smaller and the only predicted figures above 0.5% were for kittiwake (-1.04%) and puffin (-1.44%). In terms of reduced productivity, the worst case cumulative estimates predicted a decline of -2.14% for kittiwakes and -4.87% for puffins at specific colonies. Looking at estimated productivity impacts from single OWFs, predictions at specific colonies were up to -1.32% for kittiwakes and -1.73% for puffins. Decreases for guillemot, razorbill and gannet were comparably low (all below -0.5%) or unreliable. However, Searle *et al* (2014) state that the uncertainty in the magnitude of the OWF effect is likely to be large and that their results should be interpreted with considerable caution. The report does state the assumption that 1% adult mortality would approximate to a 5% reduction in productivity. This relationship though is not fully supported by the reported findings and this is attributed to stochastic noise in the model (Searle *et al* 2014). If this statement is true, such a conversion factor between adult mortality and reproductive output could be used to estimate displacement impacts that account for both potential ecological consequences.

Among waders, the annual survival rates of individually marked redshank displaced by the flooding of Cardiff Bay into the abutting and much larger Severn Estuary declined from 85% in the 2 years pre-barrage closure to 78% in the 3 following years because of a significant decline in winter survival ( $P = 0.0006$ ) (Burton *et al* 2006). There was no detectable decline in the survival rates of "control" redshank in an adjacent area of the Severn Estuary. This study provides conclusive empirical evidence that habitat loss can impact individual fitness in a bird population. Adult redshank displaced from Cardiff Bay experienced poor body

condition and a 44% increase in mortality rate. Without an increase in the recruitment of first-winter birds, such a change is likely to substantially reduce local population size ([Burton \*et al\* 2006](#)). Although this study is not directly transferable to seabirds, it indicates that displacement by habitat loss can lead to a drop in population size and as such provides support for the 'Displacement as Habitat Loss' hypothesis. It is able to indicate the direction of change – habitat loss results in reduced population size – but not to provide a numeric value transferable to seabirds.

There are other factors that may affect when a new population equilibrium will be achieved. Immigration and emigration between colonies, for example, clearly occurs in seabird populations and are important for the genetic exchange within meta-populations. Accordingly, it is likely that birds could attempt to immigrate into local populations that are being affected by displacement impacts. This could potentially trigger a situation where continued year-on-year or occasional displacement effects would impact those individuals attempting to immigrate into the population, and/or the receiving population, over the entire operational phase of an OWF. The loss of breeding seabirds from a colony due to OWF displacement impacts could attract adults in search of breeding sites to the colony and these birds could conceivably fail to breed successfully due to the habitat loss resulting from the development. This ecological trap could lead to the colony becoming a “sink” ([Watkinson & Sutherland 1995](#)) for seabirds and lead to year-on-year population effects that spill out beyond just the local neighbouring colonies. Were this scenario to be realised it could contradict the 'Displacement as Habitat Loss' hypothesis, in that the population might never reach equilibrium as birds continued to be attracted in (and impacted negatively) year-on-year.

The inter-annual variability in seabird prey resources discussed previously may be substantial ([Furness 2013](#)) and lead to a situation where displacement impacts only affect individuals during years of poor food availability. Accordingly, in those years an impact may occur, while in years of average or high prey abundance the population may be able to hold or retain its original population.

Uncertainty surrounds whether it is realistic to assume that after habitat loss, a population will equilibrate to its previous level minus the number of individuals displaced. While the Cardiff Bay study ([Burton \*et al\* 2006](#)) may provide some support for this hypothesis, other evidence does not fully support this conclusion and some ecological processes could impede such a process. If a population locally is below carrying capacity, habitat loss may not lead to a population decline and therefore it could be easy to overestimate the impact of any development on local populations and thus be overly precautionary. A contrary example could be where nest space availability at a breeding colony (caused by displacement effects from a development site) attracts individuals in search of nest sites and acts as an 'ecological sink' for immigrating seabirds. As such, the population would continue to decline in numbers due to the surrounding habitat being already at or over carrying capacity. Whereas it is not necessarily more appropriate to assess displacement impact as a year-on-year effect, the empirical case to support a 'Displacement as Habitat Loss' approach is not yet made.

### **3.1.10 Stage 10: Adjusted local numerical equilibrium of $X_E - Y_E$ individuals after A – B years**

The final stage of the assessment framework attempts to define the time period over which the local populations would achieve the new population equilibrium. As indicated by the evidence summaries 4.9 and 4.10, the length of this period will strongly depend on the apportionment of impacts into reduced reproduction and mortality, and current evidence does not allow for any recommendation on this topic. If required, a generic or arbitrary rule

could be defined to allow for this assessment to take part. Such a rule could be based on the generation time of each relevant species as that parameter is a very good descriptor of many demographic parameters such as survival (obviously) and reproductive rate.

### **3.2 Potential alternative assessment frameworks**

There were several alternative potential assessment frameworks that were considered when initially scoping this study. The assessment framework discussed extensively above and illustrated in Figure 1 focuses mainly on the displaced individuals that make up the local population. Alternative assessment frameworks were conceived that focused on the percentage habitat lost and on the prey biomass available to the displaced population.

An alternative more habitat-centred assessment framework that was considered as part of the scoping of this project is illustrated in Figure 2.

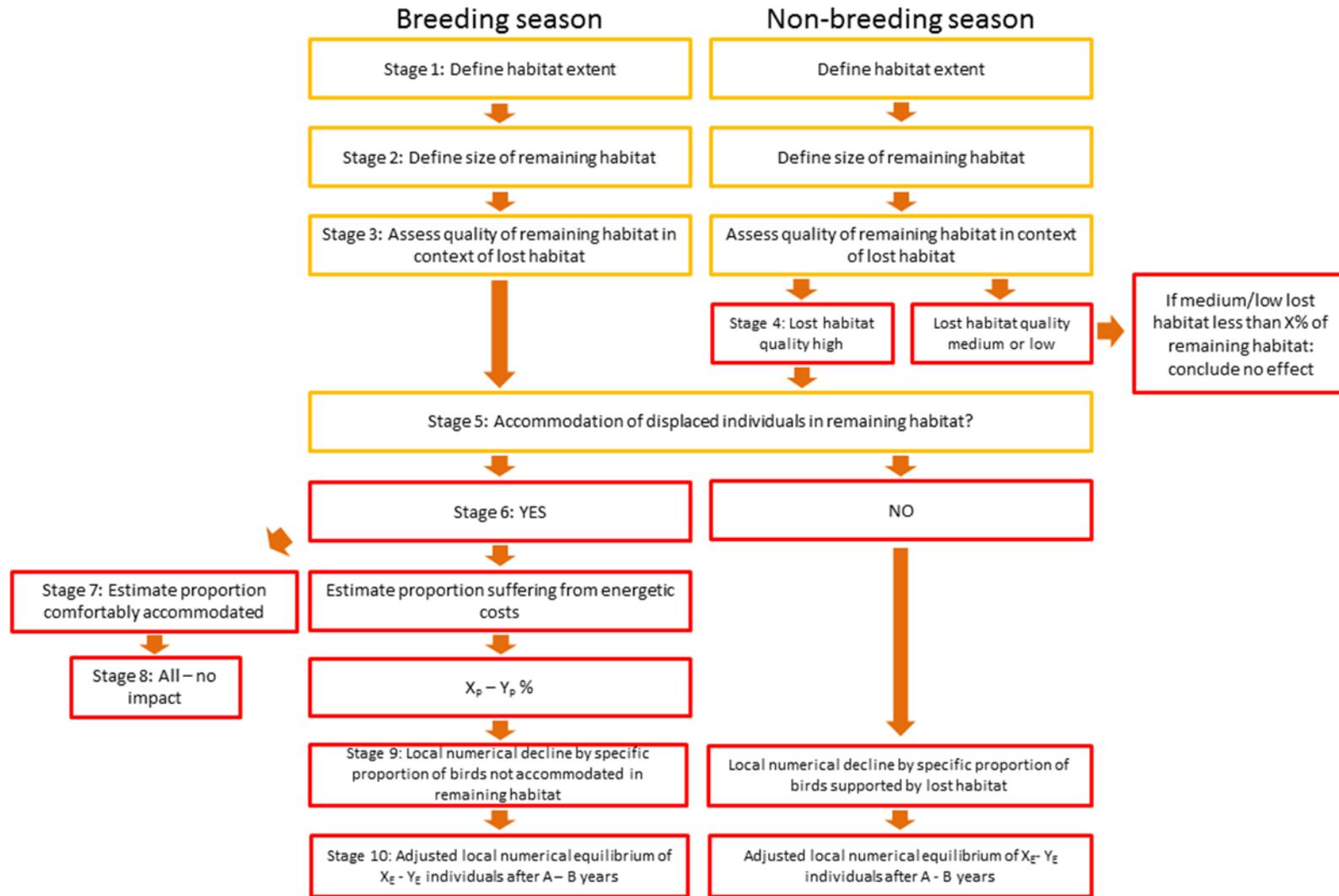


Figure 2. Alternative habitat-centred assessment framework.

This habitat-centred assessment framework was considered as a potential alternative for non-breeding season assessments. Stage 4 of the assessment (see Figure 2) indicates a potential stop point within the framework when there is a 'no effect' conclusion. This would be in the circumstance where the quality of the lost habitat can be identified to be of medium or low quality and the lost habitat accounts for, as yet undefined, low proportion of the remaining habitat. The extent of the available habitat would probably be defined by the respective species-specific BDMPS spatial area (Furness 2015). This potential stop point may be particularly relevant to those seabird species that are less habitat constrained, are far ranging and, potentially, those that spend most of their time in flight. It is less likely to be relevant to those species that are much more restricted in their habitat choice, for example, foraging in a clearly defined water depth.

As species are less restricted in their choice of foraging habitat during the non-breeding season, Stage 4 is only suggested for this season. During the non-breeding season seabirds are not, or are less, bound to the location of their breeding colonies. They are not restricted to finding prey within a foraging range that has its extent defined by a trade-off between the energy spent on foraging and provisioning chicks and that spent on themselves. In the non-breeding season seabirds are arguably less sensitive to displacement impacts as any relocation to an alternative habitat does not require subsequent regular commutes, of potentially greater distance, back to the location of the breeding colony.

In the process of scoping alternative assessment frameworks, it has been identified that the key challenge of an evidence-based understanding of the ecological processes governing the strength of any potential displacement effect, is an integral part of all the assessment frameworks trialled. For example, the habitat-centred framework (illustrated in Figure 2) from Stage 5 onwards follows the same assessment stages as the core framework extensively described above and illustrated in Figure 1. They both require a consideration of those ecological processes for which there are considerable evidence gaps (e.g. carrying capacity, density dependence, population equilibrium etc) in order to define the scale of the likely biological consequences of displacement at an individual and population level.

## 4 Assessment of assumptions underlying the 'Displacement as Habitat Loss' approach

The assumptions assessed below have been identified as described in sections 1 and 2. The assumptions 1 to 14 relate to specific stages of the proposed and assessed displacement assessment framework (Section 3) for the implementation of the 'Displacement as Habitat Loss' approach, while assumptions 15 to 18 represent additional key assumptions underlying the approach itself, while not being explicitly related to any particular stage of the assessment framework.

### 4.1 Assumption 1

*It is assumed that there is sufficient evidence on appropriate species-specific displacement buffers, i.e. available post-construction monitoring results indicate to what distance out from the footprint of OWFs the displacement of seabird species can be detected (assumption relates to Stage 1 of the assessment framework).*

#### 4.1.1 Evidence summary

Few OWFs have been in operation long enough to assess displacement effects in great detail (Furness 2013)<sup>IP1</sup> and accordingly there is only sparse evidence for species-specific displacement buffer zones for many species. The best evidence on displacement is currently available for non-breeding seabirds from studies outside the UK (Searle *et al* 2014)<sup>EP2</sup>, and results indicate varying levels of displacement behaviour among locations and species (Petersen *et al* 2006<sup>EP</sup>, 2011<sup>EP</sup>; Fox *et al* 2006b<sup>EP</sup>; Leopold *et al* 2011<sup>IP</sup>; Vanermen *et al* 2011<sup>EP</sup>, 2012<sup>EP</sup>; Leonhard *et al* 2013<sup>EP</sup>). Furness *et al* (2013)<sup>PR3</sup> provide disturbance sensitivity scores for UK seabird species in relation of OWFs which could act as proxies for species-specific displacement buffers. For some key species, especially red-throated diver and seaducks, information on displacement buffers are available. A simulation model indicated that common scoter would completely avoid OWFs in Liverpool Bay to a buffer of 2 km (Kaiser *et al* 2005)<sup>EP</sup>. Long-tailed ducks show strong evidence for displacement from the Nysted OWF post-construction and up to 2km from the OWF (Petersen *et al* 2006)<sup>EP</sup>. Walls *et al* (2012)<sup>IP</sup> indicated that displacement could occur up to 800m from Robin Riggs OWF for common scoter and red-throated diver. However, more data would be needed to confirm whether there is a significant difference, pre to post-construction. For red-throated diver Petersen *et al* (2006)<sup>EP</sup> report a marked avoidance of the OWF area and 2km and 4km buffer zones around it with D (displacement) values of 1.00 (100%) (OWF footprint), -0.77 (2 km buffer) and -0.50 (4 km buffer). Dierschke *et al* (2012)<sup>PR</sup> consider a total avoidance of OWFs and a 2km buffer to be appropriate based on a comprehensive review of studies on red-throated diver displacement. Displacement up to 1-3km is reported from the OWF Alpha Ventus (BSH 2012)<sup>EP</sup>, complete displacement up to at least 1 km is reported from Gunfleet Sands OWF (Baker 2011)<sup>IP</sup>, strong displacement effects up to at least 3km from the OWF Kentish Flats (Percival 2009, 2010, 2014)<sup>IP</sup> and strong displacement up to at least 2.5 km for North Hoyle OWF (May 2008)<sup>IP</sup> are reported for red-throated diver. Leonhard *et al* (2013)<sup>EP</sup> used a 500m exclusion zone for red-throated divers around OWFs when modelling cumulative displacement effects for the species in the Baltic Sea. Vanermen *et al* (2012)<sup>EP</sup> considered a 3km buffer as 'impacted area' by the presence of the OWFs in their study.

<sup>1</sup> IP = Internally quality assured publication

<sup>2</sup> EP = Externally quality assured publication

<sup>3</sup> PR = Peer-reviewed evidence

### 4.1.2 Conclusion

There is sufficient evidence on species-specific buffers for a few key species, especially red-throated diver, and first long-term data series become available. For most other seabird species no sufficient evidence is available and generic displacement buffers would need to be used within assessments or disturbance sensitivity indexes could be used as a proxy.

## 4.2 Assumption 2

*It is assumed that survey methodology and assessment methods currently used to estimate and monitor seabird numbers in and around OWF developments are suitable to report densities at a precision that allows the detection of significant changes in seabird numbers as a consequence of the existence and operation of OWFs (Stage 2).*

### 4.2.1 Evidence summary

There are several survey methodologies available for detecting seabird densities within OWF development areas, especially aerial surveys (Maclean *et al* 2013<sup>PR</sup>, Camphuysen *et al* 2002<sup>EP</sup>). It is standard EIA practice to estimate mean seabird densities in the footprint of the OWF and a buffer. The introduction of digital aerial surveys minimizes disturbance of seabird and is likely to produce more reliable population estimates that may also be more precise as unlike visual aerial and boat surveys digital aerial surveys do not require distance sampling to account for a declining probability of detection with distance from the survey platform (Maclean *et al* 2013)<sup>PR</sup>. Christel *et al* (2013)<sup>PR</sup> suggest Taylor's power law and mixed effect models can be used to assess spatio-temporal variability of seabird density and this was applied to aerial surveys of the Ebro Delta, Spain where a distinction between feeding and transition areas could be made. Stewart *et al* (2007)<sup>PR</sup> found that there is considerable variation with windfarm impacts but a lack of evidence owing to poor methodologies and lack of long-term data. Green (2014)<sup>EP</sup> critiqued the use of OWF impact assessment methods, stating the lack of empirical data and well-conducted field work as the leading causes. Maclean *et al* (2013)<sup>PR</sup> suggest that Round 2 OWF aerial visual surveys did not adequately detect changes in seabird numbers even when declines were in excess of 50%. Increasing the duration, frequency and extent of surveys would increase slightly the probability of detecting changes in seabird numbers, even major increases in survey effort would rarely lead to real changes being detected and there would still be the risk that false positives would be detected. The primary reason suggested for the low probability of detecting change is, as stated by Searle *et al* (2014), the strong natural fluctuation of seabird numbers over time and space as seabirds change their foraging locations depending on the abundance and location of prey (Searle *et al* 2014)<sup>EP</sup>. Species surveys over large areas are not necessarily cost-effective but spatially explicit species distribution models (SDMs) can be used to extrapolate from incomplete datasets (Braunisch and Suchant 2010)<sup>PR</sup>. Maclean *et al* (2013)<sup>PR</sup> highlights the need to explain some of the variation in bird numbers, e.g. by incorporating hydrodynamic data, as reducing the standard deviation of the modelled count data could substantially increase the power of detecting changes in seabird numbers.

### 4.2.2 Conclusion

While survey methods allow the assessment of seabird densities and derivation of population estimates, it can be challenging to detect local declines at a larger scale, and especially to differentiate local declines from natural background fluctuations.

### 4.3 Assumption 3

*It is assumed that there is sufficient evidence on the species-specific and / or species group-specific percentage / level of displacement of seabirds from OWF footprints and surrounding buffer zones and the variation in displacement rates with increasing distance from a central point (Stage 3).*

#### 4.3.1 Evidence summary

There is a considerable degree of uncertainty regarding the extent to which seabirds may be displaced (Furness 2013)<sup>IP</sup>. Current figures on percentage displacement from OWFs do exist but seem to be project specific, which may relate to differences in habitat quality (Furness 2013)<sup>IP</sup> or due to the fact that high variability in the data does not allow for statistically significant conclusions (Petersen *et al* 2006<sup>EP</sup>, Maclean *et al* 2013<sup>PR</sup>). Red-throated divers appear to show strong and persistent displacement responses to OWFs (Leonhard *et al* 2013)<sup>EP</sup>. Detailed information on density changes, pre to post-construction, for red-throated diver and decreasing intensity of displacement across the buffer zone are reported by Percival (2014)<sup>IP</sup> for the Kentish Flats OWF (-94% OWF footprint, -77% 0-0.5km buffer, -69% 0.5-1km buffer, -53% 1-2km buffer, -56% 2-3km buffer). Significant displacement was also found in some surveys at Dutch OWFs (Leopold *et al* 2011)<sup>IP</sup>. For long-tailed ducks significant displacement responses are reported (Petersen *et al* 2011)<sup>EP</sup>, with some complete avoidance suggested (Petersen *et al* 2006)<sup>EP</sup>. Common scoter showed initial strong displacement from a Danish OWF (Petersen *et al* 2006)<sup>EP</sup>, but occurred in high abundance several years later, likely as a result of an invasive shellfish colonizing the OWF area that is an important prey resource for the species (Leonhard *et al* 2013)<sup>EP</sup>. A few surveys at Dutch OWFs indicate a significant displacement of about 50% of common guillemots (2 out of 11 surveys) and razorbill (1 out of 6 surveys). A 30% displacement of guillemot and razorbill is reported for Robin Rigg OWF 1 year post-construction (Walls *et al* 2012)<sup>IP</sup>. Vanermen *et al* (2012)<sup>EP</sup> reported a significant decline of common guillemot at Blighbank OWF. Gannet showed significant displacement in some studies (Leopold *et al* 2011, Vanermen *et al* 2012), while at other sites no indications of displacement were found. In several sites gull numbers either did not change (e.g. Petersen *et al* 2006<sup>EP</sup>, Fox *et al* 2006<sup>EP</sup>, Leopold *et al* 2011<sup>IP</sup>) or increased (Vanermen *et al* 2012)<sup>EP</sup>. Some seabirds may be attracted to some OWFs, including cormorant (Leopold *et al* 2011)<sup>IP</sup>, little gull (Petersen *et al* 2006)<sup>EP</sup> and red-breasted merganser (Petersen *et al* 2006)<sup>EP</sup>.

#### 4.3.2 Conclusion

Based on the currently available evidence, that suggests strong site-specific variations in percentage displacement, it would appear best to use generic displacement categories for species, such as high, medium or low sensitivity. While a decreasing intensity of displacement with distance from the OWF footprint appears likely, and for red-throated diver an examples was available, again a generic or matrix approach would be recommended as no species-specific information is available. However it may be worth exploring the value of using an average value across all sites for species groups such as gulls, divers, auks, etc.

### 4.4 Assumption 4

*It is assumed that there is sufficient evidence on the extent (size) of specific seasonal foraging habitats used by local populations, to be able to define the extent of the remaining habitat (compared to the original habitat extent before any displacement occurred) (Stage 4).*

#### 4.4.1 Evidence summary

Seabird habitats vary according to season (breeding and non-breeding) and across species. As habitats are subject to fluctuations in prey abundance foraging locations within potential habitat change (Searle *et al* 2014)<sup>EP</sup>. Thaxter *et al* (2012a)<sup>PR</sup> suggest that breeding season foraging habitat can be identified by direct tracking of individual breeding seabirds, at-sea surveys of seabird distribution, and data for other environmental variables that should be modelled together to suggest likely area of usage associated with particular colonies. While GPS loggers provide a method for identifying the foraging sites of specific individuals, due to inter-individual variation in foraging sites tracking studies alone would require large sample sizes to be able to precisely define the size of foraging habitat of specific colonies (Langton 2013)<sup>EP</sup>. Representative generic foraging ranges can provide an extra layer of information to identify the extent of breeding season foraging habitat of seabird colonies, and will often be the best evidence available on typical or likely distances travelled by different species from a colony to a foraging area (Thaxter *et al* 2012a)<sup>PR</sup>. Accordingly foraging ranges may act as proxy for foraging habitat extent. Unfortunately we are not certain whether birds can regularly travel as far as indicated by the mean maximum or maximum foraging range and still successfully reproduce (Langton 2013)<sup>EP</sup>. Baird (1991)<sup>PR</sup> found that non-breeders attending colonies consumed more pelagic prey than breeders which ate more inshore prey, indicating that there may be a limit to the travelling distance of at least some seabirds with young. Field studies suggest that seabirds may extend their foraging ranges when prey abundance is reduced, with the consequence that foraging trip duration increases and less time is spent with chicks at the colony (Harding *et al* 2007<sup>PR</sup>, Piatt *et al* 2007<sup>PR</sup>), leading to reduced breeding success (Chivers *et al* 2012)<sup>PR</sup>. Moreover, there is evidence that certain seabird species occupy colony-specific foraging habitat with minimal overlap between foraging habitats of neighbouring colonies, as demonstrated for gannets that forage in largely mutually exclusive areas (Wakefield *et al* 2013)<sup>PR</sup>. Such evidence will need consideration when attempting to define colony-specific habitats based on generic foraging ranges. Langton (2013)<sup>EP</sup> developed a simulation model to predict the maximum range adults can forage while still achieving a specified fledging success, to allow the identification of areas at sea where successful individuals from specific colonies could be foraging. Many seabirds such as gulls take advantage of fish discards from vessels. Two different modelling methods (MARS and GLM) were used to demonstrate that fishing activity can be a major driver for gull distribution (Cama *et al* 2012). Species Distribution Models (SDMs) determined optimal feeding habitat of Kittiwakes at two colonies to specific locations. However, at the colony of Rathlin Island variation in conditions such as ocean currents caused a mismatch between predicted and observed locations (Chivers *et al* 2013)<sup>PR</sup>. For the non-breeding season, when central place foraging is no longer required, there is considerably less evidence on how to derive the extent of foraging habitat for seabird species. Species such as adult guillemots in the North Sea are known to show dispersion rather than migration movements and remain in relative proximity to their colonies all year around (Mendel *et al* 2008)<sup>EP</sup>. For such species median adult ring recovery distance to colony can give an indication of non-breeding habitat extent, which for UK adult guillemots is 357km during winter (Wernham *et al* 2002)<sup>EP</sup>. The most comprehensive information on UK non-breeding season seabird populations in specific marine regions is provided by Furness (2015)<sup>EP</sup> who define spatially distinct biologically defined minimum population scales (BDMPS) for the main UK seabird species.

#### 4.4.2 Conclusion

When combining information gathered via different methodologies, estimating the extent of foraging habitat for seabirds during the breeding season is possible. During the non-breeding season approximate marine areas / regions can presently be identified where a proportion of biogeographic populations occur, with the possible exception of some species

that show restricted dispersive movements during the non-breeding season. With the rapid development and application of geolocators and other tags the lack of detailed information during the non-breeding season is being remedied.

## 4.5 Assumption 5

*It is assumed that there is sufficient evidence to define and quantify / rank the quality of marine areas as foraging habitat for seabirds, despite strong inter-annual variability of prey resources (Stage 5).*

### 4.5.1 Evidence summary

Furness (2013)<sup>IP</sup> confirms that assessing relative habitat quality is key when trying to define whether displacement of seabirds might have population-level impacts; he notes that the displacement of non-breeding seabirds could affect survival rates if birds were to be displaced from high quality foraging habitat when their population was at carrying capacity. Birds tend to occupy habitat of high quality first and only make use of habitat of lower quality once optimal habitat is fully occupied (Fretwell & Lucas 1972<sup>PR</sup>, Newton 1998<sup>PR</sup>). This would indicate that seabird densities could act as a proxy for habitat quality, and that the variability in densities over time could indicate the habitat quality of specific areas despite strong inter-annual changes in prey abundance. Kober *et al* (2012)<sup>EP</sup> identified regularly occurring seabird hotspots based on ESAS data. Areas that held significantly higher numbers of birds than elsewhere tended to do so in at least 2 out of 3 years with sufficient data, and were assumed to be regularly occurring hotspots. Arcos *et al* (2012)<sup>PR</sup> identified foraging hotspots off the Iberian Coast for three Procellariiform species using methods incorporating field data and species distribution modelling (SDM). They concluded that stability assessment differed between species and long-term monitoring would be needed to confirm stability of Important Bird Areas (IBA). Nonetheless, quality of foraging habitat for seabirds is best indicated by high densities of prey species (Langton 2013<sup>EP</sup>, Furness 2013)<sup>IP</sup>. The composition of the seabed can indicate the suitability of an area for certain prey species, with high proportions of coarse sands, for example, indicating suitable sandeel habitat (Furness 2013)<sup>IP</sup>, a key prey species for many seabirds. According to Wright *et al* (2000)<sup>PR</sup> sediment with a silt content <4% and a gravel content <16% can be considered suitable sandeel habitat. Another relevant factor for seabird habitat quality is the propensity of the water column to stratify (Langton 2013)<sup>EP</sup>. Data on bathymetry and the speed of the peak flow of a mean spring tide, needed to quantify the potential for tidal stratification in UK waters are available in the Renewables Atlas (ABPmer 2008)<sup>EP</sup> according to Langton (2013)<sup>EP</sup>.

### 4.5.2 Conclusion

In summary, the quality of foraging habitat for seabirds can be estimated at least broadly, often through the use of proxies such as the habitat features required by seabird prey species which often do not require long term measurement. As reliable repeated data on fish distribution are rare and the abundance of fish is subject to strong inter-annual variation direct measures of foraging habitat are rarer. The relatively few long term data sets that exist make it possible to identify areas that regularly support high or low densities of seabirds, even though there can be much inter-annual variation in their numbers.

## 4.6 Assumption 6

*It is assumed that there is sufficient evidence to assess whether marine habitats are at carrying capacity for specific species and/or species groups, i.e. whether delimited habitat patches (e.g. within a foraging range of a species) are able to support additional individuals that have been displaced or not (Stage 5).*

#### 4.6.1 Evidence summary

Carrying capacity will be reached when a large enough proportion of a seabird population, in addition to fully occupying high quality habitats, occurs in poor quality habitat and so net population growth falls to zero (Newton 1998)<sup>PR</sup>. Using an approach that explored population growth in terms of density dependence a Hawaiian atoll was estimated to support a maximum of 600 pairs of breeding great frigatebird and other numbers of other seabirds (Hatfield *et al* 2012)<sup>PR</sup>. Papers that make use of the concept of carrying capacity are not uncommon. For example, regional variation in sandeel has been predicted to have a greater effect on the carrying capacity of Kittiwake than local depletion (Frederiksen *et al* 2005)<sup>PR</sup>, and it has been postulated that the carrying capacity of northern regions will need to support Guillemot, Kittiwake and Puffin populations if distribution and abundance of copepod shifts northwards in response to climate change (Frederiksen *et al* 2013)<sup>PR</sup>. However it must be borne in mind that earlier attempts at estimating carrying capacity for grey plovers (Moser 1988) and black-tailed godwits (unpublished) were empirically demonstrated to be incorrect when carrying capacity predictions were much exceeded. The difficulty of estimating carrying capacity can be demonstrated when one of the most important prey of seabirds, sandeels, between 1983-2011 has fluctuated by more than two orders of magnitude between years (ICES 2010<sup>EP</sup>, Furness 2013<sup>IP</sup>) without any noticeable change in North Sea populations of Guillemots and Razorbills. This could be taken to mean that their populations were not at carrying capacity (Furness 2013<sup>IP</sup>), were at a carrying capacity that was related to factors other than prey such as nest sites (Kokko *et al* 2004)<sup>PR</sup>, or that these long-lived seabirds have evolved to be at a carrying capacity that is much below the available prey in peak years to lessen the risk of regular population crashes in the relatively frequently occurring lower prey years. Modelling attempts can over-estimate the effect of habitat loss caused by displacement on numbers of seabirds if carrying capacity hasn't been reached (Stillman and Goss-Custard 2010)<sup>PR</sup>.

#### 4.6.2 Conclusion

In summary, it has proved possible to estimate the carrying capacity of some populations during the breeding season, helped by long-term datasets recording species' numbers and prey abundance. It is however very difficult to do so and may not be reliable due to fluctuations in prey and habitat quality in time and space, especially against a background of rapid climate change.

### 4.7 Assumption 7

*It is assumed there is sufficient evidence or theoretical knowledge to understand how density dependent processes affect individuals (e.g. increased competition for food leads to a drop in one or more vital rates like growth or survival) as a consequence of displaced birds being pushed into remaining habitat (Stage 5).*

#### 4.7.1 Evidence summary

There has been extensive modelling of the effects of density dependence based on long-term empirical data collection especially relating to waders and some other waterbirds, with Stillman and Goss-Custard (2010)<sup>PR</sup> providing a recent review of the subject. Ian Newton has carried out a broader review of the importance of density dependence for birds (Newton 1998). The following papers on the subject are heavily cited. Smaller populations tend to increase when they exclusively use high-quality habitat, known as the 'buffer' effect (Kokko *et al* 2014)<sup>PR</sup>. Using data gathered between 1949-2007 winter conditions and density dependence explained 29.7% of the variance in Velvet Scoter population growth (Hartman *et*

a/2013)<sup>PR</sup>. In Shags nest site quality declines with density and seasonal occupancy and some individuals prevent others occupying good sites (Velando and Freire 2003)<sup>PR</sup>. However, colonial breeding in seabirds can benefit individuals through enhanced protection from predators and transferring information about feeding sites (Ashbrook *et al* 2008)<sup>PR</sup>. Smaller colonies of Hutton's Shearwater are more vulnerable to predation by mink and exhibit lower breeding success (Cuthbert 2002)<sup>PR</sup>. In contrast, a trade-off between protection from predators and heightened aggression towards unattended chicks has been observed in Guillemots (Ashbrook *et al* 2010)<sup>PR</sup>. However erroneous density dependence test results can occur if estimates rather than exact counts are used (Frackleton *et al* 2006)<sup>PR</sup>.

#### 4.7.2 Conclusion

In summary, there is clear evidence suggesting that density dependence affects seabird populations during the breeding season, but less evidence during the non-breeding season. The latter is not surprising due to the greater difficulty of such studies when the birds are often more widely dispersed. There is some occasionally conflicting evidence that suggests that density dependence can have a positive effect on populations of colonial seabirds. Although there is not much evidence that directly relates displacement to density dependent effects, the conceptual understanding of how density dependence acts on birds should be sufficient to at least quantitatively estimate how it affects displacement.

### 4.8 Assumption 8

*It is assumed that there is sufficient evidence or theoretical knowledge to estimate the number or proportion of birds that could be accommodated in any remaining habitat (accommodation capacity) before any ecological consequences due to density dependent competition etc would kick in (Stage 7).*

#### 4.8.1 Evidence summary

The proportion of birds which could be accommodated in remaining habitat following displacement before density dependence has an effect would depend on various factors including its carrying capacity, the quality of habitat in the displaced region and the extent to which birds have been displaced. The scale of habitat loss and the availability and quality of remaining habitats to accommodate displaced birds will inevitably determine whether there is adverse effect to the population (Powlesland 2009)<sup>EP</sup>. Studies of geese suggest that displacement distances from on-shore wind farms are site-specific and are greatest, if good alternative (remaining) habitat is available (Percival 2005)<sup>PR</sup>. However, quantifying whether adverse effects have occurred as a result of displacement has proven difficult (Green 2014)<sup>EP</sup>. Mitigation measures could help accommodate displaced birds (Bright *et al* 2009)<sup>EP</sup>; Drewitt and Langston 2009)<sup>PR</sup>. In Canada, nationally wind farms led to 3635 ha of habitat change which could represent a loss of 5700 nests across all species (Zimmerling *et al* 2013)<sup>PR</sup>. Whereas relating displacement of an individual from its optimum feeding area to changes in its survival or reproductive success is difficult, modelling the effects and cumulative impacts can be done at the population level (Fox *et al* 2006)<sup>PR</sup>. Long-established density-dependent Leslie-matrix models have been used to model the expected rates of population size over the lifetime of projects (Green 2014)<sup>EP</sup>. The concept of source and sink habitats is interesting in the context of this assumption. For example if a local habitat, known to be a source habitat (natality > mortality) shifts toward a 'pseudo-sink' habitat, where, as a consequence of habitat loss due to displacement, immigration of displaced individuals into the remaining source habitat increases the local population size above carrying capacity and thus creates a demographic deficit (Dias 1996), this shift towards a demographic deficit would indicate that the remaining habitat cannot accommodate further displaced individuals

without negative effects on fitness. Nonetheless, the identification of source and sink habitats is challenging according to Dias (1996)<sup>PR</sup> because long-term studies are required to identify them adequately, as point measurements of demographic parameters are often misleading.

#### **4.8.2 Conclusion**

Different assumptions for which limited, or only general, evidence exists (local carrying capacity, habitat quality, density dependence) accumulate when attempting to estimate what proportion of displaced birds could be accommodated in any remaining habitat before ecological consequences occur. The concept of source and sink habitats may be useful in this context, but there is little information and no concrete examples on how to estimate precisely how birds would accommodate in remaining habitat in response to displacement.

### **4.9 Assumption 9**

*It is assumed that there is sufficient evidence or theoretical knowledge to predict if there will be an immediate impact or not (Stage 8).*

#### **4.9.1 Evidence summary**

Many of the studies investigating whether displacement has had effects on populations are related to onshore wind farms (Madders and Whitfield 2006<sup>PR</sup>; Pearce-Higgins *et al* 2009<sup>PR</sup>; Dahl *et al* 2012<sup>PR</sup>). Few studies on displacement caused by disturbance at windfarm sites are conclusive due to lack of before and after control impact (BACI) assessments (Drewitt *et al* 2006)<sup>PR</sup>. Habitat for seabirds sensitive to displacement will be lost when an OWF becomes operational and some individuals avoid the footprint and buffer zone, an effect that may also occur during the construction phase. From this perspective there clearly is an immediate impact on sensitive species in terms of habitat loss. More difficult to answer is the question over which time period the ecological consequences of habitat loss / displacement will be realised. Impacts of displacement are likely to lead to reduced juvenile survival during the breeding season and reduced body mass of adults potentially resulting in lower survival rates during the winter or not attempting to breed in the following breeding season (Furness 2013<sup>IP</sup>, Searle *et al* 2014<sup>EP</sup>, Langton 2013<sup>EP</sup>). Due to deferred breeding of many seabird species (Furness 2015)<sup>EP</sup>, but population census being based on counts of breeding pairs (Furness 2015)<sup>EP</sup>, it is likely that reduced reproduction may need several years before any reduction in numbers of breeding adults monitored would be recognized. However, monitoring the productivity at seabird colonies could overcome this problem. Adult mortality, in long-lived seabirds (Langton 2013)<sup>EP</sup>, is likely to impact local populations in a more direct way. Such mortality could occur, as breeding adults are likely to be more restricted in their choice of foraging habitat than non-breeders who do not have to remain in the vicinity of colonies. The mortality of adult birds is likely to have a population effect not buffered by several generations of immature birds.

#### **4.9.2 Conclusion**

There is little information to underpin this assumption. While an immediate impact in terms habitat loss will occur, the quantification of the induced ecological consequences is likely to depend on the quality of the local habitat. Seabird ecology may indicate that local population level effects due to displacement may be more pronounced and better identifiable in the case of adult mortality rather than reduced reproduction.

## 4.10 Assumption 10

*It is assumed that there is sufficient evidence or theoretical knowledge to define whether the ecological consequences of displacement is more likely to result in mortality or reduced reproductive outputs or in what proportion both impacts occur in parallel. Moreover, different life cycle stages may need to be considered (Stage 9).*

### 4.10.1 Evidence summary

Seasons play a huge role in seabird survival, where poor foraging conditions during the breeding season can cause adult mortality during non-breeding (Harding *et al* 2011)<sup>PR</sup>. For example little auks and Brünnich's guillemots show a sharp increase in energy expenditure between November and December caused by harsh weather conditions (Fort *et al* 2009)<sup>PR</sup>. Mortality of oystercatchers and redshank in winter, caused by severe weather was greater in first year birds, owing to poor feeding efficiency, small body size, and differences in metabolic rates (Davidson and Evans 1982)<sup>PR</sup>, similar relationships may occur in seabirds. For kittiwakes during the breeding season, adult survival varies according to age and gender. There was evidence of senescence in birds older than 12 years old, with survival rates dropping and females (82%) having a higher survival than males (78%) (Aebischer and Coulson 1990)<sup>PR</sup>. Seabird numbers are limited by food-related winter mortality and reproductive success, but there is evidence for buffering against food shortages and this varies among species (Furness 1996)<sup>PR</sup>. One report claims that displacement is unlikely to reduce survival of breeding seabirds, but reduced reproduction represents a more likely effect (Furness 2013)<sup>IP</sup>, because they buffer their survival by abandoning breeding in unfavourable conditions (Cairns 1987)<sup>PR</sup>. In this scenario, it could take decades for any population level effects to become apparent (Furness 2013)<sup>IP</sup>. For example, one study found that 8.9% of adult Great Skuas defer breeding as a buffer against environmental stress (Klomp and Furness 1992)<sup>PR</sup>. Generally long-lived species, like seabirds are thought to safeguard their own survival to allow for another breeding attempt in another year (Langton 2013)<sup>EP</sup>. Recent studies modelling the effects of displacing foraging guillemots during the breeding season concluded that displacement could result in changes to the species time/energy budgets with potential consequences for breeding performance and/or adult survival (MacDonald *et al* 2012)<sup>EP</sup>, and indicate potential for a reduction of adult provisioning rates resulting in reduced chick growth, chick survival and consequently reproductive success (Langton 2013)<sup>EP</sup>.

Nonetheless, breeding adults affected by habitat loss may reproduce at high individual costs and could suffer from diminishing body conditions at the end of the breeding season (Langton 2013)<sup>EP</sup>. Such fitness implications, carried into the non-breeding season, could well have knock-on effects for overwinter survival (Erikstad *et al* 2009)<sup>PR</sup>, Harding *et al* 2011)<sup>PR</sup>) and while less restricted in their choice of foraging habitat, seabirds cannot buffer their own survival by abandoning certain behaviour (e.g. breeding) as during the breeding season. At the end of the non-breeding season again there could be carry-over effects, when adults in more condition may not attempt to breed (Furness 2013)<sup>IP</sup>. For a wader species there is evidence for prolonged reduction in winter survival as a consequence of habitat loss (Burton *et al* 2006)<sup>PR</sup>. Following the impoundment of Cardiff Bay, some 300 redshanks were displaced from their wintering habitat. The survival rate of displaced redshank was reduced from 0.846 to 0.778 in 3 following years because of a significant decline in winter survival (Burton *et al* 2006)<sup>PR</sup>. Searle *et al* (2014)<sup>EP</sup> simulated population consequences due to displacement for seabirds breeding at Scottish SPAs and found that adult mortality increases by more than 0.5% seemed possible for specific species (for kittiwake 1.08%, for puffin 1.44%) at specific colonies/SPAs under specific prey distribution scenarios. Modelling did not provide evidence of declines in adult mortality of more than 0.5% for gannets, razorbills or guillemot at the SPAs considered. The study also modelled

potential impacts on breeding success and results were qualitatively similar, but of lower magnitude, but also harder to disentangle from effects of stochastic noise. For a puffin population of one SPA under a specific prey distribution scenario (homogenous prey) a decrease in chick survival of more than 2.5% was predicted. No species / SPA combination showed an estimated decline of more than 5% in chick survival, assumed to be roughly equivalent to 1% decrease in adult survival in demographic terms.

#### **4.10.2 Conclusion**

In summary, seabirds are resilient to environmental change. There are indications that potential displacement could be more likely to result in reduced productivity during the breeding season and increased adult mortality during the non-breeding season. It should be considered that most statements on the matter are based more on ecological theory and models than empirical evidence.

### **4.11 Assumption 11**

*It is assumed there is sufficient evidence or theoretical knowledge that local populations affected by habitat change / loss adjust to a new equilibrium (Stage 9).*

#### **4.11.1 Evidence summary**

Equilibrium in a natural population is determined by environmental conditions and, if those environmental conditions are constantly varying, then that equilibrium would have to vary in time (Wolda 1990)<sup>PR</sup>. The time required for a breeding population to reach equilibrium is determined by the relative sizes of the new and old populations, the rate of mortality and the supply of recruits to that population (Kildaw *et al* 2005)<sup>PR</sup>. Long-term seabird monitoring in the UK (JNCC 2014b)<sup>EP</sup> shows a number of patterns over the last 30 years that do or do not suggest equilibrium in the monitored population such as fulmar that has been relatively stable (although recently declining), gannet that has been steadily increasing and kittiwake that has been declining rapidly. On South Georgia, white-chinned petrel experienced 1.9% annual population decrease caused by by-catch of all age classes. It was suggested the population would be at equilibrium without the by-catch (Martin *et al* 2009)<sup>PR</sup>. A study on Australasian gannets noted that as the birds aged no increase in breeding success was reported and egg laying did not occur earlier and concluded that this was evidence that an equilibrium population had been reached (Pyk *et al* 2013)<sup>PR</sup>. Equilibrium of the red-throated diver population breeding on lakes in Montana, USA, was deemed to have been reached when the number of occupied territories stabilised (Hammond *et al* 2012)<sup>PR</sup>.

#### **4.11.2 Conclusion**

There is some evidence that populations can adjust to a new equilibrium but the time that it will take has not been predicted with confidence and under varying environmental conditions equilibrium may never be reached. Adjustment to a new equilibrium has been difficult to prove in many cases.

### **4.12 Assumption 12**

*It is assumed that there is sufficient evidence that immigration of individuals into populations declining due to displacement effects / habitat change, will not lead to a situation where year on year individuals will suffer from displacement effects, potentially making the population assessed a “sink population” that does not produce enough offspring to maintain itself in coming years without immigrants from other populations (Stage 9).*

#### 4.12.1 Evidence summary

The population occupying a 'sink' habitat is one that cannot persist without immigration of individuals from other sources (Dias 1996)<sup>PR</sup>. The strong increase in a population at the limits of that species' distributional range implies that it could function as a population sink until conditions improve to allow range expansion (Kerbirious *et al* 2012)<sup>PR</sup>. Small kittiwake colonies are more attractive to recruits than large colonies and they grow faster but they cannot produce enough young over an extended initial period (the first 70 years) and therefore the growth is supported by immigration (Porter and Coulson 1987)<sup>PR</sup>. It is not always clear which vital rate e.g. survival, productivity etc is affected the most through density-dependence, and whether this dependence holds in open populations in which immigration are important (Tavecchia *et al* 2007)<sup>PR</sup>. It was found that the largest sensitivities of equilibrium population size in short-tailed shearwater to the strength of density dependence were immigration, emigration and breeder skipping rate (Yearsley *et al* 2003)<sup>PR</sup>. Kildaw *et al* (2005)<sup>PR</sup> report that the attractiveness of established colonies to prospectors suggest that potential benefits outweigh costs of immigrating into existing colonies among seabirds, indicating that immigration into a colony affected by displacement impacts may occur. Intense competition for limited nest sites (Kildaw *et al* 2005)<sup>PR</sup> could attract seabirds to colonies that decline due to displacement impacts and therefore face less competition for high quality nesting sites. Mortality at an onshore windfarm has been shown to create a 'sink' population: A study of a population of golden eagle in California identified that collision related mortalities reduced productivity to the point where it became a 'sink' population that was dependent on immigration (Powlesland 2009)<sup>EP</sup>, though clearly this terrestrial species is not directly comparable to seabirds facing entirely different ecological and behavioural pressures.

#### 4.12.2 Conclusion

There is some evidence to suggest that 'sink' populations do occur and that these are subject to continual immigration but the existence of a 'sink' population has not been directly linked to displacement effects e.g. caused by wind farms.

### 4.13 Assumption 13

*It is assumed that there is sufficient evidence or theoretical knowledge to realistically predict time scales over which local populations affected by habitat displacement will achieve a new, adapted population equilibrium. Once achieved displacement impact can be considered as "realised" and during later years no further impact will occur (Stage 10).*

#### 4.13.1 Evidence summary

The time required for a breeding population to reach equilibrium is determined by the relative sizes of the new and old populations, the rate of mortality and the supply of recruits to that population (Kildaw *et al* 2005)<sup>PR</sup>. Long-term seabird monitoring in the UK (JNCC 2014b)<sup>EP</sup> suggests that there have been periods when particular seabird populations have 'plateaued', such as fulmar that underwent a long period of expansion, then a period of stability before declining in recent years but, in contrast, lesser lack-backed gull that in the last 30 years has shown a similar pattern of expansion and then decline did not appear to have a period of stability (a plateau) in between. The colony of Australasian gannets at Pope's Eye, Australia was monitored for 19 years since the early stage in its development, after which no further increase in breeding success occurred, suggesting equilibrium had been reached (Pyk *et al* 2013)<sup>PR</sup>.

### 4.13.2 Conclusion

There is evidence from a number of studies to suggest that equilibrium has been reached in some populations but there are few studies which have identified the timescale to reach equilibrium and none identified that follow from a sudden change in environmental conditions as would be the case for the construction of an offshore windfarm.

## 4.14 Assumption 14

*It is assumed that there is sufficient evidence to predict if habituation will occur, if at all, and to what scale and over what timescale (Stage 10).*

### 4.14.1 Evidence summary

Habituation is the capacity of an animal to become accustomed to and not react towards a repeated action or pressure such as disturbance (Immelmann 1976)<sup>PR</sup>. The possibility that wintering (non-breeding) birds might habituate to wind farm structures has been suggested, but there is little evidence and few studies of long enough duration to show that habituation does occur (Powlesland 2009<sup>EP</sup>; Hotker *et al* 2006<sup>EP</sup>; Garvin *et al* 2011<sup>EP</sup>). Determining which species will adapt and habituate to wind farms will require years of study (Garvin *et al* 2011)<sup>PR</sup>. It has been suggested that some seabird species may not be able to adapt to moving structures such as turbines (Leopold *et al* 2011)<sup>P</sup>. Hotker *et al* (2006)<sup>EP</sup> in a review of a large number of post construction monitoring studies of primarily onshore wind farms found that habituation varies among species. The review found that the following species had one or more cases with signs of habituation: Eider, common scoter, oystercatcher, lapwing, black-tailed godwit, redshank, common gull, and black-headed gull but noted the contrasting evidence between studies of the same species, stating there was “*about as many cases of birds occurring closer to wind farms (indications for the existence of habituation) over the years as those of birds occurring further away from wind farms (indications for the lack of habituation)*”. In an onshore windfarm study, pink-footed geese were displaced by a distance of >100m at the start of the study in 2000 and eventually started feeding in the windfarm area, to within 40m of the turbines at the end of the study in 2008, providing evidence for habituation (Madsen and Boertmann 2008)<sup>PR</sup>. Petersen (2006)<sup>EP</sup> suggests that common scoter show no evidence of habituation, as only flying birds were seen in the Horns Rev and Nysted OWF. Stewart *et al* (2005)<sup>EP</sup> suggest that bird abundance declines with increasing time of windfarm operation, inconsistent with some other studies that found habituation to wind farms.

### 4.14.2 Conclusion

There is conflicting evidence relating to habituation, including between different studies looking at the same species, and a lack of information relating to seabird species. Studies would need to monitor operating wind farms over the long-term i.e. >10 years in order to provide greater information on the circumstances in which habituation occurs, or not.

## 4.15 Assumption 15

*It is assumed that seabird populations achieve equilibrium after adaptation to changes in habitat extent.*

### 4.15.1 Evidence summary

Equilibrium can be achieved within a population but that may depend on factors such as density dependence, the size of the new and old colonies, and environmental stability

(Kildaw *et al* 2005)<sup>PR</sup>. Bright *et al* (2009)<sup>EP</sup> considered that for seabirds to adapt to, and recover from, displacement effects there needs to be sufficient alternative habitat available. Chivers *et al* (2013)<sup>PR</sup> managed to define specific foraging locations for Kittiwakes but one of the locations varied due to changing environmental conditions (e.g. ocean currents). The fact that equilibrium may not have been achieved became more apparent at times of changing environmental conditions such as low prey availability. After three years, the survival of individual displaced redshank, a species of wader, was still lower than prior to displacement (Burton *et al* 2006)<sup>PR</sup> showing that reaching equilibrium may take time. The *sinensis* race of great cormorant increased for many years due to a variety of changes in its habitat, including an increase in inland stocked waterbodies, an increase that was only slowed by increased shooting pressure. For this subspecies it is debatable whether equilibrium has been reached after over a decade of rapid growth. Over a longer time period, new seabird colonies can reach a stable population (Kildaw *et al* 2005)<sup>PR</sup> providing evidence that an equilibrium can be being reached after an extended period. An Australasian gannet colony took 19 years from formation to reach a stable population (Pyk *et al* 2013)<sup>PR</sup>.

#### 4.15.2 Conclusion

It is unlikely that a stable equilibrium would be achieved as long as there is sufficient or an excess of suitable habitat available and if populations are not at carrying capacity. However if the environment were to remain fairly stable over time equilibrium should be achieved.

### 4.16 Assumption 16

*It is assumed there is sufficient evidence or theoretical knowledge on the length of time (in years) over which any elevated mortality / reduced productivity as a consequence of displacement would occur until new equilibrium is achieved.*

#### 4.16.1 Evidence summary

Seabirds are long-lived species with, generally, low productivity but with indications that they can 'buffer' breeding populations against years of high mortality e.g. storm induced starvation leading to 'seabird wrecks' (Fort *et al* 2009)<sup>PR</sup> by a 'pool' of non-breeding birds that occurs in the population (Aebischer and Wanless, 1992<sup>PR</sup>; Klomp and Furness 1992<sup>PR</sup>; Cam *et al* 1998<sup>PR</sup>; Reed *et al* 2015<sup>PR</sup>). This pool of birds can be adult birds that are not currently breeding (on 'sabbatical') or sub-adult birds that can begin breeding earlier than is typical for the 'equilibrium' population. This means that in theory the length of time taken for a new equilibrium to be reached (noting the caution about whether or not equilibrium exists in the highly variable marine environment – see Assumption 15) could depend on whether or not such a 'pool' of non-breeding birds exists and this might mask elevated mortality where the measure of the population is taken as the number of breeding birds at a colony. Evidence of the time taken to reach a new equilibrium after an event of elevated mortality / reduced productivity could be inferred from one-off events such as oil spills and 'seabird wrecks' but in these cases the time measured is that for the recovery (or not) of the population back up to the previous level once the one-off cause of mortality is removed. For a non-seabird species, redshank, the sudden loss of wintering habitat had a measurable effect on over-winter survival over the following three year period of the study (Burton *et al* 2006)<sup>PR</sup>. In context of the displacement assessment framework investigated the length of time over which new population equilibrium would form would likely be approached by running population models. The model would remove the projected number of individuals dying as a result of displacement from foraging habitat, reduce the productivity of individuals projected to be affected or combine both potential biological consequences and continue model runs, under different prey availability scenarios, until the population model reached

equilibrium. Nonetheless, the evidence summary indicates that no empirical data appear to be available to verify modelling results.

#### **4.16.2 Conclusion**

The length of time before a new equilibrium is achieved would, on a theoretical basis, depend on a number of factors related to particular seabird life histories. Evidence for this time period has not been located in the literature for seabirds. In a non-seabird species, Redshank, elevated mortality was still being detected after three years.

## 5 Species examples

### 5.1 Introduction

To illustrate how the framework described in Section 3 can be applied in practice, set out below are two worked examples for species with very different degrees of risk of being vulnerable to impacts through displacement. The worked examples are for red-throated diver in the non-breeding season, potentially the species most at risk (in a UK context) of displacement impacts based on current knowledge and lesser black-backed gull in the breeding season, potentially a species with little risk (in a UK context) of displacement impacts based on current knowledge.

The framework consists of the following stages and these are used in the two species examples to structure the information:

- Stage 1: Define size of impacted area
- Stage 2: Define density in impacted area (season specific)
- Stage 3: Define individuals affected (and connectivity to colonies / region of origin)
- Stage 4: Define geographic extent of remaining habitat (season specific)
- Stage 5: Accommodation of displaced individuals in remaining habitat?
- Stage 6: YES / NO (intermediate stage)
- Stage 7: Estimate proportion comfortably accommodated / Estimate proportion suffering from energetic costs
- Stage 8: All – no impact / range of impacts
- Stage 9: Local numerical decline by specific proportion of birds not accommodated in remaining habitat / supported by lost habitat
- Stage 10: Adjusted local numerical equilibrium of  $X_E - Y_E$  individuals after A – B years

In both cases an example is taken for a generalised example of a wind farm in the southern North Sea, a location where consenting issues have arisen in relation to both red-throated diver and lesser black-backed gull.

### 5.2 Red-throated diver

#### 5.2.1 Stage 1: Define size of impacted area

The size of the impacted area is defined in two steps:

- i. The size of the development footprint. This will be development specific and does not require a particular numerical value for this example. It can be clearly defined by the developer and does not require any assumptions.
- ii. The size of the species specific buffer zone. This is added to the development footprint to derive the species-specific spatial extent of the area / habitat that would be affected by a displacement impact.

For red-throated diver there is evidence that displacement does occur and that this occurs beyond the footprint of the windfarm (Section 4). Accordingly it is necessary to define the size of the buffer. This can be done in the generic manner described in Section 3.1.1 using a ranking of sensitivity. Red-throated diver is considered a highly sensitive species based on evidence of impacts that are independent of windfarm effects e.g. Schwemmer *et al* 2011, and has been ranked as such in vulnerability scoring systems (Garthe and Huppopp 2004, Furness and Wade 2012, Furness *et al* 2013). This known sensitivity would allocate this species to the 4km displacement category in the generic system described in Section 3.1.1. For red-throated diver there is a body of evidence from post-construction wind farm

studies that show displacement from a buffer around OWFs but the size of this buffer has been given a range of estimates: 800 m (Walls *et al* 2012); > 1km (Baker 2011), 1-3km (BSH 2012); 2km (Dierschke *et al* 2012); >2.5km (May 2008); > 3km (Percival 2009, 2010, 2014) and > 4km (Petersen *et al* 2006). A buffer of 4km would be a suitably precautionary value drawn from this evidence.

### **5.2.2 Stage 2: Define density in impacted area (season specific)**

The density of red-throated diver in the wind farm footprint and its buffer would be drawn from data acquired through site specific survey over two years and from any other data that is available over longer time periods from other survey and monitoring programmes (recognising that as such data will have been collected for another purpose it is likely to be at a lower spatial resolution than that collected for a specific development study). An appropriate method such as digital aerial survey should be used for this species that is disturbed at some distance by boats. This is because there is the potential for underestimating the density of divers if such birds are missed by the boat based observer. Since the density will vary through the non-breeding season and can be expected to vary between years the mean of the maximum density in each year should be used as the density in the impacted area.

### **5.2.3 Stage 3: Define individuals affected (and connectivity to colonies / region of origin)**

The number of individuals affected is calculated firstly from the size of the impacted area multiplied (Stage 1) by the density in the impacted area (Stage 2). Secondly, that number needs to be multiplied by the strength of the displacement effect, that is the percentage of birds displaced from the footprint and buffer. For red-throated diver the current evidence is that they are displaced 100% from within the footprint but within the buffer (of whatever size is drawn from the evidence – see Stage 1 above) the percentage displacement may not be 100%. The degree of displacement (in this case 100%) also might not be expected to fall abruptly at the outer extent of the buffer – that is there is not a displacement effect ‘cliff’ – but to decline at a particular rate across all or an outer portion of the buffer. The existence of a declining pattern in degree of displacement effect with distance (as you move away from the site into the buffer and beyond) has been shown by Dierschke *et al* (2012) and Petersen *et al* (2014).

### **5.2.4 Stage 4: Define geographic extent of remaining habitat (season specific)**

The extent of ‘remaining habitat’ is defined by the area of original habitat minus the area of the development footprint and any buffer that has been added. In the case of red-throated diver in the non-breeding season, in the southern North Sea, that original habitat area could be, for example, the area of the Outer Thames Estuary SPA. Subtraction of the area of the development footprint and its buffer is a simple mathematical operation. For in-combination impact assessment studies, other development footprints and respective buffers could be included along with other activities that might create areas avoided by red-throated diver, such as regularly used shipping lanes.

### **5.2.5 Stage 5: Accommodation of displaced individuals in remaining habitat?**

This next stage is based on the determination of whether or not the remaining habitat has the potential to accommodate any red-throated diver displaced from the development footprint and its buffer. For Stages 1-4 the information required has been either factual or relatively well evidenced from OWF studies. Stage 5 requires knowledge of whether or not the area being assessed is at carrying capacity for red-throated diver. This is currently

unknown for the southern North Sea and the Outer Thames Estuary SPA. In Section 3.1.5 it is suggested that relatively stable populations could indicate a population that is close to carrying capacity during years of average prey abundance and long-term increase could indicate that the population is not at carrying capacity. Monitoring of the Outer Thames Estuary SPA in recent years has shown some above average counts (in press) which gives some weight to the suggestion that it is not at carrying capacity.

#### **5.2.6 Stage 6: YES / NO (intermediate stage)**

At this stage there is a split in the assessment framework with the decision as to which route to take being dependent on knowledge of whether or not the area of habitat is at carrying capacity. As described in Stage 5 above, this is an unknown, resulting in the assessment proceeding from this point forward either on an 'either-or' basis or a decision made on a particular route without firm evidence.

#### **5.2.7 Stage 7: Estimate proportion comfortably accommodated / Estimate proportion suffering from energetic costs**

At this stage, and from here onward, it is considered that there are too many scientific uncertainties about the red-throated diver population for the assessment framework to proceed on an illustrated example basis.

#### **5.2.8 Stage 8: All – no impact / range of impacts**

At this stage, and from here onward, it is considered that there are too many scientific uncertainties about the red-throated diver population for the assessment framework to proceed on an illustrated example basis.

#### **5.2.9 Stage 9: Local numerical decline by specific proportion of birds not accommodated in remaining habitat / supported by lost habitat**

At this stage, and from here onward, it is considered that there are too many scientific uncertainties about the red-throated diver population for the assessment framework to proceed on an illustrated example basis.

#### **5.2.10 Stage 10: Adjusted local numerical equilibrium of $X_e$ - $Y_e$ individuals after A – B years**

For this stage it is considered that there are too many scientific uncertainties about the red-throated diver population for the assessment framework to proceed on an illustrated example basis.

### **5.3 Lesser black-backed gull**

#### **5.3.1 Stage1: Define size of impacted area**

As in the example of red-throated diver above, in this example for lesser black-backed gull the size of the impacted area is also defined in two steps:

- i. The size of the development footprint. This will be development specific and does not require a particular numerical value for this example. It can be clearly defined by the developer and does not require any assumptions.

- ii. The size of the species specific buffer zone. This is added to the development footprint to derive the species-specific spatial extent of the area / habitat that would be affected by a displacement impact.

For lesser black-backed gull there is evidence that displacement does not occur, either within the windfarm or beyond the footprint of the windfarm. The weight of evidence, for the large gulls as a group, is that either the numbers of the large gulls does not change (e.g. Petersen *et al* 2006, Fox *et al* 2006, Leopold *et al* 2011) or increases, indicating attraction (Vanermen *et al* 2012).

The alternative to basing a conclusion of no displacement from this evidence from post construction monitoring studies is to apply the generic approach that relates the disturbance sensitivity score of Furness *et al* (2013) to a % displacement category (see Table 1 for an example). Applying the relationship proposed by JNCC (2014) would mean that lesser black-backed gull with a disturbance sensitivity score of 2 would be placed in the 20-40% displacement category.

With respect to defining the size of the buffer around the wind farm a similar dichotomy occurs. The post construction monitoring evidence for large gulls suggests that the buffer should be zero as there is no evidence for displacement. The generic approach described in section 3.1.1 would be that the relatively insensitive species such as gulls should have a 1 km buffer applied and within that buffer displacement would be in the category of 20-40%.

In this worked example that dichotomy means that either:

- i. the assessment ceases here because it is accepted that there is sufficient evidence that displacement does not occur, or
- ii. the assessment continues through the steps of the framework and use is made of the generic figure of 20-40% displacement within the wind farm and within a 1km buffer placed around it.

For the purposes of this worked example, scenario ii is applied.

### **5.3.2 Stage 2: Define density in impacted area (season specific)**

The density of lesser black-backed gull in the wind farm footprint and its buffer would be drawn from data acquired through site specific survey over two years and from any other data that is available over longer time periods from other survey and monitoring programmes (recognising that as such data will have been collected for another purpose it is likely to be at a lower spatial resolution than that collected for a specific development study). An appropriate method such as digital aerial survey should be used for this species that may be attracted to boats in the hope of exploiting fishery discards. Such attraction can result in an overestimate of the density of gulls in the area. Since the density will vary through the breeding season (for instance as birds exploit different food sources which, in the case of lesser black-backed gull, can include terrestrial sources (Thaxter *et al* 2011, Ross-Smith *et al* 2014)) and can be expected to vary between years the mean of the maximum density in each year should be used as the density in the impacted area.

### **5.3.3 Stage 3: Define individuals affected (and connectivity to colonies)**

The number of individuals affected is calculated firstly from the size of the impacted area multiplied (Stage 1) by the density in the impacted area (Stage 2). Secondly, that number needs to be multiplied by the strength of the displacement effect, which is the percentage of birds displaced from the footprint and buffer. For lesser black-backed gull, applying the generic approach, it is that they are displaced by 20-40% from within the footprint and by 20-40% within the 1km buffer. In the latter case account also has to be taken of whether or not

a displacement effect 'cliff' is assumed, or has been identified, to occur at the boundary of the buffer. Unlike red-throated diver where two studies have shown that there is not a displacement effect 'cliff' but a gradual decline (Dierschke *et al* 2012, Petersen *et al* 2014) for lesser black-backed gull evidence has not been located that identifies the shape of the effect at its boundary.

#### **5.3.4 Stage 4: Define geographic extent of remaining habitat (season specific)**

The extent of 'remaining habitat' is defined by the area of original habitat minus the area of the development footprint and any buffer that has been added. In the case of this example of lesser black-backed gull in the breeding season for a development in the southern North Sea, that original habitat area could be, for example, the area over which birds would forage from the Alde-Ore Estuary SPA. This could be determined either by the use of the mean maximum foraging range from Thaxter *et al* 2012a or, as the breeding birds of the Alde-Ore Estuary SPA have been the subject of a specific tracking study (Thaxter *et al* 2011, Thaxter *et al* 2012b), the foraging area determined from that site specific study. Subtraction of the area of the development footprint and its buffer is a simple mathematical operation. For in-combination impact assessment studies, other development footprints and respective buffers could be included.

#### **5.3.5 Stage 5: Accommodation of displaced individuals in remaining habitat?**

This next stage is based on the determination of whether or not the remaining habitat has the potential to accommodate any lesser black-backed gulls displaced from the development footprint and its buffer. For Stages 1-4 the information required has been either factual or relatively well evidenced from OWF studies. Stage 5 requires knowledge of whether or not the area being assessed is at carrying capacity for lesser black-backed gull. This is currently unknown for the southern North Sea. In Section 3.1.5 it is suggested knowledge of recent population trends could provide an indication of whether or not the population is at or close to carrying capacity. The lesser black-backed colony at the Alde-Ore Estuary SPA has undergone recent, large population declines with the suggestion that this is due to a number of possible factors including loss of terrestrial food supplies and predation by foxes (Ross-Smith *et al* 2014).

#### **5.3.6 Stage 6: YES / NO (intermediate stage)**

At this stage there is a split in the assessment framework with the decision as to which route to take being dependent on knowledge of whether or not the area of habitat is at carrying capacity. As described in Stage 5 above and similarly for red-throated diver, this is an unknown, resulting in the assessment proceeding from this point forward either on an 'either-or' basis or a decision made on a particular route without firm evidence.

As for the example of red-throated diver above it is considered that from this stage forward there are too many scientific uncertainties about the lesser black-backed gull population for the assessment framework to proceed on an illustrated example basis.

Stages 7 to 10 are not repeated in this example.

## 6 Recommendations on suitability of the approach

The proposed method provides a structured screening framework for reporting on potential displacement. It provides transparency on where assumptions are being made. This is an advantage over the existing matrix approach to estimating displacement that gives a range of figures without examining the underlying assumptions behind the values selected.

For species that are not displaced, and hence do not progress beyond Stage 3 of the proposed method, it does provide a transparent reporting framework.

However, development and testing of the proposed method has identified a number of stages that require specific information about the species in question that are absent from the literature. This creates issues with the method, including a large degree of uncertainty over the validity of any outputs from the method, when assumptions are made about the unknown numeric values.

The stages of the core assessment framework (Figure 1) where information is largely absent for many seabird species are:

- Stage 5: Accommodation of displaced individuals in remaining habitat
- Stage 7: Estimate proportion accommodated
- Stage 9: Local numerical decline by specific proportion of birds not accommodated in remaining habitat / supported by lost habitat
- Stage 10: Adjusted local numerical equilibrium

The development of a staged framework, with each stage having its own level of uncertainty, means that when they are put together an unacceptable degree of cumulative uncertainty can be the result. Making precautionary assumptions at each stage means that the scale of the value that is determined at the end of the process will inevitably be highly precautionary, potentially to the point that is not supportive of decision making.

Pragmatic rules such as 'if the quality of the lost habitat is medium or low and it represents less than X% of the remaining habitat, no effect can be concluded' (see Stage 4 Figure 2) may be a way to reduce the complexity of some assessment stages. In the end expert opinion is likely to be needed to estimate numerical values (replacing the X%) in the absence of empirical evidence.

The development of a framework with a series of stages does help identify where action is required to gather information by research or study to provide a better understanding of the effects of displacement at a general as well as species-specific level.

It should be noted that probably the most basic of all assumptions underlying the 'Displacement as Habitat Loss' approach - the assumption that populations adjust to a new equilibrium after the realisation of a displacement effect - is surrounded by considerable uncertainty. Even if population equilibrium exists in seabird populations, it may not be possible to reliably measure that equilibrium in the field due to strong fluctuations inherent in marine ecosystems. Accordingly, population equilibrium is unlikely to be constant and thus separating estimated displacement effects from fluctuating equilibrium values and other pressures on the population assessed may be impossible.

If the proposed method is adopted for displacement, consideration has to be given to how the adopted assumptions might apply in the case of other effects arising from the development of OWFs. Habituation might apply to the assessment of collision and to the assessment of barrier effects in the same way as proposed in the framework for

displacement. Consideration should be given to the consistency of the application of assumptions between the different types of effect that are assessed.

Based on the currently available evidence, that suggests strong site-specific variations in percentage displacement, it would appear best to use generic displacement categories for species, such as high, medium or low sensitivity.

## **6.1 Conclusion**

The staged assessment framework described in this report adds clarity to where effects could be happening. However, it introduces a number of complications to the assessment process (being particularly dependent on some seabird biology parameters that are poorly known), without providing significant benefits over the existing year-on-year mortality assessment method that currently underpins the widely used 'Displacement Matrix' approach (albeit recognising that this method also has major evidence gaps and assumptions).

That said, should significant strides be made in the empirical knowledge base underpinning some of the key assumptions/questions of the 'Displacement as Habitat Loss' approach (e.g. carrying capacity, robustly characterising when seabird populations have reached equilibrium) it may be possible for the framework to be applied to OWF assessments in the future.

## 7 Monitoring and research requirements

Set out below are brief proposals for monitoring and research actions that will progress filling some of the information gaps that have been identified through the process of testing the evidence base of the assumptions inherent in the framework (Section 4). In some cases those actions can be taken forward through post-construction monitoring at operating OWFs, in other cases it will require research studies and investigations that are independent of post-construction monitoring actions.

The proposals are:

### 1. Tracking studies

Tracking studies to define the area of the breeding season habitat as a step in estimating the area of remaining habitat after displacement has occurred. Such studies should concentrate on seabird colonies in or adjacent to areas where OWFs are planned in order to inform EIAs and Ideally should have a large sample size and, if possible, track both adult breeding birds from the colony the non-breeding birds attending the colony.

### 2. Carrying capacity

Research into the carrying capacity of local populations in or adjacent to areas where OWFs are planned. This is likely to be based on long-term data series e.g. from the Seabird Monitoring Programme (SMP), examining the population trends and relating those to prey abundance changes.

### 3. Productivity

Productivity monitoring at seabirds colonies in or adjacent to areas where OWFs have been constructed as a component of post-construction monitoring. Productivity should be modelled together with annual prey abundance to assess potential impacts on reproductive output and adult survival over the long-term as a consequence of displacement.

### 4. Habitat quality

Research into relative habitat quality (e.g. using seabird densities as a proxy). The focus should be on variability among densities over extended time periods. Areas hosting high densities showing relatively low variability may provide a realistic indication for relatively constant, high-quality habitat.

### 5. Demographic data for population models

Acquiring the demographic data for, and developing, population models for species most likely to be affected by displacement.

### 6. Species-specific displacement buffers

For most seabird species, the principal exception being red-throated diver, there is little or no evidence from peer reviewed sources of the degree of displacement from within OWFs and the degree and extent of displacement around OWFs. Further post-construction monitoring studies should be able to provide such quantitative measures. A BACI approach is applicable where there is reliable pre-construction information and a gradient approach taken for sites where pre-construction information of insufficient quality.

**7. Survey protocols with sufficient precision to detect significant change**

The high variability of seabird populations occupying any particular sea area between years poses a challenge to any survey design and the method applied to detect and identify the significance of change. Approaches could include more intensive sampling, measuring and relating changes to environmental variables and alternative statistical approaches.

**8. Defining the size of the seasonal specific foraging area for a defined population**

For most species foraging ranges have been defined for the breeding season and these can be used to identify the breeding season foraging area. For the non-breeding season there is not an extensive evidence base. Tracking studies with small tags that can operate over extended timescales offer a means to follow birds over the course of one or several years and hence define the sea areas used, for what periods, outside the breeding season (e.g. as for kittiwake). If such studies can be conducted in sufficient quantity then they may offer a means to define the foraging areas used by birds from particular colonies.

**9. Habituation**

Studies would need to monitor operating wind farms over the long-term (i.e. >10 years) in order to provide greater information on the circumstances in which habituation does, or does not, occur.

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