Departmental brief:

Northumberland Marine potential Special Protection Area (pSPA)

Natural England

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Contents

Summary

This departmental brief sets out the scientific case for the classification of the Northumberland Marine potential Special Protection Area. Northumberland Marine pSPA is proposed to protect important areas of sea used for a variety of purposes, including maintenance and foraging behaviours by the qualifying interest features of a number of already-classified SPAs: Coquet Island, Farne Islands, Lindisfarne and Northumbria Coast, as well as potential additional features identified by a review of the seabird populations of those SPAs. Northumberland Marine pSPA qualifies under Article 4 of the Birds Directive (2009/147/EC) for the following reasons (summarised in Table 1):

- The site regularly supports more than 1% of the Great Britain breeding populations of five species listed in Annex I of the EC Birds Directive. Therefore the site qualifies for SPA Classification in accordance with the UK SPA selection guidelines (stage 1.1).
- The site regularly supports more than 1% of the biogeographical population of two regularly occurring migratory species not listed in Annex I of the EC Birds Directive. Therefore the site qualifies for SPA Classification in accordance with the UK SPA selection guidelines (stage 1.2).
- The site regularly supports an assemblage of more than 20,000 individual breeding seabirds. Therefore the site qualifies for SPA Classification in accordance with the UK SPA selection guidelines (stage 1.3).

Table 1 Summary of qualifying ornithological interest in Northumberland Marine pSPA

Species	Count (period)	% of subspecies or population	Interest type	Selection criteria
Sandwich tern *Sterna sandvicensis*	4,324 individuals (2010-2014)[1]	19.66% of GB population[6]	Annex 1	Stage 1.1
Common tern *Sterna hirundo*	2,572 individuals (2010-2014)[1]	12.86% of GB population[6]	Annex 1	Stage 1.1
Arctic tern *Sterna paradisaea*	9,564 individuals (2010-2014)[1]	9.02% of GB population[6]	Annex 1	Stage 1.1
Roseate tern *Sterna dougallii*	160 individuals (2010-2014)[2]	93.02% of GB population[6]	Annex 1	Stage 1.1
Little tern *Sternula albifrons*	90 individuals (2010-2014)[2]	2.37% of GB population[6]	Annex 1	Stage 1.1
Puffin *Fratercula arctica*	108,484 individuals (2008-2013)[1,3]	1.05% of biogeographic population[7]	Regularly occurring migratory species	Stage 1.2
Guillemot *Uria aalge*	65,751 individuals (2010-2014)[1,4]	1.72% of biogeographic population[8]	Regularly occurring migratory species	Stage 1.2
Internationally seabird assemblage of over 20,000 individuals (including the 7 qualifying species listed above plus: great cormorant, European shag, black-headed gull and black-legged kittiwake as main components of the assemblage)	214,669 Individuals (2010-2014)[5]			Stage 1.3

Seabirds that undertake maintenance and/or foraging behaviour within Northumberland Marine pSPA include those that breed at existing SPAs in Northumberland, including species and overall seabird assemblages that are not currently classified features of these SPAs but are present in qualifying numbers. Specifically these are; Lindisfarne, Northumbria Coast, Farne Islands and Coquet Island SPAs. Accordingly the numbers listed in the table above are summed across the relevant site specific population estimates.

[1] Data from: Seabird Monitoring Programme (SMP) and colony managers (Pairs multiplied by 2 to arrive at breeding adults; this rule applies to all species listed within the table, with the exception of guillemot.)

[2] Data from: directly from colony managers (Pairs multiplied by 2 to arrive at breeding adults; this rule applies to all species listed within the table, with the exception of guillemot.)

[3] Results of puffin censuses from the Seabird Monitoring Programme (SMP)

[4] Guillemots are counted as “individuals on land”; this is multiplied by a correction factor of 0.67 (Harris 1989) to translate to breeding pairs and multiplied by 2 to yield an estimate of the number of breeding adult individuals.

[5] With exception of puffin for which censuses in 2008, 2009 and 2013 have been used.

[6] GB breeding populations derived from Musgrove *et al*. (2013)

[7] Biogeographic populations of 5,176,257 pairs (10,352,514 breeding individuals) derived from UK SPA and Ramsar Scientific Working Group (2014) paper*: International Population Estimates for some seabird species*. Figure derived in line with Mitchell *et al.* 2004 on the basis that puffins which used to be considered to be of the race *Fratercula arctica grabae* are now combined with those of the nominate race of *F.a.arctica* and a biogeographic population estimate derived by summing birds breeding in: France, GB, Isle of Man and Channel Islands, All-Ireland, all of Norway, Iceland and Russia (but excluding birds listed as *F.a.arctica* in Mitchell et al (2004) and breeding in Canada, USA and Greenland.

[8] Birds breeding at the Farne Islands and hence included within the Northumberland Marine pSPA are assumed to belong to the nominate race of *Uria aalge aalge* in line with UK SPA and Ramsar Scientific Working Group (2014) paper*: International Population Estimates for some seabird species* in which a population midpoint estimate of 1,909,417 pairs (rounded to 3,820,000 individuals) is given.

1. Assessment against SPA selection guidelines

The UK SPA Selection Guidelines requires that SPA identification should be determined in two stages (Stroud *et al*. 2001). The first stage is intended to identify areas that are likely to qualify for SPA status. The second stage further considers these areas using one or more of the judgements in Stage 2 to select the most suitable areas in number and size for SPA classification (Stroud *et al*. 2001).

1.1. Stage 1

Under stage 1 of the SPA selection guidelines (JNCC, 1999), sites eligible for selection as a potential SPA must demonstrate one or more of the following:

1) an area is used regularly by 1% or more of the Great Britain (or in Northern Ireland, the all-Ireland) population of a species listed in Annex I of the Birds Directive (2009/147/EC) in any season;

2) an area is used regularly by 1% or more of the biogeographical population of a regularly occurring migratory species (other than those listed in Annex I) in any season;

3) an area is used regularly by over 20,000 waterbirds (waterbirds as defined by the Ramsar Convention) or 20,000 seabirds in any season

4) an area which meets the requirements of one or more of the Stage 2 guidelines in any season, where the application of Stage 1 guidelines 1, 2 or 3 for a species does not identify an adequate suite of most suitable sites for the conservation of that species.

Northumberland Marine pSPA qualifies under stage 1(1) because it regularly supports greater than 1% of the GB population of five species listed in Annex I of the Birds Directive (*i.e*. Sandwich tern, common tern, Arctic tern, roseate tern and little tern). In addition, the site qualifies under stage 1 (2) because it regularly supports over 1% of the biogeographical populations of two regularly occurring migratory birds (puffin and guillemot), and under stage 1 (3) by regularly supporting a breeding seabird assemblage of over 20,000 individuals, including great cormorant, European shag, black-headed gull and black-legged kittiwake as main components (see table 1).

1.2. Stage 2

The Northumberland Marine pSPA is assessed against Stage 2 of the SPA selection guidelines in Table 2. It should be noted that in applying the SPA selection guidelines, Stroud *et al.* (2001) note that a site which meets only one of these Stage 2 judgments is not considered any less preferable than a site which meets several of them, as the factors operate independently as indicators of the various different kinds of importance that a site may have. In fact, the pSPA meets most of the Stage 2 criteria indicating the high value of the site.

Table 2 Assessment of the bird interest against stage 2 of the SPA selection guidelines

Feature	Qualification	Assessment
1. Population size & density[1]	✓	Arctic tern, common tern and roseate tern - largest in UK Sandwich tern - 2nd largest in UK Seabird assemblage **-** 2nd largest in England Puffin - 3rd largest in UK Guillemot - 4th largest in UK Little tern - 11th largest in UK
2. Species range	✓	Southernmost SPA with puffin and Arctic tern as qualifying features in England.

3. Breeding success	✓	In 2013 puffin breeding success on the Farne Islands was relatively high at 0.93 fledglings per nest (SMP website) compared with the average for Britain and Ireland of 0.71 per nest during 1986 to 2005 (Mavor *et al.* 2008). Between 2010 and 2014, average Arctic tern productivity on Coquet Island was high at 0.97 fledglings per nest (SMP website) compared to the UK population as a whole, where between 1986 and 2013, annual average productivity has only risen above 0.40 once, in 2000, and in most years is below 0.30 (http://jncc.defra.gov.uk/page-2896)
4. History of occupancy[2]	✓	All primary source colonies have a long history of occupancy by breeding seabirds: Farne Islands since 17th C, Coquet island has held larger tern species since 19th C and Northumbria Coast has held little terns since 1950s and Arctic terns since 1980.
5. Multi-species area	✓	Five qualifying Annex 1 features, two qualifying regularly occurring migrants and a breeding seabird assemblage of which a further four species occur in nationally important numbers.
6. Naturalness		No longer applicable, following ruling from the SPA and Ramsar site Working Group
7. Severe weather refuge		Not relevant to breeding species.

[1] Note that these rankings should only be considered indicative of the relative importance of the pSPA as it is based on comparison of the sum of the most recent 5 year mean populations of each tern species at the source SPAs (as listed in Table 1) with the historical populations of each species at each SPA in the UK as listed in Stroud *et al.* (2001).
[2] Source: The Atlas of Breeding Birds of Northumbria (Day *et al*., 1995)

2. Rationale and data underpinning site classification

In 1979 the European Community adopted Council Directive 79/409/EC on the conservation of wild birds known as the Birds Directive. This has been amended subsequently as Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds. The Birds Directive provides for protection, management and control of naturally occurring wild birds within the European Union through a range of mechanisms. One of the key provisions is the establishment of an ecologically coherent network of protected areas. Member States are required to identify and classify in particular the most suitable territories in size and number for rare or vulnerable species listed in Annex I to the Directive (Article 4.1) and for regularly occurring migratory species (under Article 4.2). These sites are known as Special Protection Areas, referred to as SPAs in the UK. Guidelines for selecting SPAs in the UK are derived from knowledge of common international practice and based on scientific criteria (JNCC 1999).

The task of identifying all of the UK's terrestrial sites is largely complete, and the rationale described by Stroud *et al*. (2001). Stroud *et al.* (2001) describe a network of 243 sites in the UK, some of which include areas used by inshore non-breeding waterbirds, for example in estuaries. However, this suite of sites does not address the requirement for the implementation of conservation measures in the wholly offshore environment in which many birds access resources that are critical for their survival and reproduction. Johnston *et al*. (2002) describe a process consisting of three strands by which SPAs might be identified for marine birds under the Birds Directive *i.e*. the identification of:

i. seaward extensions of existing seabird breeding colony SPAs beyond the low water mark;
ii. inshore feeding areas used by concentrations of birds (e.g. seaduck, grebes and divers) in the non-breeding season; and
iii. offshore areas used by marine birds, probably for feeding but also for other purposes.

Since then, a fourth strand was added to the work conducted by the Joint Nature Conservation Committee (JNCC) to address the need for:

iv. other types of SPA http://jncc.defra.gov.uk/page-4184 that would identify some important areas for marine birds that may not be included within the above three categories and will be considered individually

Under the first of these 4 strands, the Joint Nature Conservation Committee (JNCC) has recommended that the boundary of certain existing seabird colony SPAs be extended into the marine environment and produced generic guidance to implement this measure (McSorley *et al*. 2003, 2005, 2006; Reid & Webb 2005). The seaward extensions of existing boundaries in these cases will include waters vital for ensuring that some of the essential ecological requirements of the breeding seabird populations are met (e.g. preening, bathing, displaying and potentially local foraging). The distance of the extension is dependent upon the qualifying species breeding within the SPA. Generic seaward extensions to the boundaries of existing SPAs have been implemented at 31 sites in Scotland and are under consideration at the Flamborough and Filey Coast pSPA (Natural England 2014). It is proposed here that generic boundary extensions that could be applied directly to the Coquet Island SPA and Farne Islands SPA are instead incorporated within the Northumberland Marine pSPA. However, these generic boundary extensions recommended by JNCC in the case of certain species are not influenced by or meant to encompass the principal foraging areas used by the species for which they are identified or any other species at the colonies concerned.

All five species of tern that breed in the UK (Arctic *Sterna paradisaea*, common *S. hirundo,* Sandwich *S. sandvicensis,* roseate *S. dougallii* and little tern *Sternula albifrons*) are listed as rare and vulnerable on Annex I of the EC Birds Directive and thus are subject to special conservation measures including the classification of Special Protection Areas (SPAs). Within the UK there are currently 57 breeding colony SPAs where at least one species of tern is protected. However, additional important areas for terns foraging at sea have yet to be identified and classified as marine SPAs to complement the existing terrestrial suite. Since 2007, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of such areas under the fourth work strand as, given the likely extent of these areas, these cannot be addressed by application of the generic “maintenance” extensions approach and are not covered by the work on identifying inshore non-breeding aggregations or important offshore areas.

This departmental brief sets out all of the information supporting the identification of the qualifying features of the Northumberland Marine potential SPA and the definition of its proposed boundaries on the basis of the areas of sea identified, under the first and fourth strands of JNCC’s work, as being most important to the auk and tern populations that are among the qualifying features of this new pSPA i.e. the populations of auks and terns at the existing Coquet Island, Farne Islands, Northumbria Coast and Lindisfarne SPAs.

In the process by which a site becomes fully classified as an SPA, Ministerial approval has to be given to undertake formal consultation on the proposal to classify the site. At this stage in the process a site becomes known as a potential SPA (pSPA). Within this Departmental Brief, and others being prepared at the same time, sites currently under consideration include both new sites (such as Northumberland Marine) and existing sites (such as Poole Harbour SPA) which are being extended and/or having new features added. For the purpose of clarity in this and other Departmental briefs, sites are referred to as “SPA” when referring to the existing classified site. Where reference is made to an entirely new site, or to an extended site, or to a site including new features being proposed it will be referred to as pSPA since the site (if new), or any additional extent or feature is not yet fully classified.

SPA site selection guidelines have been applied to the most up to date information for the site. However, these contemporary data reveal that at some source SPA colonies some species are no longer present in qualifying numbers (either through declines or because the relevant threshold has increased). It is not clear whether anthropogenic influences have affected the populations at the site. Defra policy indicates that in these circumstances the feature should be retained until such time as the reasons for the reduction in population can be established. Natural England therefore considers that these species should be retained on the citation of the source SPAs, and the level of ambition set out in the conservation objectives for the species maintained, until we have evidence to support the conclusion that declines are a result of natural processes and that the SPA is no longer suitable for this species.

2.1. Data collection – defining the suite of species and numbers of those supported by the Northumberland Marine pSPA.

The size of each of the populations of birds supported by the Northumberland Marine pSPA have been derived as the sum of the numbers of those qualifying species breeding at the existing SPAs from which the individuals recorded at sea within the Northumberland Marine pSPA are most likely to originate. For each of those SPAs, the numbers have been taken to be the most recently available from the Seabird Monitoring Programme (SMP) website (http://jncc.defra.gov.uk/smp/) i.e. within the last 5 years, unless otherwise indicated. Where possible, this dataset has been augmented by information requested directly from colony managers for roseate tern and little tern.

2.2. Data collection – defining the boundary of Northumberland Marine pSPA

The overall boundary of the Northumberland Marine pSPA has been drawn to encompass the sea areas identified under the first and fourth strands of JNCCs work programme as being most important to support the auks and terns which are already qualifying features of several source colony SPAs (or are proposed to be added as new qualifying features to them). The work done to identify the areas important to each species differed and was conducted separately.

3. Site status and boundary

By virtue of encompassing the principal sea areas of importance to support the foraging of populations of five Annex 1 species of tern (little tern, Arctic tern, common tern, roseate tern, Sandwich tern) at three existing SPAs where the numbers of one or more of those species exceed SPA selection guidelines (JNCC 1999) (and little terns at Lindisfarne whose numbers currently do not exceed SPA selection guidelines), Northumberland Marine pSPA itself qualifies by regularly supporting those nationally important numbers of breeding Annex 1 species. Also by encompassing the principal sea areas identified as of importance for the maintenance behaviour of guillemot and puffin at two existing SPAs where the numbers of these species exceed SPA selection guidelines (JNCC 1999) the Northumberland Marine pSPA itself qualifies by regularly supporting those nationally important numbers of regularly occurring migratory species.

Furthermore, given the presence of assemblages of European importance of over 20,000 breeding seabirds at both the Farne Islands SPA and Coquet Island SPA, the Northumberland Marine pSPA itself qualifies by regularly supporting an assemblage of European importance of over 20,000 breeding seabirds. Little tern, Arctic tern, common tern, roseate tern, Sandwich tern, guillemot and puffin, which are all present in internationally important numbers, are all main components of the pSPA assemblage. So too are: great cormorant *Phalacracorax carbo*, European shag *Phalacracorax aristotelis*, black-headed gull *Chroicocephalus ridibundus* and black-legged kittiwake *Rissa tridactyla* which are present within the two principal source colonies of Coquet Island and Farne Islands SPAs in numbers which, when combined across these source SPAs, either exceed 2,000 individuals (10% of the minimum qualifying assemblage of 20,000 individuals) or 1% of the GB breeding population.

In addition, fulmar, great black-backed gull *Larus marinus*, lesser black-backed gull *Larus fuscus*, herring gull *Larus argentatus* and razorbill are also present within the two principal source colonies of Coquet Island and Farne Islands SPAs and are therefore considered to constitute part of the assemblage of the Northumberland Marine pSPA, albeit not as main components, due to their relatively low numbers.

It is proposed that the potential SPA (pSPA) should be known as Northumberland Marine pSPA to reflect its broad geographical location and marine nature.

3.1. Coastal boundary

The landward boundary is located on the Northumberland coast and surrounding waters. The boundary starts just south of Berwick-Upon-Tweed, at Scremerston in the north, and ends approximately 75 km to the south just beyond Blyth. The northernmost extremity is determined by the modelled usage of Sandwich terns foraging from the Farne Islands SPA, whereas the southernmost extremity is determined by the modelled usage of Sandwich terns foraging from Coquet Island SPA. JNCC guidelines on selecting marine SPAs (Webb & Reid 2004: Annex B) states that where the distribution of birds is likely to meet land, landward boundaries should be set at Mean High Water (MHW) “unless there is evidence that the qualifying species make no use of the intertidal region at high water”. Observations indicated that little terns forage both in the intertidal zone and subtidal zone (Parsons *et al.* 2015). The likelihood of use of such areas by all larger tern species is supported by information in the scientific literature. A review of tern foraging ecology (Eglington 2013) notes that all five species of tern considered here routinely forage in areas of shallow water. Thus, along this stretch of coast the boundary will follow the Mean High Water. This means that in the areas where the intertidal zone is already classified as SPA (at Lindisfarne and Northumbria Coast SPAs) there will be an overlap in SPA designations. This is required to ensure the intertidal habitats used by foraging terns (not all of which are features of the Lindisfarne and Northumbria Coast SPAs) are protected.

3.1. Seaward boundary

The seaward boundary consists broadly of three arcs running north to south with the most seaward parts of the these arcs lying in an approximately north-easterly direction from the Farne Islands, Long Nanny and Coquet Island colonies. These seaward extremities lie at 20km, 15.4km and 14km respectively from the mainland shore and are determined by the modelled usage of Arctic terns foraging from each of these three colonies.

3.2. Boundary around offshore islands

Around the Farne Islands and Coquet Island the boundary has been drawn at Mean Low Water, avoiding overlap with the existing island SPAs, which already encompass both the islands and the fringing intertidal habitats.

3.3. Description of the boundary determination

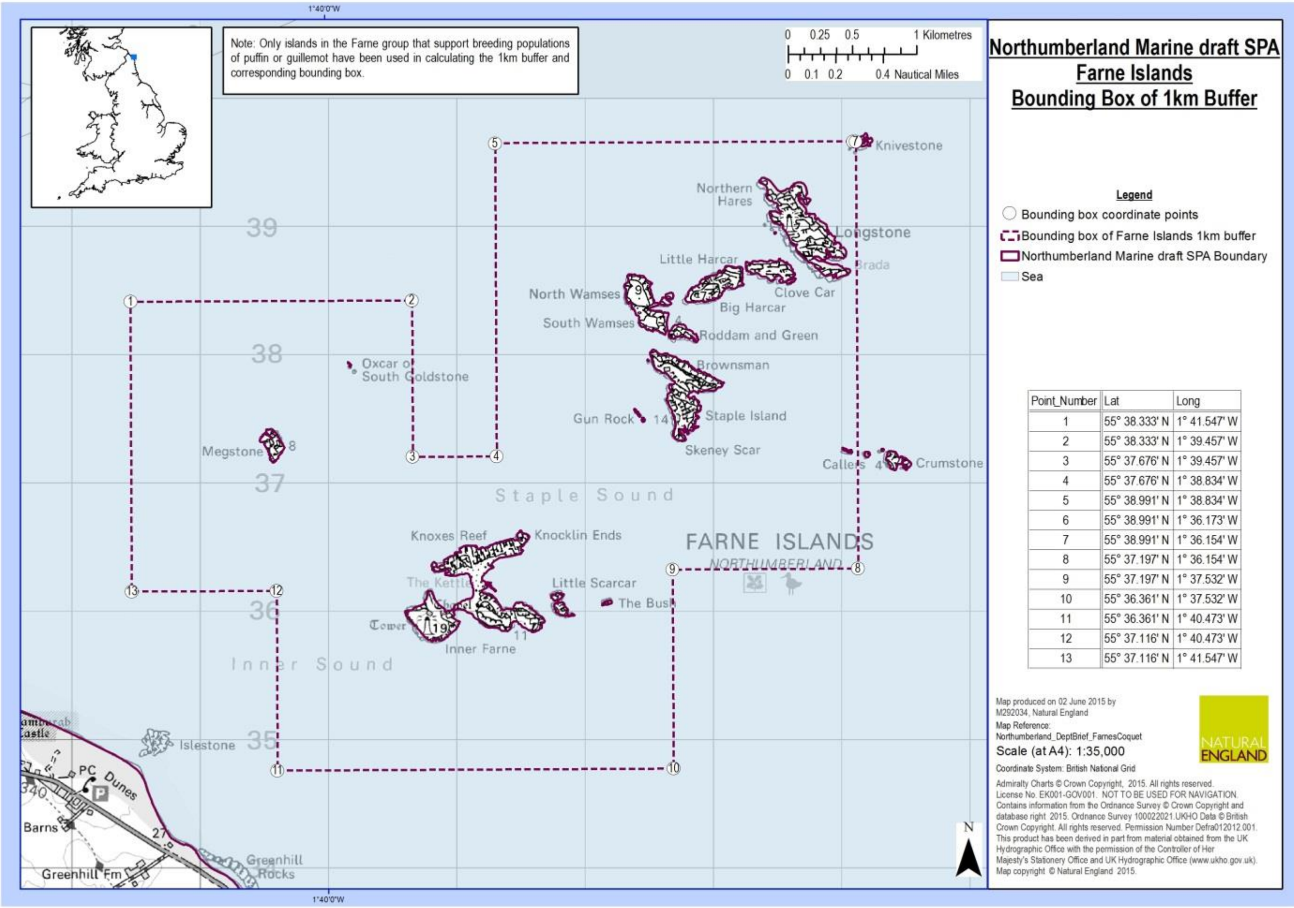
This section describes how the final composite boundary for the site has been arrived at. Northumberland Marine Area pSPA will encompass several terrestrial sites. This section explains how the final boundary has been drawn as a composition of the extensions for the auk and tern species and their relation to each existing terrestrial site.

3.3.1. Extensions for maintenance behaviours

JNCC guidance regarding seaward maintenance extensions (McSorley *et al.* 2003, 2006) recommends that all existing colony SPAs at which guillemot *Uria aalge,* razorbill *Alca torda* and puffin *Fratercula arctica* are qualifying species, be extended by 1km from mean low water mark. On that basis, a 1km generic seaward extension is proposed here in regard of two auk species i.e. puffin and guillemot which are either qualifying features or main assemblage components of existing colony SPAs at the Farne Islands (both species) and Coquet Island (puffin only) or are proposed as being so (see below).

Natural England is recommending the amendment the Farne Islands pSPA to clarify the inclusion of terns and guillemot as a qualifying feature, and the addition of a seabird assemblage of over 20,000 seabirds with puffin as a named assemblage component to the Farne Islands pSPA and the Coquet Island pSPA (Coquet Island and Farne Islands Departmental Brief, *published, 2015*). Therefore, the Northumberland Marine pSPA will encompass a generic 1 km seaward extension around the Farne Islands pSPA for the puffin and guillemot and a generic 1 km seaward extension around Coquet Island pSPA for the puffin. It should be noted that when a species covered in JNCC's guidance regarding generic seaward extensions (McSorley *et al*. 2003, 2005) does not occur in sufficient numbers to be a main assemblage component at an SPA, then it is not considered when identifying generic seaward extension boundaries (i.e. razorbill *Alca torda* at the Farne Islands pSPA and fulmar *Fulmarus glacialis* at both the Farne Islands pSPA and Coquet Island pSPA).
The two generic seaward extension areas around the existing island SPAs sit wholly within the proposed boundary of the Northumberland Marine pSPA (Figures 1 and 9). The details of the derivation of these generic seaward extension areas are described in Annex 4.

a)



b)

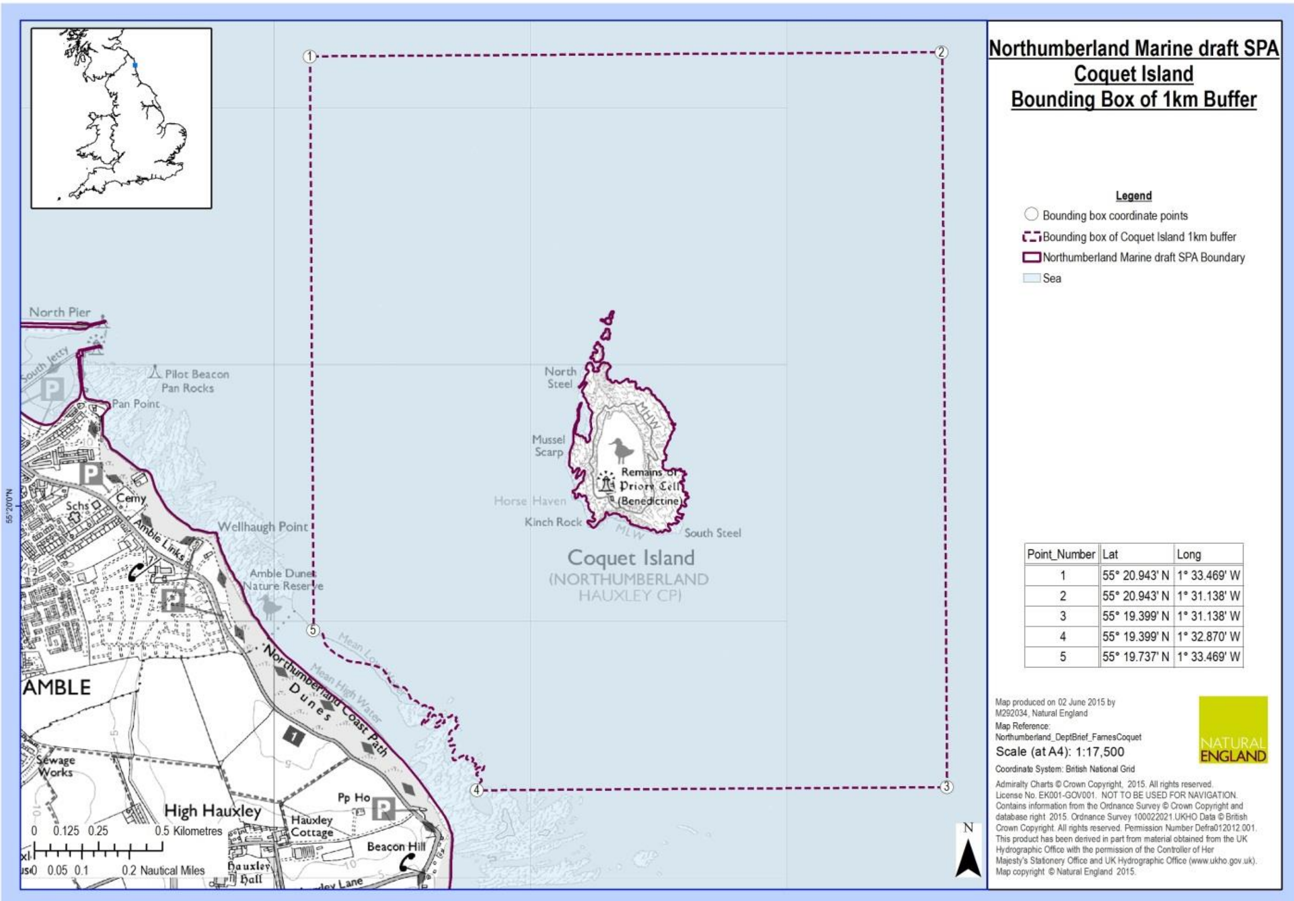


Figure 1. Application of generic seaward extensions of 1 km around a) Farne Islands (for puffin and guillemot) and b) Coquet Island (for puffin). Note that the boundaries are drawn in relation only to those islands within the Farne Islands group which support breeding guillemot and/or puffin.

3.3.2. Identification of important marine areas for little tern

Of the six species of tern which regularly breed in Great Britain, little tern is the smallest in size and has the most limited foraging range: mean range of 2.1 km, mean of recorded maxima of 6.3 km and maximum ever recorded in the literature being 11 km (Thaxter *et al.* 2012). In the light of this evidence, JNCC, in agreement with the SNCBs, decided that the most effective method to determine the extent of the areas most heavily used for foraging by breeding little terns would be to undertake a programme of shore based observations and of boat-based transects of offshore areas around colonies, and to use the resultant distribution data directly in setting the alongshore and seaward boundaries respectively. Accordingly, between 2009 and 2013, JNCC coordinated a programme of survey work to identify important foraging areas for little terns at a number of UK little tern colonies. These surveys were conducted during the chick rearing period in each year and comprised repeated shore-based counts of little terns seen at a series of observation stations at increasing distances from the colony locations, and repeated boat based surveys along transects across the waters offshore from the colonies. These surveys sought to establish the distances both alongshore and offshore that little terns were travelling to feed.

In total, 70 shore-based surveys were undertaken at 14 little tern colonies around the UK with a total of 7,006 registrations of little terns at various points along the shore. Twenty three boat-based transect surveys were undertaken across waters near eight colonies around the UK with a total of 781 registrations of little terns at various distances offshore.

The following two sections outline in brief the survey work and boundaries identified at the two little tern colonies which fall within the area of the Northumberland Marine pSPA. Further general information on little tern survey programme is presented in Annex 5.

3.3.2.1. Northumbria Coast SPA

Nine shore-based surveys were undertaken at the colony at Newton Links/ Long Nanny within the Northumbria Coast SPA in 2009 (2), 2010 (5) and 2011 (2). They recorded a total of 518 tern passes. Two boat-based surveys were completed in 2011 on which a total of 22 little terns were sighted.

Based on site-specific survey data the maximum alongshore extent of little tern observations from the Northumbria Coast SPA colony i.e. the colony at Newton Links/Long Nanny, was 5,000m north and 6,000m south, and the mean maximum seaward extent was 2,598m (Figure 2).

The little tern foraging area does not define the boundary of the proposed Northumberland Marine pSPA as it sits entirely within the boundary identified on the basis of the work on the larger tern species.

Long Nanny, Northumbria Coast SPA
Estimates of foraging extent

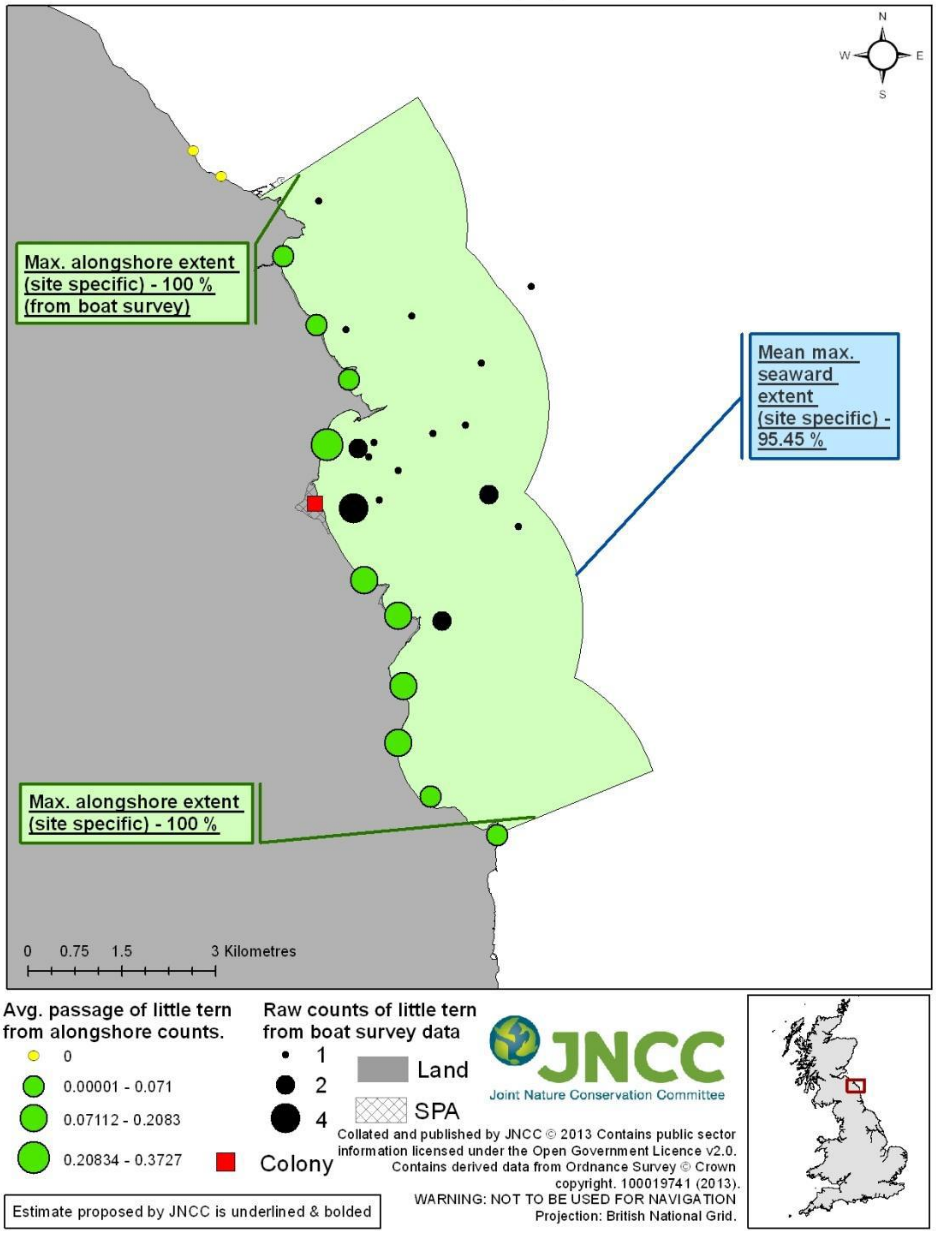


Figure 2. Application of site specific alongshore and site specific seaward extents to define boundaries to little tern foraging areas at the Long Nanny colony. The percentage values given in the labels indicate the site specific percentage of little tern observations within the shore-based (alongshore) dataset and boat-based (seaward) dataset captured within the proposed alongshore and seaward boundaries. Note that the 95.45% of sightings recorded within the seaward boundary is not the measure used to define where that boundary should be, but is simply a consequence of setting the boundary at the mean of the survey specific maxima and thereby excluding sightings of a few individuals seen furthest offshore – most likely on the survey day that yielded the furthest ever seaward sightings.

3.3.2.2. Lindisfarne SPA

Only six shore-based surveys were undertaken at Lindisfarne SPA in 2012 (3) and 2013 (3) and recorded a total of 53 tern passes. In summary, the maximum alongshore extent recorded was 3,000m north and 4,000m south of the colony. However, due to the low number of little tern sightings on these surveys (n=53), this was assessed as insufficient to justify a site-specific approach to boundary definition. The alongshore foraging extent that would be relevant to birds from this colony was set to be the generic value derived from all of the surveys at all of the colonies (n=13) i.e. 3,900m (see Annex 5).

No boat-based surveys were undertaken at the Lindisfarne SPA, because there were a low number of breeding pairs during the survey period, meaning it would have been unlikely to obtain sufficient observations. Therefore, the seaward foraging extent that would be relevant to birds from this colony of little terns was set to be the generic seaward extent value derived from all of the surveys at all of the colonies (n=7), i.e. 2,176m (see Annex 5).

In summary, given the absence of site-specific information on seaward extent and of poor data for alongshore extent, the identification of the sea areas most likely to be heavily used by birds from this colony is best based on the generic extents for both these parameters (Figure 3).

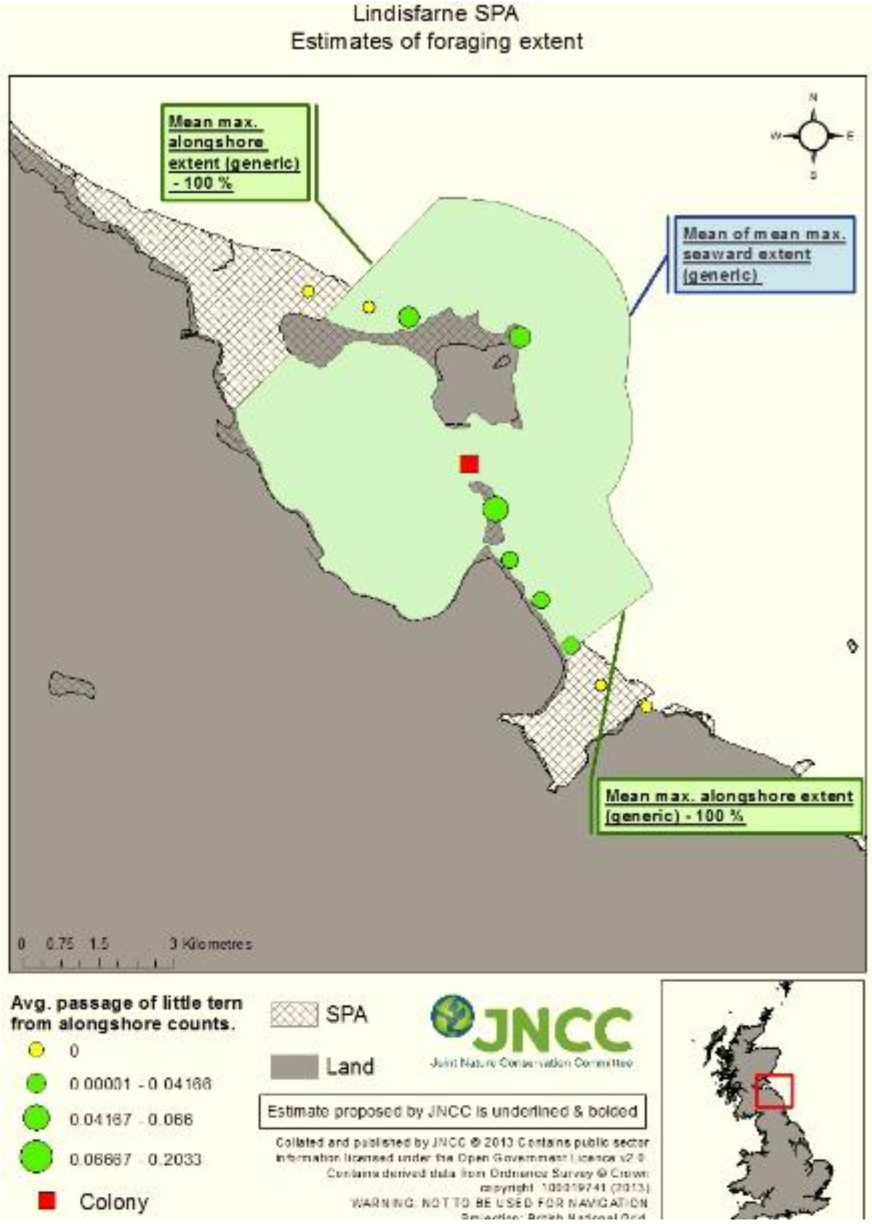


Figure 3. Application of generic alongshore and generic seaward extents to define boundaries to little tern foraging areas at the Lindisfarne SPA.

The little tern foraging area does not define the boundary of the proposed Northumberland Marine pSPA as it sits entirely within the boundary identified on the basis of the work on the larger tern species.

3.3.3. Identification of important marine areas for larger terns

The four larger species of tern which breed regularly in Great Britain have recorded mean foraging ranges between 4.5 km and 12.2 km and maximum recorded foraging ranges between 15.2 km and 49 km (Thaxter *et al.* 2012). In the light of these larger areas of interest, JNCC, in agreement with all of the SNCBs, decided that the most effective method to determine the extent of the areas most heavily used by breeding terns of the four larger species would be different to that employed for little terns. In this case the approach was to undertake a programme of boat-based visual tracking of foraging birds and the use of resultant information on foraging locations chosen by the birds in conjunction with information on the habitat characteristics of those locations relative to other areas available to the birds, to construct habitat association models of tern usage. These models were used to predict tern usage patterns across the full extent of the sea areas which the literature on foraging ranges suggested the birds may have available to them and to identify within those areas, the ones that were most heavily used and by inference, most important to the birds. Accordingly, between 2009 and 2013 JNCC coordinated a programme of visual tracking work to identify important foraging areas for larger terns at a number of UK colonies. These surveys were conducted during the chick rearing period in each year and comprised repeated days of observations of individual terns whose tracks were followed by boat as they left the colony to forage.

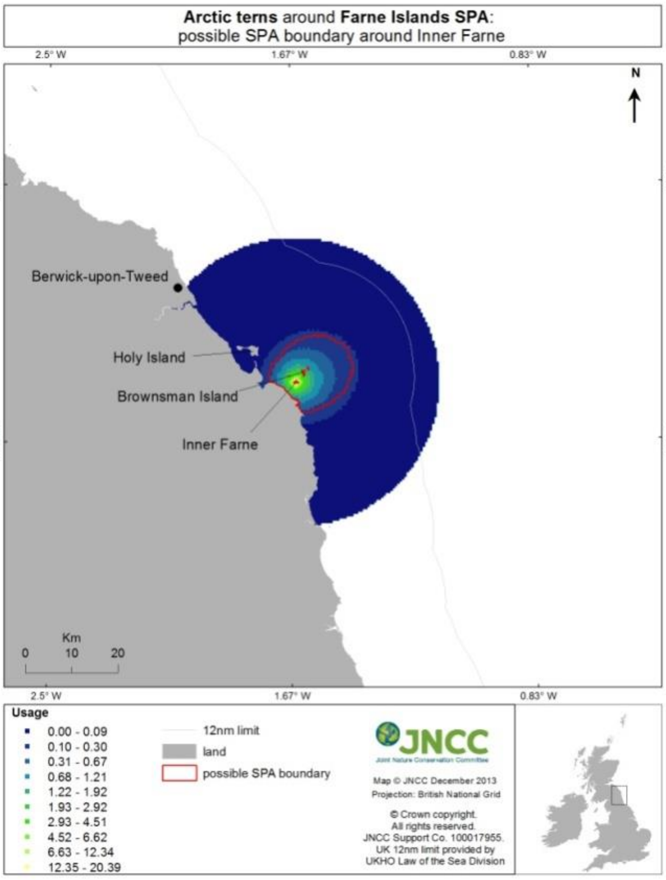
The total number of tracks obtained was 1004 including 55 tracks (6%) for roseate tern (2 SPAs), 184 tracks (18%) for Arctic tern (6 SPAs, 1 non-SPA), 381 tracks (38%) for common tern (7 SPAs, 1 non-SPA) and 384 tracks (38%) for Sandwich tern (5 SPAs, 1 non-SPA), with multiple years of data collected at five of the ten JNCC study colony SPAs. In addition, visual tracking data were obtained through a data-sharing agreement with ECON Ecological Consultancy Ltd for two SPAs: Ynys Feurig, Cemlyn Bay and The Skerries SPA (136 Sandwich, 2 common and 1 Arctic tern tracks, all collected in 2009) and North Norfolk Coast SPA (108 Sandwich and 24 common tern tracks collected 2006-2008). This gave a total of 1275 tracks available to the project, although not all data were used in the modelling; only those where foraging behaviour was recorded.

The following two sections outline in summary the survey work and boundaries identified at the three colonies which fall within the Northumberland Marine pSPA. Further general information on the programme of survey work is presented in Annex 6.

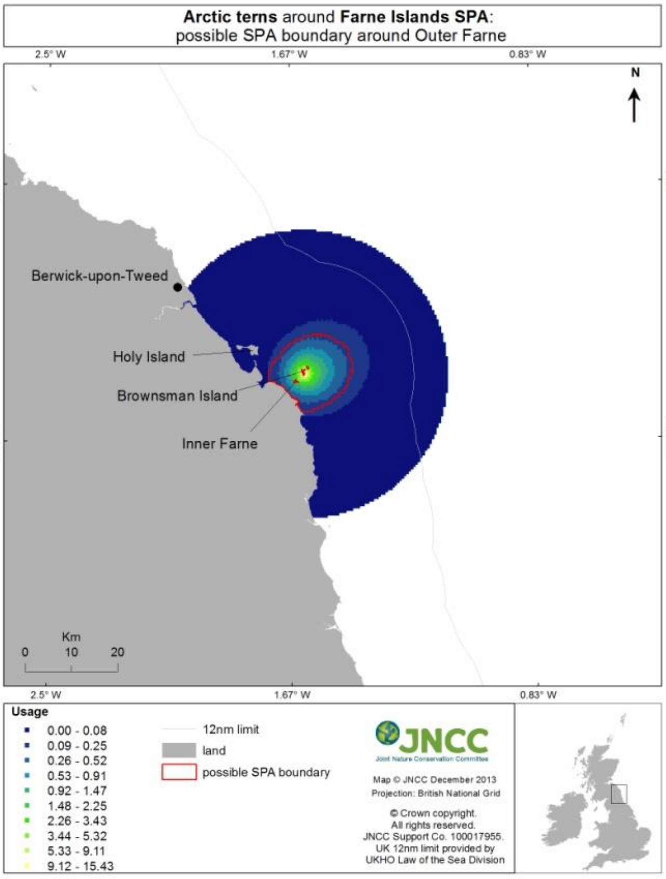
3.3.3.1. Farne Islands SPA

The species of interest here were Arctic, common and Sandwich terns. A total of 81 tern tracks (Arctic (42), common (3), Sandwich (36)) was obtained for these species over a single survey season in 2010, with two separate periods of tracking carried out timed to coincide with incubation (mid-May) and chick-rearing (mid-June). Only in the case of Arctic tern was it possible to construct a reliable model of the distribution of foraging locations based on the data collected at this site in isolation (phase 1 model). In the case of the other two species, it was necessary to make use of models derived from data collected at all of the other colonies (phase 2 model) included within the survey programme to predict areas of significant usage around the Farne Islands.

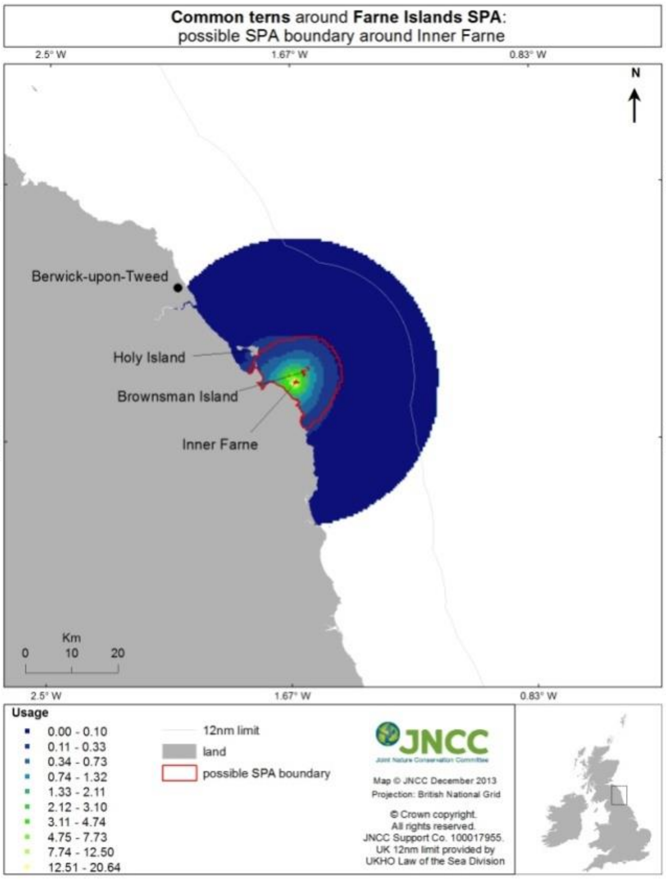
The model-generated predictions of relative usage by each tern species are shown in Figure 4. In each figure, the red line indicates the limit to the most important areas of usage, as determined by application of the maximum curvature approach to the usage data to identify a threshold level of usage (see Annex 6). In each of the images in Figure 4 the full extent of the area over which predictions of usage were made was defined by the limit of the dark blue circle shown. This reflects the constraint imposed on the modelling by use of a radius the size of the overall mean maximum distance to colony derived from tracking data held by JNCC, ECON Ecological Consultancy Ltd (for Scolt Head, Blakeney Point and Cemlyn Bay only) and Thaxter *et al.* (2012). It can be seen in every case that very substantial areas of sea within that wider area which are distant to the colony and/or distant from the shore are predicted to have very little or no usage by foraging terns.



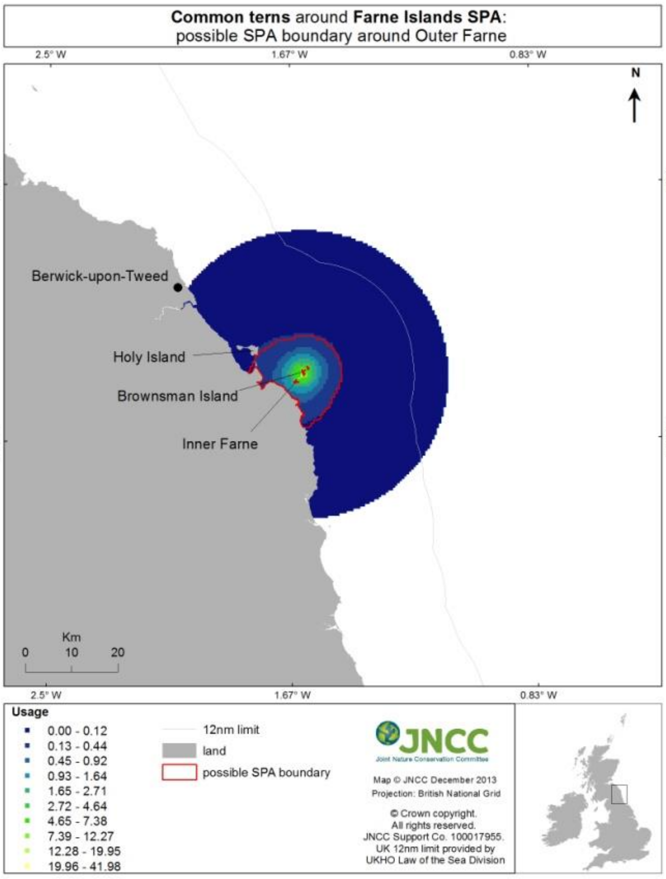
a)



b)



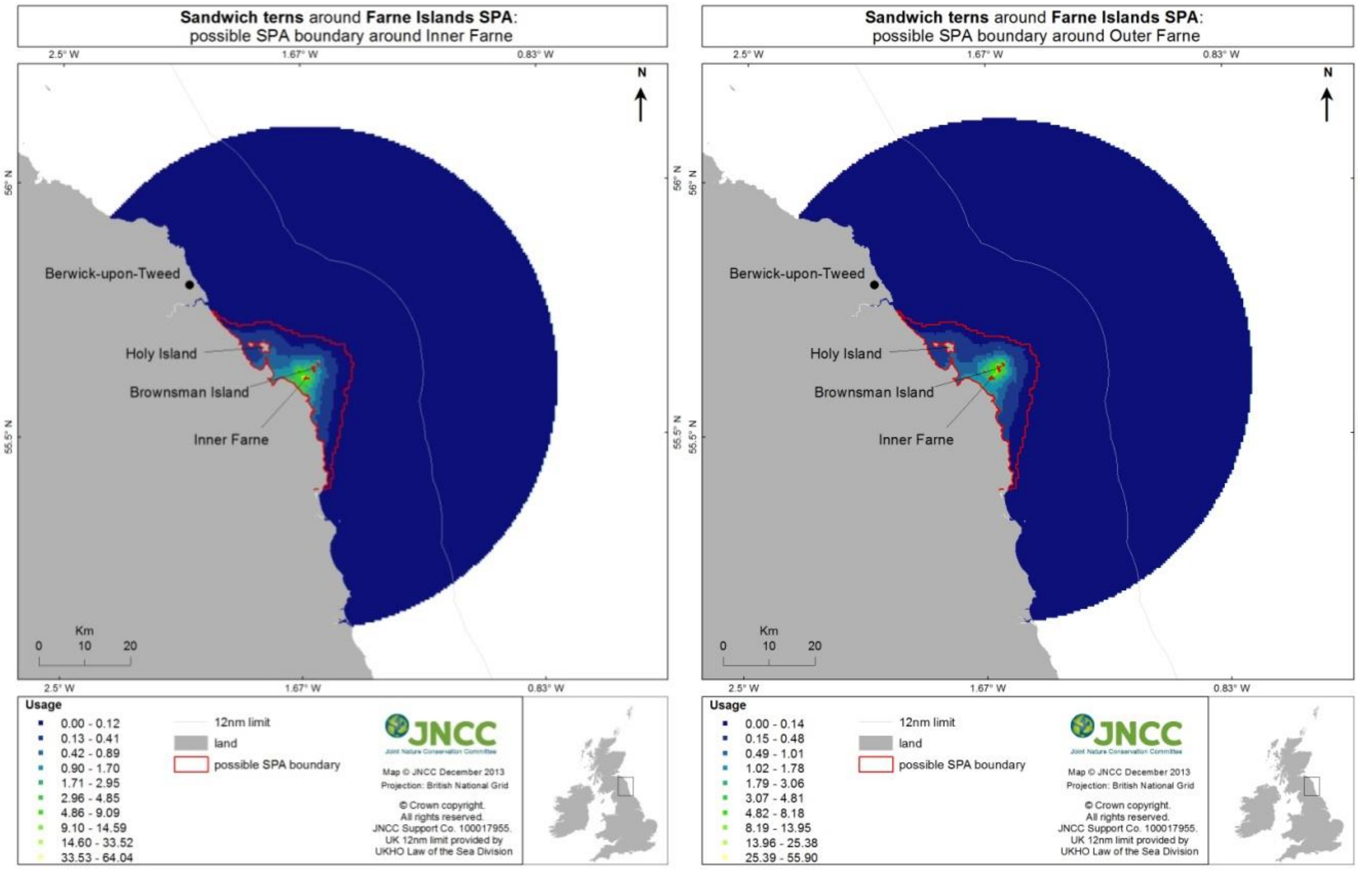
c)



d)

Figure 4. Model predictions of tern usage overlaid with maximum curvature derived limits to areas of most importance (red lines) around the Inner and Outer Farne Islands for (a, b) Arctic and (c, d) common terns. A single boundary has been produced for each species but is shown twice here as the usage predictions for Inner and Outer Farne islands differ slightly (due to different distance to colony values for the prediction

grid). Source: Win *et al.* (2013).The figure legends which refer to “possible SPA boundaries” should be disregarded.



e) f)

Figure 4 (cont). Model predictions of tern usage overlaid with maximum curvature derived limits to areas of most importance (red lines) around the Inner and Outer Farne Islands for (e, f) Sandwich terns. A single boundary has been produced but is shown twice here as the usage predictions for Inner and Outer Farne Islands differ slightly (due to different distance to colony values for the prediction grid). Source: Win *et al.* (2013). The figure legends which refer to “possible SPA boundaries” should be disregarded.

Based on models of tern usage derived from: i) site-specific survey data of Arctic terns and ii) generic survey data regarding the other two species, a composite of the modelled foraging ranges of terns from the Farne Islands SPA is shown in Figure 5, indicating a potential SPA boundary for foraging terns from this SPA alone. This defines the seaward boundary of the northern part of the Northumberland Marine pSPA.

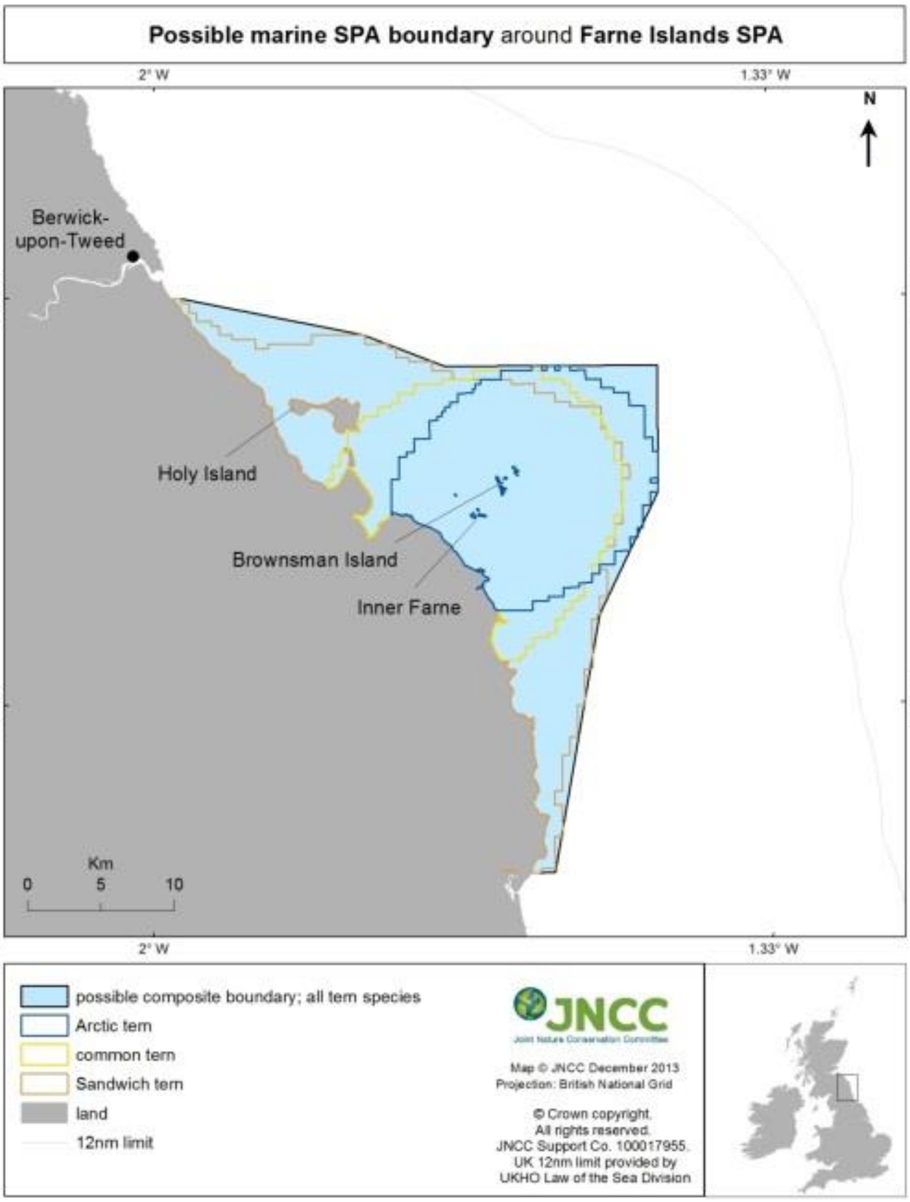


Figure 5. Proposed simple, composite boundary drawn around the species specific complex boundaries derived for each of the three larger tern species at the Farne Islands SPA. Source: Win *et al.* (2013).

3.3.3.2. Coquet Island SPA

The species of interest here were Arctic, common, roseate and Sandwich terns. A total of 374 tern tracks (Arctic (104), common (90), Roseate (53), Sandwich (127)) was obtained for these species over three survey seasons from 2009-2011. Tracking work was timed to coincide with the chick-rearing season, except in 2010 when data were also collected during the incubation period in May.

It was possible to construct reliable models of the distribution of foraging locations based on the data collected at this site in isolation (phase 1 models) for each of the four species.

The model-generated predictions of relative usage by each tern species are shown in Figure 6. In each figure, the red line indicates the limit to the most important areas of usage as determined by application of the maximum curvature approach to the usage data to identify a threshold level of usage (see Annex 6). In each of the images in Figure 6 the full extent of the area over which predictions of usage were made was defined by the limit of the dark blue circle shown. This reflects the constraint imposed on the modelling by use of a radius the size of the overall mean maximum distance to colony derived from tracking data held by JNCC, ECON Ecological Consultancy Ltd (for Scolt Head, Blakeney Point and Cemlyn Bay only) and Thaxter *et al.* (2012). It can be seen in every case that very substantial areas of sea within that wider area which are distant to the colony and/or distant from the shore are predicted to have very little or no usage by foraging terns.

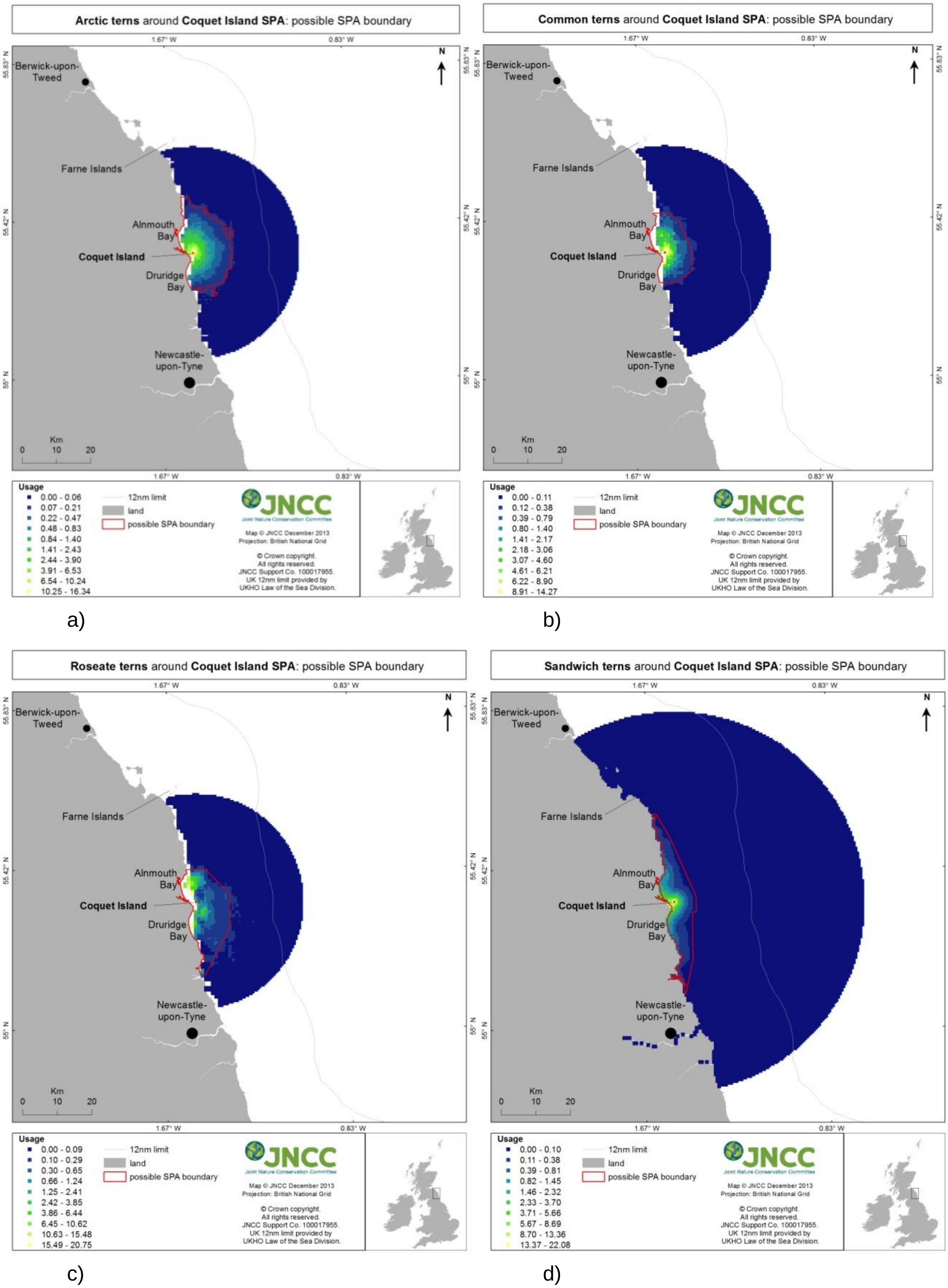


Figure 6. Model predictions of tern usage overlaid with maximum curvature derived limits to areas of most importance (red lines) around Coquet Island for a) Arctic, b) common, c) roseate and d) Sandwich terns. Source: Win *et al.* 2013. The figure legends which refer to "possible SPA boundaries" should be disregarded.

Based on models of tern usage derived from site-specific survey data of each of the four species, a composite of the modelled foraging ranges of terns from Coquet Island SPA is shown in Figure 7, indicating a potential SPA boundary for foraging terns from this SPA alone. This defines the seaward boundary of the southern part and much of the central part of the Northumberland Marine pSPA.

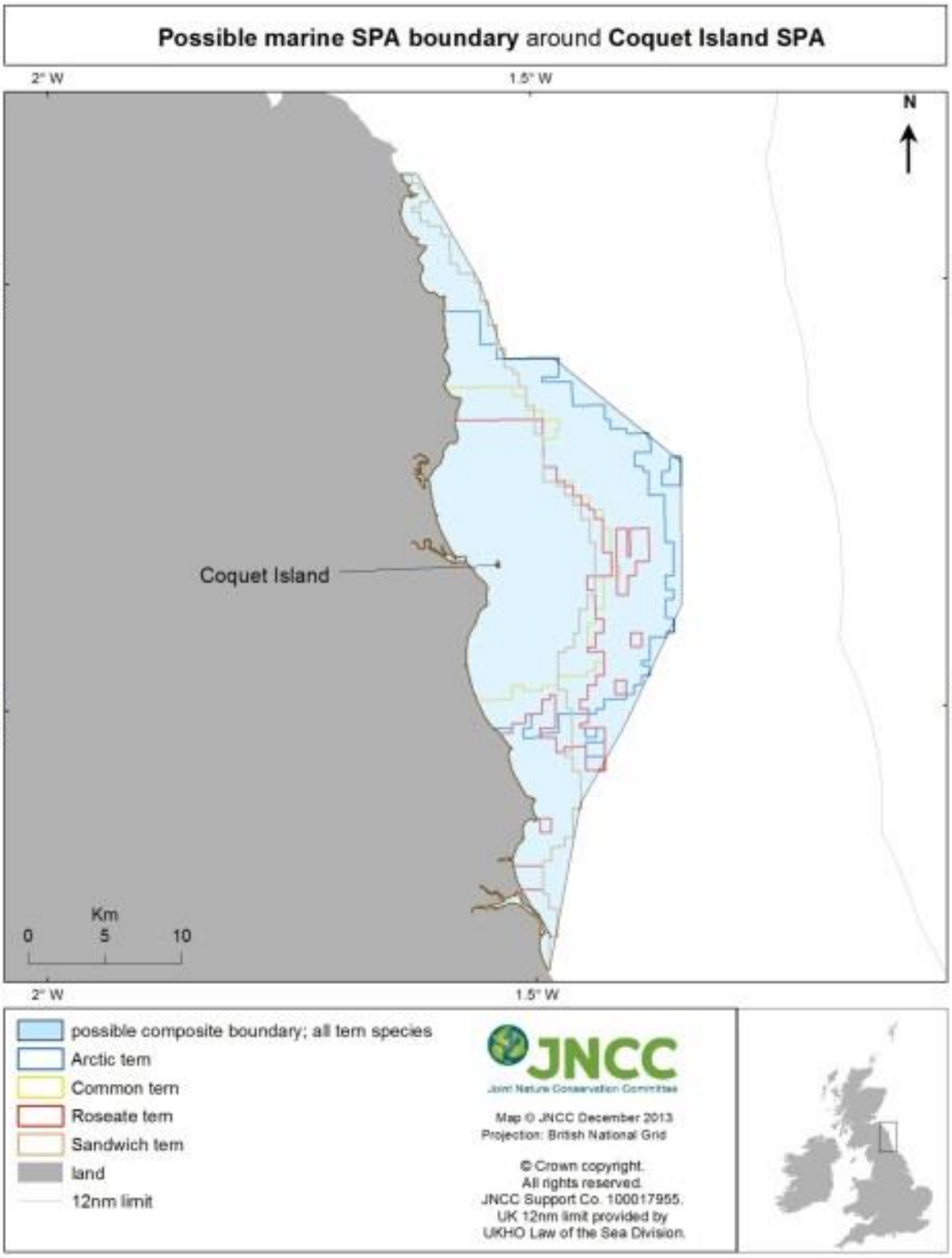


Figure 7. Proposed simple, composite boundary drawn around the species specific complex boundaries derived for each of the four larger tern species at Coquet Island SPA. Source: Win *et al.* (2013).

3.3.3.3. Northumbria Coast SPA (Newton Links/Long Nanny)

At the time at which the larger tern survey programme was devised by JNCC, Arctic tern were not a qualifying feature of the Northumbria Coast SPA. Accordingly, no site specific survey work on Arctic terns was conducted at this site. However, in the light of increases in the size of this colony in recent years, Natural England is recommending the inclusion of Arctic tern as a qualifying feature of the Northumbria Coast pSPA (Northumbria Coast pSPA departmental brief *unpublished*,). In the absence of site-specific survey data in this instance, it was necessary to make use of models derived from data collected at all of the other colonies included within the survey programme (phase 2 model) to predict areas of significant usage by Arctic terns originating from the Northumbria Coast SPA (Figure 8). This indicates a potential SPA boundary for foraging terns from this SPA alone. This defines the seaward boundary of the central part of the Northumberland Marine pSPA.

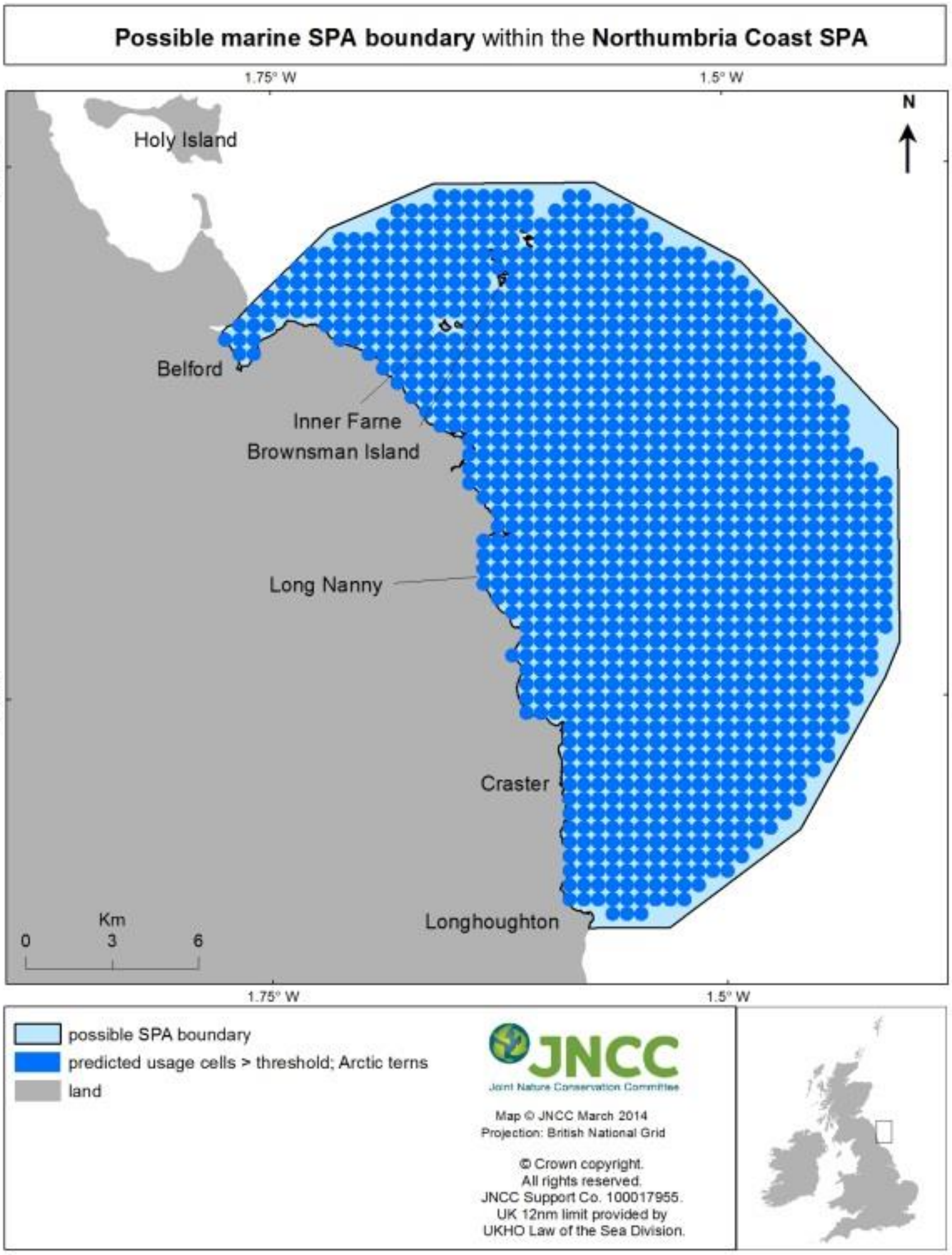


Figure 8. Proposed simple boundary drawn around the cells within which predicted Arctic tern usage levels centred on the Long Nanny colony exceeded the threshold level identified by application of the maximum curvature methodology to the predicted usage surface (see Annex 6).

3.3.4. Composite boundary of Northumberland Marine pSPA

The seaward extent of the pSPA is determined by, from north to south, the modelled foraging distributions of: Sandwich tern and Arctic terns at the Farne Islands SPA; Arctic terns at Northumbria Coast SPA and Arctic, roseate and Sandwich terns at Coquet Island SPA (Table 2). The overlapping nature of these foraging ranges is also evident, meaning that some sea areas will be supporting birds of the same species from different SPAs. The generic seaward maintenance areas around both Coquet Island SPA and the Farne Islands SPA are contained entirely within the composite boundary of the pSPA as too are the extents of the little tern foraging areas around the Lindisfarne SPA and Northumbria Coast SPA.

Table 2. The type of model (site specific (phase 1) or generic (phase 2)) which were applied to each of the larger tern species at each source colony. Those models which generated predicted usage maps that

contributed to the composite seaward boundary of the Northumberland Marine pSPA are highlighted in bold.

Species	Colony	model type
Arctic tern	**Farne Islands**	**phase 1**
Arctic tern	**Northumbria Coast**	**phase 2**
Arctic tern	**Coquet Island**	**phase 1**
Common tern	Farne Islands	phase 2
Common tern	Coquet island	phase 1
Roseate tern	**Coquet island**	**phase 1**
Sandwich tern	**Farne Islands**	**phase 2**
Sandwich tern	**Coquet Island**	**phase 1**

4. Location and habitats

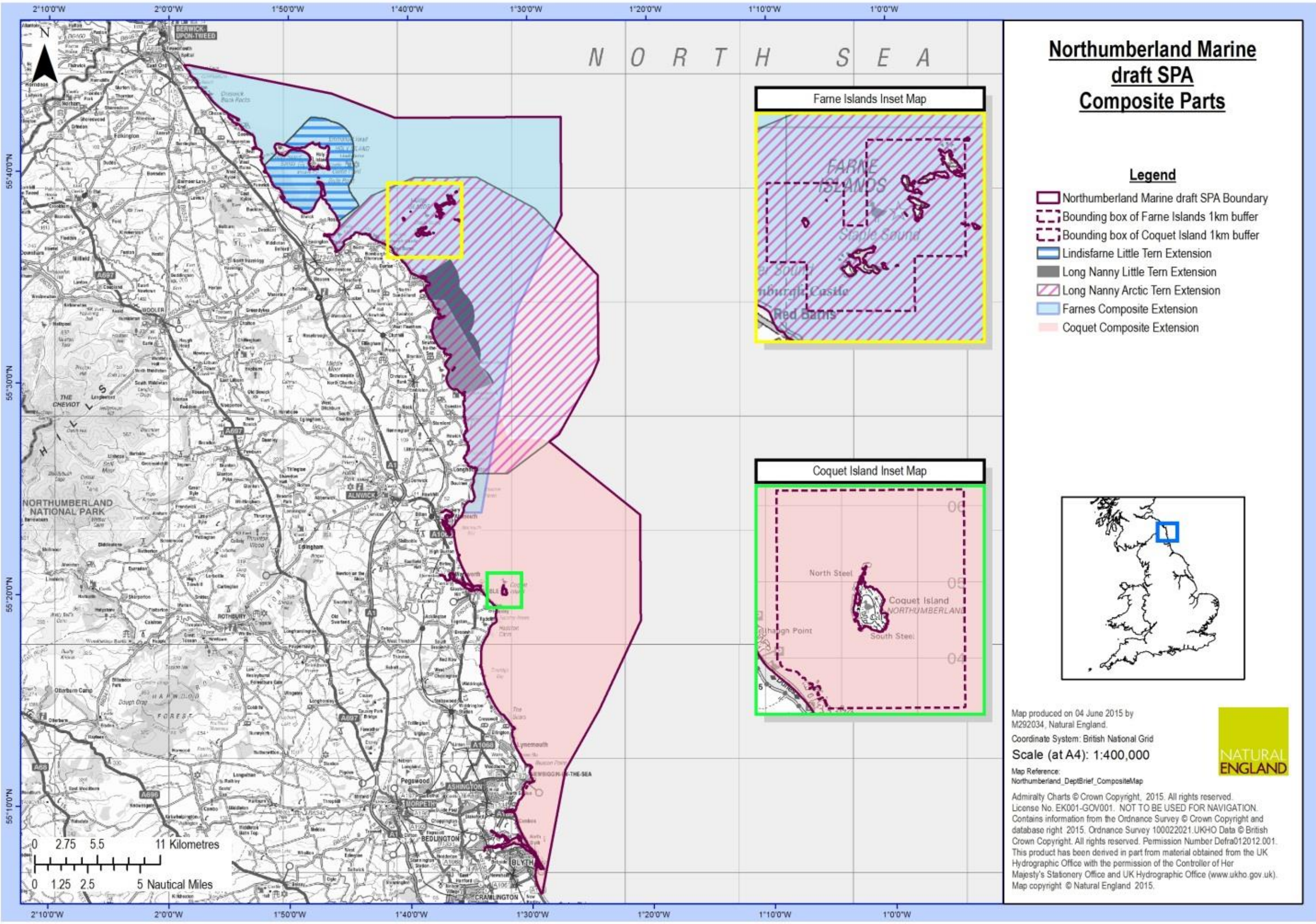


Figure 9. Summary map of Northumberland Marine proposed SPA

The Northumberland Marine pSPA is located on the Northumberland coast between the Scottish border and the Tyne Estuary.

The Northumberland coast includes a range of coastal habitats including rocky shores, intertidal sandflats and mudflats (such as those at Budle Bay and Lindisfarne), and an extensive stretch of mobile sand dune systems.

The site supports a wide range of marine habitats. It has high habitat diversity and a large number of marine invertebrate species with a northern distribution. In the intertidal zone, the varied geology and topography supports contrasting foreshore communities on the coal and shale seams, limestones and sandstones.

Extensive intertidal sand and mud flats occur, particularly in the northern part of the site: Fenham Flats, Ross Sands, Budle Bay and the coast adjacent to the north of Holy Island form the most extensive area of intertidal sand and mud in North East England. These extensive areas support one of the largest intertidal beds of the narrow leaved seagrass *Zostera angustifolia* and dwarf seagrass *Zostera noltii* on the east coast of the UK. The intertidal zone provides important habitats for a wide range of wintering waterbirds.

The Northumberland Marine pSPA surrounds two important offshore breeding locations for breeding seabirds; the Farne Islands SPA and Coquet Island SPA. The Farne Islands SPA consists of a group of rocky offshore islands and stacks lying between 1.5 - 4 miles off the Northumberland Coast at Seahouses. The islands are formed of the resistant quartz dolerite, the most easterly outcropping of the Great Whin Sill which on a number of islands retains a capping of boulder clay and peaty soils. Coquet Island SPA is located a mile offshore from Amble on the Northumberland Coast and consists of a flat, grassy platform and sandstone cliffs. The plateau is covered by a peaty soil supporting fescue grassland, sea campion *Silene maritima* and thrift *Armeria maritima*.

The subtidal environment across the site includes a wide range of habitats. Between the shore and 40 m depth there are extensive rocky exposures interspersed with sandy bays of clean, fine sand. Extensive rocky reefs occur, notably around the Farne Islands and together with the wide variety of associated intertidal reefs, these are the most biodiverse known on the North Sea coast. The majority of the coast is at least moderately exposed to wave action. Offshore at about 30 m depth, the seabed becomes predominantly sedimentary and further offshore, at depths beyond 60 m, the sediments generally consist of very fine sand with varying proportions of silt.

The proposed pSPA boundary overlaps and is adjacent to various other sites which have been notified or designated under either or both British and European conservation legislation. The intertidal elements of the pSPA are largely underpinned by the Northumberland Shore SSSI that runs intermittently from the border between Scotland and England to Durham, and by the Lindisfarne SSSI. The boundary includes large bays such as Embleton, Beadnell and Budle, and extends into rivers and estuaries, such as the Coquet, Aln and Blyth. Parts of these rivers and estuaries are also SSSI, and the Aln Estuary is a designated Marine Conservation Zone (MCZ).

The Berwickshire and North Northumberland Coast SAC also overlaps the boundary in the northern half of the pSPA. The SAC is designated for a wide range of intertidal and subtidal habitats, including large shallow inlets and bays, intertidal mudflats and sandflats and rocky reefs, as well as grey seals *Halichoerus grypus*. The SAC also has some of the best example of clean sand intertidal sandbanks and is an important area of dwarf seagrass *Zostera noltii*.

The southern portion of the draft boundary overlaps with the northern part of the Coquet to St Mary’s proposed Marine Conservation Zone (pMCZ), subject to a public consultation by Defra between January and April 2015. The pMCZ is being primarily being recommended for both inter- and subtidal rocky habitats but also includes significant areas of soft sediment habitats.

5. Assessment of ornithological interest

5.1. Survey information and summary

Details of the site-specific survey work carried out to characterise the foraging areas used by breeding adult terns originating from the existing SPAs (Lindisfarne, Farne Islands, Coquet Island and Northumbria Coast) are detailed in Annexes 5 and 6. The purpose of those surveys was to provide a robust empirical dataset on which to base predictive models of habitat usage around these and other tern colonies, and not to derive estimates of the number of individuals of any species at sea within the boundaries of the pSPA at any point in time. Accordingly, the numbers detailed in Table 1 and in the following sections represent the numbers of breeding individuals of each species which will make use of the resources provided by the supporting habitats within the pSPA during the course of the breeding season rather than the numbers that will be present within the site boundary at any particular point in time.

The counts of the breeding seabirds at the colonies within existing SPAs which are those most likely to be the origin of birds within the pSPA have been provided by the national Seabird Monitoring Programme (SMP). In addition, this dataset has been augmented by information requested directly from colony managers for roseate tern and little tern. These data, which provide once-yearly estimates of the populations of each breeding seabird species at each of the source colonies, have been averaged across the 5 most recently available years (2010-02014 unless otherwise stated in the following sections) to yield a 5 year mean for each species at each colony. These same count data have also been summed across source colonies in each year and averaged across the 5 most recently available years (2010-2014 unless otherwise stated) to give an overall 5 year mean for the pSPA.

5.2. Annex 1 species

5.2.1. Common tern *Sterna hirundo*

The breeding population of common terns in Great Britain is estimated to be 10,000 pairs (Musgrove *et al.* 2013), representing at least 2% of the Northern & Eastern European breeding population (50,000 pairs derived by division by 3 of the upper estimate of 1,500,000 individuals: AEWA 2012). A significant proportion of the British population breeds in Scotland. Coastal colonies in England are concentrated in the north-east, East Anglia, at a few localities along the south coast, and in the north-west (Mitchell *et al.* 2004).

Common terns breed not only around coasts but, unlike the other tern species which breed in the UK, also breed frequently beside inland freshwater bodies.

5.2.1.1. Lindisfarne SPA

Common tern is not a qualifying feature of this SPA. The last available count data are from 2009 (20 occupied nests or 40 individuals). No count data are available for this species at this site since 2010. This population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging range from Lindisfarne SPA is not considered in defining the boundary of the pSPA.

5.2.1.2. Farne Island SPA

Common tern is a qualifying feature of this SPA. The citation (1983 revision) lists 183 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 230 pairs as a 5-year mean (1993-1997) at the time representing 1.9% of the GB breeding population. Between 2010 and 2014 the Farne Islands SPA supported an average of 97 breeding pairs of common tern (counted as occupied nests) representing 194 breeding adults (1 occupied nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 0.97% of the GB breeding population.

5.2.1.3. Coquet Island SPA

Common tern is a qualifying feature of this SPA. The citation (1983 revision) lists 1,100 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 740 pairs as a 5-year mean (1993-1997) at the time representing 6% of the GB breeding population. Between 2010 and 2014 Coquet Island SPA supported an average of 1,189 breeding pairs of common tern (counted as occupied nests) representing 2,378 breeding adults (1 occupied nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 11.89% of the GB breeding population.

5.2.1.4. Long Nanny at Northumbria Coast SPA

Common tern is not a qualifying feature of this SPA. The species does not currently breed within this SPA.

5.2.1.5. Northumberland Marine pSPA

The principal common tern breeding colonies supported by the Northumberland Marine pSPA during the breeding season are located at both the Farne Islands and Coquet Island. The combined individual common tern figures from these colonies and supported by the pSPA (averaged between 2010 and 2014) is 2,572 breeding adults which constitutes 12.86% of the GB breeding population. Historically, there have also been small numbers of common tern breeding at Lindisfarne. Given the lack of recent population estimates and the fact that common tern is not a qualifying feature of the Lindisfarne SPA no contribution to the numbers of common tern supported by the Northumberland Marine pSPA has been included from Lindisfarne SPA. Modelling of common tern foraging locations indicates that the most heavily used foraging areas for common terns from the Farne Islands stretch north to the south shore of the Holy Island of Lindisfarne and south to Beadnell Bay, as well as offshore. The modelling also indicates that the most heavily used foraging areas for common tern from the colony at Coquet Island extend north to Alnmouth and south to the southern end of Druridge Bay, as well as offshore.

5.2.2. Arctic tern *Sterna paradisaea*

The breeding population of Arctic terns in Great Britain is estimated to be 53,000 pairs (Musgrove *et al*. 2013), representing at least 2.9% of the European & North Atlantic breeding population (1,800,000 pairs being the maximum estimate given in Mitchell *et al*. (2004): AEWA (2012) only give an estimate of in excess of 1,000,000 individuals for the Western Eurasian breeding population – from which a % value cannot be derived). Arctic terns have a strongly northerly distribution in the UK, with the breeding population concentrated on Shetland, Orkney and north and west Scotland (Mitchell *et al.* 2004). Apart from three large colonies in Northumberland (Coquet Island, the Farne Islands and Newton Links/Long Nanny), they are a rare breeding bird in England.

5.2.2.1. Lindisfarne SPA

Arctic tern is not a qualifying feature of this SPA. The last available count data are from 2011 (70 occupied nests or 140 individuals). No count data are available for this species at this site since 2012. This population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging range from Lindisfarne SPA is not considered in defining the boundary of the pSPA.

5.2.2.2. Farne Islands SPA

Arctic tern is a qualifying feature of this SPA. The citation (1983 revision) lists 4,000 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 2,840 pairs as a 5-year mean (1993-1997) at the time representing 6.5% of the GB breeding population. Between 2010 and 2014 the Farne Islands SPA supported an average of 2,003 breeding pairs of Arctic tern (counted as occupied nests) representing 4,006 breeding adults (1 occupied nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 3.78% of the GB breeding population.

5.2.2.3. Coquet Island SPA

Arctic tern is a qualifying feature of this SPA. The citation (1983 revision) lists 700 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 700 pairs as a 4-year mean (1993 & 1995-1997) at the time representing 1.6% of the GB breeding population. Between 2010 and 2014 Coquet Island SPA supported an average of 1,230 breeding pairs of Arctic tern (counted as occupied nests) representing 2,460 breeding adults (1 occupied nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 2.32% of the GB breeding population.

5.2.2.4. Long Nanny at Northumbria Coast SPA

Arctic tern is not currently a qualifying species of the Northumbria Coast SPA. However, between 2010 and 2014 Long Nanny at Northumbria Coast SPA supported an average of 1,549 breeding pairs of Arctic tern (counted as occupied territories or nests) representing 3,098 breeding adults (1 nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 2.92% of the GB breeding population. Accordingly, Natural England will in due course prepare a departmental brief recommending its addition as a new qualifying feature at this site.

5.2.2.5. Northumberland Marine pSPA

The principal Arctic tern breeding colonies supported by the Northumberland Marine pSPA during the breeding season are located at the Farne Islands, Coquet Island and at Long Nanny on the Northumbria Coast. The combined individual Arctic tern figures from these colonies and supported by the pSPA (averaged between 2010 and 2014) is 9,564 breeding adults which constitutes 9.02% of the GB breeding population. Arctic tern are also known to breed at Lindisfarne, although there have been no counts of this species at Lindisfarne since 2011. Furthermore, Arctic tern is not a qualifying feature of the Lindisfarne SPA. Accordingly, no contribution to the numbers of Arctic tern supported by the Northumberland Marine pSPA has been included from Lindisfarne SPA. Modelling of Arctic tern foraging locations indicates that the most heavily used foraging areas for Arctic terns from all three colonies are broadly circular in shape, centred on each of the colonies and stretching roughly equal distances in all directions, including out to sea.

5.2.3. Roseate tern *Sterna dougallii*

The breeding population of roseate terns in Great Britain is estimated to be 86 pairs (Musgrove *et al*. 2013), representing at least 4.5% of the European breeding population (1,900 pairs derived by division by 3 of the upper estimate of 5,700 individuals: AEWA 2012). The roseate tern is the UK’s rarest regularly-breeding seabird, and it is restricted to a very small number of colonies (Mitchell *et al.* 2004). The Seabird 2000 census recorded 56 apparently occupied nests in the UK, with 36 of these being in England (Mitchell *et al.* 2004). However, the population has increased since then: a recent maximum of 92 pairs bred in England in 2009, of those 90 pairs were on Coquet Island. Elsewhere only single pairs breed in the UK and then only sporadically.

5.2.3.1. Lindisfarne SPA

Roseate tern is a qualifying feature of this SPA. The citation (dated 1990) lists 4 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states >0 pairs as a count in the late 1990s but does not provide an estimate of the % of the GB breeding population. The last known record of breeding here was in

1991. Given its complete absence over the last 25 years or more, this population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging range from Lindisfarne SPA is not considered in defining the boundary of the pSPA.

5.2.3.2. Farne Islands SPA

Roseate tern is a qualifying feature of this SPA. The citation (1983 revision) lists 13 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) gives no figure for this species. This species has not bred on the Farne Islands since 2009. Given its complete absence over at least the last 6 years, this population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging range from Farne Islands SPA is not considered in defining the boundary of the pSPA.

5.2.3.3. Coquet Island SPA

Roseate tern is a qualifying feature of this SPA. The citation (1983 revision) lists 29 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 31 pairs as a 5-year mean (1993-1997) at the time representing 48.4% of the GB breeding population. Between 2010 and 2014 Coquet Island SPA supported an average of 80 breeding pairs of roseate tern (counted as breeding pairs) representing 160 breeding adults. This represents 93.02% of the GB breeding population.

5.2.3.4. Long Nanny at Northumbria Coast SPA

Roseate tern is not a qualifying feature of this SPA. The species does not currently breed within this SPA.

5.2.3.5. Northumberland Marine pSPA

The only extant roseate tern breeding colony supported by the Northumberland Marine pSPA during the breeding season is located at Coquet Island SPA. Accordingly, the roseate tern breeding population supported by the pSPA (averaged between 2010 and 2014) is the 160 breeding adults on Coquet Island which constitutes 93.02% of the GB breeding population. Modelling of roseate tern foraging locations indicates that the most heavily used foraging areas for roseate terns from Coquet Island stretch from Boulmer in the north to Cambois in the south, as well as offshore.

5.2.4. Sandwich tern *Sterna sandvicensis*

The breeding population of Sandwich terns in Great Britain is estimated to be 11,000 pairs (Musgrove et al. 2013), representing about 19.3% of the Western Europe/West Africa breeding population (57,000 pairs derived by division by 3 of the upper estimate of 171,000 individuals: AEWA 2012). In the UK, the species is restricted to relatively few large colonies, most of which are on the east coast of Britain with a few smaller ones on the south and north-west coasts of England and in Northern Ireland. Colonies are mostly confined to coastal shingle beaches, sand dunes and offshore islets (Mitchell *et al.* 2004).

5.2.4.1. Lindisfarne SPA

Sandwich tern is not a qualifying feature of this SPA. There are no records of breeding at this site since at least 2008.

5.2.4.2. Farne Island SPA

Sandwich tern is a qualifying feature of this SPA. The citation (1983 revision) lists 4,000 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 2,070 pairs as a 5-year mean (1993-1997) at the time representing 14.8% of the GB breeding population. Between 2010 and 2014 the Farne Islands SPA supported an average of 862 breeding pairs of Sandwich tern (counted as occupied nests) representing 1,724 breeding adults (1 occupied nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 7.84% of the GB breeding population.

5.2.4.3. Coquet Island SPA

Sandwich tern is a qualifying feature of this SPA. The citation (1983 revision) lists 1,500 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 1,590 pairs as a 5-year mean (1993-1997) at the time representing 11.4% of the GB breeding population. Between 2010 and 2014 Coquet Island SPA supported an average of 1,300 breeding pairs of Sandwich tern (counted as occupied nests) representing 2,600 breeding adults (1 occupied nest = 1 pair *i.e*. no correction factor applied to counts of nests). This represents 11.82% of the GB breeding population.

5.2.4.4. Long Nanny at Northumbria Coast SPA

Sandwich tern is not a qualifying feature of this SPA. The species does not currently breed within this SPA.

5.2.4.5. Northumberland Marine pSPA

The only Sandwich tern breeding colonies supported by the Northumberland Marine pSPA during the breeding season are located at the Farne Islands and Coquet Island. The combined individual Sandwich tern figures from these colonies and supported by the pSPA (averaged between 2010 and 2014) is 4,324 breeding adults which constitutes 19.66% of the GB breeding population. Modelling of Sandwich tern foraging locations indicates that the most heavily used foraging areas for Sandwich terns extend less far out to sea but further along the coast than those of other large tern species. The most heavily used foraging areas for Sandwich terns from the Farne Islands stretch as far north as Scremerston and south beyond Boulmer, whereas the areas used by Sandwich terns from Coquet Island stretch north to Seahouses (overlapping with birds from the Farnes) and south into the Blyth Estuary and Blyth Bay.

5.2.5. Little tern *Sternula albifrons*

The breeding population of little terns in Great Britain is estimated to be 1,900 pairs (Musgrove *et al*. 2013), representing about 10.3% of the Eastern Atlantic breeding population (18,500 pairs derived by division by 3 of the upper estimate of 55,500 individuals: AEWA 2012). Breeding occurs in scattered colonies along much of the east and west coasts of Britain, from the north of Scotland to (and including) the south coast of England (Mitchell *et al.* 2004). The greater part of the population occurs in south and east England from Dorset to Norfolk (Mitchell *et al.* 2004). All British little terns nest on the coast, utilising sand and shingle beaches and spits, as well as tiny islets of sand or rock close inshore (Mitchell *et al.* 2004).

5.2.5.1. Lindisfarne SPA

Little tern is a qualifying feature of this SPA. The citation (dated 1990) lists 42 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 15 pairs as the 5-year mean (1992-1996) at the time representing 0.6% of the GB breeding population. Between 2010 and 2014 the Lindisfarne SPA supported an average of 14 breeding pairs of little tern (counted as pairs) representing 28 breeding adults. This represents 0.73% of the GB breeding population.

5.2.5.2. Northumbria Coast SPA (Newton Links/Long Nanny)

Little tern is a qualifying feature of this SPA. The citation (compilation date 2000) lists 40 pairs. The Natura 2000 Standard Data Form (JNCC, updated 1999) states 40 pairs as the 5-year mean (1992/3-1996/97) at the time representing 1.7% of the GB breeding population. Between 2010 and 2014 the Northumbria Coast SPA supported an average of 31 breeding pairs of little tern (counted as pairs) representing 62 breeding adults. This represents 1.64% of the GB breeding population.

5.2.5.3. Northumberland Marine pSPA

The only little tern breeding colonies supported by the Northumberland Marine pSPA are located at Lindisfarne and at Newton Links/Long Nanny on the Northumbria Coast. The combined individual little tern figures from these colonies and supported by the pSPA (averaged between 2010 and 2014) is 90 breeding adults which constitutes 2.37% of the GB breeding population. Observations of the little terns at Long Nanny indicate that the principal foraging areas lie between Shoreston in the north and the southern limit of Embleton Bay in the south. Application of the generic alongshore limits to the foraging areas of little terns at Lindisfarne indicate that the principal foraging areas around this colony extend from the westernmost point of Holy Island to the mouth of Budle Bay.

5.3. Regularly occurring migratory species

5.3.1. Puffin *Fratercula arctica*

Mitchell *et al.* (2004) note that Stroud *et al.* (2001) use the total population of the subspecies *F.a.grabae* as the relevant biogeographic population to which puffins in the UK belong. However, they also note that this subspecies is now considered to be indistinct from the nominate subspecies *F.a.arctica* and present a total population for *F.a.arctica* which includes populations of both of these previously recognised subspecies. UK SPA and Ramsar Scientific Working Group (2014) presents a figure of 5,176,257 pairs for this population which is derived in line with the figures in Mitchell *et al.* (2004) by summing birds breeding in: France, GB, Isle of Man and Channel Islands, All-Ireland, Faroes, all of Norway, Iceland and Russia (although excluding birds listed as *F.a.arctica* in Mitchell *et al*. (2004) and breeding in Canada, USA and Greenland). On the basis of this figure, the breeding population of puffin in Great Britain, which is estimated to be 580,000 pairs (1,160,000 breeding individuals) (Musgrove *et al.* 2013), represents about 11.2% of the north east Atlantic biogeographic population of the subspecies *F. arctica arctica*. The majority of UK birds nest in north and west Scotland (Mitchell *et al*. 2004). In England, puffins breed only at a few colonies on coasts and islands, with the only significant colonies being in the north-east (Mitchell *et al*. 2004).

5.3.1.1. Farne Islands pSPA

The largest colony in England is on the Farne Islands. The SPA citation (revision dated 1983) states 14,000 pairs. Over the last two censuses (conducted in 2008 and 2013) the Farne Islands supported an average of 38,399 pairs of puffin (counted as "occupied burrows") representing 76,798 individuals (1 occupied burrow = 1 pair *i.e*. no correction factor typically applied to auk counts are applied to counts of burrows). This constitutes 0.74% of the north east Atlantic biogeographic population (UK SPA and Ramsar Scientific Working Group 2014). This also constitutes 6.62% of the GB breeding population. As the Farne Islands support a nationally important population of puffin (over 1% of GB), it is proposed that puffin is identified as a main component of a qualifying assemblage of over 20,000 breeding seabirds in the Farne Islands pSPA.

5.3.1.2. Coquet Island pSPA

The SPA citation (revision dated 1983) makes no mention of puffins. However, Coquet Island now holds the second largest colony in England. During the last three censuses (conducted in 2008, 2009 and 2013) Coquet Island supported an average of 15,843 pairs of puffin (counted as "occupied burrows") representing 31,686 individuals (1 burrow =1 pair *i.e*. no correction factor typically applied to auk counts are applied to counts of burrows). This constitutes 0.31% of the north east Atlantic biogeographic population (UK SPA and Ramsar Scientific Working Group 2014). This also constitutes 2.73% of the GB breeding population. As Coquet Island supports a nationally important population of puffin (over 1% of GB), it is proposed that puffin is identified as a main component of a qualifying assemblage of over 20,000 breeding seabirds in the Coquet Island pSPA .

5.3.1.3. Northumberland Marine pSPA

The pSPA supports the maintenance behaviour requirements of the puffin breeding colonies on the Farne Islands and Coquet Island. The combined means of the populations at these two colonies (each averaged over the censuses within 5 years of the most recent one in 2013) and supported by the pSPA is 54,242 pairs (108,484 breeding adults), which constitutes 1.05% of the north east Atlantic biogeographic population (UK SPA and Ramsar Scientific Working Group 2014). As the Northumberland Marine pSPA supports an internationally important population of puffin (over 1% of biogeographic population) it is proposed that puffin is identified as a qualifying feature of the pSPA.

5.3.2. Guillemot *Uria aalge*

The breeding population of guillemots in Great Britain is estimated to be 880,000 pairs (1,313,433 individuals) (Musgrove *et al*. 2013), representing about 31% of the North Atlantic population (Mitchell *et al*. 2004). Breeding colonies are distributed widely around the coast of Britain, with the exception of the southeast of England from Sussex to Lincolnshire. The largest colonies of over 10,000 individuals are found from Northumberland north up the east coast of mainland Scotland to Shetland and the Hebrides, with outliers on Flamborough Head in Yorkshire and Skomer Island off Pembrokeshire (Mitchell *et al*. 2004). Nesting is confined to areas safe from mammalian predators such as sheer cliffs and offshore islands. It is considered that two distinct subspecies occur within the British Isles, with those breeding in Scotland and Northumberland belonging to the nominate race *U.a.aalge* while those in the rest of England, Wales and all of Ireland belong to the southern race *U.a.albionis* (UK SPA and Ramsar Scientific Working Group 2014). The population size of the nominate *U.a.aalge* in the northern UK has recently been estimated to be 591,017 pairs which represents 30.8% of the north-east Atlantic population of this subspecies (1,917,167 pairs : UK SPA and Ramsar Scientific Working Group (2014) (excluding birds breeding in eastern North America and Greenland due to the lack of any evidence of trans-Atlantic movements).

5.3.2.1. Farne Islands pSPA

The SPA citation (revision dated 1983) states 6,000 pairs. Between 2010 and 2014 the Farne Islands supported an average of 49,068 guillemots (counted as “individuals on land”) representing 32,875 pairs (correction factor of 0.67 applied to counts of individuals, Harris 1989; Musgrove *et al*. 2013) which equates to 65,751 breeding adults. This constitutes 1.72% of the North East Atlantic biogeographic population of the nominate race *U.a.aalge* (UK SPA and Ramsar Scientific Working Group 2014). As the Farne Islands support an internationally important population of guillemot (over 1% of biogeographic population), it is proposed that guillemot is identified as a qualifying feature of the Farne Islands pSPA.

5.3.2.2. Northumberland Marine pSPA

The pSPA supports the maintenance behaviour requirements of the guillemot breeding colony on the Farne Islands. The guillemot figures from this colony between 2010 and 2014 average 32,875 pairs or 65,751 breeding adults (correction factor of 0.67 applied to counts of individuals, Harris 1989; Musgrove *et al*. 2013) This constitutes 1.72% of the North East Atlantic biogeographic population of the nominate race *U.a.aalge* (UK SPA and Ramsar Scientific Working Group 2014). As the Northumberland Marine pSPA supports an internationally important population of guillemot (over 1% of biogeographic population) it is proposed that guillemot is identified as a qualifying feature of the pSPA.

5.3.3. Breeding seabird assemblages

5.3.3.1. Farne Islands pSPA

Adding up the most recent population estimates for each species (i.e. the 5 year mean population figure from 2010-2014 for all species except puffin for which the average of censuses in 2008 and 2013 are used) yields a total of 163,819 individual breeding seabirds supported by the Farne Islands pSPA. This qualifies as an internationally important assemblage of over 20,000 seabirds. In the context of SPA qualification, this seabird assemblage includes: Arctic tern, common tern, roseate tern, Sandwich tern and guillemot, i.e. the species described above which qualify as features in their own right. In addition, the breeding seabird assemblage includes puffin (76,798 breeding adults), great cormorant *Phalacrocorax carbo* (230 breeding adults), European shag *Phalacrocorax aristotelis* (1,677 breeding adults) and black-legged kittiwake *Rissa tridactyla* (8,241 breeding adults), all of which are present in nationally important numbers *i.e*. exceeding 1% of the national population (6.62%, 1.37%, 3.11% and 1.11% of the GB breeding populations respectively).It is therefore proposed that these species are identified as main components of the assemblage in line with Stage 1.3 of the SPA Selection guidelines (Stroud *et al*. 2001) (Table 3).

As well as the qualifying named features of the pSPA and the main assemblage components identified above, the assemblage also includes the following species: fulmar *Fulmarus glacialis*, black-headed gull *Chroicocephalus ridibundus*, great black-backed gull *Larus marinus*, lesser black-backed gull *Larus fuscus*, herring gull *Larus argentatus* and razorbill *Alca torda*. Although these species do not occur in numbers that meet the qualifying criteria for them to be listed as main components of the assemblage, these migratory species are still considered part of the designated assemblage of all migratory and Annex 1 species protected by the pSPA in accordance with the SPA selection guidelines (Stroud *et al*. 2001) (Table 3).

5.3.3.2. Coquet Island pSPA

Adding up the most recent population estimates for each species (i.e. the 5 year mean population figure from 2010-2014 for all species except puffin for which the average of censuses in 2008, 2009 and 2013 are used) yields a total of 47,662 individual breeding seabirds supported by the Coquet Island pSPA. This qualifies as an internationally important assemblage of over 20,000 seabirds.. In the context of SPA qualification, this seabird assemblage includes: Arctic tern, common tern, roseate tern and Sandwich tern i.e. the species described above which qualify as features in their own right. In addition, the breeding seabird assemblage includes: Puffin (31,686 breeding adults), and Black-headed gull (7,772 breeding adults) which are present in nationally important numbers *i.e*. exceeding 1% of the national population (2.73% and 2.99% of the GB breeding population respectively). It is therefore proposed that these species are identified as main components of the pSPA assemblage in line with Stage 1.3 of the SPA Selection guidelines (Stroud *et al*. 2001) (Table 3).

As well as the qualifying named features of the SPA and the main assemblage components, the assemblage also includes the following species: fulmar, herring gull, lesser black-backed gull and black-legged kittiwake. Although these species do not occur in numbers that meet the qualifying criteria for them to be listed as main components of the assemblage, these migratory species are still considered part of the designated assemblage of all migratory and Annex 1 species protected by the pSPA in accordance with the SPA selection guidelines (Stroud *et al*. 2001) (Table 3).

5.3.3.3. <u>Northumberland Marine pSPA</u>

In the same way as the total numbers of each of the species detailed under sections 4.2 and 4.3 that are supported by the pSPA have been determined by summing the numbers of those species at each of the contributing colony SPAs, so the size of the overall breeding seabird assemblage supported by the Northumberland Marine pSPA has also been determined by summing numbers of all relevant species across the contributing colony SPAs. Thus, the Northumberland Marine pSPA supports an internationally important assemblage of 214,669 individual breeding seabirds. In the context of SPA qualification, this seabird assemblage includes: Arctic tern, common tern, roseate tern, Sandwich tern, little tern, puffin and guillemot i.e. the species described above which qualify as features in their own right. In addition the breeding seabird assemblage of the pSPA includes: great cormorant, European shag, black-headed gull and black-legged kittiwake all of which are present in nationally important numbers *i.e*. exceeding 1% of the national population. It is therefore proposed that these species are identified as main components of the assemblage within the Northumberland Marine pSPA in line with Stage 1.3 of the SPA Selection guidelines (Stroud *et al*. 2001) (Table 3).

As well as the qualifying features of the pSPA (Table 1) and the main assemblage components, the assemblage also includes the following species: fulmar, great black-backed gull, lesser black-backed gull, herring gull and razorbill. Although these species do not occur in numbers that meet the qualifying criteria for them to be listed as main components of the assemblage, these migratory species are still considered part of the designated assemblage of all migratory and Annex 1 species protected by the pSPA in accordance with the SPA selection guidelines (Stroud *et al*. 2001) (Table 3).

In deriving all of the numbers presented in this section we have wherever possible used 5-year mean population estimates at each of the contributing colony SPAs using count data from the most recent available 5 year period *i.e*. 2010-2014. In the 2 cases where this has not been possible i.e., puffin at Coquet Island and Farne Islands we have used an average of the most recent censuses at those colonies conducted within 5 years of the most recent census in 2013 i.e. those in 2008, 2009 and 2013 at Coquet Island and those conducted in 2008 and 2013 on the Farne Islands.

Table 3. Breeding seabird species supported by the Northumberland Marine pSPA which do not occur in numbers sufficient to qualify in their own right under SPA selection guidelines (stage 1.1 or 1.2) but where a) their numbers equal or exceed 1% of the GB breeding population or exceed 2,000 individuals and are therefore proposed to be identified as main component species within the qualifying breeding seabird assemblage identified under SPA selection guidelines (stage 1.3), or b) are present in even smaller numbers but are nonetheless included in the assemblage identified under stage 1.3.

a)

Species	source colony	Population (5 year mean from2010-2014 unless indicated)	% GB population
great cormorant	Farne Islands	115 pairs (230 breeding adults)	1.37%
European shag	Farne Islands	839 pairs (1,677 breeding adults)	3.11%
black-headed gull	Farne Islands & Coquet Island	4,373 pairs (8,745 breeding adults)	3.36%
black-legged kittiwake	Farne Islands & Coquet Island	4,334 pairs (8,667 breeding adults)	1.17%

b)

Species	source colony	Population (mean 2010-2014 unless indicated)	Comment
Fulmar	Farne Islands & Coquet Island	341 pairs (682 breeding adults)	Populations do not reach numbers of national importance (1% of GB breeding population) or 2,000 individuals i.e. 10% of the minimum number of breeding seabirds required to comprise an assemblage feature. Therefore, these species are not main components of the breeding seabird assemblage feature
Great black-backed gull	Farne Islands	13 pairs (27 breeding adults)	
Lesser black-backed gull	Farne Islands & Coquet Island	726 pairs (1,452 breeding adults)	
Herring gull	Farne Islands & Coquet Island	836 pairs (1,672 breeding adults)	
Razorbill	Farne Islands	286 pairs (572 breeding adults)	

4. Comparison with other sites in the UK

A comparison of the numbers of each of the named qualifying features of the Northumberland Marine pSPA, derived by summing the most recent population estimates from the source colonies, with the most recent populations supported by other SPAs in the UK which also have these same species as named qualifying features in their own right, is presented in Table 4. As the source colony SPAs of Farne Islands SPA and Coquet Island SPA continue to exist in their own right, they are included in this table. This leads to duplication of numbers of birds with those tabulated for Northumberland Marine pSPA (acknowledging the difference in time periods between derivations of these numbers).

Table 4. Comparison of the numbers of individuals (and pairs) of each of the named features of the Northumberland Marine pSPA with those at other SPAs identified by the JNCC SPA review as supporting those features. Figures for the Northumberland Marine pSPA are, with the exception of puffin, based on count data between 2010 and 2014 inclusive. Figures for all other sites are taken from Stroud *et al*. (2001).

Species	**Site**	**Individuals (pairs)**[1]	**Rank**[2,3]	**Comments**
Sandwich tern *Sterna sandvicensis* (breeding)	North Norfolk Coast	6,914 (3,457)	1st of 17	
	Northumberland Marine	4,324 (2,162)	2nd of 17	
	Farne Islands	4,140 (2,070)	3rd of 17	
	Coquet Island	3,180 (1,590)	4th of 17	
	Ythan Estuary, Sands of Forvie and Meikle Loch	1,200 (600)	5th of 17	
Common tern *Sterna hirundo*	Northumberland Marine	2,572 (1,286)	1st of 23	
	Firth of Forth Islands	1,600 (800)	2nd of 23	

[1] Stroud *et al. (2001)* notes: Data from the JNCC/RSPB/ Seabird Group's Seabird Colony Register have been used. These comprised the best available, whole colony counts for the period 1993-1997 or earlier. These data have been supplemented with additional census data for some sites provided by country agencies (especially in Scotland) and/or as a result of more recent surveys of particular species.

[2] Note that these rankings should only be considered indicative of the relative importance of the pSPA as they are based on comparison of the sum of the most recent 5 year mean populations of each species at the source SPAs (as listed in Table 1) with the historical populations of each species at each SPA in the UK as listed in Stroud *et al.* (2001). The number of sites ranked is based on the number of sites listed for each species in Stroud *et al*. (2001) and included from that list are SPAs contributing to the total presented for the Northumberland Marine pSPA, and adding one site to account for the pSPA itself. For brevity, only the top 5 ranked sites are tabulated for each species, except where the Northumberland Marine pSPA position in the rank order is lower than this – in which case all sites down to that rank position are tabulated.

[3] These rank orders to not take account of numbers currently being considered in the context of other pSPAs in the United Kingdom.

	Coquet Island	1,480 (740)	3rd of 23	
	Strangford Lough	1,206 (603)	4th of 23	
	Glas Eileanan	1,060 (530)	5th of 23	
Arctic tern *Sterna paradisaea*	Northumberland Marine	9,564 (4,782)	1st of 18	
	Farne Islands	5,680 (2,840)	2nd of 18	
	Papa Westray (North Hill and Holm)	3,900 (1,950)	3rd of 18	
	Ynys Feurig, Cemlyn Bay and The Skerries	2,580 (1,290)	4th of 18	
	Pentland Firth Islands	2,400 (1,200)	5th = of 18	
	West Westray	2,400 (1,200)	5th = of 18	
Roseate tern *Sterna dougallii*	Northumberland Marine	160 (80)	1st of 8	
	Coquet Island	64 (31)	2nd of 8	
	Firth of Forth Islands	18 (9)	3rd of 8	
	Larne Lough	12 (6)	4th of 8	
	Farne Islands	6 (3)	5th = of 8	
	Ynys Feurig, Cemlyn Bay and The Skerries	6 (3)	5th = of 8	
Little tern *Sternula albifrons*	North Norfolk Coast	754 (377)	1st of 28	
	Great Yarmouth North Denes	440 (220)	2nd of 28	
	Chichester and Langstone Harbours	200 (100)	3rd of 28	
	Humber Flats, Marshes and Coast	126 (63)	4th of 28	
	The Dee Estuary	112 (56)	5th of 28	
	Chesil Beach and The Fleet	110 (55)	6th = of 28	
	Hamford Water	110 (55)	6th = of 28	
	Benacre to east Bavents	106 (53)	8th of 28	
	Solent and Southampton Water	98 (49)	9th of 28	
	Alde – Ore Estuary	96 (48)	10th of 28	
	Northumberland Marine	90 (45)	11th of 28	
Guillemot *Uria aalge*	Handa	152,206 (76,103)	1st of 35	
	East Caithness Cliffs	143,018 (71,509)	2nd of 35	
	Fowlsheugh	80,280 (40,140)	3rd of 35	
	Northumberland Marine	65,750 (32,875)	4th of 35	
	Noss	61,238 (30,619)	5th of 35	
Puffin *Fratercula arctica*	St Kilda	310,000 (155,000)	1st of 22	
	Shiant Isles	152,200 (76,100)	2nd of 22	

	Northumberland Marine	108,484 (54,242)	3rd of 22	
	Foula	96,000 (48,000)	4th of 22	
	Sule Skerry and Sule Stack	86,760 (43,380)	5th of 22	

6. Conclusion

It can be seen from the evidence presented above that Northumberland Marine is in the top 10 sites for all the proposed features (apart from little tern where it is 11th) and that the site features meet the required selection criteria for classification as an SPA.

We are confident that the proposed boundaries encompass the extent of area required by the qualifying species concerned and have been placed around the qualifying concentrations themselves, using the best available data on their distribution and using the best scientific evaluation of the limits of the species distribution.

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Annex 2 Special Protection Area (SPA) Citation

EC Directive 79/409 on the Conservation of Wild Birds potential Special Protection Area (SPA)

Name: Northumberland Marine pSPA

Counties/Unitary Authorities: Northumberland, North Tyneside

Boundary of the SPA:

The landward boundary of the pSPA covers the coastline from Scremerston near Berwick-Upon- Tweed in the north to Blyth in the south. Along this stretch of coast the boundary will follow the Mean High Water mark except around the existing Coquet Island SPA and Farne Islands SPA where the boundary will be defined by the Mean Low Water mark so as to abut the existing boundaries of those 2 SPAs where terns are already features. The seaward boundary extends up to 18 km out to sea on the basis of an analysis which identified areas of sea with the characteristics typical of areas used most heavily for foraging by breeding terns at existing colonies.

Size of SPA: The SPA covers an area of 88,687 ha.

Site description:

Northumberland Marine pSPA is located on the Northumberland coast between Blyth and Berwick-Upon-Tweed. The coastal parts of the site consist of sandy bays separated by rocky headlands backed by dunes or soft and hard cliffs. There are extensive areas of inter-tidal rocky reef, long sandy beaches at Beadnell, Embleton and Druridge Bay and extensive sand and mud flats at Budle Bay and Fenham Flats at Lindisfarne. Discrete areas of intertidal mudflats and estuarine channels are also included where the site extends into the Aln, Coquet, Wansbeck and Blyth estuaries. The open coast habitats extend into the subtidal zone, where large shallow inlets and bays and extensive rocky reefs are present. Further offshore, soft sediments predominate.

Qualifying species:

The site qualifies under **Article 4** of the Birds Directive (2009/147/EC) for the following reasons (summarised in Table 1):

- The site regularly supports more than 1% of the Great Britain breeding populations of five species listed in Annex I of the EC Birds Directive. Therefore the site qualifies for SPA Classification in accordance with the UK SPA selection guidelines (stage 1.1).
- The site regularly supports more than 1% of the biogeographical population of two regularly occurring migratory species not listed in Annex I of the EC Birds Directive. Therefore the site qualifies for SPA Classification in accordance with the UK SPA selection guidelines (stage 1.2).

Table 1 Summary of qualifying ornithological interest species in Northumberland Marine pSPA

Species	Count (period)	% of subspecies or population	Interest type
Sandwich tern *Sterna sandvicensis*	4,324 individuals (2010-2014)[1]	19.66% of GB population[5]	Annex 1
Common tern *Sterna hirundo*	2,572 individuals (2010-2014)[1]	12.86% of GB population[55]	Annex 1
Arctic tern *Sterna paradisaea*	9,564 individuals (2010-2014)[1]	9.02% of GB population[5]	Annex 1
Roseate tern *Sterna dougallii*	160 individuals (2010-2014)[2]	93.02% of GB population[5]	Annex 1

Little tern *Sternula albifrons*	90 individuals (2010-2014)[2]	2.37% of GB population[5]	Annex 1
Puffin *Fratercula arctica*	108,484 individuals (2008-2013)[1,3]	1.05% of biogeographic population[6]	Regularly occurring migratory species
Guillemot *Uria aalge*	65,751 individuals (2010-2014)[1,4]	1.72% of biogeographic population[7]	Regularly occurring migratory species

Seabirds that undertake maintenance and/or foraging behaviour within Northumberland Marine pSPA include those that breed at existing SPAs in Northumberland. Specifically these are; Lindisfarne, Northumbria Coast, Farne Islands and Coquet Island SPAs. Accordingly the numbers listed in the table above are summed across the relevant site specific population estimates.

[1] Data from: Seabird Monitoring Programme (SMP) and colony managers (Pairs multiplied by 2 to arrive at breeding adults; this rule applies to all species listed within the table, with the exception of guillemot.)
[2] Data from: Directly from colony managers (Pairs multiplied by 2 to arrive at breeding adults; this rule applies to all species listed within the table, with the exception of guillemot.)
[3] Results of puffin censuses from the Seabird Monitoring Programme (SMP)
[4] Guillemots are counted as "individuals on land"; this is multiplied by a correction factor of 0.67 (Harris 1989) to translate to breeding pairs and multiplied by 2 to yield an estimate of the number of breeding adult individuals.
[5] GB breeding populations derived from Musgrove *et al*. (2013)
[6] Biogeographic populations of 5,176,257 pairs (10,352,514 breeding individuals) derived from UK SPA and Ramsar Scientific Working Group (2014) paper*: International Population Estimates for some seabird species*. Figure derived in line with Mitchell *et al.* 2004 on the basis that puffins which used to be considered to be of the race *Fratercula arctica grabae* are now combined with those of the nominate race of *F.a.arctica* and a biogeographic population estimate derived by summing birds breeding in: France, GB, Isle of Man and Channel Islands, All-Ireland, all of Norway, Iceland and Russia (but excluding birds listed as *F.a.arctica* in Mitchell et al (2004) and breeding in Canada, USA and Greenland.
[7] Birds breeding at the Farne Islands and hence included within the Northumberland Marine pSPA are assumed to belong to the nominate race of *Uria aalge aalge* in line with UK SPA and Ramsar Scientific Working Group (2014) paper*: International Population Estimates for some seabird species* in which a population midpoint estimate of 1,909,417 pairs (rounded to 3,820,000 individuals) is given.

Assemblage qualification:

The site qualifies under **article 4.2** of the Directive (2009/147/EC) as it used regularly by over 20,000 seabirds in any season:

During the breeding season (2010-2014), the area supports 214,669 individual seabirds including: great cormorant (230 breeding adults), European shag (1,677 breeding adults), black-headed gull (8,745 breeding adults) and black-legged kittiwake (8,667 breeding adults), all of which are present in nationally important numbers (1.37%, 3.11%, 3.36% and 1.17% of the UK populations respectively) and therefore are named as key assemblage components.

Principal bird data sources:

Colony counts from JNCC Seabird Monitoring Programme contributed by colony managers: Natural England (Lindisfarne SPA), National Trust (Northumbria Coast and Farne Islands SPA) and RSPB (Coquet Island SPA), supplemented by most up to date counts in some instances from those colony managers.

Annex 3 Sources of bird data

Source of Data	Data provider	Subject	Date produced	Method of data collection	Verification
JNCC larger tern survey report	JNCC	Empirical survey data on the foraging locations of breeding terns tracked from several UK colonies and the identification of important foraging areas around colonies using habitat association models	2009-2011	Visual tracking of individual terns from boat-based survey platform	Verification by JNCC and external peer review of final report
JNCC little tern survey report	JNCC	Empirical survey data on the sightings of little terns along the shore and at sea at several UK colonies and definition of alongshore and seaward limits to important foraging areas around colonies	2009-2013	Shore-based counts from fixed vantage points and boat-based transects at sea	Verification by JNCC and external peer review of final report
Seabird Monitoring Programme	JNCC and site manager	Lindisfarne breeding seabird data	2010-2014	Standard methodology	Verified by site manager and JNCC
Seabird Monitoring Programme	JNCC and site manager	Farne Islands breeding seabird data	2008-2014	Standard methodology	Verified by site manager and JNCC
Seabird Monitoring Programme	JNCC and site manager	Coquet Island breeding seabird data	2008-2014	Standard methodology	Verified by site manager and JNCC
Seabird Monitoring Programme	JNCC and site manager	Northumbria Coast breeding seabird data	2010-2014	Standard methodology	Verified by site manager and JNCC

Annex 4 Detailed information on the derivation of the generic seaward extensions around Farne Islands and Coquet Island SPAs and their inclusion within the Northumberland Marine pSPA.

Introduction and background

Generic seaward extensions to existing breeding seabird colony SPAs represent one of three strands under the Marine Natura 2000 project to identify additional marine areas under the Birds Directive (Johnston *et al*. 2002).

The purpose of the generic marine extension to breeding seabird SPAs is the inclusion of marine areas supporting those seabirds for which an SPA is designated. Breeding and non-breeding adult seabirds use marine waters immediately adjacent to their colonies for displaying, washing, preening and potentially feeding (Tasker & Leaper 1993; Harding & Riley 2000). Targeted at-sea surveys (McSorley *et al*. 2003, 2006) demonstrate significant use and clear ecological dependence by certain seabird species on these waters. Marine extensions to existing seabird breeding colony SPAs will protect this important habitat, support the seabird colonies, and ensure compliance with Article 4 of the Birds Directive. Given that essential maintenance activities (e.g. washing, preening) were found to be non-site-specific (while feeding areas will differ between colonies), JNCC has produced generic guidance on how far extensions should be made into the marine environment, depending on which breeding species are present and informed by evidence of seabirds engaging in non-site specific behaviours and analysis of the dispersion of individuals in the vicinity of the breeding colonies (McSorley *et al*. 2003, 2005, 2006; Reid & Webb 2005). The guidance includes the following recommendations:

- colony SPAs for which puffin, guillemot, or razorbill are qualifying species/features be extended by 1 km;
- colony SPAs for which gannet or fulmar are qualifying species/features be extended by 2 km;
- colony SPAs for which Manx shearwater is an interest feature be extended by at least 4 km, or more where the available evidence warrants it[1] and
- marine SPA extensions for maintenance behaviours are not appropriate for the following species: great cormorant, skuas, gulls, black-throated diver, great crested grebe, Slavonian grebe, common scoter, red-necked phalarope.

This generic approach was approved by JNCC (in committee paper JNCC 05 P15B, December 2005 (Reid & Webb 2005)). In September 2009, Scottish Ministers classified 31 marine extensions based on the generic JNCC guidance. A proposal to include a generic marine extension within the potential SPA at Flamborough Head and Filey Coast is currently being consulted upon (Natural England 2014).

Generic marine extensions can be applied whether a species qualifies in its own right (Article 4.2 – SPA selection guideline 1.2) or as a main component of the qualifying assemblage of 20,000 seabirds (Article 4.2 – SPA selection guideline 1.3) (SNH 2008a,b).

The Farne Islands SPA supports internationally important numbers of guillemot (>1% of the biogeographic population) and nationally important numbers of puffins (>2,000 individuals). Coquet Island also supports nationally important numbers of puffins (>2,000 individuals).

On the basis of these numbers, JNCC recommendations (McSorley *et al.* 2003, 2006), suggest that the sea areas within 1 km of both colonies should be considered for inclusion within marine extensions around these two existing SPAs. However, given that all such sea areas are identified within this Departmental Brief as being important tern foraging areas, and are therefore included within the proposed boundary to

[1] Usage of marine areas around Manx shearwater colonies differed between the surveyed colonies (4 km (Skomer), 6 km (Rum) and 9 km (Bardsey)). Site specific information needs to be considered when determining an appropriate size of marine extensions for Manx shearwater breeding SPAs (Reid & Webb 2005). Therefore marine extensions to Manx shearwater SPAs should be site specific and do not derive from generic guidance.

the Northumberland marine pSPA, Natural England proposes that these generic marine extensions around the Farne Islands SPA and Coquet Island SPA are included within the Northumberland Marine pSPA rather than being treated as extensions to the existing SPAs. This approach ensures the protection of these sea areas for the features to which they are relevant while avoiding unnecessary overlapping of the pSPA with the boundaries of the existing island SPAs, were these to be extended in line with the generic guidance.

It is important to point out that although the distance of the extension boundary from shore is determined on the basis of a limited range of species, the extended site becomes an integrated whole and therefore automatically represents the full feature list of the original terrestrial site. Following JNCC guidance (McSorley *et al*. 2003, 2005, 2006; Reid & Webb 2005), generic marine colony extensions are based on non-site-specific behaviour. The guidance is based on the distribution of certain seabird species engaging in maintenance behaviours which was found to be independent of particular site characteristics across the sampled colonies. Within those ecologically important waters adjacent to the colonies birds are nevertheless protected regardless of what activity they are engaged in.

Distribution of species at the source colonies justifying the size and shape of generic marine extension areas included within the Northumberland Marine pSPA.

In the Flamborough Head and Filey Coast Departmental Brief (Natural England 2014), considerable attention was paid to the distributions of birds across Seabird Monitoring Programme count sectors within the terrestrial limits of the pSPA and the influence this might have in defining the size and shape of the marine extension around that colony. While this issue is considered in the following section, in the current case, it is concluded that such detailed analysis is largely academic. This conclusion is made on the grounds that any minor variation in the sea areas which might merit identification as marine extensions around the Farne Islands and Coquet Island, dependent upon the precise distribution of puffin and guillemot nesting locations within these SPAs, is of less import than in the case of the Flamborough Head and Filey Coast pSPA given that the entire sea area around both the Farne Islands and Coquet Island SPAs are already proposed to be included within the boundaries of the Northumberland Marine pSPA by virtue of its importance in supporting tern foraging activity.

Extent and Shape of the generic marine extension

1.1. Generic guidance

The JNCC guidance recommends that, for the species of interest on the Farne Islands and Coquet Island, boundaries should be extended by 1 km from the mean low water mark (MLWM) (McSorley *et al*. 2003).

The shape of the generic 1 km marine extension areas around the Farne Islands and Coquet Island is informed by JNCC guidance and experience from the implementation in Scotland (eg SNH 2008a,b) and current proposals at Flamborough Head and Filey Coast pSPA (Natural England 2014).

McSorley *et al*. (2003) states that boundaries should be defined by a rectilinear polygon, drawn along parallels of latitude and meridians of longitude and using the minimum number of lines. Those vertices are defined in degrees and minutes to two decimal places. Diagonal lines between two points can be used where this provides a more easily identified or more practical boundary (Johnston *et al*. 2004). However, McSorley *et al*. (2003) also note that other simple shapes and alignments may be used where practical e.g. required by local circumstances and should take into account appropriate geography (e.g. landmarks) (SNH 2008a,b).

SNH (2008a,b) identified a number of parameters to be considered in identifying the boundary for a marine extension. Of these, the following are of relevance in the current case:

1. The existing SPA boundary is realigned the appropriate extension distance (1, 2, or 4km) into the marine environment and then refined to a “best-fit” polygon with a minimum number of vertices defined in latitude and longitude whilst taking into account appropriate geography and connectivity.

2. The extended boundary includes the seabed, water column and surface.

3. Adjacent island SPAs may have overlapping boundary extensions.

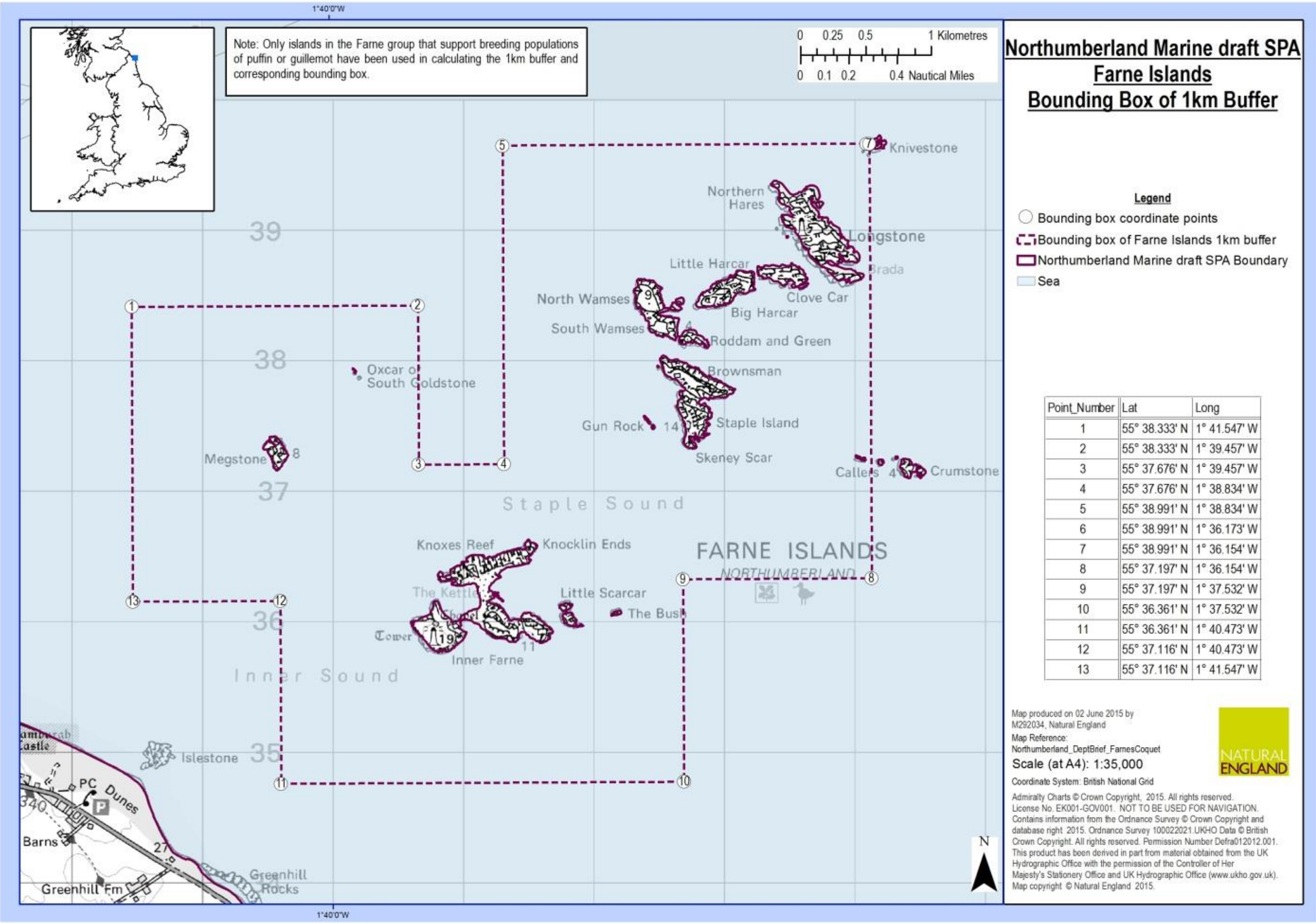
4. Almost contiguous extended intra-site boundaries will be coalesced, using judgement on proximity.

5. Where there are obviously discontinuous sections of an SPA and discontinuous interest features, the appropriate distance extension for the interest features represented in the section is applied.

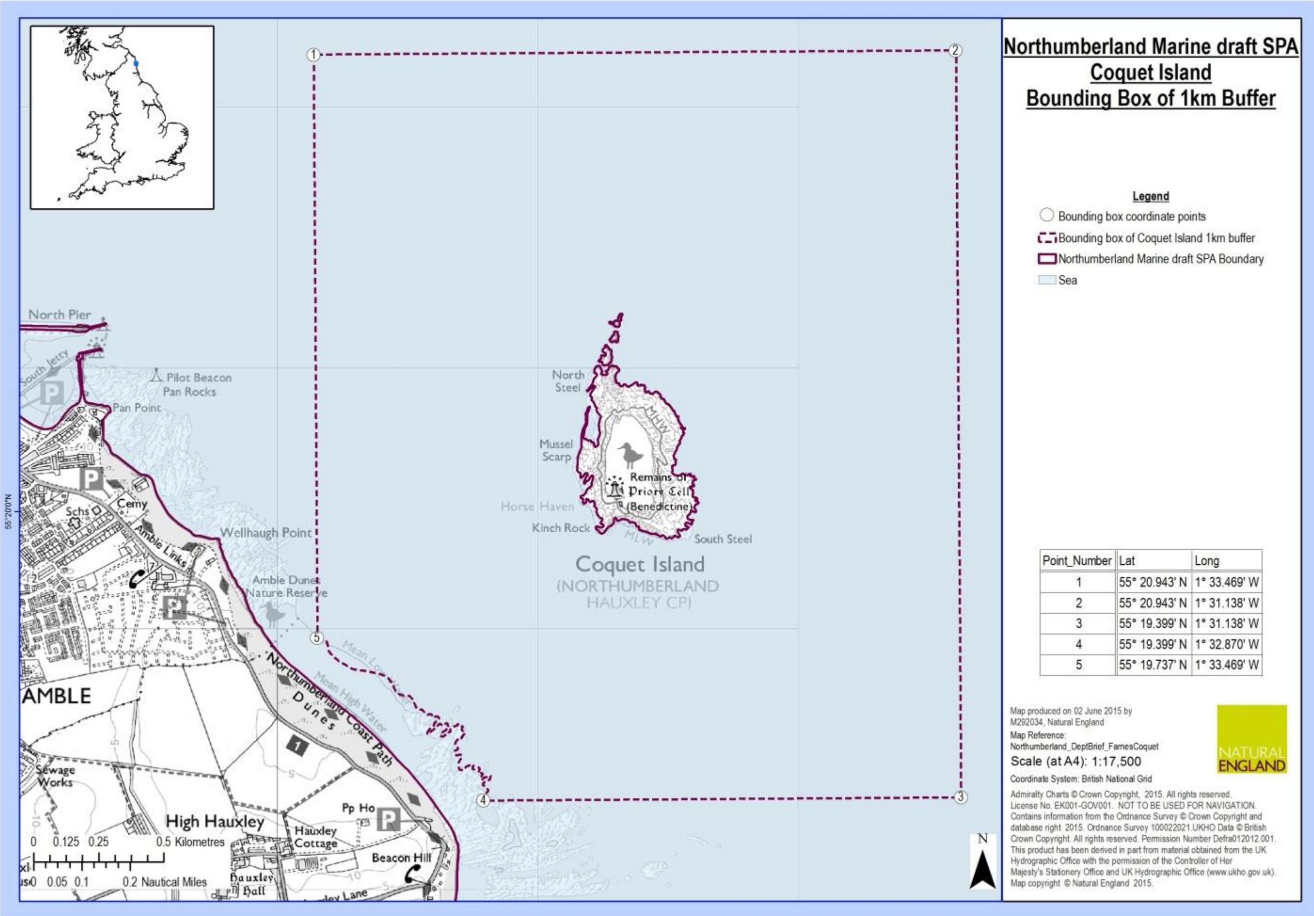
In the light of these parameters, the proposed boundaries to the areas which qualify for consideration as marine extensions around the two source colony SPAs are set out in Figure 1. The seaward boundaries to these areas, which lie entirely within the boundaries of the Northumberland marine pSPA as a whole, have for simplicity been derived from "best-fit" polygons defining a minimum of a 1 km buffer around each of the islands which constitute the Farne Islands group and Coquet Island (Figure 1) and which hold the relevant interest features i.e. guillemots or puffins. These boundaries have been coalesced in cases where the extended intra-site boundaries are almost contiguous or overlapping. The landward limit of the pSPA around these islands, and hence the landward limit of these marine extension areas is set at Mean Low Water Mark (MLWM), and hence the areas contained within these boundaries do not overlap the existing statutory protected sites.

The seabed, water column and surface are included in the proposed extension.

Figure 1:



a)



b)

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Annex 5 Detailed information on the definition of little tern foraging areas and seaward boundary definition.

1. Background and overview

All five species of tern that breed in the UK (Arctic *Sterna paradisaea*, common *S. hirundo,* Sandwich *S. sandvicensis,* roseate *S. dougallii* and little tern *Sternula albifrons*) are listed as rare and vulnerable on Annex I of the EU Birds Directive and thus are subject to special conservation measures including the classification of Special Protection Areas (SPAs). Little terns nest on sand or shingle beaches, islets and spits, often very close to the high water mark and are among the rarest seabird species breeding in the UK. There are currently 28 breeding colony SPAs designated within which little terns are protected. The marine areas they use while foraging to provide their young have not yet been identified and classified as SPAs to complement the existing terrestrial suite. Since 2009, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of such areas.

This annex gives an overview of the survey and analytical work carried out by and on behalf of JNCC between 2009 and 2013 for the little tern. This work focussed on those colony SPAs which have been regularly occupied[1] by significant numbers of little tern pairs over the last 5-10 years (13 colony SPAs). Shore based and boat based survey work was undertaken which allowed characterisation of the distances that little terns fly from their colony in order to forage. Boundaries of important foraging areas were drawn based on the distances which little terns fly along the coast, and distances which they fly out to sea. A full and detailed description of the analysis can be found in the JNCC report on this work (http://jncc.defra.gov.uk/pdf/Report_548_web.pdf). A different approach was deemed appropriate for large terns as they search for food over a much wider area and further from the coast and breeding colony than little terns. An overview of that work is described in Annex 6 and a full and detailed description of that analysis can be found in the JNCC report on that work (http://jncc.defra.gov.uk/page-6644).

2. Data collection

The study aimed to provide three years of colony specific data for all regularly occupied breeding SPAs of little terns. However logistics, colony failure, and other factors meant the data coverage for each colony varied. Surveys were timed to coincide as far as possible with chick rearing, which is the period of greatest energetic demand to the species during the breeding season and therefore critical to the maintenance of the population.

Two types of survey (boat- and shore-based observations) were applied in order to estimate both seaward as well as alongshore (coastal) extent of little tern foraging areas.

2.1. Seaward extent of little tern distribution (boat-based survey)

Boat-based surveys were carried out to assess how far out at sea foraging little terns would range (*i.e*. to confirm their maximum seaward foraging extent). Surveys involved the boats travelling along a series of parallel lines through a survey area around each colony. These surveys extended to 6 km from the coast to approximate the mean maximum foraging range as revealed from the literature (e.g. Thaxter *et al.* 2012) and preliminary JNCC observations. Two methods of recording little terns along a transect line were employed: (i) Instantaneous counts undertaken systematically at pre-determined points (between 300 m and 1800 m apart). The instantaneous count area was an 180º arc either ahead of, or off one side of, the boat depending on viewing conditions. All birds seen within this arc (out to a maximum estimated distance of 300 m) were recorded, along with the distance and bearing of the sighting and information on behaviour; (ii) Continuous counts of any little terns observed between the instantaneous points were also recorded to provide an[2] index of relative abundance. Although observers recorded behaviour (foraging/flying), restricting the analysis to just foraging observations would have limited the sample size. Therefore, all records (foraging and not foraging) were included in the analyses.

[1] 'Regularly occupied' was defined where the mean peak breeding numbers of the most recent five years at the time of assessment equalled or exceeded the 1% of the national population. Colony counts were provided by the Seabird Monitoring Programme (www.jncc.defra. gov.uk/page-1550) and direct from site managers.

2.2. Alongshore extent of little tern distribution (shore-based surveys)

Shore-based observations aimed to assess to what extent little terns forage away from their colony along the coastal strip. Observation points were chosen at 1 km intervals to either side of the colony, up to a distance of 6 km along the coast, according to the mean maximum foraging range indicated by the literature. If preliminary observations found birds going further than 6 km, more observation points were added at successive 1 km intervals. Birds were counted within a distance of 300 m to either side of the observation point (resulting in a 180° arc). The shore based counts recorded passage rate and foraging use and if possible snapshot counts at one minute or two minute intervals were also recorded. The aim of the snapshot counts was to provide information on the intensity of foraging at each observation point. Ideally, counts at different observation points were done concurrently, lasting at least 30 minutes at each observation point. This time is based on the mean foraging trip duration for little terns lasting 16–29 minutes according to Perrow *et al.* (2006). However, in some cases this was not possible due to time constraints and/or logistical difficulties. In order to account for this difference in effort between observation points the shore-based count data were standardised to the number of birds observed per minute at each observation point. Care was taken to cover a range of tidal states, as variations in water levels between the times of high and low water are likely to play a significant role in determining the foraging locations of terns.

To ensure that the data were comparable between sites the samples were analysed as a proportion of the total birds counted (per minute) at the first count point (usually 1 km) in either direction alongshore from the colony. Each side of the colony was analysed as a separate sample. This approach assumes that 100% of birds leaving the colony in a particular direction reach the first count point, and that all birds reaching subsequent count points have passed through (and had been counted at) point one on their way.

3. Data analysis

The density of little terns within each survey area was relatively small, leading to small numbers of observations within boat transects and shore based count points. This was particularly evident at the colonies with fewer breeding pairs. Given this, techniques successfully used for defining boundaries to areas of importance for other seabird and waterfowl species i.e. interpolation based on analyses of transect data to yield density maps (e.g. O'Brien *et al.* 2012) could not be used in this case. Furthermore, the small foraging range of the little terns precluded application of the habitat association modelling approach used in the case of the work on larger terns (Annex 6). Accordingly, JNCC developed a method for boundary delineation which would work with this type of data.

The approach developed to boundary setting was based on use of simple metrics that could be derived from the boat-based and shore-based survey data collected at each site. At colonies where sufficient data were available, site-specific survey data were used to determine the values of these metrics. Analysis found that colony size and density had only a weak effect on the extent of little tern foraging ranges, so in the case of colonies where there were insufficient or no data, averages of all the colony specific values were used to define seaward and alongshore boundaries. These options are set out in more detail below.

3.1 Site-specific options

For colonies with sufficient data to describe either or both seaward and alongshore extents, the following site-specific metrics were used to define boundaries:

A) Seaward extent

The **site-specific seaward** extent of foraging areas was determined by the **mean of the maximum extents** of little tern observations from repeated surveys at that site.

Using the mean of the maximum seaward observations across repeated surveys aims to represent the maximum foraging distance used by an average little tern on an average day. Within a given survey day maximum extent is used because there were relatively few survey data available and additional sampling effort would likely extend the observed maximum range. The mean of these maximum extents was used in order to express the variability of extents between samples. This approach avoids the risk of outliers dictating the extent, as would be the case if the 'maximum extent' ever observed at a site was used.

B) Alongshore extent

The **site-specific alongshore** extent of foraging areas was determined by the **maximum extent** of alongshore distribution at a site.

Using the maximum alongshore observation was considered appropriate to avoid a potential bias towards underestimation of the distances travelled alongshore that would have arisen from use of any other metric because there were: i) relatively few survey data available at each site, ii) a tendency for count points furthest away from the colony to receive slightly less counting effort, and iii) instances in which little terns were observed at the furthermost observation point alongshore. Furthermore, there appeared to be very few outliers in these datasets such that there was a lower risk of the alongshore extent being unduly influenced by outliers than in the case of the defining the seaward extent.

3.2 Generic options

For colonies with insufficient or missing data, generic options were applied to define either or both seaward and alongshore extents, based on the averages of the relevant values derived at each of the colonies for which sufficient data were available to determine site-specific values.

A) Seaward extent

The **generic seaward** extent of foraging areas was determined by the **mean** of the **mean maximum extent** obtained from site-specific datasets.

B) Alongshore extent

The **generic alongshore** extent of foraging areas was determined by the **mean** of the **maximum alongshore extent** obtained from site-specific datasets.

The validity of using these averages across sites to define the generic values for both seaward and alongshore extent at colonies with insufficient or missing data was explored by examination of the relationships between the cumulative numbers of little tern observations and increasing distance out to sea and alongshore, pooled across all sites (see next section).

3.3 Derivation of site specific and generic seaward and alongshore extents

A summary of the seaward extents as estimated from boat-based transect surveys at each colony, together with the generic seaward foraging extent derived from these values is set out in Table 1.

Table 1. Values of the maximum seaward observation of little terns on each survey at each SPA surveyed. The number of values in the 2nd column indicates the number of boat-based surveys yielding independent estimates of maximum seaward extent of occurrence at each colony. The values in the 3rd column are the site specific average of the values in the 2nd column. The value in the final row is the average of the site specific mean values.

SPA colony	Maximum seaward observation per survey (m)	Mean of maximum seaward observations (m)
Teesmouth and Cleveland Coast	1564,5661,4504,1357,4153	3448
Solent & Southampton water	492, 1620	1056
North Norfolk Coast	2077, 2129, 1946	2051
Hamford Water	2487, 1065	1776
Great Yarmouth and North Denes	800[1], 3120[1], 3770[1], 1390[2], 1730[2], 3780[2]	2430
Northumbria Coast	2185, 3011	2598
Dee estuary	1674, 2070	1872
Generic (mean value) applied to sites with insufficient data	-	2176

1. Derived from birds breeding at the North Denes colony; 85% kernel contours.
2. Derived from bird breeding (radio-tracking; 85% kernel contours) or assumed to be breeding (boat transects) at Winterton colony.

A summary of the alongshore extents as estimated from shore-based surveys at each colony, together with the generic alongshore foraging extent derived from these values is set out in Table 2.

Table 2. Values of the distance of the observation point furthest alongshore (in each direction) from each colony at which little terns were observed on any survey at that colony in any year. The value in the final row is the average of the site specific values.

SPA colony	Maximum alongshore extent from the colony in each direction (km)
Ythan Estuary, Sands of Forvie and Meikle Loch	2, 5.35
Dee Estuary	3, 3
Northumbria Coast	5, 6
Humber Estuary	6, 6
North Norfolk Coast	7, 7
Teesmouth & Cleveland Coast	5, 5
Gibraltar Point	2, N/A
Great Yarmouth North Denes	5, 4
Hamford Water	4, 3
Solent & Southampton water	1, N/A
Morecambe Bay	7, 2
Lindisfarne	3, 4
Chesil Beach and The Fleet	1, 0.5, 1
Generic (mean value) applied to sites with insufficient data	3.9

The relationships between the cumulative numbers of little tern observations with increasing distance out to sea and alongshore, pooled across all sites are presented in Figures 1 and 2. These have been used to assess the appropriateness and degree of precaution associated with the use of the generic values of 2.2 km offshore and 3.9 km alongshore to define the boundaries in the case of colonies with insufficient or missing data.

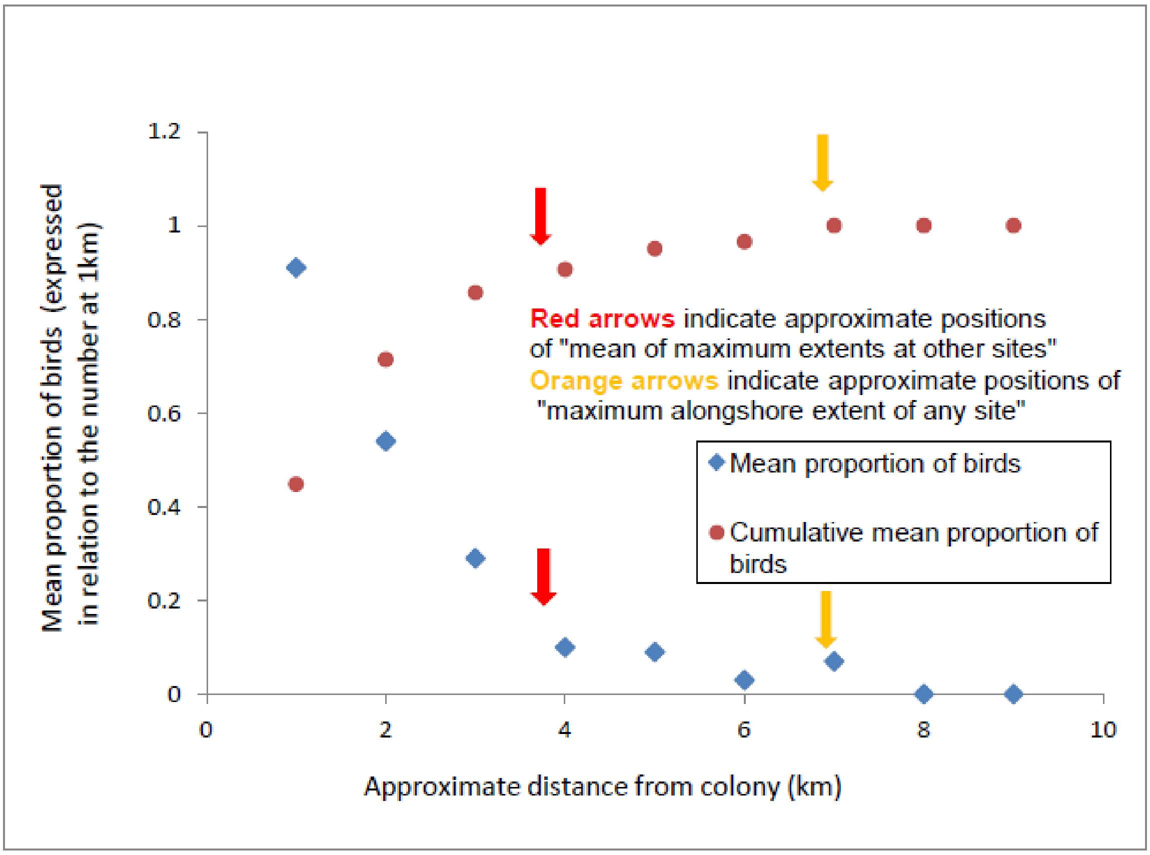

Figure 1: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing distances alongshore from the colony. Each blue point represents the mean proportional usage at each distance band from the colony averaged across colonies. The proportion at each distance (blue dots) is expressed relative to the number at the 1 km mark. The mean proportion of birds at 1 km is less than 1.0 because, in a few cases, no birds were observed at 1 km. The red arrows indicate the values at the generic mean of the maximum site-specific alongshore extent (3.9 km) whereas the yellow arrows indicate the values at the greatest site-specific maximum alongshore extent recorded (7 km at North Norfolk Coast and Morecambe Bay). Source: Parsons *et al.* (2015).

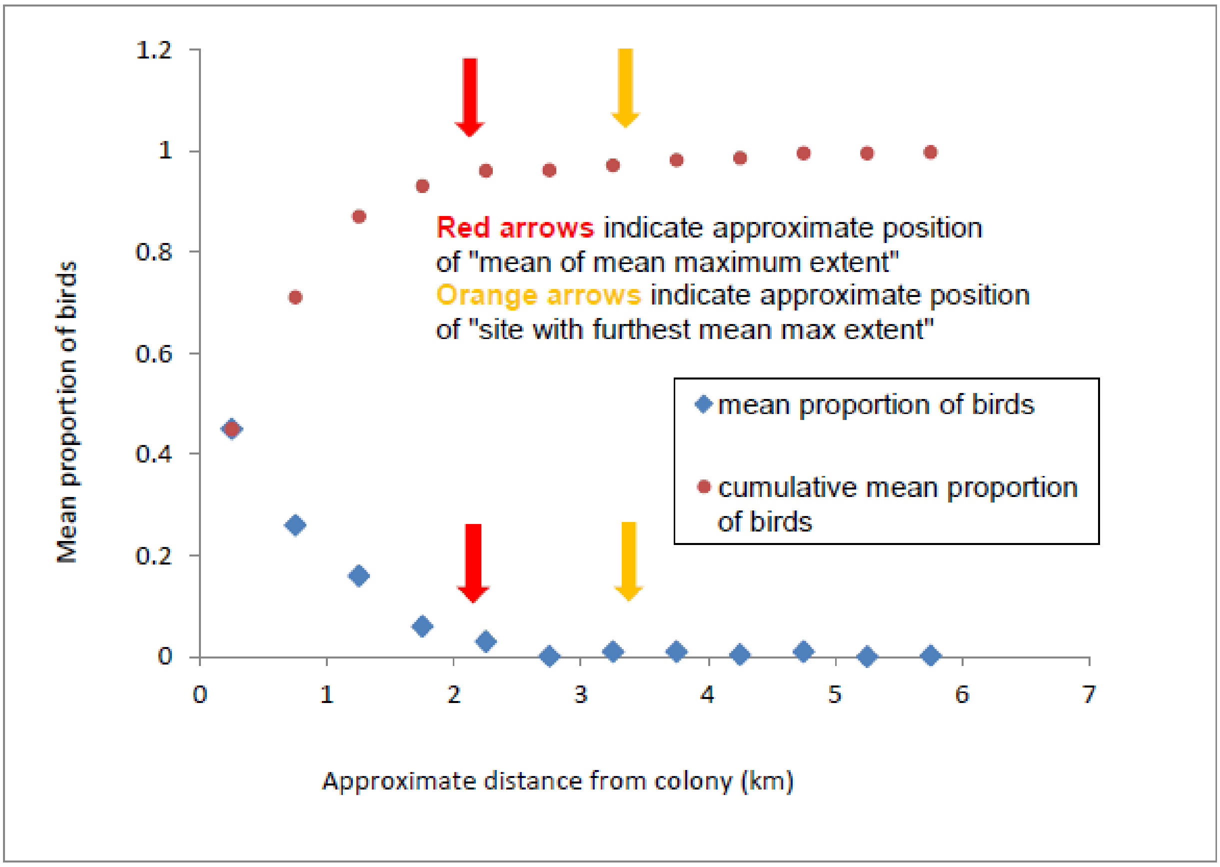

Figure 2: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing seaward distances from mean high water mark. Each blue point represents the mean proportional usage at each distance band from mean high water mark averaged across colonies. The red arrows indicate the values at the generic mean of the mean maximum site-specific seaward extent (2.2 km) whereas the yellow arrows indicate the values at the greatest of the site specific mean maximum seaward extents (3.4 km at Teesmouth and Cleveland Coast). Source: Parsons *et al.* (2015).

These figures demonstrate the nature of the relationship of increasing cumulative usage with increasing distance from colony. For alongshore (Figure 1) approximately 0.86 of all recorded usage occurred within 3.9 km from the colony, this being the mean of maximum extents at other sites and used as the generic value to define alongshore boundaries at colonies with insufficient or missing data. In comparison, at 7 km from the colony (i.e. the maximum distance of any observation station from any colony) all recorded usage was encompassed. For offshore extent (Figure 2), approximately 0.97 of all recorded usage occurred within 2.18 km of the coast, this being the "mean of the site specific mean maximum extents" at other sites and used as the generic value to define seaward boundaries at colonies with insufficient or missing data. In comparison, at 3.4 km which is the greatest of the site specific mean maximum seaward extents, 0.99 of all recorded usage at all sites was encompassed.

From these analyses it can be seen that in order to capture all recorded usage in an alongshore direction (1.0 at 7 km) and almost all recorded usage in a seaward direction (0.99 at 3.4 km) there would need to be a considerable increase in the distances being considered for defining the generic boundaries over those proposed (i.e. a further 3.1 km alongshore in each direction and a further 1.2 km offshore). On the simplifying assumption that alongshore and seaward limits define a rectangle lying parallel to the coast and with the landward edge centred on the colony, the sea area encompassed by these greater limits would be approximately 2.8 times that encompassed by the narrower limits proposed. The analyses suggest, however, that the gain in terms of the inclusion of additional areas of significant little tern activity would be relatively modest as the proportion of bird observations included within the narrower generic boundaries proposed already capture 0.86 and 0.97 of recorded usage alongshore and offshore respectively. It would seem to be overly precautionary for an estimate of foraging extent to encompass all or nearly all observations, given that at any one site this would probably result in significant areas of very low tern usage being included in the estimate. Therefore, the average of the site specific maximum alongshore extents (3.9 km) and the average of the site specific mean maximum seaward extents (2.2 km) have been adopted for a generic estimation of foraging extent at colonies with insufficient or missing data. Use of these values is, on the basis of the analyses, likely to encompass areas of high to moderate use by breeding adult little terns during chick-rearing while excluding areas which are likely to have very low usage at that stage of the season.

4 Boundary delineation

At each colony SPA, an assessment was made on the quality and quantity of data available for defining seaward extent and alongshore extent. If the quality or quantity was felt to be insufficient (eg no data or low numbers of birds observed, or few surveys, or data from only one year), then the generic option was applied at that colony. Judgement was applied rather than strict adherence to numerical thresholds for quantity of data. If the data at a site was felt to be sufficient, then the site-specific options, as described above, were applied at that colony.

Alongshore boundaries for little tern foraging areas were simply drawn as straight lines perpendicular to the coast at the distances of the site specific or generic alongshore extent on each side of the colony. Site specific alongshore boundaries were allowed to differ between the shores on either side of a colony if the data indicated this to be appropriate, whereas generic alongshore boundaries were drawn equidistant on both sides of a colony. These lines were then joined up using a line parallel to the coast and drawn at a distance defined either by the site specific or generic seaward extent. Observations indicated that little terns forage both in the intertidal zone and subtidal zone, so the landward limit of foraging extents has been taken to Mean High Water.

An example of a potential boundary around little tern foraging areas based on the approach described above is shown in Figure 3.

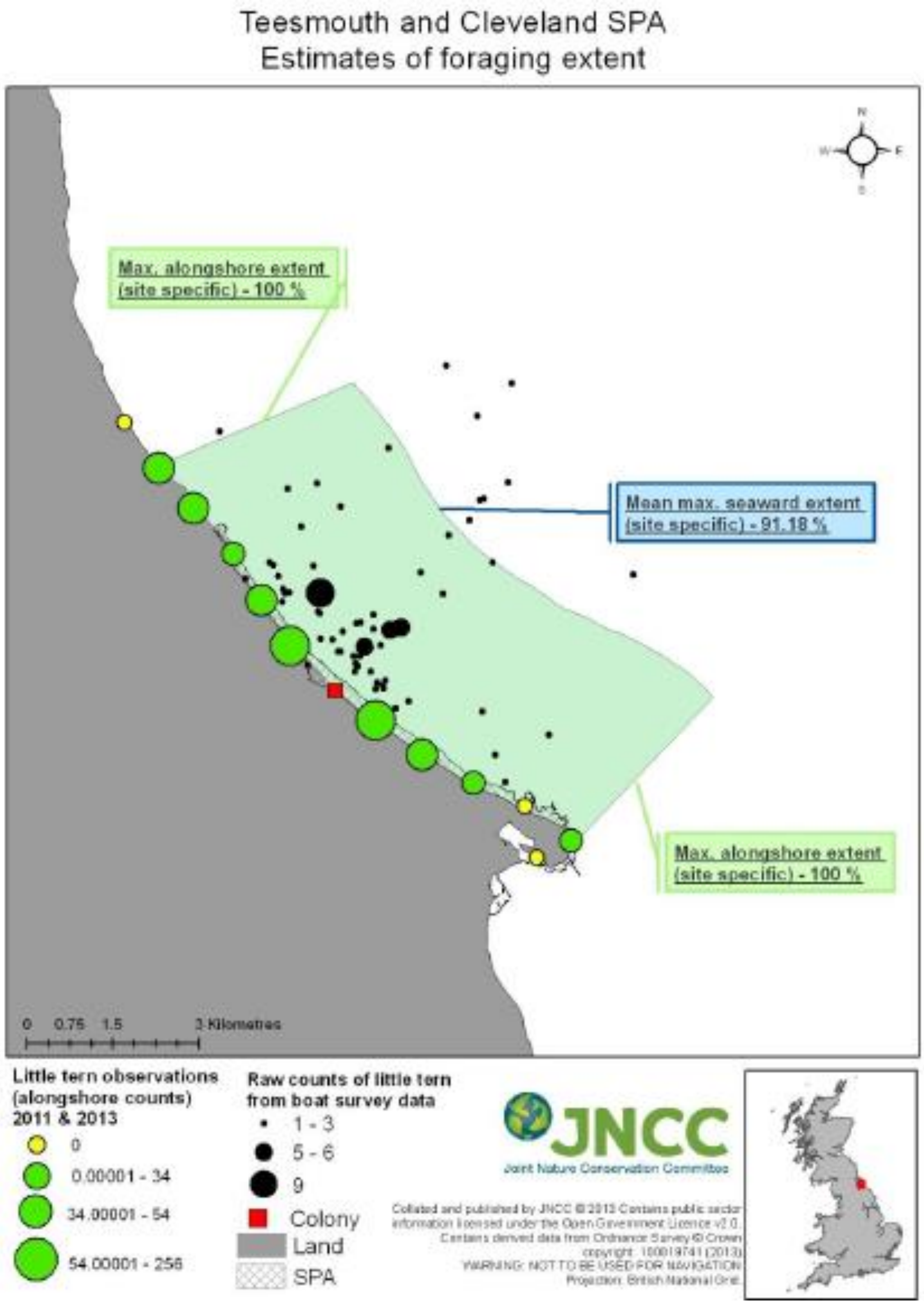

Figure 3. An example of the application of site specific alongshore and site specific seaward extents to define the boundaries to little tern foraging areas at the Teesmouth and Cleveland SPA. The % values given in the labels indicate the site specific % of little tern observations within the shore-based (alongshore) dataset and boat-based (seaward) dataset captured within the alongshore and seaward boundaries.

5 Conclusion

The aim of this work was to quantify usage of the marine environment by little terns around their breeding colony SPAs in the UK. The foraging extents identified by this study derive from information gathered over multiple years using site-specific information where possible. Most information derives from data collected between 2009 and 2013, a combination of shore-based observation (to determine the alongshore extent of use) and boat-based transect surveys (to establish the seaward extent). At one SPA - Great Yarmouth North Denes – these data were supplemented by information from radio tracking, collected in 2003-6 (Perrow and Skeate 2010).

Collection of site-specific data was attempted at most currently occupied SPAs, though in many cases data on seaward or alongshore extent could not be collected, and at others, no or few usable data were collected, either due to colony failure (caused by tidal inundation, predation or disturbance) or simply too few breeding pairs for sufficient observations to be detected by surveys.

Therefore, methods were required which aim to quantify foraging extent under a range of cases of data availability: i) where there are good data for both parameters; ii) where there are no site-specific survey data; iii) where data on seaward and/or alongshore extent are deficient.

For colonies with sufficient data on seaward extent, the mean of the maximum seaward extent of little tern observations from repeat surveys at that site has been used. Using the mean of repeat surveys aims to represent average usage and is therefore moderately conservative, and avoids the risk of outliers having a large influence on extent, as would be the case if the alternative – maximum distance offshore at which a single little tern was ever observed at a site – were used. For colonies with sufficient data on alongshore extent, the maximum distance alongshore at which terns were observed has been used, on the basis that because there are relatively few survey data at each site, and the tendency for furthest count points to have received slightly less effort on average, further survey would probably have extended the estimates of range. Because of this, it was judged that choosing the maximum extent at a site would not be excessively precautionary nor would the influence of outliers pose significant risk of over-estimation of extent.

For colonies with no or insufficient data, a method to derive generic extents was developed, based on data collected at other colonies. This aimed to weigh the risks of being overly precautionary (over-estimate foraging extent) or overly conservative (under-estimate foraging extent). Analyses indicated that use of the average across sites of the site specific means of the maximum recorded seaward extents captured 0.97 of all recorded tern observations, while use of the average across sites of the site specific maximum recorded alongshore extent captured 0.86 of all recorded tern observations. This suggested that use of these values at colonies with insufficient data to derive site-specific boundaries to little tern foraging areas would be likely to encompass areas of high to moderate use while excluding areas which are likely to have very low usage during the chick-rearing period.

The colony SPAs selected for study were those assessed to be currently occupied. This, however leaves a number of SPAs where little tern is a feature, where it was judged that little terns are no longer regularly breeding in significant numbers (as well as those currently occupied SPAs where no or few data could be collected). The assessment of occupation of such sites may change with time. This study has provided generic extents that could be applied following changed assessments.

The methods to estimate foraging extents are derived from field surveys and analyses of a nature appropriate to the data and the ecology of the little tern. Habitat modelling, such as that undertaken for the larger tern species (Annex 6) is not appropriate for the little tern, due to the combined effects of their more restricted inherent foraging range and the limited availability of habitat data at a suitable resolution or inshore locations.

The foraging extents of little tern estimated in this study fall within the range identified for little tern in a recent review of foraging ranges (Thaxter *et al.* 2012). That study identified the mean extent of the three studies included in the review as 2.1 km, with the mean of maxima across studies as 6.3 km. The work by JNCC, on a larger number of colonies, gave a mean maximum extent of 2.2 km, with a range of 1.1-3.4 km (for seaward extent) and a mean maximum of 3.9 km, with a range of 0.5-7 km (for alongshore extent). Eglington (2013), in a literature review of foraging ecology of terns, concluded that most studies, including those citing anecdotal information, reported a foraging radius less than 4 km from the colony, which accords with the results of JNCC's work.

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Abstract available at:
http://www.researchgate.net/publication/236034521_Seabird_foraging_ranges_as_a_preliminary_tool_for_identifying_candidate_Marine_Protected_Areas/file/3deec515ec5e3a2218.pdf

Annex 6 Detailed information on the definition of larger tern foraging areas and seaward boundary definition.

1. Background and overview

All five species of tern that breed in the UK (Arctic *Sterna paradisaea*, common *S. hirundo,* Sandwich *S. sandvicensis,* roseate *S. dougallii* and little tern *Sternula albifrons*) are listed as rare and vulnerable on Annex I of the EU Birds Directive and thus are subject to special conservation measures including the classification of Special Protection Areas (SPAs). Within the UK there are currently 57 breeding colony SPAs for which at least one species of tern is protected. However, additional important areas for terns at sea have yet to be identified and classified as marine SPAs to complement the existing terrestrial suite. Since 2007, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of such areas.

The work described here aimed to detect and characterise marine feeding areas used by terns breeding within colony SPAs. Given that at least one of five species of terns occur as an interest feature within 57 colony SPAs spread across the UK, it was recognised that resource and time constraints would preclude the detailed site-specific surveys at all colony SPAs over several years that, in an ideal world, would provide the most robust empirically based characterisation of marine feeding areas used by terns breeding within every colony SPA. Accordingly a statistical modelling approach was adopted which used data collected from a sub-sample of colonies to a) characterise the types of marine environment that are used by foraging terns, and b) use this information to identify potential feeding areas around all colony SPAs.

This annex gives an overview of the survey and analytical work carried out by and on behalf of JNCC between 2009 and 2013 for the four larger tern species (*Sterna* species). A full and detailed description of the analysis can be found in the JNCC report on this work (http://jncc.defra.gov.uk/page-6644). A different approach was deemed appropriate for little terns as they search for food in a much more restricted area closer to the coast and to the breeding colony. An overview of that work is described in Annex 5 and a full and detailed description of that analysis can be found in the JNCC report on that work (http://jncc.defra.gov.uk/pdf/Report_548_web.pdf). For the modelling analysis aspect of the project, JNCC worked collaboratively with Biomathematics and Statistics Scotland (BioSS)[3].

2. Data collection

To acquire information on the at-sea foraging distributions of breeding terns, three years of targeted data collection were carried out or commissioned by JNCC around selected tern colonies from 2009 to 2011, using the visual-tracking technique[4] (see BOX 1 for details). The majority of the data were collected during the chick-rearing period (June to early July), a highly demanding period for breeding adult terns due to food gathering for chick feeding and rearing. The need to regularly return to the colony results in a higher number of foraging trips within a generally more restricted foraging range. Accordingly, areas used during this period are considered as crucial for overall survival and are thus high priority for site-based conservation.

[3] BioSS are one of the Main Research Providers for strategic research in environmental, agricultural and biological science funded by the Scottish Government's Rural and Environment Science and Analytical Services Division.
[4] PERROW, M. R., SKEATE, E. R. and GILROY, J. J. (2011). Visual tracking from a rigid-hulled inflatable boat to determine foraging movements of breeding terns. *Journal of Field Ornithology*, 82(1), 68-79.

BOX 1.
Observers on-board a rigid-hulled inflatable boat (RIB) followed individual terns during their foraging trips. An on-board GPS recorded the boat's track, which was used to represent the track of the bird. Observations commenced immediately adjacent to the SPA colony. The actual starting position was varied to capture the full range of departure directions of the birds. Observers maintained constant visual contact with the bird (by maintaining the RIB c.50-200 m from the bird*) and recorded any incidence of foraging behaviours, along with their associated timings. Behaviours could then be assigned to a distinct location within the GPS track by matching the timings.
* This distance was found to be optimal in terms of maintaining visual contact whilst minimising disturbance to the bird

Existing information on tern foraging ranges (Thaxter *et al.* 2012) suggest that the larger terns are capable of foraging as far as 30 km (Arctic, common and roseate terns) or 54 km (Sandwich terns) from their colonies. Accordingly, models were used to generate predicted distributions out to these maximum foraging ranges around the colonies of interest. To do so, information on habitat conditions across these areas was gathered from various sources to be fed into the habitat models as so-called 'environmental covariates'. Such environmental covariates were chosen for their potential to explain the observed tern distribution data. Due to a lack of information on actual prey distributions (e.g. sandeels, clupeids such as herring and sardine, zooplankton), environmental covariates which could relate to the occurrence or availability of these prey species such as water depth, temperature, salinity, current and wave energy, frontal features, chlorophyll concentrations, seabed slope and type of sediment as well as distance to colony (as a proxy for energetic costs) were used instead.

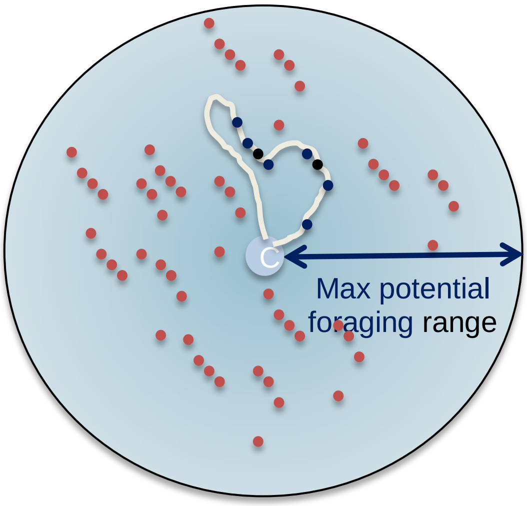
3. Data preparation and analysis

Prior to analysis within the habitat models, data had to be prepared and processed into a suitable format. Each track of a tern comprised periods of time when the bird was clearly not engaged in either actively searching for prey or in active foraging but appeared to be in transit to or from the colony or between areas of search at sea. As the aim of this work was to characterise important foraging areas and inclusion in the modelling of locations passed over in transit would, with terns being central place foragers (meaning they must travel to and from their nest site on each trip), almost certainly lead to a bias towards high usage of areas close to the colony, data from commuting periods (i.e. parts of the bird track where no foraging behaviour[5] was recorded) were removed from the modelling analysis.

In order to identify the preferred type of area used for feeding, the environmental conditions found at foraging locations had to be compared with conditions found at locations which were not used for foraging. The analysis therefore compared observed foraging presence locations with foraging absence locations (see BOX 2 for more detail on how these were defined) to characterise the kind of environment used for foraging by the terns.

[5] Foraging behaviour was defined as an instance of circling slowly actively searching for food in the water below, diving into the water, or dipping into the water surface.

Box 2.
Given that the data is collected by tracking individual birds rather than from transect surveys, we do not have a comprehensive picture of where the terns did not forage, but instead we do know where a particular bird did forage throughout a feeding trip. During that trip, it did not (choose to) feed anywhere else. There is an infinite number of possible 'non-foraging locations' where that tern could have gone to forage, so to provide something meaningful for the comparison analysis, we took a sample of non-foraging locations to which that individual might have gone from within the maximum published foraging range of each species.



The figure shows an example of the observed foraging locations (blue) along one bird track. Although an individual can (choose to) conduct a foraging trip to anywhere within the maximum foraging range, each location at which it forages on a given trip (i.e. the blue dots) is at least partly dependent upon the locations at which it has already foraged while on that trip i.e. one location follows another – the bird does not move about at random across the entire foraging range between successive foraging events on any given trip. Accordingly, to retain this within trip structure in the comparison of "presence " locations with "absence" locations, for each trip, matching sets of "absence " locations (red dots) were generated at random starting points within the maximum published foraging range of each species[6], These matching tracks therefore retained the number and spatial structure of observed foraging locations within each bird's track. 'Absence' locations represented areas available to the foraging bird but where the bird was absent at the time of recording. Twelve replicate "absence tracks" were generated for each actual trip. Subsequently, the resulting data sets to be used in the habitat models consisted of both 'foraging' and matching sets of 'absence' points for each individual foraging trip, as well as respective X and Y co-ordinates and values of the environmental covariates associated with each point

The environment that the terns use for foraging was characterised by analysis of the presence and matching absence data in relation to a suite of environmental covariates (see BOX 3 for details). This analysis was then 'reversed' and the modelled relationships between tern usage and the environmental covariates used, in conjunction with maps of environmental conditions or habitats around tern colonies, to identify those areas with characteristics suggesting that they are likely to be used for foraging, either by other terns at the same colony, or by terns at other colonies (see Figure 1).

[6] Species specific maximum foraging range from our own data and those identified in THAXTER, C.B., LASCELLES, B., SUGAR, K., COOK, A.S.C.P., ROOS, S., BOLTON, M., LANGSTON, R.H.W. & BURTON, N.H.K. 2012. Seabird foraging ranges as a preliminary tool for identifying candidate Marine Protected Areas. *Biological Conservation*. **156**: 53-61.

Box 3.

Extensive investigative analysis showed that logistic Generalised Linear Models (GLMs) were the appropriate statistical tool to identify habitat preferences of foraging terns based on observational data, and to generate predicted foraging distributions around colonies where data were missing. GLMs quantify the relationship between environmental covariates and tern foraging locations within a defined area, and by simply reversing this relationship, they are able to calculate the relative likelihood of a tern foraging (or not) at any location based on the values of the environmental covariates at that location.

As part of the development of the final GLMs used in the analysis, we ascertained that the relationship between tern foraging usage and environmental covariates was consistent between years, warranting the combination of data from all years of the study in the final models. Moreover, environmental covariates were ranked based on their biological meaningfulness, while also taking into account of the suitability and robustness of the data sets for making predictions of foraging use. Selection of which environmental covariates were included in the final model was based on this ranking combined with a standard statistical approach which trades off model complexity with goodness-of-fit to the underlying data.

In order to make a smoothed map of predicted foraging distribution, a 500 m by 500 m grid was created to cover the published foraging range for each colony of interest. Predictions of foraging likelihood were then made to each grid-cell based on the environmental conditions at the centre points of each cell. These predictions were then rescaled to provide a measure of relative foraging density within each grid-cell.

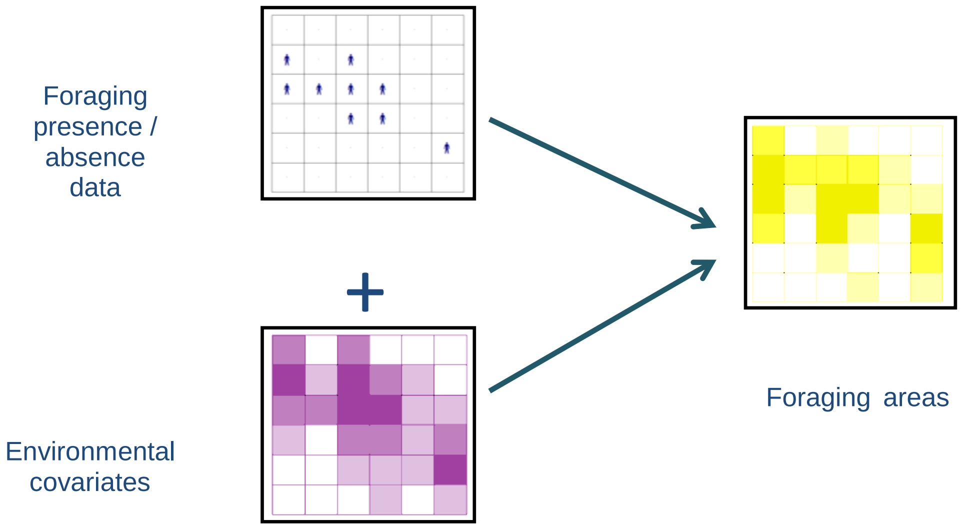


Figure 1. Simplified, schematic representation of the process of modelling distributions based on environmental information, using a single covariate distribution map in the example.

For each species of tern, there were two types of analysis: for colonies where we had collected sufficient data, the data from that colony only was used in the analysis, providing a colony-specific relative foraging density map (phase 1 analysis in Figure 2).

For colonies where we had insufficient data to produce a colony-specific relative foraging density map, all data for that species was combined to produce a UK wide analysis which could be used to produce foraging density maps around any tern colony in the UK, based on the environment and habitat conditions around those colonies (phase 2 analysis in Figure 2).

The process of analysis in this way involves creating a statistical model, and it is this model which characterises the environment that the terns use for foraging.

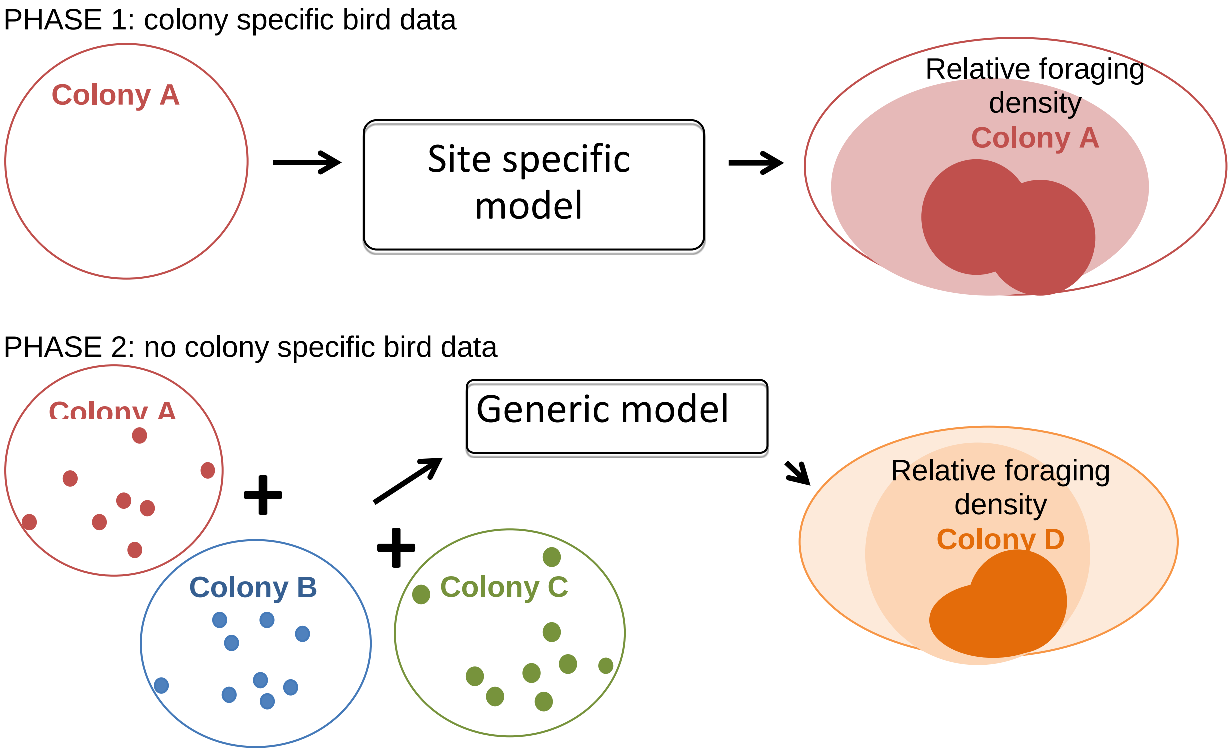


Figure 2. Simplified, schematic representation of the process whereby empirical observations of tern foraging locations around a colony were either: used to build predictive, site-specific models of tern usage that generated relative foraging density maps around that colony (phase 1 analyses); or combined with observations of tern foraging locations around other study colonies to build predictive, generic models of tern usage that generated relative foraging density maps around poorly studied or unstudied colonies (phase 2 analyses).

In order to have confidence in the robustness of the habitat association model predictions of tern usage, which are based on samples of tern tracks, it is important to consider the degree to which the sample datasets on which the models are based can be considered representative of all of the foraging locations which would have been visited across all foraging trips by all birds from a colony across an entire chick-rearing period.

Accordingly, an analysis was carried out to assess whether sufficient birds had been tracked to capture the foraging areas of the populations at individual colonies (although as discussed below this was not the primary objective of the tracking work). This analysis was conducted on data derived from three years of tracking from the Coquet Island colony of Arctic, Sandwich and roseate terns and two years of tracking from the common tern colony at the Imperial Dock (Leith). A recently published and peer-reviewed method for the analysis of tracking data was used for the analysis (see Soanes *et al*. 2013). This method examines the home range of birds derived from tracks, based on the time spent in individual predefined grid cells. All of the cells ever visited represent the total area of use, whilst other fractions of the total area of use, determined by ranking the cells in order of the amount of time spent within them were also examined i.e. the area of active use (95%) and the core foraging area (50%).

These areas are derived for samples of the pooled track data to produce results based on the use of 1 individual, 2 individuals, 3 individuals, etc… randomly sampled from the pool of available tracks in the dataset. Models are then fitted to the resulting data to examine the relationship between sample size and the total area of use, area of active use and the core foraging area. Parameters derived from these models can then be used to estimate the numbers of tracks required to capture different percentages of the area of interest (e.g. 50%, 75% and 95% of the total, active and core areas of use) given a specific colony size, thus providing an indication of how sufficient the sampling is.

The full details of the analyses are presented in Harwood & Perrow (2013). In summary, the analyses revealed that the available samples of tracks described between 45% and 68% of the total area of use, 50% and 73% of the area of active use and between 72% and 83% of the core foraging area for the four species (Table 1).

Table 1. Percentages of the predicted total (100%), active (95%) and core foraging (50%) areas based on colony size, resulting from the actual sample sizes achieved. Source: Harwood & Perrow (2013)

Tern species	Sample size (number of tracks)	% of total area of use (CI)	% of area of active use (CI)	% of core foraging area (CI)
Common (Leith)	121	68.1 (66.4-69.8)	72.7 (71.1-74.3)	73.8 (72.0-75.6)
Arctic (Coquet)	91	44.8 (40.3-49.2)	49.9 (45.5-54.0)	72.4 (68.6-75.9)
Sandwich (Coquet)	117	51.4 (48.3-54.4)	54.8 (51.7-57.7)	71.9 (69.1-74.6)
Roseate (Coquet)	50	67.9 (62.8-72.5)	72.2 (67.4-76.5)	83.3 (78.4-87.5)

Thus, although the sampling effort captured no more than two thirds of the total area of use in any case, it should be noted that the total area of use is unlikely to be described fully by any reasonable amount of tracking effort; as this would require every movement of every individual in a colony to be constantly monitored. However, the surveys did provide sufficient data to account for a large proportion of the core foraging area, which is a key metric for investigating habitat association. This provides reassurance that, even when a relatively small proportion of the colony population is sampled, the data are likely to represent well the core foraging areas of the colony population as a whole.

Furthermore, it should be borne in mind that the objective of the tracking work was not to gather a comprehensive body of tracks from which to determine directly a potential boundary around important foraging locations. Rather, the goal was to gather a representative sample of tracks from which to construct a habitat association model to identify areas with the characteristics of important foraging locations i.e. to identify not just those locations where foraging was observed within the necessarily limited empirical dataset on which the models were based, but also to identify other locations where relatively high levels of usage by foraging terns might be expected based on their characteristics.

With that in mind, for each model produced, an assessment was made of how good this model would be at making predictions of tern foraging around the same colony (for colony specific analysis) or around other colonies (for UK wide analysis). This assessment was made using a technique called cross-validation.

Cross-validation involves omitting a sub-set of data (the validation set), and refitting the chosen model to the remaining data (the training set). Predictions, in this case of tern foraging locations, generated by models based on each training set are then compared with the validation set – which in this case comprises the actual tern foraging locations not used in building the model. Comparisons can be done by various scoring methods; three were used to avoid reliance on a single method, but for simplicity only one of these i.e. the Area Under the Curve (AUC) score, is presented in this annex. The AUC score represents the discriminatory ability of a model as follows: > 0.9, excellent; 0.8-0.9, good; 0.7-0.8, moderate; 0.6-0.7, poor; and 0.5-0.6, unsuccessful (Swets 1988).

Phase 1 model performance was assessed in two ways: by investigating how well each site and species specific model predicted: (i) validation data for omitted individuals and (ii) validation data for omitted years. The former analyses were conducted for any species/colonies with at least 50 tracks that could be sub-sampled while the latter analyses were conducted for any species/colonies with more than one year of data with at least five tracks in each.

The main concern regarding the use of Phase 2 models was ensuring the models performed well when extrapolated to new areas. Therefore, model selection for Phase 2 was based on the ability of models to predict data from new colonies. The predictive ability of models consisting of all combinations of the candidate covariates was tested using cross-validation, by omitting each colony in turn and developing a model using data from the remaining colonies. Using a UK wide analysis based on data from three tern colonies (such as colonies A, B and C in Figure 2) as an example: The cross validation analysis is undertaken, creating a model which predicts tern foraging locations, based on data from only two of the three colonies, which is then used to make predictions of tern foraging locations around the third colony. Those model predictions are compared with the data that were actually collected around the third colony to see how similar they are; how well does the prediction match what the data tells us (Figure 3). This process is repeated with all possible combinations of two colonies going into the analysis, and testing the output on the third, or 'left-out', colony, to give an overall estimate of how well the model performs when making predictions to a 'new' colony.

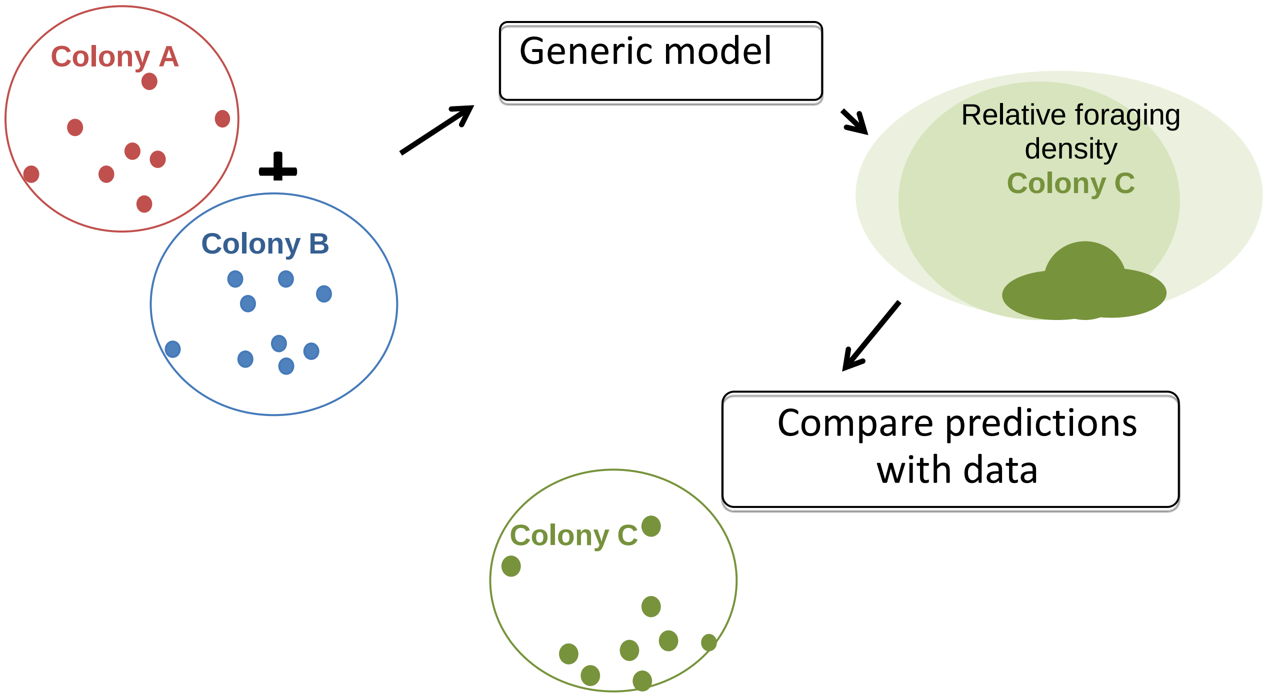


Figure 3. Schematic representation of the cross-validation process, using an example where we have data for three colonies A, B and C, of which data from two at a time (A and B in this diagram) are used to build a predictive model, the predictions of which are then tested by comparison with empirical data from the other colony (C in this case).

The cross-validation results for testing the ability of the Phase 1 models to predict validation data from individuals omitted from the models are shown in Table 2, while the results for testing the ability of the models to predict validation data from omitted years are shown in Table 3. On the basis of the average AUC scores of the Phase 1 models tested, two models performed moderately well, two were good and two were excellent in their ability to predict validation data for omitted individuals (Table 2). Of those tested for their ability to predict validation data for omitted years, based on the average AUC score, one performed poorly, two performed moderately well, three were good and two were excellent (Table 3). The cross-validation results for the Phase 2 models are summarised in Table 4. They showed that, when predicting data from new colonies, the final Arctic tern generic models performed moderately well, common tern generic models were good, and Sandwich tern generic models were excellent. For all species, the final Phase 2 models performed better than simple models containing only distance to colony.

Table 2. The results of cross-validation of Phase 1 models, testing the ability of the models to predict validation data from omitted individuals tracked at the same colony.

Species	SPA Colony	Average AUC score
Arctic tern	Coquet Island	0.796
Common tern	Coquet Island	0.845
	Imperial Dock Lock	0.741
Sandwich tern	Coquet Island	0.915
	North Norfolk	0.884
	Ynys Feurig, Cemlyn Bay and The Skerries	0.939
	Ythan Estuary, Sands of Forvie and Meikle Loch	0.990

Table 3 The results of cross-validation of Phase 1 models, testing the ability of the models to predict validation data from a different year of survey omitted from the model building phase.

Species	SPA colony	Number of combinations of years that comprised either training or test datasets	Average AUC score
Arctic tern	Coquet Island	9 (2009, 2010 & 2011)	0.71
	Outer Ards	4[1] (2009, 2010 & 2011)	0.72
Common tern	Coquet Island	9 (2009, 2010 & 2011)	0.84
	Imperial Dock Lock	2 (2009 & 2010)	0.68
	Larne Lough	4[1] (2009, 2010 & 2011)	0.87
Roseate tern	Coquet Island	4[1] (2009, 2010 & 2011)	0.84
Sandwich tern	Coquet Island	9 (2009, 2010 & 2011)	0.92
	Larne Lough	9 (2009, 2010 & 2011)	0.98

[1] In these cases there were insufficient tracks in 2010 for this year to be used as a test dataset or as a training dataset on its own.

Table 4. The results of cross-validation of Phase 2 models based on the AUC score for (a) Arctic, (b) common and (c) Sandwich terns. For each species the final model chosen (based on all three different cross-validation scores, rather than just the AUC score) is shown in bold. In addition, a model containing only distance to colony and the model which maximised the AUC score are shown for comparison. Note that the selection of the final models was based not just on these relative AUC scores but also their performance when judged using two alternative metrics. For the full cross-validation results for all the other models tested, and for all three scores, see Potts *et al*. 2013c.

(a)

Arctic terns	AUC score for each test colony			
Model	Coquet Island	Farne Islands	Outer Ards	Average AUC
Distance to colony	0.790	0.753	0.700	0.747
Distance to colony, bathymetry	**0.789**	**0.762**	**0.713**	**0.755**
Distance to colony, bathymetry, shear stress current	0.786	0.774	0.713	0.758

(b)

Common terns	AUC score for each test colony						
Model	North Norfolk	Coquet Island	Cemlyn	Larne Lough	Imperial Dock Lock	Glas Eileanan	Average AUC
Distance to colony	0.923	0.801	0.916	0.819	0.655	0.746	0.810
Distance to colony, bathymetry, distance to shore	**0.931**	**0.813**	**0.913**	**0.788**	**0.665**	**0.761**	**0.812**
Distance to colony, slope	0.930	0.805	0.908	0.853	0.670	0.749	0.819

(c)

Sandwich terns	AUC score for each test colony						
Model	North Norfolk	Coquet Island	Larne Lough	Sands of Forvie	Farne Islands	Cemlyn	Average AUC
Distance to colony	0.877	0.850	0.963	0.898	0.889	0.866	0.884
Distance to colony, bathymetry	0.878	0.899	0.979	0.962	0.956	0.907	0.920
Distance to	**0.821**	**0.911**	**0.979**	**0.973**	**0.970**	**0.907**	**0.916**

colony, bathymetry, distance to shore

4. Boundary Delineation

The maps created from outputs of the GLM models in Phases 1 and 2 are essentially a series of grid squares, each with an associated measure of relative foraging density, and indicates how likely the area within that square is to be used by feeding terns compared to other squares. There is no clear threshold in these relative density values to distinguish between 'important' and 'not important'. This kind of problem occurs in most of the marine SPA analysis JNCC has undertaken and details on how this problem has been tackled is available at http://jncc.defra.gov.uk/pdf/SAS_Defining_SPA_boundaries_at_sea. In order to identify important foraging areas for terns and draw a boundary around them, a cut-off or threshold value has to be found and only those grid squares with a usage value above this cut-off would be included within an SPA boundary. One well established way of doing this is to generate a list of every grid cell within an area of interest, ranked in decreasing order by its predicted level of usage and from that list generate a cumulative relationship between the level of bird usage captured within an area and the size of that area as, starting with the most heavily used grid cell each one in turn is added. This process invariably leads to a cumulative curve which, provided a sufficient area has been surveyed and includes some areas of relatively limited usage, gradually approaches an asymptote *i.e*. exhibits gradually diminishing returns in terms of levels of bird usage captured as the area considered increases. An objective and repeatable method to identifying a threshold value of diminishing returns on such cumulative curves is called maximum curvature (O'Brien *et al.* 2012). This method identifies at what point on the cumulative curve disproportionately large areas would have to be included within the boundary to accommodate any more increase in, in this case, foraging tern usage.

As the maximum curvature technique is sensitive to the size of the area to which it is applied, the analysis was based on a common area unit for each species. A species-specific mean maximum foraging range (*i.e*. the furthest that an average individual forages from a colony) was determined using all available data[7], resulting in 30km for Arctic, 20km for common, 32km for Sandwich and 21 km for roseate tern. Any grid cells outside the mean maximum foraging ranges were excluded prior to maximum curvature analysis.

In example of a maximum curvature boundary drawn tightly around the modelled usage distribution of Artic terns from Coquet Island is shown in Figure 4.

[7] The global mean maximum foraging range was calculated using all available tracking data (those collated for Thaxter *et al*. 2012, JNCC's tern project data, and data collected by Econ Ecological Consultancy Ltd). THAXTER, C.B., LASCELLES, B., SUGAR, K., COOK, A.S.C.P., ROOS, S., BOLTON, M., LANGSTON, R.H.W. & BURTON, N.H.K. 2012. Seabird foraging ranges as a preliminary tool for identifying candidate Marine Protected Areas. *Biological Conservation*. **156**: 53-61.

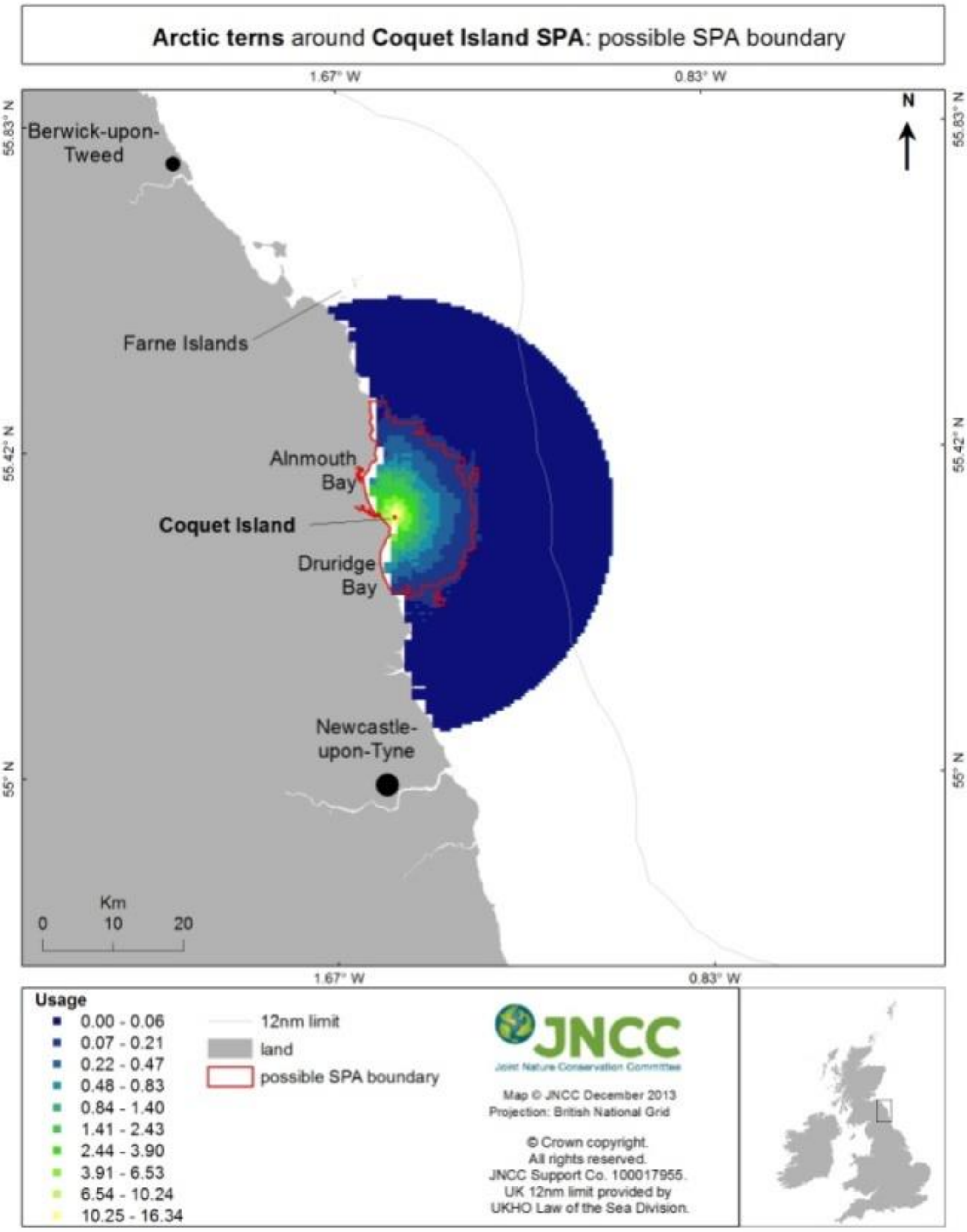


Figure 4 Maximum curvature derived boundary (red line) overlaid on map of model predictions of usage by Arctic terns around Coquet Island. The extent of the dark blue circle of model predictions of usage is 30 km - the global mean maximum distance to colony, calculated using tracking data held by JNCC; ECON Ecological Consultancy Ltd and Thaxter *et al.* 2012. These values were used to constrain the usage data used before Maximum curvature analysis was applied. Source: Win et al (2013).

Finally, boundaries were then drawn, in as simple a way as possible, around all the cells within which tern usage exceeded the maximum curvature threshold, as described in the JNCC document at http://jncc.defra.gov.uk/pdf/SAS_Defining_SPA_boundaries_at_sea.

In several pSPAs, boundaries are composites derived by application of maximum curvature methods to model predictions of usage of several interest features. In such cases, the composite boundary to the pSPA is derived by the combination of those stretches of the feature specific boundaries which together ensure that all of the important areas identified within the feature-specific boundaries are included within the whole.

5. Conclusion

Delineation of the boundaries around areas of sea that are most heavily used by seabirds have, in several existing marine SPAs, been based on maps of the relative density of birds derived directly from empirical at sea surveys of bird distribution. However, such an approach was not followed in the current project for a number of reasons. First, with tern foraging being predominantly close to shore and with the need to consider colonies all around the United Kingdom, existing data sources eg the European Seabirds at Sea (ESAS) database (http://jncc.defra.gov.uk/page-1547) were not fit for purpose. For this approach to have been followed, a significant programme of bespoke, near-shore at sea transect surveys around the UK would have been required. Furthermore, as the objective of the work was to identify foraging areas of importance to birds originating from existing SPA colonies it was necessary that survey methods could identify the origin of each bird seen at sea. Conventional at sea transect surveys cannot provide this information with any certainty, particularly when considering sightings of birds in sea areas that may be many kilometers from possible source colonies. Accordingly, a programme of boat-based tracking of breeding terns was identified as being the most suitable approach to gathering the necessary information on at sea tern foraging distributions. In an ideal world, such tracking would have been carried out on each species at every colony of interest around the UK with the intention of collating sufficiently large numbers of tracks to allow delineation of a boundary to important areas of use of each species at each colony directly from maps of relative intensity of occurrence. However, given the scale of the task (41 breeding colony SPAs have one or more of the larger tern *Sterna* species as a feature) and the inevitable limitations to survey effort that could be deployed, it was recognized that a targeted survey programme leading to development of predictive models would be the most pragmatic, cost-effective and indeed reliable approach to this project.

This project collected and collated a substantial amount of data on the distributions of terns at sea and to our knowledge represents the largest available resource of tracking data for breeding terns. The data collected/collated consisted of up to three years of survey around eleven colony SPAs and a total of almost 1300 tracks were available to the project across the four species. Geographical coverage across the UK was maximised within the constraints of the time available, logistics and resources. This ensured that data were obtained across a large range of covariate values, and that inter-colony variation could be captured as much as possible for the generic models.

The datasets collected and modelling carried out within this project allowed the development of site-specific models for 16 species/SPAs as well as generic models for each species that were used to extrapolate geographically for 30 species/SPAs. Thus the project delivered predictions of relative distributions of the larger tern species around the full complement of 32 colony SPAs in the UK which were deemed to be recently regularly occupied (46 species/SPA models in total).

Distributions predicted by the Phase 1 models generally matched the underlying data well, but also occasionally identified areas of use which were not captured by the tracking data. This is one of the key advantages of using a habitat modelling approach as it allows extrapolation into areas which were not sampled, but which are predicted to be used based on the suitability of the environment. Interpolation based only on raw data would risk overlooking the potential importance of some areas if they had not happened to be used at the time of tracking by the individuals that were sampled. A habitat modelling approach also allowed us to apply generic models which benefit from pooling data across multiple colonies, gaining strength from increased sample sizes which are able to identify broad, consistent preference relationships across multiple colonies.

All of our models predicted highest usage around the colony, with usage generally declining with increasing distance from the colony. This pattern accords well with what we might expect from central place foragers. For Arctic and common terns, the pattern of usage generally radiated out from the colony in all directions out to sea. For Sandwich terns, usage was in most cases confined to a relatively narrow coastal area either side of the colony. In all cases, there was negligible use of areas distant from the colony; more than half of the maximum potential foraging range was predicted to be virtually unused. The majority of usage was also confined to an area less than that encompassed by the mean maximum foraging ranges (as recorded in this study as well as those in Thaxter *et al.* (2012)). So although a simple approach such as applying a mean maximum foraging range radius around the colony, would correctly identify areas being used (and be a simpler method to explain) and could have been used in boundary setting, it would also include large areas of relatively low importance. The habitat modelling approach, although relatively complex, provides more realistic estimates of the relative importance of the areas within the maximum and mean maximum foraging ranges.

It might be considered that boundaries determined directly from empirically derived maps of the distributions of terns around each colony would have had a smaller degree of uncertainty associated with them than ones derived, as in this project, on the basis of model predictions of bird usage patterns, which in the case of some species and colonies are derived entirely from models of the association between bird usage and environmental covariates which have been derived elsewhere. However, this need not be the case. As noted above, the modelling approach has the advantage of allowing extrapolation of predicted usage levels into sea areas which may not be seen to be sampled (by the birds) in what will always be a necessarily limited sample dataset. Furthermore, the cross-validation of both site specific and generic models has indicated that the pooling of data across years and colonies has allowed models of tern usage to be built which are relatively robust to variations in tern foraging behaviour in time and space. For these reasons it is considered that this project has generated proposed boundaries which have degrees of uncertainty that are acceptable, and certainly need not be considered to be any worse than if it had been possible to apply more conventional approaches.

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Annex 7 Implementation of Evidence standards within Boundary Making decision process

Decision-making processes within NE are evidence driven and the Natural England strategic evidence standard, and supporting guidance were followed. In particular, the four principles for the analysis of evidence set out in the Natural England Standard *Analysis of Evidence* have been adhered to. These two standards documents can be downloaded from the following web-links:

Strategic Evidence Standard:
http://publications.naturalengland.org.uk/publication/7699291?category=3769710

Analysis of Evidence Standard:
http://publications.naturalengland.org.uk/publication/7850003?category=3769710

An explanation follows as to how the principles within the *Analysis of Evidence* standard have been applied in defining the set of qualifying features and boundary of the Northumberland Marine pSPA.

1.) ***The evidence used is of a quality and relevance appropriate to the research question or issue requiring advice or decision***

Quantification of Northumberland Marine pSPA interest feature population sizes.

In order to determine the suite of species present within the Northumberland Marine pSPA which meet the SPA selection guidelines, and to determine the nature and size of the breeding seabird assemblage supported by this site, bird count data at each of the existing coastal SPAs from which birds within the marine pSPA may originate were used as the evidence base. These data were as follows:

1. Data from JNCC's Seabird Monitoring Programme (SMP) (http://jncc.defra.gov.uk/smp/)

   a) Farne Islands and Coquet island data 2010-2014 for all species, plus results of puffin censuses on Coquet Island in 2008 & 2009 and on Farne Islands in 2009. All of these counts are assessed as "accurate"
   b) Northumbria Coast: Arctic tern count data 2010-2014. Counts in 2010-2012 are assessed as "estimates" whereas counts in 2013 and 2014 are assessed as "accurate"

2. Data from colony managers supplemented the SMP data where this was not available, in the following instances:

   a. Lindisfarne count data 2010-2014 (all species). All of these counts are assessed as "accurate"
   b. Northumbria Coast: little tern count data 2010-2014.
   c. Roseate tern and little tern data which were not accessible via the SMP webpages.

   These data were made available by the National Trust (Farne Islands SPA and Northumbria Coast SPA) and the RSPB (Coquet Island SPA).

3. Data from Natural England staff pilot surveys (2014) and contracted verification surveys (2015) of the use of estuarine habitats by large tern species

The count data taken from the SMP database is the best available information. In addition, the 2013 SMP data has been checked by JNCC. The count data which were obtained directly from the colony managers is source information that will in due course become part of the SMP database. As such, it too is the best available information. Natural England contacted Jane Lancaster at Natural Trust to determine how the 'estimates' for Arctic tern data at Northumbria Coast between 2010-2012 were arrived at. These estimates were based on a single walk-through survey of the Arctic tern colony, with every Arctic tern nest recorded, the nests being marked individually to prevent double counting. The results of these surveys were then rounded to the nearest 50, hence being estimates rather than accurate counts. The 'accurate' counts in 2013 and 2014 used the same methodology, but in these years the total number of nests/pairs was not rounded to the nearest 50.

Establishment of extent of marine pSPAs using tern tracking data

Webb & Reid (2004) provide a series of guidelines for the selection of marine SPAs for aggregations of inshore non-breeding waterbirds. This guidance does not directly consider the evidence requirements for the selection of marine SPAs focussed on the principal foraging areas used by breeding seabirds. However, a number of the issues and principles covered in Webb & Reid (2004) nonetheless have some relevance in this context. Accordingly, the following section describes in broad terms a comparison of the quality and relevance of the tern evidence base with the guidelines produced by Webb & Reid (2004).

Webb & Reid (2004) note that the guidelines for selecting SPAs in the United Kingdom are described in Stroud *et al*. (2001), and are adequate and competent for application to site selection in the inshore environment for inshore non-breeding waterbird aggregations. However, given that the type and quality of data which underpins the Northumberland Marine pSPA differs from those used in identifying sites for terrestrial birds and aggregations of non-breeding waterbirds, it is necessary to consider their adequacy and relevance.

Webb & Reid (2004) set out seven criteria to assess the adequacy of count data. Although not all of direct relevance in the current case these criteria are set out in Table 1 with accompanying comments regarding the tern tracking and modelling work.

Table 1 Criteria for inshore SPA data adequacy.

Criterion	Adequacy of JNCC led larger tern surveys	Adequacy of JNCC led little tern surveys
Experience of observers	All tracking of terns was undertaken either by JNCC staff or experienced contractors commissioned by JNCC to do the work.	All observations of terns were undertaken either by JNCC staff or experienced contractors commissioned by JNCC or volunteer counters who received training in the shore-based observation techniques.
Systematic surveys	Tern tracking was conducted in as systematic a way as possible. Tracking at each colony was carried out during well-defined periods of the breeding season (chick-rearing) in one or more years. Tracking was undertaken in accordance with a field protocol established by JNCC. In the context of tern tracking, the movements of birds is an essential component of the technique and not a source of systematic bias in the survey results as it may be in conventional transect surveys.	Boat-based survey work followed systematic transect survey designs that were appropriate to each colony and were followed on repeated surveys. Shore based survey work used systematic series of observation stations and a standard recording protocol which was used repeatedly at each colony.
Completeness	The aim of the tracking survey method was not to cover all of the areas sea to consider for inclusion in the pSPA, but to ensure that the tracking effort was sufficient to capture tern usage across a representative proportion of that area on the basis of which reliable habitat association models could be constructed and used to predict tern usage patterns across the wider area – including those areas in which no direct observations of terns were made.	Boat-based transects extended up to 6km offshore and alongshore survey stations were positioned at 1km intervals up to at least 6km in either direction from the colony (and where necessary, further). With the mean maximum foraging range reported to be 6.3km, the survey areas gave virtual complete coverage of the likely areas of greatest importance.
Counting method	The larger tern tracking work did not involve counting of birds or use of such information to derive population estimates for the pSPA. However, the modelling is based on samples of tracks of relatively few individual terns from each colony rather than surveys of the distribution of terns (of unknown origin)	At sea observations included instantaneous counts at predetermined distances along transects at which all terns in flight within 300 m in an 180º arc of the boat were recorded. Between these points, continuous records of all little terns seen were also made to provide an index of relative abundance. During shore-based observations, terns

	around the colony. Cross-validation tests of the models' predictions and analysis of sample adequacy both suggest that the results of the models, although based on the samples of tracks, are robust.	recorded within 300 m of the observation point were recorded during timed observation periods. Counts at each station were standardised to birds/minute and expressed as proportions of the value recorded at the 1 km observation station to standardise across sites.
Quality of sampling	Cross-validation tests of the models' predictions and analysis of sample size adequacy both suggest that the results of the models based on the samples of tracks are robust.	This was affected by the low numbers of birds at many colonies and the frequent breeding failures. At colonies with 5 or more shore-based surveys yielding records of 200 or more terns, this was deemed sufficient to derive site-specific along shore boundaries. At colonies with at least 2 boat-based surveys yielding at least 20 tern sightings this was deemed sufficient to derive site-specific seaward boundaries. At colonies where these criteria were not met, a generic approach was used by pooling sample data across sites to yield better-evidence based estimates of limits.
Robustness of population estimate	Not applicable as the tern tracking work was not used to generate a population estimate	Not applicable as the tern observation work was not used to generate a population estimate
External factors affecting the survey	Tracking was constrained by weather, e.g. tracking could not take place with sea state ≥3 and during rain. Thus, tracking data were gathered only under favourable weather conditions.	Although the aim was to collect data from most currently occupied SPAs, in many cases data on seaward or alongshore extent could not be collected due to colony failure (caused by tidal inundation, predation or disturbance) or simply too few breeding pairs for sufficient observations to be detected by surveys. Accessibility to count points in all parts of the possible extent of a foraging area limited the ability to provide site-specific alongshore extents in some cases.

Webb & Reid (2004) also discuss the issue of establishing sufficient evidence in the case of marine SPAs to establish regularity of use, which is a key element of the SPA selection guidelines. The tern tracking work was never intended to establish regularity of use of certain sea areas by particular species around particular colonies. The aim of that work was simply to capture sufficient representative information on tern foraging behaviour to allow reliable habitat association models to be constructed and used to generate maps of areas of principal usage. The results of the cross validation of those models' predictions, in which data from different years were used as test datasets, suggests a relatively high degree of consistency in usage patterns between years i.e. regularity of use of those most important areas (Wilson *et al.* 2014). However, no formal tests of the regularity of use of the sea areas within the pSPA boundary have been made. Regularity of use of the pSPA has been reasonably inferred from the continued existence of the site's named features in qualifying numbers in each of the existing coastal SPAs from which birds within the marine SPA are most likely to originate.

Webb & Reid (2004) discuss the issue of boundary placement. They note that the principles for defining boundaries for terrestrial SPAs in the UK are described in Stroud *et al.* (2001) thus (emphasis added):

*"The first stage of boundary determination involves **defining the extent of area required by the qualifying species concerned**. These scientific judgements are made in the light of the ecological requirements of the relevant species that may be delivered by that particular site, and the extent to which the site can fulfil these requirements. This follows a **rigorous assessment of the best-available local information regarding distribution, abundance and movements of the qualifying species**. It may also involve the **commissioning of special surveys** where the information base is weak. Following this stage, every attempt is made to define a boundary that is identifiable on the ground and can be recognised by those responsible for the management of the site. This **boundary will include the most suitable areas for the qualifying species** identified in the first stage……"*

The larger tern tracking and little tern observations were conducted to define the extent of the area required by these species on the basis of specially commissioned surveys that generated the best available local information regarding distribution, abundance and movements of these qualifying species.

Webb & Reid (2004) discuss the principles of setting both landward and seaward boundaries of marine SPAs.

In regard of setting landward boundaries they note that “*Where the distribution of birds at a site is likely to meet land, a boundary should usually be set at the mean high water mark (MHW)……. unless there is evidence that the qualifying species make no use of the intertidal region at high water*.”

The landward boundary of the pSPA has been drawn at MHW along the mainland coast in the light of: i) model predictions of the usage of such areas by foraging larger terns, ii) existing evidence which indicates that all five species of tern considered here routinely forage in areas of shallow water (Eglington 2013), iii) observations that little terns forage in the intertidal zone (Parsons et al. 2015) and iv) to ensure protection of these areas as supporting habitat for tern species within the pSPA in locations where these species are not already features of the existing SPA with which the overlap will occur (eg Sandwich, common and Arctic tern at Lindisfarne SPA). The “landward” boundary of the pSPA around the offshore islands has been drawn at MLW to avoid overlap with parts of the existing SPAs where the same individuals of the same species are already named features and afforded protection in those areas already.

Webb & Reid (2004) set out a recommended method for defining the seaward boundary of SPAs for inshore non-breeding waterbirds on the basis of analysing bird data from aerial or boat-based sample surveys using spatial interpolation combined with spatial analysis. They note exceptions to this method which include the case in which “*habitat data are also used in combination with bird distribution data to determine boundaries*”. This is the approach which has been used in the tern work which has determined the seaward boundary of this pSPA.

Webb & Reid (2004) describe spatial interpolation methods by which survey sample data can be used to generate maps of species probability of occurrence or abundance. This involves use of a “….*suite of modelling techniques in which the probability of bird occurrence or the total number of birds present is estimated at unsampled locations (usually in grid cells) using information on the presence or absence, or the number of birds recorded at sampled locations*”. This is the principle underlying the modelling of the tern tracking data, albeit that the nature of the statistical models used is somewhat different to those considered by Webb & Reid (2004). As such, the principle of the method which has been used to define the seaward boundary of the pSPA is entirely in line with the recommendation of Webb & Reid (2004).

Webb & Reid (2004) conclude by discussing the method by which a boundary should be drawn around the parts of a site identified as being most important. They refer to Webb *et al.* (2003) which sets out a method for classifying grid cells so that the most important ones for a species on any given survey are highlighted. In that method, the grid cells are ranked from lowest predicted bird abundance to highest, and the cumulative population calculated from lowest ranked grid cell to highest. The highest ranking grid cells were selected such that they comprised 95% of the total population. The analytical approach which has been applied to the grid-based, modelled predictions of tern usage to define the most important areas to include within the pSPA boundary (Win *et al.* 2013) follows the basic ranking principle outlined by Webb *et al.* (2003). However, the application of the maximum curvature technique to such cumulative usage curves in the current case (Win *et al*. 2013) reflects the advances in the details of this analytical method by JNCC since then (O’Brien *et al.* 2012).

Thus, in summary, although Webb & Reid (2004) does not directly address the issue of data requirements in regard of establishing marine SPAs for breeding seabirds, many aspects of the collection and analysis of the tern tracking work which has been used to define the location and extent of the Northumberland Marine pSPA can be seen to be in accord with the guidelines set out in that document.

Establishment of the extent of the Northumberland Marine pSPA

The extent of and boundary to the Northumberland Marine pSPA is determined by the extent of the model generated predictions of which areas of sea are most heavily used by foraging terns originating from three source colonies. The boundary of the pSPA is a composite of several discrete, but in some case overlapping, areas identified by the modelling as being most heavily used by different species of tern from different colonies (Table 2). Some of these species and colony-specific areas of use have been derived from models based on at sea records of the foraging locations of the particular species at the particular colonies i.e. site-specific models (eg Arctic terns at the Farne Islands). Other species and colony-specific areas of use have been derived from models based on at sea records of the foraging locations of the particular species but at other colonies around the UK i.e. generic models (eg Sandwich terns at the Farne Islands). The quality and relevance of the evidence provided in both of these ways is discussed in the following section.

Table 2. Species and source colonies, the foraging areas of which combine to define the alongshore and seaward limits of the Northumberland Marine pSPA boundary, and the nature of the model on which those areas of usage were based.

Species	source colony	Model type	tern tracks contributing to model	number of sites contributing to model	number of tern site/years of data underlying model
Sandwich tern	Farne Islands	Generic	277	5	11
Arctic tern	Farne Islands	site specific	32	1	1
Arctic tern	Newton Links/Long Nanny	Generic	149	4	10
Arctic tern	Coquet Island	site specific	91	1	3
Roseate tern	Coquet Island	site specific	40	1	3
Sandwich tern	Coquet Island	site specific	90	1	3
All			679		

The adequacy and relevance of these various models and of the modelling approach in general, was addressed by JNCC in 3 ways (Wilson *et al.* 2014):

i) Cross-validation of site specific models
ii) Cross-validation of generic models
iii) Adequacy of sample size data

A summary of the results of the cross-validation of both site specific and generic models of larger tern usage is presented in Annex 6, as is a summary of the analysis addressing the adequacy of the sample sizes.

2.) The Analysis carried out is appropriate to the evidence available and the question or issue under consideration

Natural England has incorporated the 1 km marine extension around the Farne Islands SPA and Coquet Island SPA into the Northumberland Marine pSPA through following the generic guidance recommended by JNCC and implemented at 31 Scottish and 1 English site (Flamborough & Filey Coast pSPA). Therefore, the inclusion of maintenance extensions within the pSPA cannot be considered a novel approach, but an application of a well-established approach. The site-specific evidence – qualifying breeding populations of common guillemot (Farne Islands SPA only) and puffin on the terrestrial SPAs– required to justify the proposed 1 km marine extension around these colonies is provided by the SMP (http://jncc.defra.gov.uk/smp/) and National Trust/RSPB survey data. These data were compared to established site selection criteria (JNCC 1999), meaning the analysis is entirely appropriate.

The other major analyses which underpin the pSPA are: i) the boat-based and shore-based observations of little terns, ii) the habitat-association based modelling of larger tern usage patterns and ii) identification of threshold levels of predicted larger tern usage which were used to define the site boundary.

The very restricted foraging range of little terns precluded the use of the predictive habitat association modelling approach that was used for the larger terns. Accordingly, it was appropriate to gather empirical evidence on little tern distributions from which to determine directly the boundaries to the areas of greatest usage by foraging birds at each colony. At colonies where evidence was lacking or insufficient it was considered appropriate to make use of data gathered at other colonies to determine “generic” boundaries which, comparison with all available data indicated, would capture a very significant proportion of total usage (see Annex 5).

The habitat association modelling approach is a novel one which has not been used in defining the extent or boundaries of any marine SPA to date. However, the decision to adopt a habitat association modelling approach was the subject of discussion between JNCC and all other statutory nature conservation bodies over many years and agreement to follow this approach informed the design of the survey programme coordinated by JNCC since 2009. For the modelling analysis part of the project JNCC worked collaboratively with their statistical advisors Biomathematics and Statistics Scotland (BioSS).

Although the method by which the grid-cell based maps of predicted bird distribution were drawn up in this case differed in detail from more conventional spatial interpolation and spatial analysis considered by Webb & Reid (2004), the way in which the resultant maps of predicted bird distribution were analysed to determine threshold levels of predicted tern usage, and hence to define the site boundary, (i.e. maximum curvature analysis) represents application of an established method used at other marine SPAs (O’Brien *et al.* 2012) and is thus entirely appropriate to the evidence available.

Following completion of the work on both larger terns and little terns, JNCC commissioned external peer review of both pieces of work. Those peer reviews did not highlight any significant issues with the appropriateness of the analyses which were not resolved by subsequent discussion between the reviewers and JNCC. Further details of the external peer review are provided in section 5 of this Annex.

*3.) **Conclusions are drawn which clearly relate to the evidence and analysis***

The conclusions regarding the list of features and their reference population sizes within the pSPA are based on application of the SPA selection guidelines issued by JNCC (JNCC 1999) to the best and most recent count data from the principal source breeding colonies from which birds within the pSPA are most likely to originate (SMP data & National Trust/RSPB survey data). As such the conclusions in this respect clearly relate to the best available evidence.

The conclusions regarding the extent of the recommended seaward maintenance extensions around both the Farne Islands SPA and Coquet Island SPA, which are encompassed within the Northumberland Marine pSPA, are based on the evidence gathered on maintenance behaviour of SPA seabirds. Species-specific analysis of the use of waters surrounding colonies was carried out by JNCC to inform the recommended extent of colony extensions (McSorley *et al*. 2003, 2005), and the conclusions set out in this brief have been derived from this. These conclusions have already been applied to breeding seabird colony SPAs in Scotland (with 31 sites in Scottish waters being consulted on in 2008 and classified in 2009) and England (Flamborough and Filey Coast pSPA).

The conclusions regarding the drawing of the landward boundary of the pSPA along the mainland coast at MHW are based upon the evidence provided in the form of models of predicted usage by foraging larger tern species. In several instances these models (common tern – generic model, sandwich tern - generic model and site specific models at Coquet and Farne Islands) included distance from shore as a significant covariate with a negative coefficient indicative of highest use being closest to shore and therefore in many instances inclusive of intertidal areas. The use of intertidal areas between MLW and MHW by foraging little terns is recorded in Parsons *et al.* (2015). That the use of such areas by all larger tern species is also likely is supported by information in the scientific literature. A review of tern foraging ecology (Eglington 2013) notes that all five species of tern considered here routinely forage in areas of shallow water. There is no reason on the basis of that review to consider it likely that that these birds will not forage over intertidal areas. Accordingly, in this respect too, the conclusions clearly relate to the best available evidence.

The conclusions regarding the drawing of the seaward boundary of the pSPA are based upon the evidence provided in the form of models of predicted usage by foraging larger tern species and the application of a standard analytical method, already well-established for use in marine SPA boundary setting i.e. maximum curvature (O’Brien *et al*. 2012), to the models’ outputs. The validity and robustness of the outputs of the site specific and generic models used to underpin the boundary analysis of the pSPA have been established by the process of cross-validation described in Annex 6. Thus, the conclusions in this respect clearly relate to the best available analysis of the best available evidence.

Since the modelling work was completed by JNCC, the Department of the Environment, Northern Ireland (DoENI) commissioned in 2014 a programme of land-based and at-sea surveys to verify the extents of tern foraging activity at three sites in Northern Ireland i.e. Larne Lough, Strangford Lough and Carlingford Lough. At each of these sites, the same generic predictive models, as already described in this Departmental Brief, had also been used to generate relative usage maps for at least one species of larger tern (and in some cases for all species) and hence to determine proposed site boundaries. In summary, this work (Allen & Mellon Environmental Ltd 2015) confirmed the presence of terns (mainly Sandwich) to the furthermost alongshore limits of the areas searched and in one case beyond the limit of the modelled alongshore boundaries. The work provided some evidence that the larger terns do feed further out to sea than the limits of the modelled boundaries. However, the use of the threshold setting approach to the predicted relative usage maps does not deny that terns may forage beyond that limit. The work also provided some evidence that the very intense use of localised hotspots of activity recorded in or close to the entrances to the loughs were not as clearly identified as such by the models. However, the proposed boundaries in each of the three sites did contain the hotspots within the lough entrances. Thus, these verification surveys provide: confirmation that hotspots of usage near colonies are contained within modelled boundaries, some evidence that proposed boundaries, based on model predictions, may be somewhat conservative in regard of their seaward limits, and no evidence that their alongshore or seaward extents are in any way excessive.

4.) ***Uncertainty arising due to the nature of the evidence and analysis is clearly identified, explained and recorded.***

Count data

The UK SMP is an internationally recognised monitoring scheme coordinated by JNCC in partnership with others (e.g. statutory nature conservation bodies, the RSPB and other colony managers as data providers, etc.). It collects data according to standardised field methods (Walsh *et al.* 1995). SMP data are verified by the JNCC seabird team. Therefore, there is high confidence in SMP data. The majority of the data which has been used in determining the size of the populations of each of the species considered for inclusion as features of the pSPA is based on counts which are on the SMP database and so justify high confidence.

RSPB survey data are verified and quality assured by the RSPB count coordinator and site manager and Natural Trust data is verified and quality assured by the site manager. Both are professional organisations with long-standing experience of seabird monitoring, and surveys are conducted by trained surveyors. There is therefore high confidence in RSPB and National Trust survey data. Accordingly, even the most recent count data referred to in this Departmental Brief can be considered to justify high confidence.

One particular issue with the count data requires further consideration, namely the lack of consecutive counts of puffin. Due to the complexity and costs of puffin burrow surveys these are not carried out yearly by all colony managers, but are surveyed as a minimum on a 5-yearly basis as part of a UK-wide puffin census. Given this constraint to the availability of population estimates for puffins, the most recent of these censuses at both Coquet Island and the Farne Islands in 2008, 2009 and 2013 have been used in our assessment. This is the best available evidence.

Landward boundary

The issue regarding the confidence in the evidence base upon which the decision to draw the landward boundary of the pSPA to MHW along the coast has been made, is discussed in the previous section.

Seaward boundary

The position of the seaward boundary of the pSPA is the principal source of uncertainty in the identification and characterisation of the site. The position of the seaward boundary of the pSPA has been determined on the basis of outputs of statistical models which in some cases are based on tern behaviour at colonies in other parts of the United Kingdom. Accordingly, it is almost inevitable that there is a greater degree of uncertainty regarding the robustness of the boundary location than if it had been derived directly from a comprehensive site-specific set of observations of tern foraging locations. However, provided the models are empirically evidence based, and shown to be robust via cross validation, the modelling approach brings with it a robustness which may exceed that which might be achieved from reliance on a limited empirical dataset of tern foraging locations. It is considered that the cross-validation analyses and sample-size sufficiency analyses indicate that proposed boundaries generated by the modelling approach have degrees of uncertainty that are acceptable, and certainly need not be considered to be any worse than if it had been possible to apply more conventional approaches. This issue is discussed fully in Annex 6.

5.) Independent expert review and internal quality assurance processes

Independent expert review

Natural England's standard in quality assurance of use of evidence, including peer review, (http://www.naturalengland.org.uk/images/operationalstandardsforevidence_tcm6-28588.pdf) has been followed in determining the level of independent expert review and internal quality assurance required in relation to Natural England's analysis of the evidence for this site and the way that the boundary has been drawn up. Independent expert review is to be adopted where there is a high novelty or technical difficulty to the analysis.

The application of a generic marine extension for maintenance activity and the analysis of the colony count data are considered not to be novel or technically difficult because:

a) The generic marine extension approach has already been applied at 31 sites in Scotland and 1 in England which was informed by reviewed evidence provided through JNCC.
b) The colony count data have been analysed and assessed in the same way as carried out for other terrestrial SPAs in England and conforms to the SPA selection guidelines (JNCC 1999).

For the reasons set out at a) and b) above Natural England believes these elements of the recommendation and associated analysis are not contentious and therefore independent expert review of how it has applied the evidence in drawing up these maintenance extension boundaries is not being sought.

In contrast, the derivation of the alongshore extent and seaward boundary to the pSPA is based on an entirely novel approach, never used before in SPA designation, and has entailed considerable technical difficulty in the analyses. In recognition of this, JNCC commissioned independent expert review of both the larger tern and little tern programmes of work. A representative of Natural England, along with those of all other country statutory nature conservation bodies, was involved by JNCC in setting the terms of reference for the review work, in nominating potential reviewers for JNCC to consider approaching, and in the selection of those who carried out the reviews.

The larger tern modelling work was reviewed by two independent scientists (Dr Mark Bolton of the British Trust for Ornithology and Dr Norman Ratcliffe of the British Antarctic Survey). In summary, both reviewers raised two primary issues with the data collection and its analyses. These related to: i) the focus of the tern tracking work during the chick-rearing phase of the breeding season and ii) to the details of the way in which control points denoting tern absence were generated to match track locations where terns were recorded and the use of that information to determine terns' preference for each location and the conversion of that preference pattern into a pattern of tern usage. In regard to the first issue, JNCC acknowledged that the focus of the tracking work was only on the chick-rearing period, partly in order to ensure that sufficient data were gathered during that one period, but also in recognition of the need to focus attention on the identification and protection of those sea areas which are of most importance to the birds when their ability to buffer themselves against adverse environmental conditions by foraging further from the colony is most limited by time and energy constraints and their need to provision their chicks. The report (Wilson *et al*. 2014) was amended to acknowledge the fact that the modelled boundaries are unlikely to fully capture areas of importance during the incubation phase of the breeding cycle. The second point of concern raised by the reviewers led to extended discussion between the reviewers, JNCC and BiOSS. As part of this process, independent advice was sought from Dr Geert Aarts (AEW Wageningen University). In summary, the conclusion of those discussions, agreed by all, was that the methods used by JNCC and BioSS were sound and appropriate, but that further clarification was needed in the text of the report. As a result of these discussions, the relevant section of the report (Box 1 in Wilson *et al.* 2014) was amended.

The reports on the little tern field work methodology and results and subsequent boundary setting work were also put out to independent peer review by JNCC. One main point made by the peer reviewer(s) was that the boat and shore-based observations should have been corroborated more extensively with data from radio tracking or even habitat modelling. JNCC did in fact use radio tracking, at one site, where it confirmed the results of their techniques. JNCC did not consider it to be necessary or even practicable to apply this approach more widely. JNCC considered that habitat modelling was not possible, given the small range of the species and the limited availability of environmental data over that range. JNCC noted that it would have been prohibitively expensive to collect their own environmental data, even at a few sites, and with unknown chance of "success". The other main point made by the peer reviewers (in accord with the same suggestion made by the peer reviewers of the larger tern work) was for data to have also been collected during the incubation period. However, as noted above in regard of work on larger terns, it was decided at the outset of the work that the priority should be on the chick-rearing period, because it is probably at this time when little terns face the greatest energetic demands. The focus was on chick-rearing for biological reasons but also logistical ones; JNCC noted that there would have been a risk of obtaining too few data during both incubation and chick-rearing if both periods were studied. One reviewer asked for greater reference to the findings of other studies but JNCC considered this aspect to be sufficient. A number of improvements were made to text, tables and figures by JNCC, on the recommendation of the reviewer, and some additional text was included in the Discussion to serve as a Conclusion to the report.

In the light of Natural England's involvement with the review process conducted by JNCC and in the light of its outcomes, Natural England did not consider it necessary to initiate its own independent expert review of the reports prepared by JNCC.

Internal peer review and quality assurance

A representative of Natural England has been involved in the entire history of the larger and little tern monitoring and modelling work programme since its inception. Since late 2009, this role was fulfilled by Dr Richard Caldow (Senior Environmental Specialist: Marine Ornithology). Accordingly, Natural England has, in conjunction with Scottish Natural Heritage (SNH), Natural Resources Wales (NRW) and Department of the Environment Northern Ireland (DoENI), been in a position to review and provide quality assurance of the programme of JNCCs work and its findings from start to finish as detailed below.

JNCC evidence reports relating to marine SPA identification go through an extensive internal and external QA process. This has applied to all of the main strands of analysis (ESAS analyses to identify offshore hotspots of usage, inshore wintering waterbird work, larger tern work, and little tern work).

The general approach and survey methods are subject to internal and external discussion, often in workshop format. External discussion can involve organisations such as SNCBs who will use the outputs, academics and other researchers in the field. Once an approach and survey method has been agreed and data collection has started, interim reports are prepared which are subject to internal and SNCB review. Analysis of data is subject to discussions (and workshops if appropriate) internally and with academics and statistical contractors if appropriate. For particularly challenging analyses (such as larger tern modelling work) statistical contractors may undertake significant portions of exploration and development work, and/or of final analysis. Finally, once all the data has been collected and analysed, JNCC prepare an extensive report which has contributions from several JNCC staff, undergoes several rounds of JNCC and SNCB comment, and is finally signed off at JNCC Grade 7 level. At this stage it goes to SNCBs for use in their own work in parallel with going to external peer review, where a minimum of 2 reviewers are sought. Reviewers are usually sought with knowledge of the species ecologies and/or statistical and technical understanding, with reviewers sought to complement each other (for example with differing expertise, from differing types of organisation). JNCC then respond to peer reviews, making changes to 'final' reports if appropriate. Only if peer review comments are significant and fundamental is further grade 7 sign off sought before publishing as part of the JNCC report series.

Departmental Briefs are drafted by an ornithologist with support from the site lead who provides the local site specific detail (in this case Richard Caldow with support from Tim Dixon, Martin Kerby and Katie Finkill-Coombs). The document is then quality assured by the marine N2K National Project Management team as well as selected members of the Project Board. The brief is circulated for external comments from Defra Marine Policy Officer, JNCC ***, Marine Protected Area Technical Group and UK Marine Biodiversity Policy Steering Group. The briefs are also sent to Natural England Board members for early sight of SPA proposals. The amended briefs are then reviewed and approved by the Marine N2K Project Board, Marine Director and relevant Area Managers and subsequently by the Natural England Chief Scientist. The brief is then signed off as required by our Non-Financial Scheme of Delegation and in accordance with our Quality Management Standard by a representative of the Senior Leadership Team with delegated authority before being submitted to Defra.

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