Departmental brief:

Outer Thames Estuary potential Special Protection Area

Natural England and JNCC

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Contents

Summary

Outer Thames Estuary potential Special Protection Area (pSPA) detailed in this Departmental Brief is proposed to protect important areas of coast and sea used for a variety of purposes by the qualifying features. The new pSPA enlarges the existing Outer Thames Estuary SPA (classified solely for non-breeding red-throated divers *Gavia stellata*) to include three new areas identified for foraging terns breeding at other (already classified) SPAs on shore; these are parts of the Rivers Yare and Bure, a small riverine section at Minsmere, and both estuarine and marine areas around Foulness. The pSPA therefore comprises areas for foraging breeding seabirds and non-breeding waterbirds. The feature of the existing SPA is retained, and new qualifying features are added based on a review of up-to-date bird abundance information. The total area of the Outer Thames Estuary pSPA is approx. 391,910 ha (3919 km^2).

The two species of tern relevant to the pSPA are common tern *Sterna hirundo* and little tern *Sternula albifrons*. From north to south, the adjacent SPAs with these tern species as qualifying features (all little tern unless stated) are: Great Yarmouth North Denes SPA; Breydon Water SPA (common tern only); Benacre to Easton Bavents SPA; Minsmere – Walberswick SPA; Alde-Ore Estuary SPA; Foulness SPA (common tern and little tern); and Thanet Coast & Sandwich Bay SPA. In addition to these, common and little terns breeding at Scroby Sands, a sand bank completely contained within the pSPA, and other coastal nesting locations functionally linked to terrestrial SPAs, are included in determining the abundance of terns at the site.

However, Sandwich terns at the Alde-Ore Estuary and Foulness SPAs are not included in determining the details of the pSPA because the feature has been absent at these SPAs for too long to merit influencing the size and shape of the site (Wilson *et al*. 2014). Marine extensions to Hamford Water SPA are the subject of a separate Departmental Brief and do not influence the Outer Thames Estuary pSPA, whilst small numbers of little terns at Colne Estuary, Blackwater Estuary and Medway Estuary and Marshes SPAs are not expected to forage within the marine pSPA based on generic foraging models (Parsons *et al*. 2015).

This Departmental Brief makes use of the most recent available estimates of the population sizes of these species at these sites to derive the populations of birds supported by the pSPA. However, in respect of the existing classified (terrestrial) SPAs, this Departmental Brief does not make any proposal to add or remove qualifying features, amend baseline population figures, or alter site boundaries.

This Departmental Brief sets out the scientific case for the classification of the Outer Thames Estuary pSPA. This 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 three species listed in Annex I of the Birds Directive. Therefore, the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.1).

Table 1 Summary of qualifying ornithological interest in Outer Thames Estuary pSPA

Species	Count (period)	% of subspecies or population	Interest type	Selection criteria	Status of feature
Little tern *Sternula albifrons* (in breeding season)	746 individuals (2011 – 2015)	19.64% of GB population	Annex 1	Stage 1.1	New
Common tern *Sterna hirundo* (in breeding season)	532 individuals (2011 – 2015)	2.66% of GB population	Annex 1	Stage 1.1	new
Red-throated diver *Gavia stellata* (in non- breeding season)	6,466 individuals (1989 – 2006/07)[1]	38.0% of GB population	Annex 1	Stage 1.1	From existing SPA

[1] Citation value from original Outer Thames Estuary SPA classification, 2010

1. Assessment against SPA selection guidelines

The UK SPA selection guidelines require 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.

Outer Thames Estuary pSPA qualifies under stage 1(1) because it regularly supports greater than 1% of the GB population of three Annex I species; two in the breeding season (little tern, common tern) and one in the non-breeding season (red-throated diver).

1.2. Stage 2

Outer Thames Estuary 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. The pSPA meets most of the Stage 2 criteria indicating the different kinds of importance the site holds.

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

Feature	Qualification	Assessment
1. Population size & density	✓	The site supports comfortably the largest aggregation of red-throated divers in the UK (O'Brien *et al*. 2008). It also supports foraging areas for nearly 20% of the GB population of little terns, and nearly 3% of the GB population of common terns.
2. Species range	✓	The pSPA is the main non-breeding area for red-throated divers in the UK, and is the most south-easterly of sites classified or under consideration. Similarly, south east England supports the bulk of the UK's breeding little terns (Mitchell *et al*. 2004) and the pSPA provides for foraging in this crucial part of their range.
3. Breeding success	✓	Little tern productivity at some colonies contributing to the pSPA has exceeded the UK average of 0.51 chicks per pair (Cook & Robinson 2010) occasionally (e.g. Winterton 2012, 2013; Benacre to Easton Bavents 2014: RSPB data). Common tern productivity is estimated to fluctuate nationally between an average 0.7 and 0.3 (Wilson *et al.* 2014); productivity at Breydon Water SPA exceeds this average in most years (RSPB data) and is likely to be especially high

		(perhaps 1.7 chicks per pair) at Foulness SPA. The pSPA directly contributes to productivity, as food resources are contained within it.
4. History of occupancy	✓	Large aggregations of red-throated divers began to be discovered through a programme of aerial surveys between 2001 and 2006 (O'Brien *et al*. 2008). Therefore there is a history of occupancy dating back almost 15 years, although it is highly likely divers were present before our knowledge developed. Breeding little terns and common terns have bred at locations adjacent to the pSPA for many years, meaning several sites were classified as SPAs from the early 1990s. There is every reason to believe the foraging areas within the pSPA would have been used for an equal period, given the foraging ranges of the relevant terns are unlikely to have changed significantly.
5. Multi-species area	✓	Three features qualify in total.
6. Naturalness	N/A	No longer applicable, following ruling from the SPA & Ramsar Scientific Working Group.
7. Severe weather refuge	**?**	No data are available to determine whether the pSPA acts as a severe weather refuge for red-throated divers. Numbers of divers within the pSPA do fluctuate, but the reasons are imperfectly understood.

2. Rationale and data underpinning site classification

In 1979, the European Community adopted Council Directive 79/409/EC on the conservation of wild birds (EEC, 1979) 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. This 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 the most suitable territories in size and number for rare or vulnerable species listed in Annex I (Article 4.1) and for 'regularly occurring migratory species' under Article 4.2 of the Directive. These sites are known as Special Protection Areas (SPAs) in the UK. Guidelines for selecting SPAs in the UK were derived from knowledge of common international practice and based on scientific criteria (JNCC, 1999).

According to Stroud *et al.* (2001), the task of identifying a coherent network of terrestrial sites in the UK is largely complete, comprising of 243 sites of which some include areas used by inshore non-breeding waterbirds, for example in estuaries. However, the JNCC's SPA Selection Guidelines do not review requirements of birds using 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:

Strand 1: seaward extensions of existing seabird breeding colony SPAs beyond the low water mark;
Strand 2: inshore feeding areas used by concentrations of birds (e.g. seaduck, grebes and divers) in the non-breeding season; and
Strand 3: 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:

Strand 4: other types of SPA (JNCC, 2011) that would identify some important areas for marine birds that may not be included within the above three categories and will be considered individually

To implement conservation measures under **Strand 1**, the JNCC produced generic guidance (McSorley *et* *al.* 2003, 2005, 2006; Reid & Webb 2005) to extend the seaward extent of SPA boundaries from seabird colonies. The seaward extensions of existing boundaries in these cases 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. However, these generic boundary extensions 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. 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). However, in line with the recommendations of Reid & Webb (2005), generic extensions have only been implemented at sites holding certain seabird species, none of which occur as breeding birds within the existing SPAs which border the Outer Thames Estuary pSPA. Reid & Webb (2005) note that no evidence has been found that any of the five species of tern which breed regularly in Great Britain make significant use of waters around their colony for maintenance activity (McSorley *et al.* 2003) and conclude that generic guidance for extension of colony SPAs for this purpose is not appropriate in the case of terns.

The original Outer Thames Estuary SPA was classified under **Strand 2** in 2010. Classification was for the marine area supporting a peak mean value of 6,466 red-throated divers in the non-breeding season (JNCC, 2011). As no boundary changes are proposed for this species, and as insufficient contemporary data are available to revise the citation value, this Departmental Brief will not focus on the scientific case for inclusion of this species. The starting position is that this original feature is retained, and all further justification relates to tern foraging areas (which mainly overlap red-throated diver non-breeding areas).

All five species of tern that regularly breed in the UK (Arctic tern *Sterna paradisaea*, common tern *S. hirundo,* Sandwich tern *S. sandvicensis,* roseate tern *S. dougallii* and little tern *Sternula albifrons*) are listed 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 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 **Strand 4** 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 due to difficulties in identification of terns and to limited survey coverage closer to shore (terns have limited foraging ranges compared to other seabird species).

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 Solent & Dorset Coast pSPA) and existing sites (such as Hamford Water 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 SPAs when referring to existing classified sites. Where reference is made to an entirely new site, or to an extended site, or to a site including new features being proposed (such as Outer Thames Estuary), it will be referred to as pSPA since the site (if new), or any additional extent or feature is not yet fully classified.

This Departmental Brief sets out information supporting the identification of the qualifying features of the Outer Thames Estuary pSPA and definition of its proposed boundaries. This is based upon the areas of sea identified as being most important to the tern populations that comprise the qualifying features of this new marine SPA, i.e. terns breeding at the existing Great Yarmouth North Denes, Breydon Water, Benacre to Easton Bavents, Minsmere – Walberswick, Alde-Ore Estuary, Foulness and Thanet Coast & Sandwich Bay SPAs, as well as some functionally linked nesting locations.

SPA site selection guidelines have been applied to the most up to date data for the site.

2.1. Data collection – defining the suite of breeding features and numbers supported by the Outer Thames Estuary pSPA

The size of each of the populations of terns supported by the Outer Thames Estuary pSPA, and which exceed the SPA qualifying thresholds, have been derived as the sum of the numbers of those species at each of the existing SPAs from which the individuals recorded at sea within the pSPA are most likely to originate. Citation figures from existing SPAs have not been used to calculate the Outer Thames Estuary pSPA population. These figures are considered out of date and therefore inappropriate for use in defining the sizes of the populations of these species supported by the entirely new pSPA. Therefore, for each of the source SPAs, the numbers are the most recently available from the Seabird Monitoring Programme (SMP) database (i.e. within the last five years), unless otherwise indicated. Where necessary and possible, this dataset has been augmented by information requested directly from colony managers, from relevant reports (Parsons *et al*. 2015; Norfolk Bird & Mammal Reports), from the national bird ringing scheme, and from the LIFE+ little tern project.

The pSPA population calculation excluded: i) numbers of any terns that may forage within Outer Thames Estuary pSPA, but derive from breeding colonies that are situated outside of existing SPAs, apart from those with strong evidence of functional linkage between SPAs and alternate nesting locations; ii) numbers of terns at existing SPAs which are not qualifying features of these sites and not currently present in numbers exceeding SPA selection criteria thresholds at those sites; iii) numbers of terns at existing SPAs which, although qualifying features of those sites are no longer present in such numbers at those particular sites, and do not meet selection criteria when summed across all source SPAs that might contribute to the pSPA (e.g. Sandwich tern). These exclusions were made to ensure that the size and shape of the pSPA were determined by the foraging requirements of the large numbers of birds originating from the principal source colonies and not unduly influenced by the inclusion of areas of sea that might be used only by relatively small numbers of birds from colonies that do not meet SPA selection criteria thresholds.

2.2. Defining the boundary of Outer Thames Estuary pSPA

The overall boundary of the Outer Thames Estuary pSPA is largely unchanged from the existing SPA, defined according to the distribution of non-breeding red-throated divers (O’Brien *et al*. 2012). However, some additional nearshore areas are proposed to allow for tern foraging requirements. The work done to identify important areas for little and larger tern species differed and was conducted separately (Wilson *et al*. 2014; Parsons *et al*. 2015). These separate pieces of work are described in brief in the following two sub-sections. The overall site boundary was drawn as a composite of the separate species-specific boundaries and this is described in section 3.4.

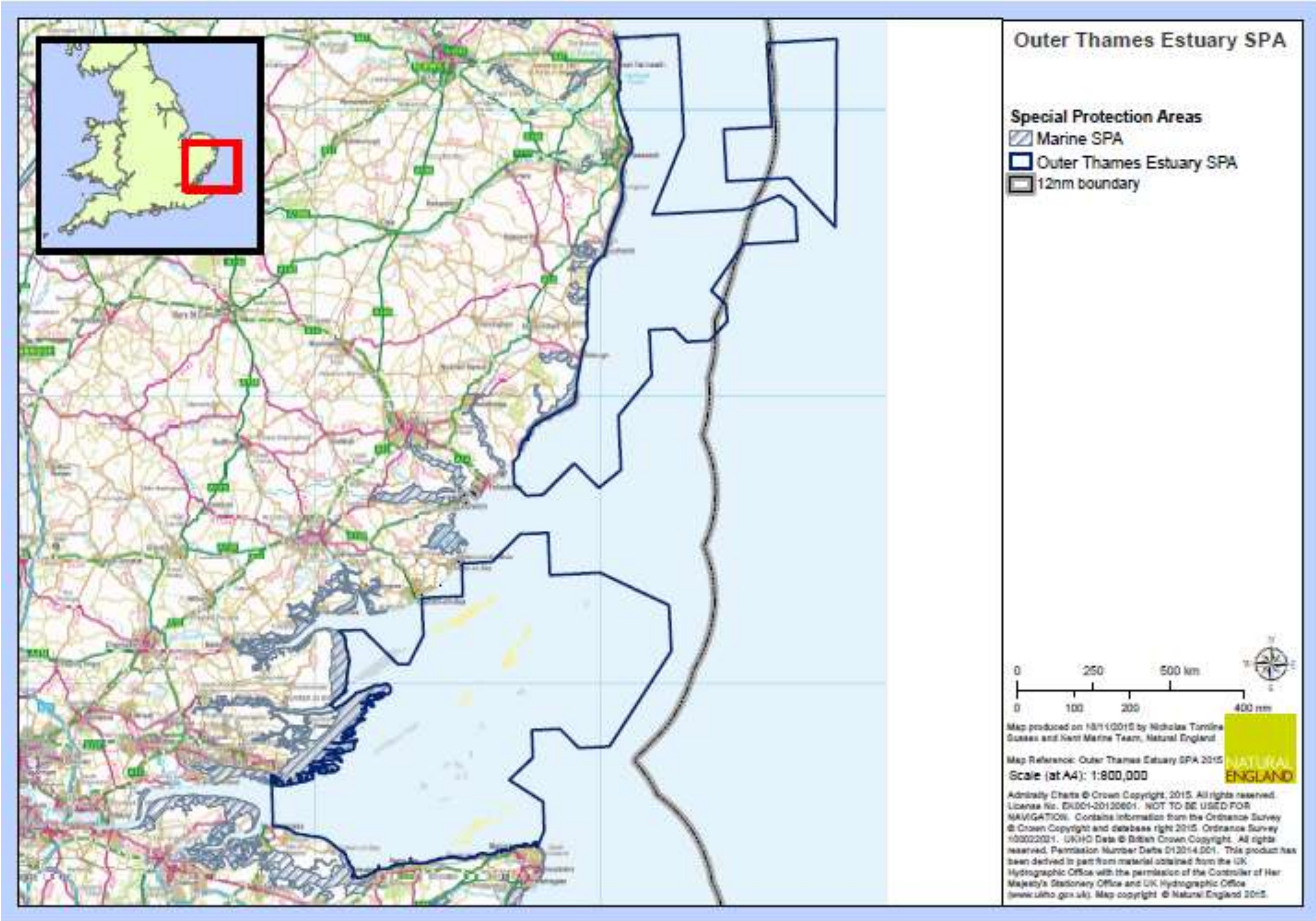
3. Site Status and Boundary

3.1. Existing Boundary

The total area of the existing Outer Thames Estuary SPA is approx. 379,268 ha (3792 km^2).and is divided into three main areas (refer to Figure 1):

- The outer estuary (east of a line north from Sheerness, Kent to Shoebury Ness, Essex);
- A separate area extending south along the coast from East Norfolk (from Caister-on-Sea) to Woodbridge, Suffolk; and
- An area lying offshore slightly further north.

Figure 1. Existing Outer Thames Estuary SPA boundary



Generally, the landward boundary of the existing SPA follows the Mean Low Water (MLW) mark or the seaward boundaries of existing coastal SPAs along most of its length (whichever is the further seaward). The coastal SPAs which directly abut the site from north to south are:

- Great Yarmouth North Denes SPA
- Benacre to Easton Bavents SPA
- Minsmere-Walberswick SPA
- Alde-Ore Estuary SPA
- Crouch and Roach Estuaries SPA
- Dengie SPA
- Foulness SPA
- Southend and Benfleet Marshes SPA
- Thames Estuary and Marshes SPA
- Medway Estuary and Marshes SPA
- The Swale SPA, and
- Thanet Coast and Sandwich Bay SPA

Intertidal mudflats and sandbanks separated from the mainland coast by subtidal areas at MLW are within the existing SPA boundary, except where they are within the boundaries of existing coastal SPAs.

The offshore boundary of the site is largely within the 12 nautical mile (nm) zone; however a significant component of the northern section does extend beyond the 12 nm limit. The total area of the existing Outer Thames Estuary SPA is currently approx.. 379,268 ha (3792 km^2).

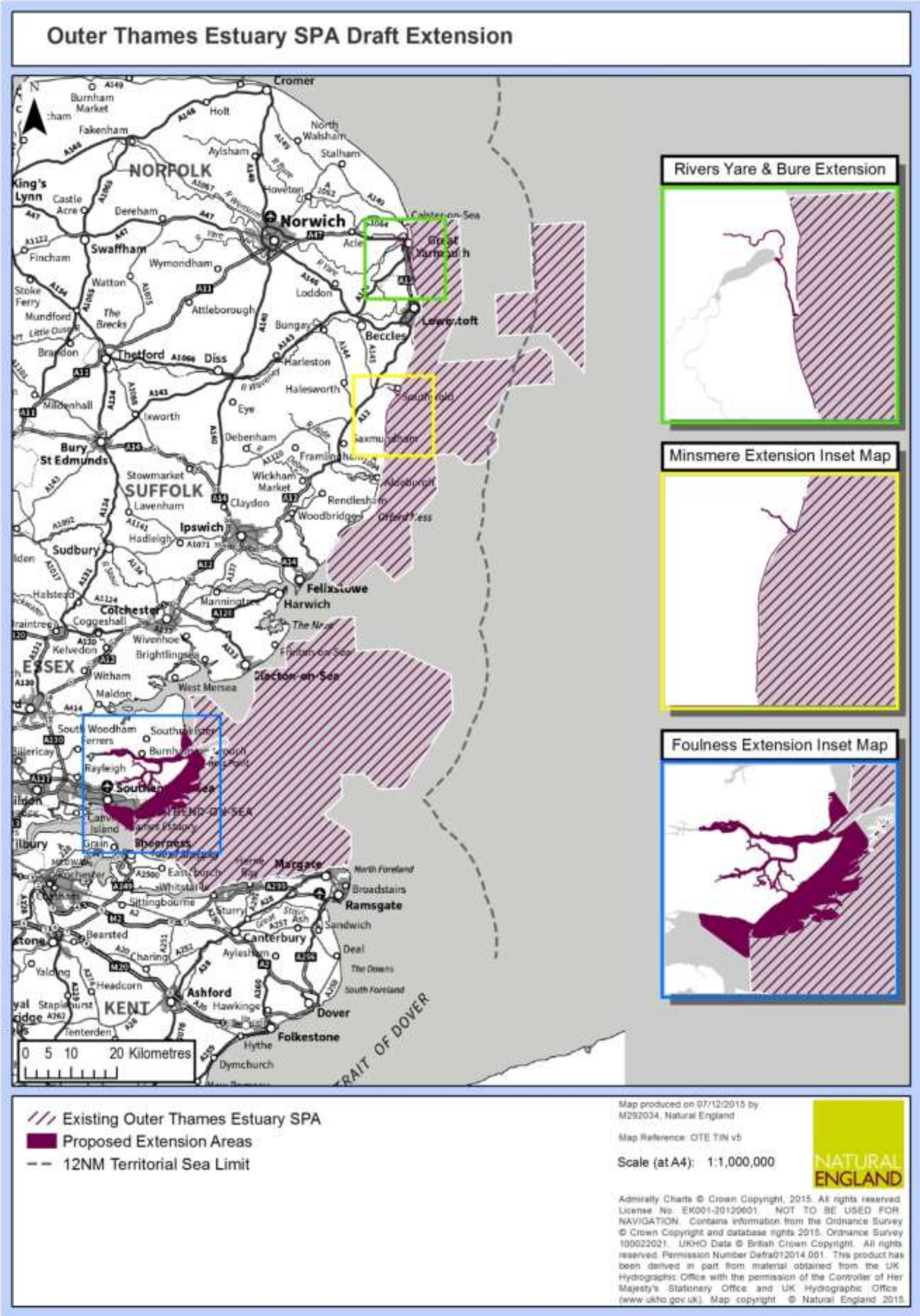
3.2. Outer Thames Estuary pSPA boundary

The total area of the Outer Thames Estuary pSPA is approx.391,909 ha (3919 km^2) - refer to Figure 1a.

The proposed boundary changes to the existing Outer Thames Estuary SPA are based upon projected foraging areas of common terns and little terns breeding within several qualifying coastal SPAs.

The proposed boundary change has been drawn to encompass the qualifying foraging areas of tern species overlaid with maximum curvature derived limits, and has excluded areas that do not support qualifying densities.

Figure 1a - map showing the existing Outer Thames Estuary and the three proposed extensions



3.3. Seaward boundary of the pSPA

There will be no changes to the existing eastern seaward boundary of the Outer Thames Estuary SPA in proposing the boundary extension. The boundary is proposed to extend seaward southwards from the Southend coast driven by the distribution of common terns (Annex 1a). Further information on the extension will be discussed below in section 3.4.

3.4. Landward boundary of the pSPA

The proposed landward boundary of the pSPA is driven by the distribution of both common and little terns which extends in places into the inter-tidal zone (Annex 1a).

Further information on the extension locations are discussed below.

3.4.1. Identification of important marine areas for little terns

Of the five species of tern which regularly breed in Great Britain, little tern is the smallest 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 light of this evidence, JNCC, in agreement with all of the Statutory Nature Conservation Bodies (SNCBs), decided that the most effective method to determine the extent of the area's most heavily used for foraging by breeding little terns would be to undertake a programme of shore based observations and of boat-based transects 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 around 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 little tern observations. Twenty three boat-based transect surveys were undertaken across waters near eight colonies around the UK with a total of 781 little tern observations.

The following sub-sections summarise survey work and boundaries identified at little tern colonies that are qualifying features of SPAs located adjacent to the Outer Thames Estuary pSPA. Further general information on the little tern survey programme is presented in Parsons *et al*. (2015) and Annex 4.

3.4.1.1. <u>Great Yarmouth North Denes SPA</u>

Three shore-based surveys were undertaken in 2013 which collected 937 little tern observations. Two boat-based surveys were also completed in 2013 and recorded 202 little tern observations. These data were supplemented by radio-tracking data collected at the site in preceding years (Perrow & Skeate 2010; Parsons *et al*. 2015). The total number of observations for both shore and boat-based surveys was judged to be sufficient to justify a site-specific approach to boundary definition. The alongshore foraging extent for this colony was set to be 5 km to the north and 4 km to the south. The mean of maximum seaward foraging extents for this colony of little terns was 2.43 km (Figure 2; Parsons *et al*. 2015).

The little tern foraging area is mostly contained within the existing Outer Thames Estuary SPA boundary with the exception of the coastal areas up to Mean High Water (MHW) and therefore the proposed pSPA boundary will be extended to incorporate this area (Annex 1b). However, the northern extent of the foraging areas from Great Yarmouth North Denes SPA overlaps with the proposed Greater Wash pSPA. Because of the tendency for little terns to switch nesting preferences between two colonies within the Great Yarmouth North Denes SPA (at Winterton and North Denes), and because it is not possible to definitively assign foraging areas exclusively to one pSPA, birds at this colony contribute to totals for both pSPAs. This recognises that they could be foraging in either marine pSPA area at any given time.

3.4.1.2. Minsmere - Walberswick SPA

No data were collected for this SPA, as breeding terns were absent during the study period (Parsons *et al*. 2015). It was therefore not possible to apply a site-specific foraging boundary, and instead a generic approach was applied. The alongshore and seaward foraging extents for this colony were set to be the generic values derived from all of the surveys at all of the colonies, i.e. 3.9 km alongshore and 2.18 km seaward (Figure 3). This generic foraging area is mostly contained by the existing Outer Thames Estuary SPA boundary, although the pSPA boundary is proposed to extend inland along the River Blyth to encompass Blythburgh Water, a tidal lagoon directly adjacent to northern parts of the Minsmere – Walberswick SPA. A further expansion along the coast to MHW northwards to Southwold and southwards to Leiston is proposed to incorporate the foraging area (Annex 1c).

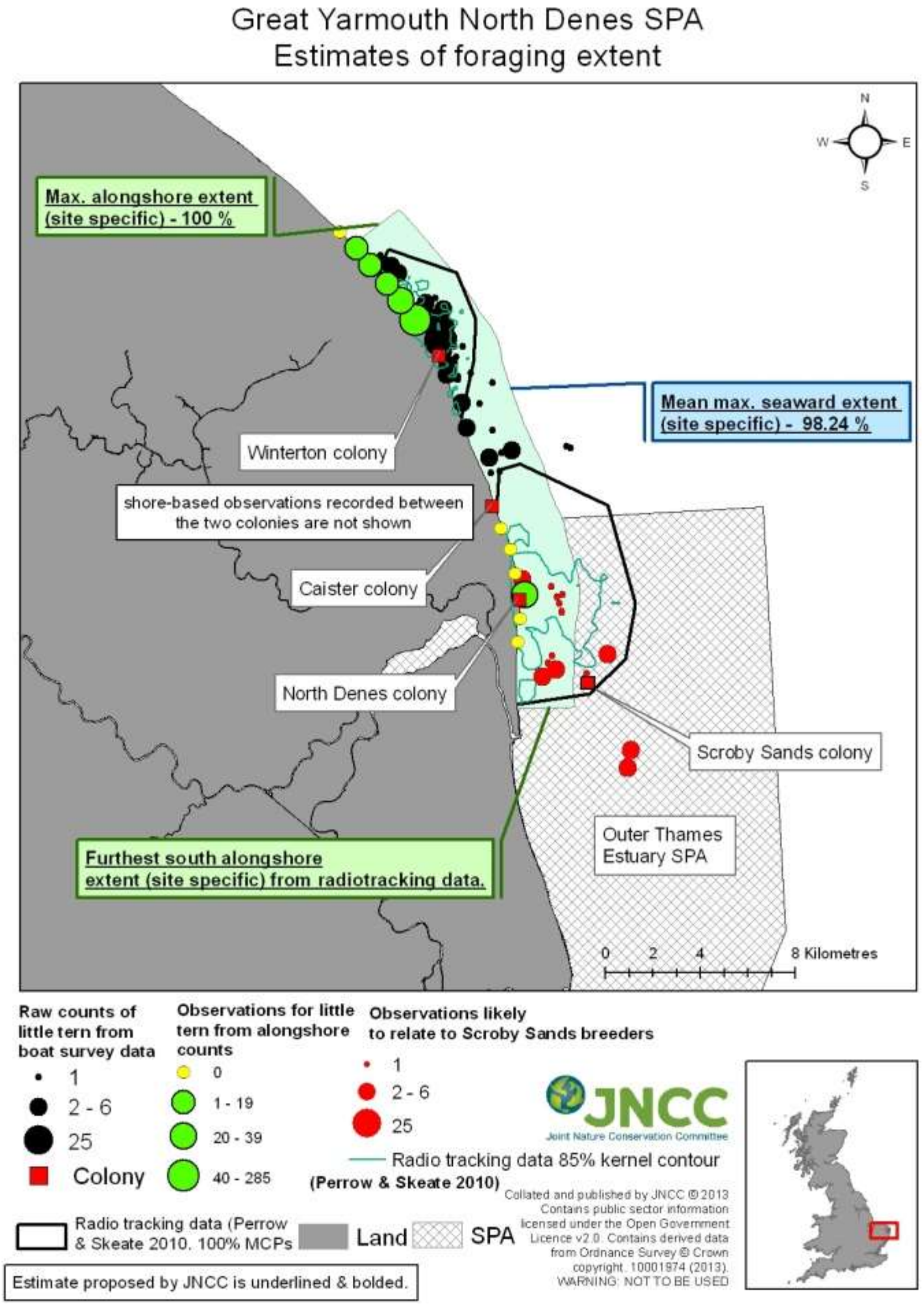


Figure 2. Application of site-specific alongshore and seaward extents to define boundaries for little tern foraging areas around colonies within Great Yarmouth North Denes SPA

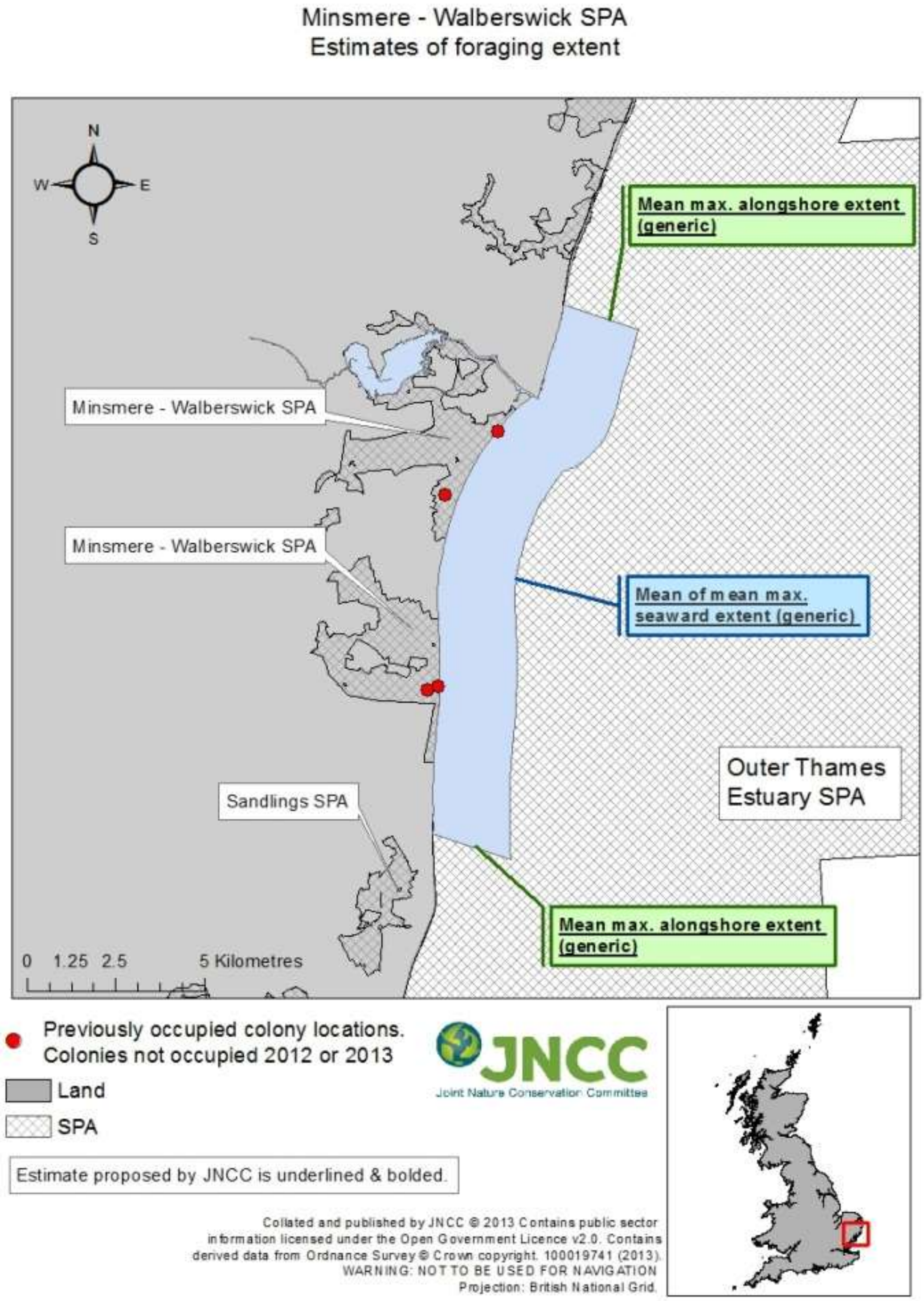


Figure 3. Application of generic alongshore and seaward extents for Minsmere – Walberswick SPA.

3.4.1.3. Alde-Ore Estuary, Benacre to Easton Bavents, Foulness and Thanet Coast & Sandwich Bay SPAs

The Alde-Ore Estuary, Benacre to Easton Bavents, Foulness and Thanet Coast & Sandwich Bay SPAs were amongst a group of sites listed as not regularly occupied (defined as supporting an average of 1% of the GB population in the most recent five year period: Parsons *et al*. 2015). Consequently, no attempt was made to collect data at these sites, or to fit models of expected foraging areas. However, the Outer Thames Estuary pSPA boundary directly abuts these existing SPAs, and therefore the foraging areas of little terns at these sites are by default within the pSPA. Thus, whilst tern foraging areas do not alter the boundary of the pSPA, any terns breeding at these sites do contribute to the abundance total within the site.

3.4.1.4. Scroby Sands

In addition to the above SPAs, the Outer Thames Estuary pSPA contains a breeding colony not currently protected within any SPA citation; Scroby Sands. This is an exposed sand bank lying approximately 6 km offshore from Great Yarmouth, south of the Scroby Sands Offshore Wind Farm, in an area known as South Scroby. There is some evidence that breeding little terns interchange between Great Yarmouth North Denes SPA and South Scroby (section 5.2), meaning Scroby Sands may be considered functionally linked land, and justifying the extension of protection to the Outer Thames Estuary pSPA. When breeding at this offshore site, the foraging area used by little terns is highly likely to be entirely contained within the Outer Thames Estuary pSPA, based on foraging range (Thaxter *et al*. 2012; Parsons *et al*. 2015).

The proposal is that terns at this colony should contribute to the Outer Thames Estuary pSPA abundance total and be recognised as part of the pSPA, because it is contained entirely within the existing SPA boundary and because of the likely connectivity with Great Yarmouth North Denes SPA.

3.4.1.5. Hamford Water, Blackwater Estuary, Colne Estuary and Medway Estuary & Marshes SPAs

Parsons *et al*. (2015) identified Hamford Water SPA as supporting enough terns (between 30 and 45 pairs) to include in their survey programme. Five boat-based surveys took place over 2012 and 2013, with three shore-based surveys also in 2013. Sufficient data were collected to derive a site-specific foraging tern boundary around the SPA, and this is the subject of a separate Departmental Brief.

The Blackwater Estuary, Colne Estuary and Medway Estuary & Marshes SPAs were amongst the group of sites listed as not regularly occupied (Parsons *et al*. 2015). Consequently, no attempt was made to collect data at these sites, or to fit models of expected foraging areas.

When applying the maximum extent of the generic models (3.9 km) in an arc around the location of tern colonies within these SPAs (Old Hall Marshes / Tollesbury Wick; Colne Point; and Deadman's Island, respectively), there is either no overlap or only negligible overlap with the Outer Thames Estuary pSPA boundary. Little terns at these sites are thus not expected to routinely forage within the Outer Thames Estuary pSPA boundary and therefore do not contribute to the abundance total of the pSPA.

3.4.2. Identification of important marine areas for larger terns

The four larger species of tern (common, Arctic, Sandwich and roseate) 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). JNCC, in agreement with all of the SNCBs, decided that the most effective method to determine the extent of the area's most heavily used by larger breeding terns 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. The resultant information on foraging locations chosen by the birds was combined with information on the habitat characteristics of those locations relative to other areas available to construct habitat association models of tern usage. These models were used to predict species specific tern usage patterns around breeding colony SPAs. Usage predictions were made out to the maximum recorded foraging range from each colony. This process of producing usage predictions around colonies for which tracking data had been gathered had colony (and species) specific analysis which produced a smoothed map of foraging usage around the colony. In Phase 2, analysis of pooled data across colonies (species specific) produced generic models which allowed production of maps of smoothed foraging usage around colonies for which no (or insufficient) data were available.

In order to draw a boundary around the most important foraging areas for terns from each colony of interest, a cut-off or threshold value of usage has to be found and only those areas in which usage exceeds that cut-off value included within a possible SPA boundary. An objective and repeatable method to identifying a threshold value, based on the law of diminishing returns, is maximum curvature (O'Brien *et al.* 2012). This method identifies a threshold value below which disproportionately large areas would have to be included within the boundary to accommodate any more increase in, in this case, foraging tern usage. Further details of this work are given in Annex 5.

To gather the empirical data necessary for the modelling, JNCC coordinated a programme of visual tracking work between 2009 and 2011 to identify important foraging areas 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.

Visual tracking was carried out or commissioned by JNCC at 10 of 32 colony SPAs which were deemed to be recently regularly occupied (Wilson *et al.* 2014). Survey effort was prioritised at these 10 sites on the basis of several considerations including: maximising geographical coverage across each species' range, logistical ease of boat-based work, and maximising likely sample sizes (e.g. larger/multi-species colonies with recent successful breeding seasons). As a result no boat-based tracking work was undertaken on the south coast of England.

The total number of tracks obtained was 1,004 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 1,275 tracks available to the project, although not all data were used in the modelling; incomplete tracks or those which recorded no foraging behaviour were excluded.

The following three sub-sections summarise the application of generic boundaries, derived from the modelling of tracking data at other UK tern colonies, to each of the two relevant larger tern colonies within the Outer Thames Estuary pSPA. Further general information on these surveys is presented in Annex 5.

3.4.2.1. Breydon Water SPA

Breeding common terns are qualifying features of Breydon Water SPA. Generic models of foraging behaviour, generated from pooled data obtained from surveys of tern colonies across the UK as described in section 3.4.2, were used to generate boundaries around the SPA. The predictor variables used in the generic models to generate usage patterns of common tern at this SPA were: i) distance to colony, ii) distance to shore, and iii) bathymetry. These variables predicted highest usage around the colony, generally decreasing with increasing distance from it. This means that for the common tern nesting colony located at Breydon Water, only the lower River Yare and part of the River Bure are predicted by the model to be used for foraging by the terns.

The model-generated predictions of relative usage by common terns, together with the boundary drawn around all of the areas in which predicted usage exceeded the threshold identified by application of the maximum curvature approach (to define a limit to the extent of the most important areas) are shown in Figures 4 and 5. The extent of the area of prediction was defined by the limit of the dark blue circles shown (Fig. 4). This reflects the constraint imposed on the modelling by use of a radius the size of the global mean maximum foraging distance from 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.

The predicted usage boundaries largely sit within the existing boundaries of the Outer Thames Estuary pSPA, and thus do not influence it greatly, except along the coast northward to Caister-on-Sea and southward to South of Corton, where the boundary is extended to incorporate the gap between MLW (where the existing Outer Thames Estuary SPA boundary is currently drawn to) and MHW. Also, the Outer Thames Estuary pSPA boundary will be extended inland along the River Yare to meet the existing Breydon Water SPA boundary, and along the lower part of the River Bure approximately to Runham, thus providing no gap in protection across the predicted usage area (Annex 1b).

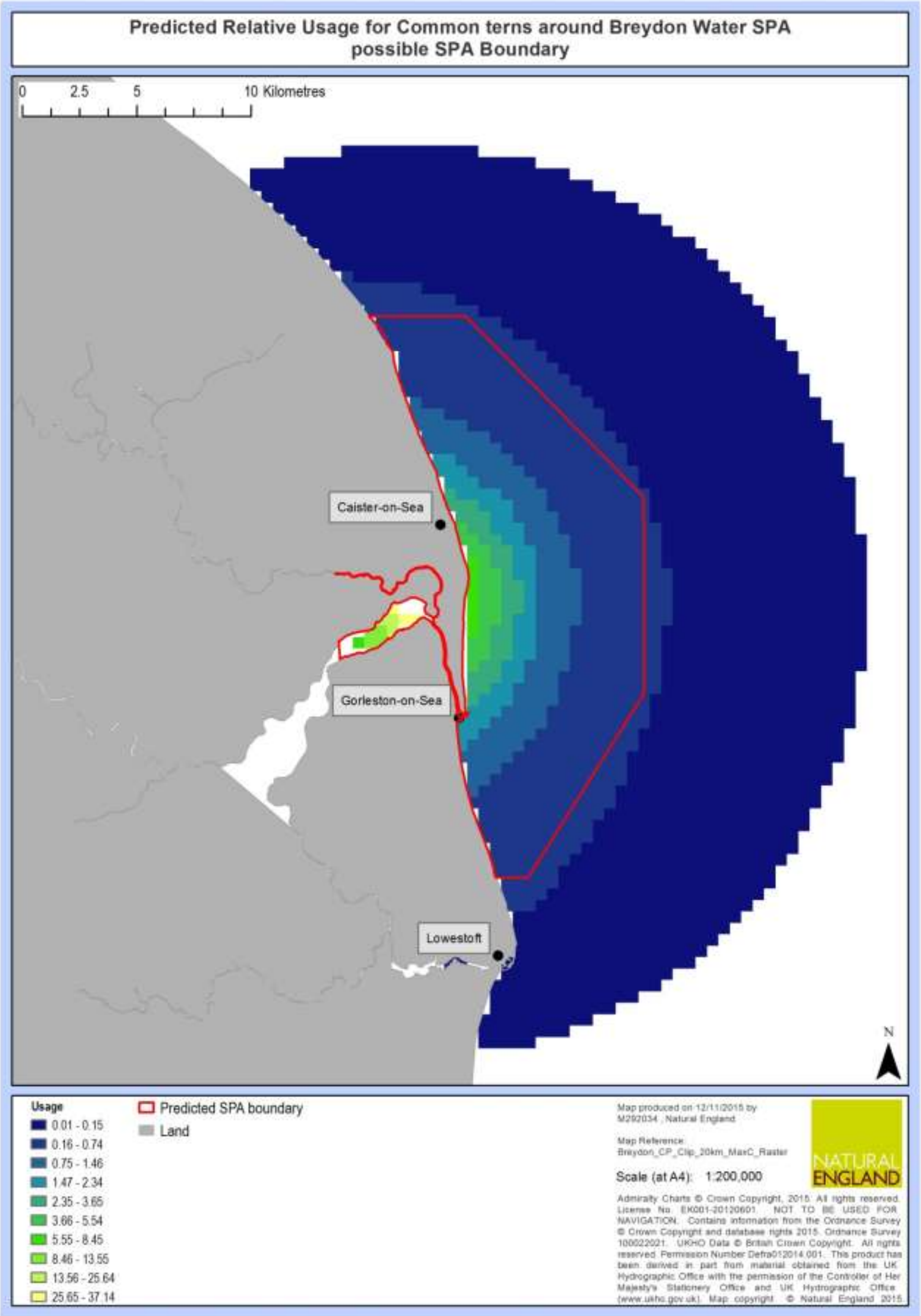


Figure 4. Model predictions of common tern usage overlaid with maximum curvature derived limits to areas of most importance around the Breydon Water SPA. Source: Win *et al.* (2015).

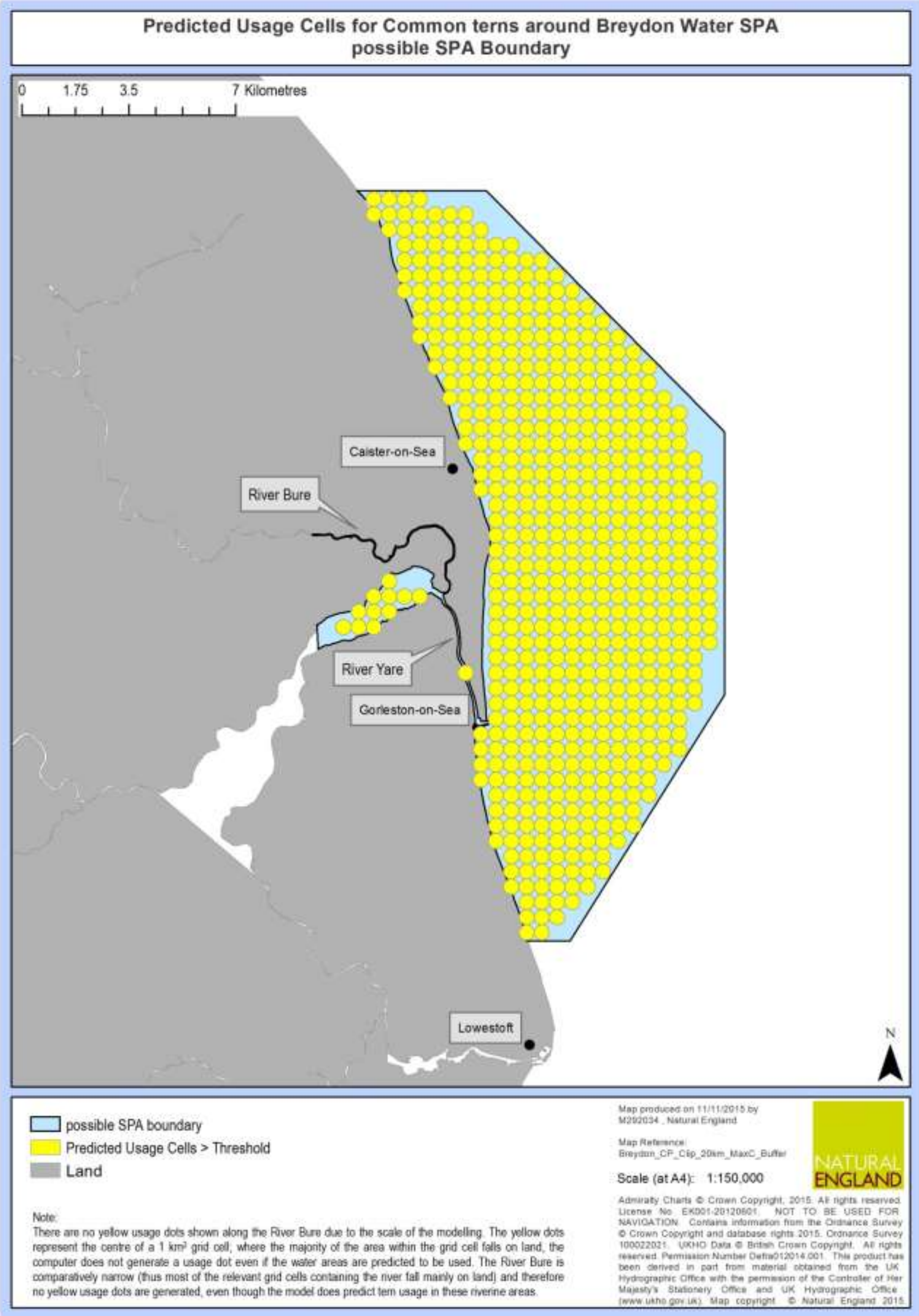


Figure 5 Proposed boundary drawn around the cells within which predicted usage levels by common terns, exceeded the threshold level identified by application of the maximum curvature methodology to the

predicted usage surfaces (see Annex 5). Source: Win *et al.* (2015).

3.4.2.2. Foulness SPA

Breeding common terns are qualifying features of Foulness SPA. Generic models of foraging behaviour, generated from pooled data obtained from surveys of tern colonies across the UK, were used to generate boundaries around the SPA. The predictor variables used in the generic models to generate usage patterns of both species of tern at this SPA were: i) distance to colony, ii) distance to shore, and iii) bathymetry. Predicted usage levels for both species were highest around the colony, generally decreasing with increasing distance from each colony.

The model-generated predictions of relative usage by common terns, together with the boundary drawn around all of the areas in which predicted usage exceeded the threshold identified by application of the maximum curvature approach (to define a limit to the extent of the most important areas), are shown in Figures 6 and 7. The extent of the area of prediction was defined by the limit of the dark blue circles shown (Figure 6). This reflects the constraint imposed on the modelling by use of a radius the size of the global mean maximum foraging distance from 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 which are distant to the colony and/or distant from the shore are predicted to have very little or no usage by foraging terns, therefore these areas have not been included in the proposed boundary.

The predicted usage boundaries largely sit within the existing boundaries of the Outer Thames Estuary SPA, but the pSPA boundary is influenced by the new predicted foraging area. Firstly, it includes the estuarine areas (up to Mean High Water) of the Crouch and Roach Estuaries SPA, approximately as far inland as South Fambridge. As common terns are not a feature of this SPA, which extends down to MLW, the Outer Thames Estuary pSPA will overlap with the Crouch and Roach Estuaries SPA in the relevant intertidal areas (Figure 6). Additionally, the Outer Thames Estuary pSPA boundary will extend seaward to the south and west, overlapping with part of Benfleet & Southend Marshes SPA and then northwards where it will overlap Dengie SPA (none have common terns as a qualifying feature) and also parts of Foulness SPA itself (which does have common terns as a qualifying feature); this is necessary to provide protection in all of the predicted foraging usage areas. Finally, the predicted usage model extends the existing Outer Thames Estuary SPA boundary to the west as far as Westcliffe-on-sea along the Southend coast (Annex 1d).

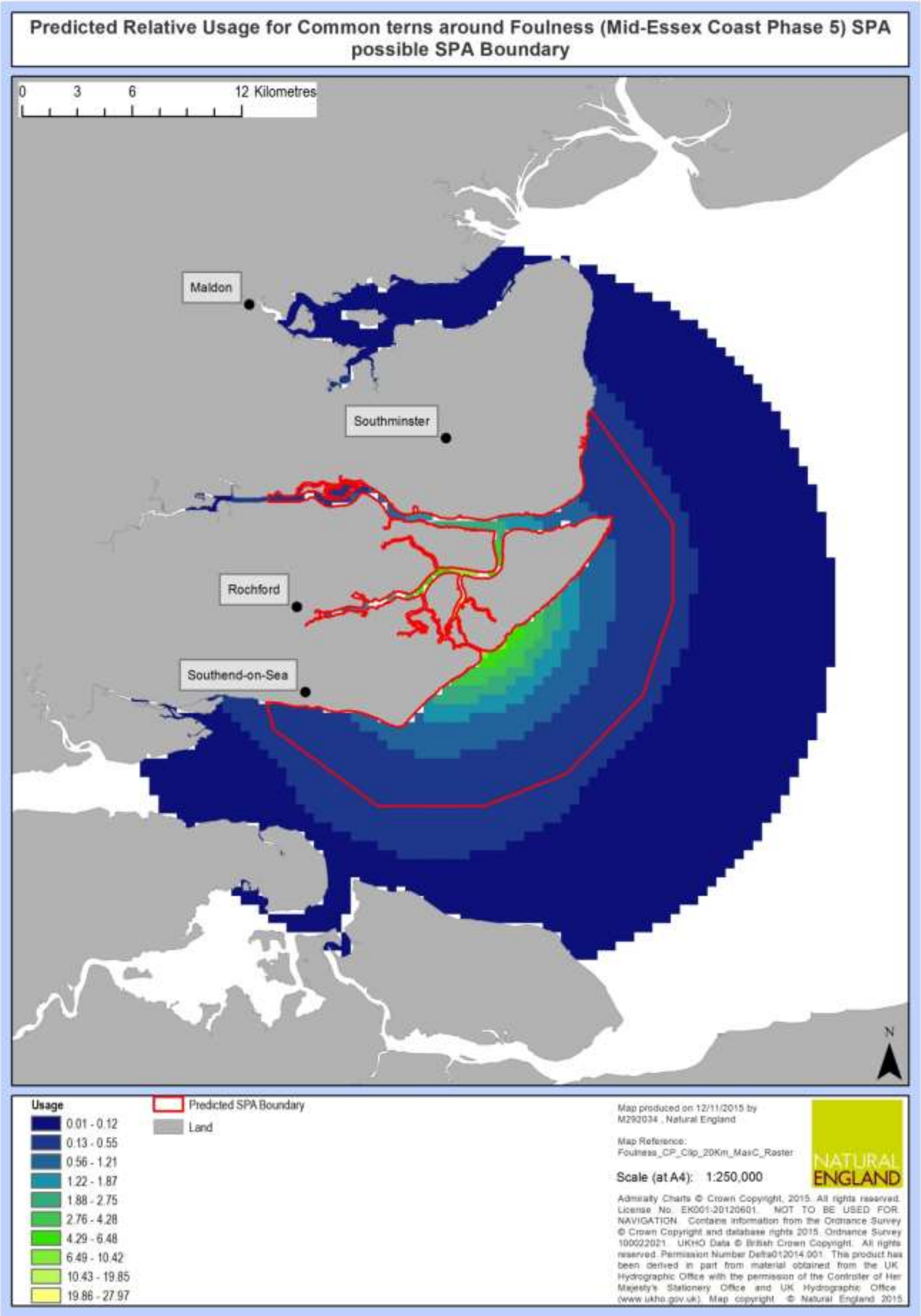


Figure 6. Model predictions of common tern usage overlaid with maximum curvature derived limits to areas

of most importance around Foulness SPA. Source: Win *et al.* (2015).

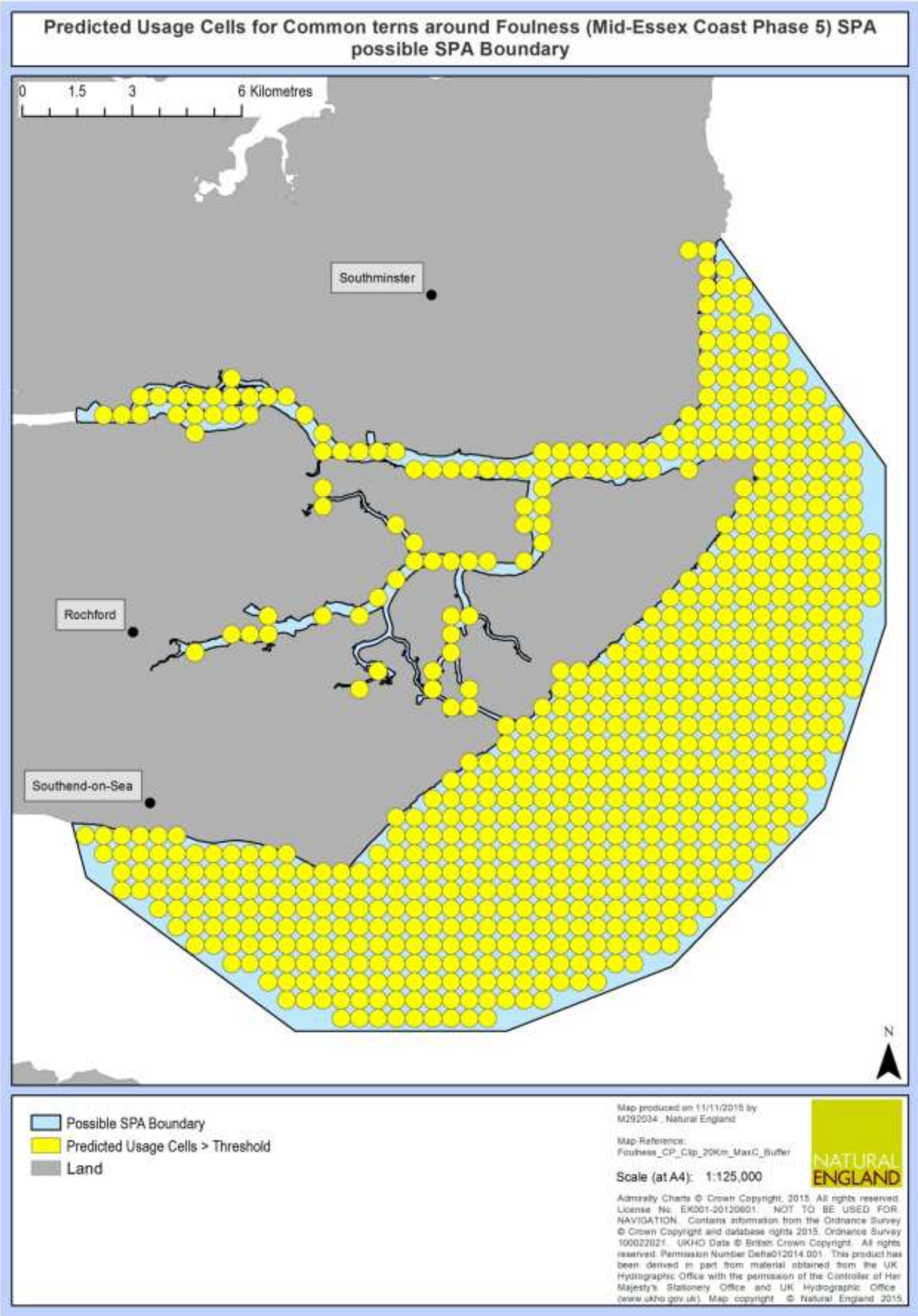


Figure 7. Proposed boundary drawn around the cells within which predicted usage levels by common

terns, centred on the source colony, exceeded the threshold level identified by application of the maximum curvature methodology to the predicted usage surfaces (see Annex 5). Source: Win *et al.* (2015).

3.4.2.3. Sandwich terns – Alde-Ore Estuary and Foulness SPAs

Breeding Sandwich terns are a feature of these SPAs, but they are not considered to be regularly occupied in recent years (Wilson *et al*. 2014). Generic foraging models have not been applied to their parent SPA colonies, and so they do not influence the pSPA boundary; likewise they do not contribute to the total number of terns which the pSPA is expected to support; neither do the Sandwich terns sporadically breeding at Scroby Sands.

3.4.3. Composite boundary of Outer Thames Estuary pSPA

The seaward and alongshore extent of the Outer Thames Estuary pSPA (Annex 1a) is almost entirely determined by the boundaries of the existing Outer Thames Estuary SPA, defined according to the distribution of non-breeding red-throated divers (O'Brien *et al*. 2012). The new areas are:

a. The inclusion of the River Blyth to encompass Blythburgh Water, a tidal lagoon directly adjacent to the northern parts of Minsmere-Walberswick SPA in addition to include MHW areas up the coast (to Southwold) and down the coast (to Leiston) to provide continuous coverage for little terns foraging from this SPA.
b. The inclusion of the River Yare channel, to abut the eastern boundary of the existing Breydon Water SPA, and the lower River Bure, to provide continuous SPA coverage for common terns foraging from this SPA;
c. The inclusion of coastal areas up to MHW up the coast (to Caister-on-Sea) to provide coverage for little terns from Great Yarmouth North Denes foraging from this SPA, and common terns foraging from Breydon Water SPA.
d. The inclusion of coastal areas up to MHW down the coast (to just south of Corton) to provide coverage for common terns from Breydon Water foraging from this SPA.
e. The inclusion of the estuarine areas up to Mean High Water within the Crouch and Roach Estuaries, overlapping the existing Crouch and Roach Estuaries SPA in the intertidal area, to provide SPA coverage for common terns foraging from the existing Foulness SPA;
f. The inclusion of a small additional marine area along the south Essex coast and overlapping part of the Foulness SPA, to the west of the existing Outer Thames Estuary SPA boundary, to provide coverage for common terns foraging from the existing Foulness SPA.

In total, the additional area encompasses 12,642 ha, an increase of 3.3% from the existing SPA area.

Given that the parts of the proposed boundary of the pSPA listed above are determined on the basis of predictions of common tern usage patterns generated by a generic model, rather than a model based on observations of common terns in the Outer Thames Estuary, it is appropriate to consider the reliability of that evidence base. Annex 5 describes the process of cross-validation by which the robustness of each generic model was assessed using standard statistical criteria. This assessment involved assessing the ability of each species-specific, generic model to predict the observed distribution of terns of the species of interest at colonies which were (in the cross-validation process) excluded in turn from building the model. This demonstrated that of the three species-specific, generic models, the Sandwich tern model was the most reliable, with an average test statistic for this cross-validation process that was classed as indicative of the model being "excellent". By the same measure, the generic common tern model was judged to be "good" i.e. better than other possible classes of "moderate", "poor" or "unsuccessful". This analysis indicated that there is reasonable consistency between colonies around the UK in the characteristics of sea areas which hold the highest relative densities of foraging common terns. Accordingly, there is a correspondingly high degree of confidence that the boundary of this pSPA, being partly dependent upon the predicted usage patterns of common terns, is founded on a reliable evidence base, albeit not one derived directly from birds at the colonies in question.

4. Location and habitats

The Thames Estuary is located in the southern part of the North Sea on the east coast of England, between the counties of Essex (on the north side) and Kent (on the south) and extends as a broad opening into the North Sea. The Outer Thames Estuary extends northwards to Caister-on-Sea in Norfolk.

The Outer Thames Estuary pSPA consists of areas of shallow and deeper water (ranging from 0-50 m below sea level), high tidal current streams and a range of mobile sediments. Large areas of mud, silt and gravelly sediments form the deeper water channels, the main ones representing the approach route to the ports of London and as such being continually disturbed by shipping and maintenance dredging. Sand in the form of sandbanks separated by troughs predominates in the remaining areas and the crests of some of the banks are exposed at MLW; Cross Sand, Scroby Sands, Helm Sand, Newcombe Sand, Aldeburgh Napes, Aldeburgh Ridge, North Ship Head and Bawdsey Bank; in the southern part of the site the main sandbanks are Kentish Flats, West and East Barrow, Ray Sand, Foulness Sands, Maplin Sands, Chapman Sands, Southend Sands and Yantlet Flats, Long Sand, Margate Sand and Kentish Knock.

The proposed boundary overlaps various other sites which have been notified or designated under either British or European conservation legislation, such as SSSIs and SPAs. The proposed boundary will overlap with the following coastal SPAs;

- Crouch and Roach Estuaries SPA;
- Dengie SPA;
- Foulness SPA; and
- Benfleet & Southend Marshes SPA

These overlapping areas comprise of inter-tidal mud, sand and saltmarsh in addition to creeks which are key areas where the terns forage. The Outer Thames Estuary pSPA also overlaps with several existing SACs including from north to south;

- Essex Estuaries SAC: designated for a wide range of characteristic marine and estuarine sediment communities; subtidal areas have rich invert fauna.The SAC also has extensive mudflats and sandflats with extensive growths of eelgrass *Zostera spp.* on the open coast.
- Thanet Coast SAC: designated for chalk, having the longest continuous stretch of coastal chalk in the UK with subtidal chalk reefs which extend into the intertidal zone.

Furthermore, the boundary overlaps the following MCZs:

- Blackwater, Crouch, Roach and Colne Estuaries MCZ which is primarily designated for native oyster and native oyster beds.
- Thames Estuary rMCZ which is recommended for designation of the intertidal and subtidal sediments as well as species such as tentacle lagoon worm, European eel and Smelt.
- Medway Estuaries MCZ, which is primarily designated for intertidal and subtidal mud.
- Swale Estuary pMCZ; which is subject to public consultation by Defra. The pMCZ is primarily being recommended for subtidal habitats (mud and mixed sediments).
- Thanet Coast MCZ which is primarily designated for further extensions of chalk reef, intertidal *Sabellaria spinulosa* and also the stalked jellyfish (*Lucernoriopsis cruxmelitensis*).

The seabed in the area of the Norfolk and Suffolk coast is of a similar composition to that in the main estuary with large shallow areas of mud, sand, silt and gravely sediments but, in the absence of main port areas with approaches inside the SPA, there are consequently fewer disturbances through shipping or dredging.

5. Assessment of ornithological interest

5.1. Survey Information and summary

SPA site selection guidelines have been applied to the most up to date information for the site.

Counts of breeding seabirds (and / or young) at the colonies within the existing SPAs (which are also those most likely to be the origin of birds within the marine foraging areas of the pSPA) are from the national Seabird Monitoring Programme (SMP). This dataset has been augmented by information from colony managers and the LIFE+ little tern project (all through RSPB), the Foulness Area Bird Survey Group, data collected for the national bird ringing scheme (administered by the British Trust for Ornithology) by the ringing group at Foulness, and relevant editions of the Norfolk Bird & Mammal Report.

Parameters adopted in transforming numbers of young common terns ringed into numbers of breeding adult pairs at Foulness SPA are outlined in Annex 7.

Details of the work carried out to characterise the foraging areas used by breeding adult terns within the Outer Thames Estuary pSPA are above in sections 2 and 3 and in Annexes 4 and 5.

Data on non-breeding red-throated divers are unchanged from the Outer Thames Estuary SPA citation and the N2K standard data form (JNCC, 2011), outlined in O'Brien *et al*. (2012).

5.2. Annex I species

5.2.1. Breeding season

5.2.1.1. <u>Little tern *Sternula albifrons*</u>

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).

Little terns are a qualifying feature of Great Yarmouth North Denes, Benacre to Easton Bavents, Minsmere – Walberswick, Alde-Ore Estuary, Foulness and Thanet Coast and Sandwich Bay SPAs. Little terns are notoriously transitory in their nesting habits (Brown & Grice 2005) and may move between different colonies in response to factors including disturbance and predation. Because of this habit, the estimates for Great Yarmouth North Denes SPA include figures from Caister (< 1 km from the SPA boundary), Eccles and Scroby Sands (both approximately 6 km from the SPA boundary), all of which are thought to be functionally linked to colonies protected within the Great Yarmouth North Denes SPA. This view is supported on the basis of little variation between the summed totals from year to year (Figure 5.2a), particularly between 2011 and 2014, when little terns were all but absent from North Denes, instead breeding predominantly at Winterton and Scroby Sands. If the Benacre – Easton Bavents SPA is also considered, including an apparently functionally linked site at nearby (< 1 km from SPA boundary) Kessingland, the collective number of little tern pairs averages 392, with a standard deviation of just 40 pairs (2009 – 2015). This provides strong evidence of functional linkage between this group of sites, and provides justification for including data from each of them within the total number of little terns expected to use the Outer Thames Estuary pSPA. Recent shifts to Benacre and Kessingland may reflect a response to targeted site management here, and possibly beach accretion.

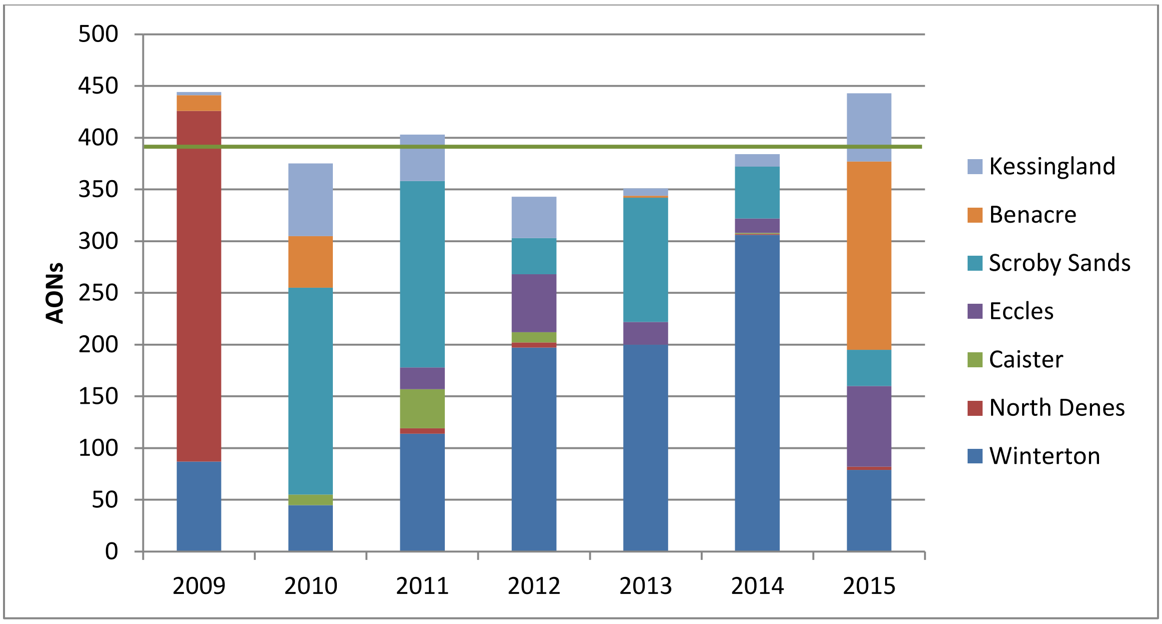


Figure 8. Little tern numbers (Apparently Occupied Nests, AONs, equivalent to adult pairs) at five locations either within or thought to be functionally linked to the Great Yarmouth North Denes SPA (Winterton, North Denes, Caister, Eccles and Scorby Sands) and two either within or thought to be functionally linked to the Benacre to Easton Bavents SPA (Benacre, Kessingland). Green horizontal line shows average for period 2009 – 2015.

Although there is a suggestion of similar functional linkage between little terns breeding within the Alde-Ore Estuary SPA and the sandbanks at the mouth of the River Deben (known as the Deben Knolls), current data suggest only sporadic breeding and do not allow comparable demonstration of linkage with sufficient confidence.

Combined, the SPAs listed and their associated functionally linked nesting sites currently contribute a five year average of 373 pairs (Table 3). This represents 19.64% of the GB population. The pSPA will thus offer protection of foraging areas to a very significant proportion of little terns breeding in Great Britain.

5.2.1.2. 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 15% of the Southern & Western European breeding population (67,000 pairs derived by division by 3 of the upper estimate of 200,000 individuals and rounded to nearest 1,000: 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.

Common terns are a qualifying feature of Foulness and Breydon Water SPAs. The species still nests at both sites. At Foulness SPA, the five year mean (2011 – 2015) of 17.5 pairs derives from counts of adult pairs and counts of ringed young breeding at New England Creek (Annex 7). The five year mean at Breydon Water SPA for the same period is 104 pairs.

Common terns also breed on the sandbanks at Scroby Sands, along with little terns. It is likely that the common terns nesting here are functionally linked to the Breydon Water SPA population; as numbers at Breydon Water have declined since Scroby Sands has become exposed, numbers at Scroby Sands have generally increased (Figure 9). The average number of common tern pairs for the two areas combined is 235, with a standard deviation of 54.5 pairs (2009 – 2015). This suggests annual variation is limited, especially with the apparently anomalous large count in 2013, and provides evidence of functional linkage between Breydon Water SPA and Scroby Sands. This provides justification for including data from each of them within the total number of common terns expected to use the Outer Thames Estuary pSPA.

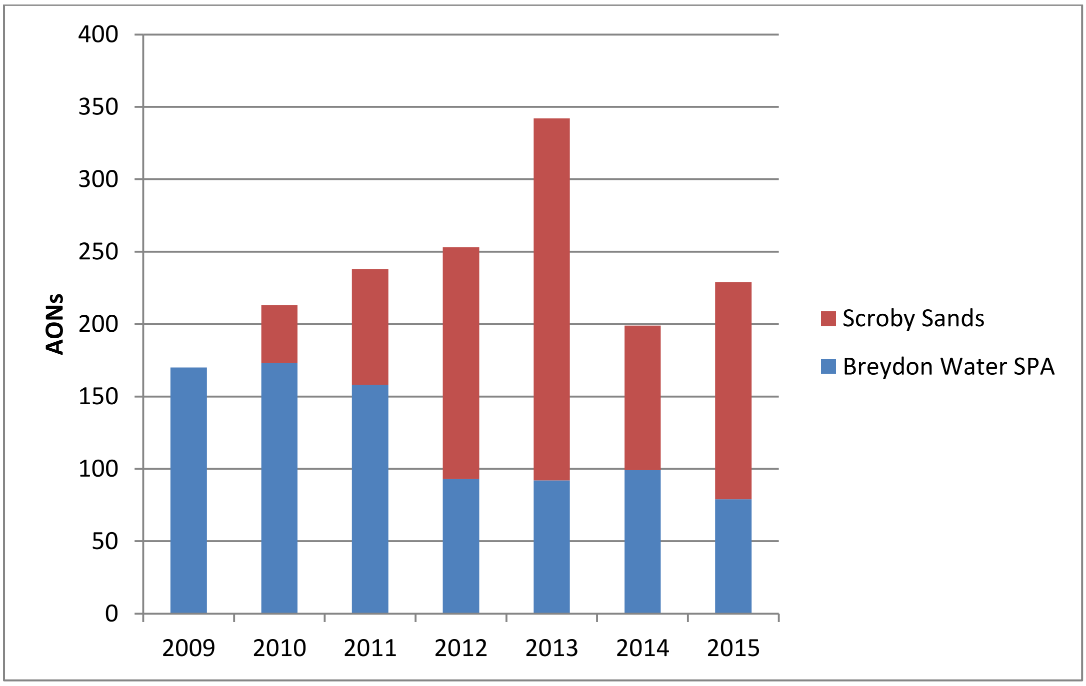


Figure 9 Common tern numbers (Apparently Occupied Nests, AONs) at Scroby Sands and Breydon Water SPA 2009 – 2015.

Combined, Foulness SPA, Breydon Water SPA, and the associated functionally linked nesting site at Scroby Sands currently contribute a five year average of 266 pairs (Table 5.2). This represents 2.66% of the GB population. The pSPA will thus offer protection of foraging areas to a significant proportion of common terns breeding in Great Britain.

5.2.2. Comparison of counts for breeding sites

Current data used for the pSPA total are presented here, alongside values from SPA citation forms and N2K Standard Data Forms (Table 3). These are for comparison purposes within this Brief; it is the current data that informs the classification of the site.

Table 3. Counts of terns (pairs) contributing to the Outer Thames Estuary pSPA total, and current five-year means (2011 – 2015), including likely functionally linked breeding sites within SPA totals. Sandwich terns presented for information only (see section 4.2.4). Grey cells indicate where the species is not a feature of the SPA.

	Little tern			Common tern			Sandwich tern		
SPA	*Current*	*SPA citation*	*N2K data form*	*Current*	*SPA citation*	*N2K data form*	*Current*	*SPA citation*	*N2K data form*
Great Yarmouth North Denes	314	277	220						
Breydon Water				252.2	155	155			
Benacre to Easton Bavents	57.6	39	21						
Minsmere-Walberswick	0.8	32	28						
Alde-Ore Estuary	0.8	No data	48				No data	No data	170
Foulness	0	73	>24	17.5	186	220	0	267	320
Thanet Coast and Sandwich Bay	0	30	6						
Current five-year mean (sum)	373.2			266.2					

5.2.3. Non-breeding season

5.2.3.1. Red-throated diver *Gavia stellata*

The non-breeding population of red-throated divers in Great Britain is estimated to be 17,000 individuals (Musgrove *et al.* 2013), mostly distributed in marine areas in the south east of England (O'Brien *et al*. 2008). The original Outer Thames Estuary SPA boundary was determined for red-throated divers, using visual aerial survey data, Kernel Density Estimation and Maximum Curvature Analysis (Natural England 2010 (http://publications.naturalengland.org.uk/publication/3233957); O'Brien *et al*. 2012).

The Outer Thames Estuary pSPA boundary remains largely unchanged from the original SPA classification, and the peak mean value of 6,466 individuals is also unchanged.

5.2.4. Species not currently meeting SPA selection guidelines

Although Sandwich terns are a breeding feature of the existing Alde-Ore Estuary and Foulness SPAs, their continued absence at these sites means their foraging requirements were neither directly measured nor modelled, and they make no contribution to the Outer Thames Estuary pSPA total. Although Sandwich terns are recorded sporadically on Scroby Sands, the species is not present regularly in abundances exceeding the stage 1.1 selection guideline (four year peak mean 70.5 pairs *cf*. 1% GB population threshold of 110 pairs (Musgrove *et al*. 2013); derived from counts of 0 (2012), 2 (2013), 250 (2014) and 30 (2015): data source – RSPB). Thus Sandwich terns are not currently a feature of the Outer Thames Estuary pSPA. This may require review in future if populations recover at the terrestrial breeding sites.

6. Comparison with other sites in the UK

Breeding season

A comparison of the numbers of terns within the Outer Thames Estuary pSPA, derived by summing the most recent five year colony counts 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 6. As the source colony SPAs 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 Outer Thames Estuary pSPA (acknowledging the difference in time periods between derivation of these numbers).

Table 6. Comparison of the average numbers of individuals (and pairs) of each of the features of the Outer Thames Estuary pSPA (2011 – 2015) with those at other SPAs identified (Stroud *et al*. 2001) as supporting those features.

Species	Site	Individuals (pairs)[2]	Rank[34]	Comments
Common tern *Sterna hirundo*	Dungeness to Pett Level SPA	376 (188)	=16th of 23	
	Outer Thames Estuary pSPA	532 (266)	=11th of 23	
	Ythan Estuary, Sands of Forvie and Meikle Loch SPA	530 (265)	13th of 23	
Little tern *Sternula albifrons*	Outer Thames Estuary pSPA	779 (389)	1st of 28	
	North Norfolk Coast	754 (377)	2nd of 28	
	Great Yarmouth North Denes	440 (220)	3rd of 28	

Non-breeding season

The Outer Thames Estuary SPA, when classified, supported 38% of the GB population (five year peak mean of 6,466 birds); the only other classified SPA in the UK (Liverpool Bay SPA) supported 5.4% (five year peak mean of 922 birds). The only other SPA for the species in the UK is the Firth of Forth SPA, supporting 90 individuals.

The Outer Thames Estuary pSPA is therefore the highest ranked site in the UK.

[2] 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.

[3] 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 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 Outer Thames Estuary pSPA, and adding one site to account for the pSPA itself.

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

7. Conclusion

The evidence presented in this Departmental Brief sets out the scientific case for SPA classification, based on peer-reviewed models of tern foraging requirements and red-throated diver distributional data. The proposed boundary changes only slightly in comparison to the original Outer Thames Estuary SPA, and is still largely determined by aggregations of red-throated divers.

The pSPA is internationally important for three species. It will remain the most abundant site in the UK for red-throated divers, and will provide foraging habitat for a combined total of little terns exceeding the single most abundant breeding colony total (being comprised of birds from six source SPA colonies). Also, it will support internationally important numbers of foraging common terns from two source SPA colonies.

In conclusion, the site qualifies as per the original Outer Thames Estuary SPA, with the addition of little tern and common tern features to protect the marine foraging areas used by birds breeding along the adjacent coastline.

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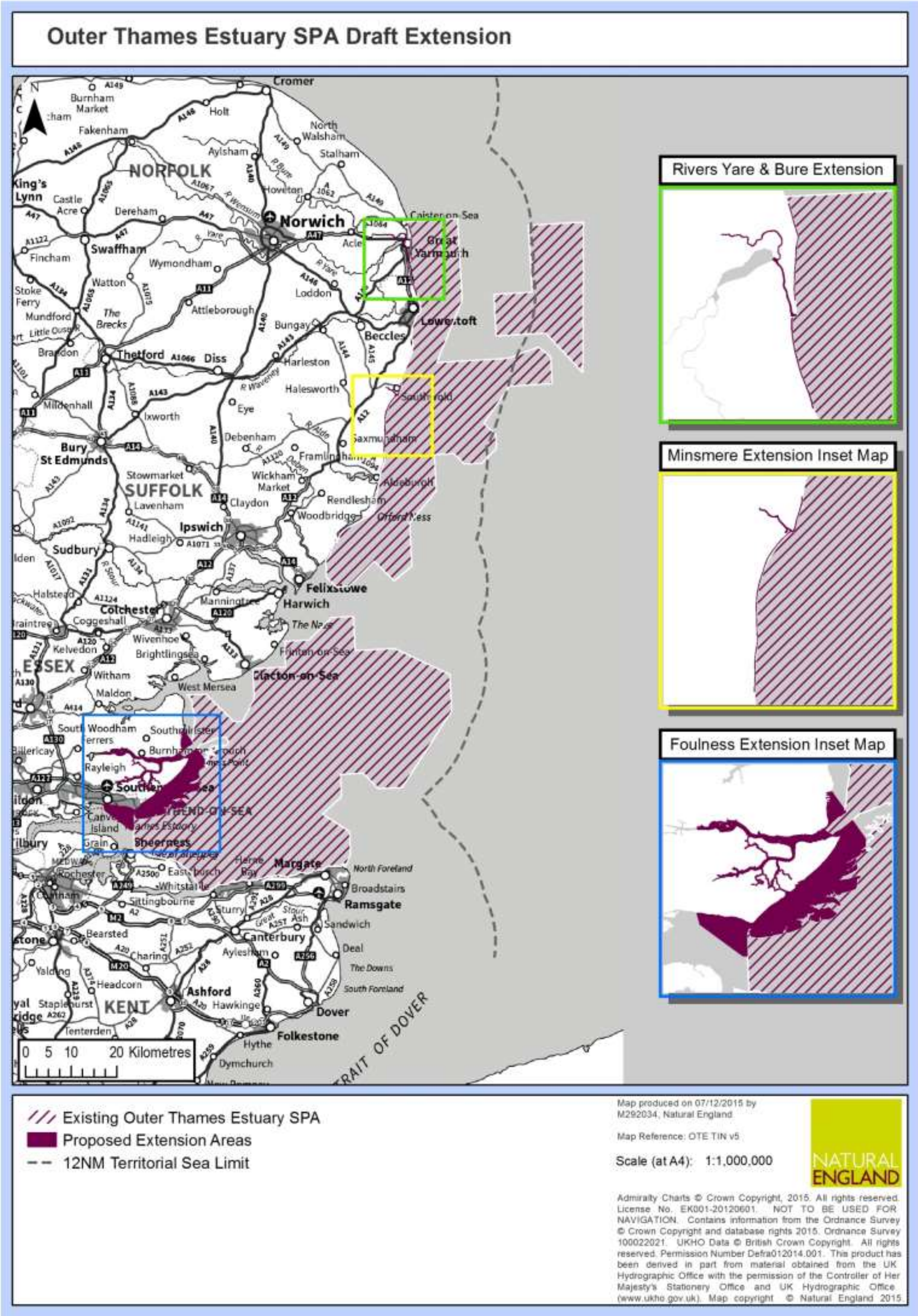
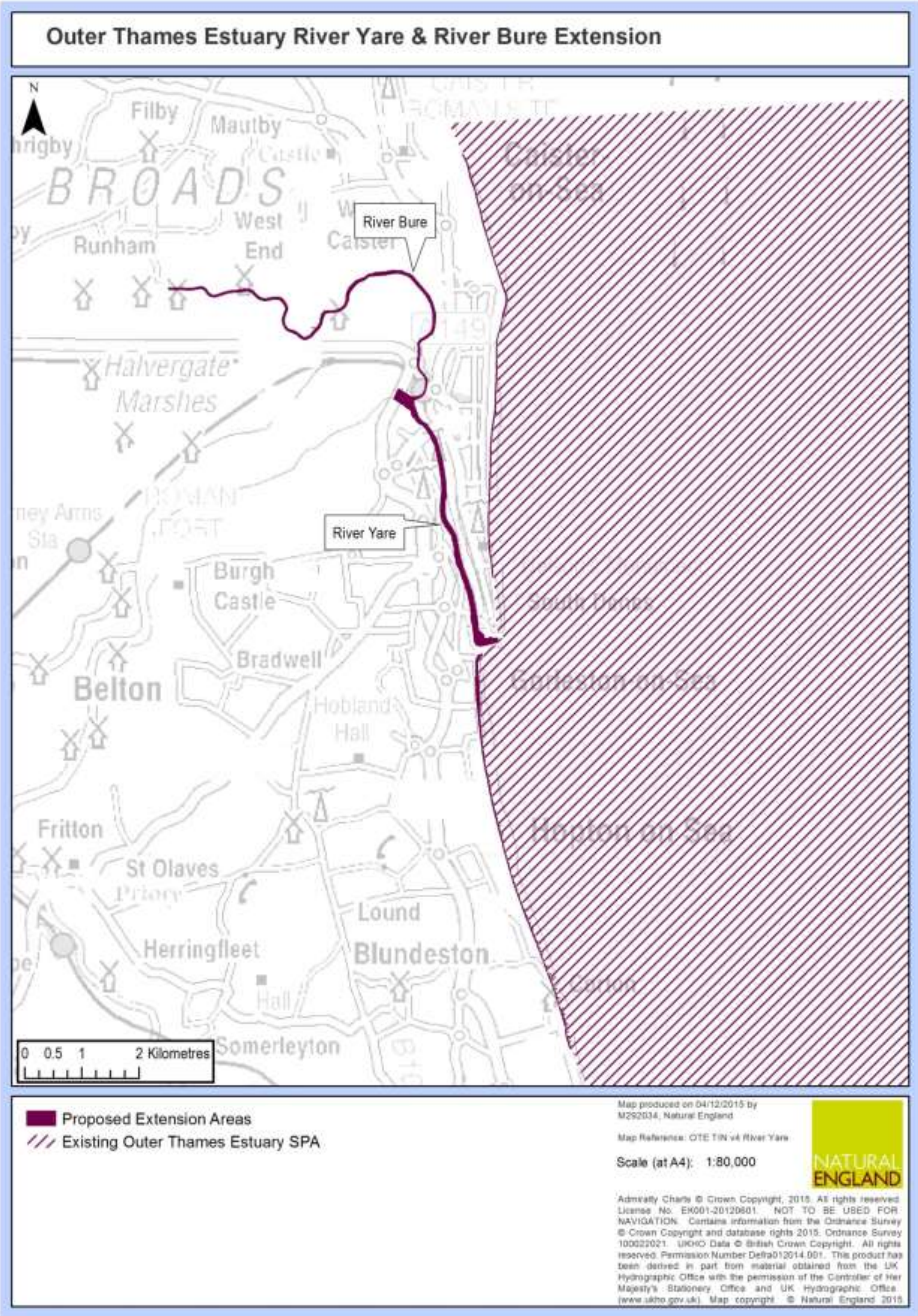
Annex 1 Site maps

Annex 1a



Annex 1b


Outer Thames Estuary River Yare & River Bure Extension
N
Filby
Mautby
BROADS
West End
Runham
Caister
River Bure
Halvergate Marshes
River Yare
Burgh Castle
Bradwell
Belton
Hobland Hall
Caister-on-Sea
South Denes
Gorleston-on-Sea
Hopton on Sea
Fritton
St Olaves
Herringfleet
Lound
Blundeston
Somerleyton
0 0.5 1 2 Kilometres
Proposed Extension Areas
Existing Outer Thames Estuary SPA
Map produced on 04/12/2015 by M292034, Natural England
Map Reference: OTE TIN v4 River Yare
Scale (at A4): 1:80,000
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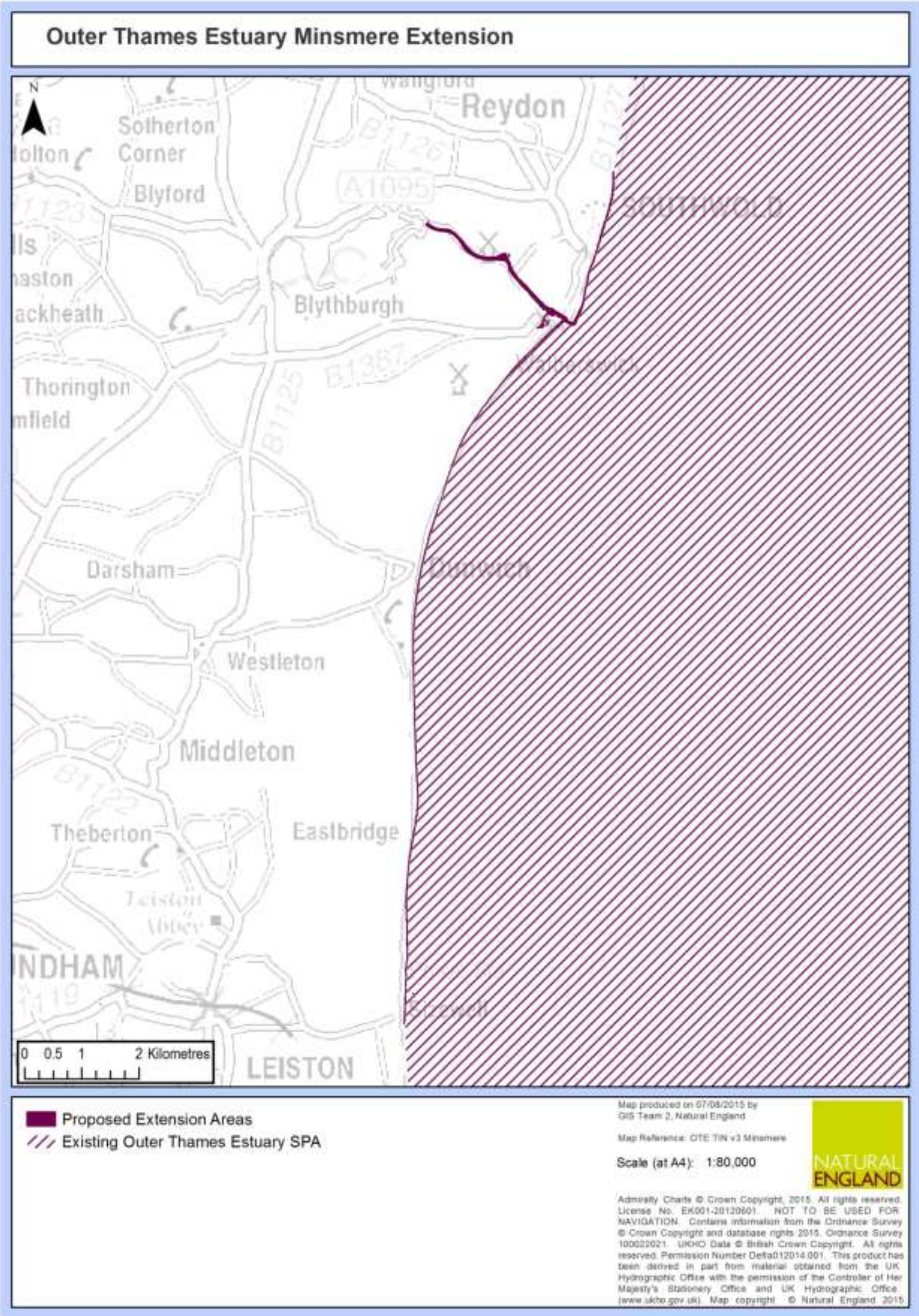
Annex 1c



Outer Thames Estuary Minsmere Extension
N
Reydon
Sotherton
Corner
Blyford
A1095
B1126
SOUTHWOLD
Blythburgh
B1387
Walberswick
Thorington
B1125
Darsham
Dunwich
Westleton
Middleton
B1122
Theberton
Eastbridge
Leiston
Abbey
Sizewell
LEISTON
0 0.5 1 2 Kilometres
Proposed Extension Areas
Existing Outer Thames Estuary SPA
Map produced on 07/08/2015 by GIS Team 2, Natural England
Map Reference: OTE TIN v3 Minsmere
Scale (at A4): 1:80,000
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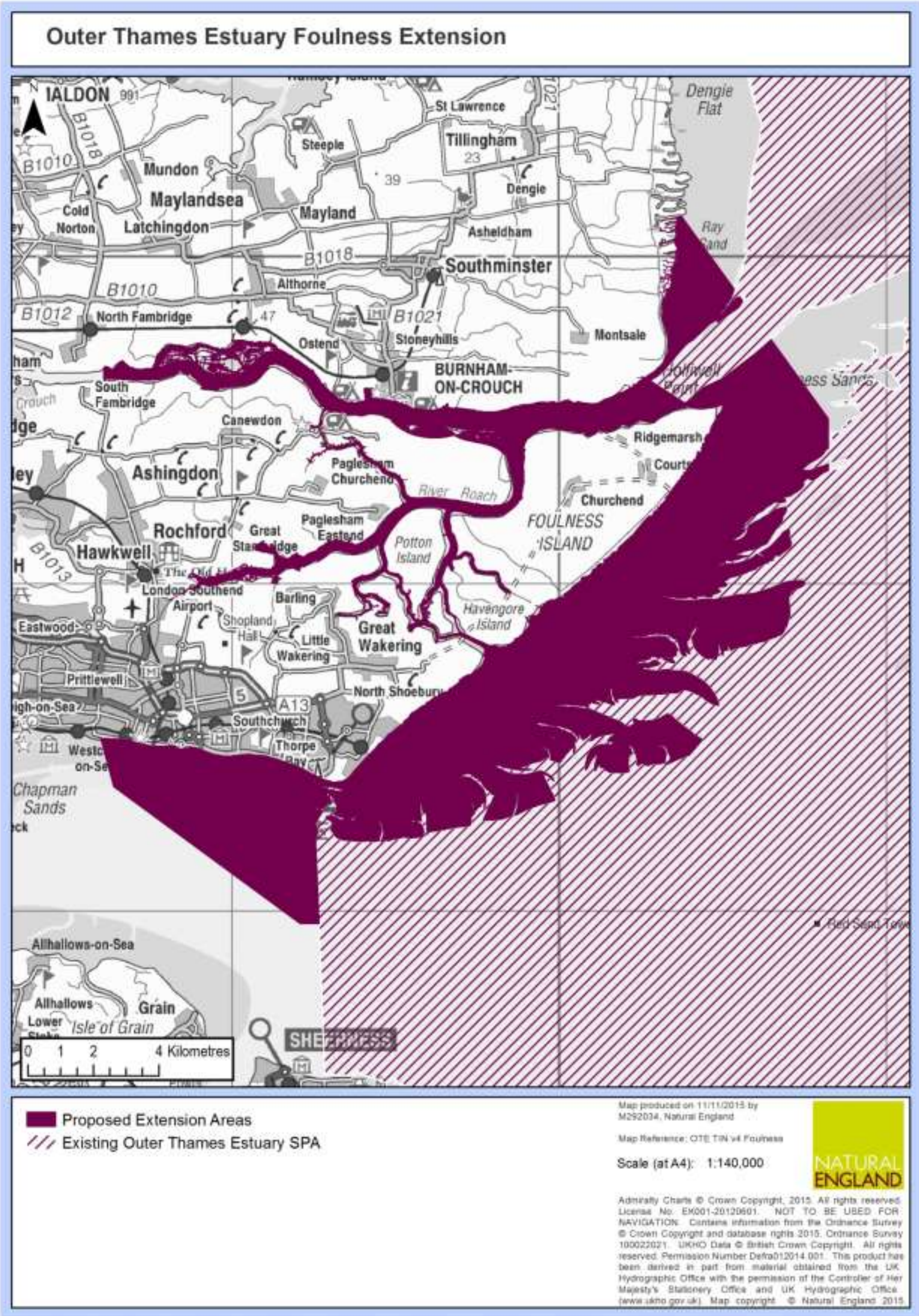
Annex 1d



Outer Thames Estuary Foulness Extension
MALDON
St Lawrence
Steeple
Tillingham
Mundon
Maylandsea
Mayland
Dengie
Cold Norton
Latchingdon
Asheldham
Dengie Flat
Ray Sand
Southminster
Althorne
North Fambridge
Ostend
Stoneyhills
Montsale
BURNHAM-ON-CROUCH
South Fambridge
Canewdon
Ridgemarsh
Paglesham Churchend
Courts
Ashingdon
River Roach
Churchend
Rochford
Great Stambridge
Paglesham Eastend
Potton Island
FOULNESS ISLAND
Hawkwell
London Southend Airport
Barling
Havengore Island
Eastwood
Shopland Hall
Little Wakering
Great Wakering
Prittlewell
North Shoebury
Southchurch
Thorpe Bay
Chapman Sands
Allhallows-on-Sea
Allhallows
Grain
Isle of Grain
0 1 2 4 Kilometres
Proposed Extension Areas
Existing Outer Thames Estuary SPA
Map produced on 11/11/2015 by M292034, Natural England
Map Reference: OTE TIN v4 Foulness
Scale (at A4): 1:140,000
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Annex 2 Site Citation

EC Directive 79/409 on the Conservation of Wild Birds

potential Special Protection Area (SPA)

Name: Outer Thames Estuary pSPA

Counties/Unitary Authorities:

Norfolk, Suffolk, Essex, Kent

Boundary of the pSPA:

The pSPA is divided into three main areas: the main part of the site is the outer part of the estuary, located between a line eastwards just north of Walton on the Naze, Essex in the north, to approximately Foreness Point seaward in the south, reflecting the existing SPA boundary. This area however extends inland to Westcliffe-on-sea along the Southend coast and down the River Roach and as far west as South Fambridge on the River Crouch. A separate area extends south along the coast of east Norfolk from Caister-on Sea in the north to offshore Felixstowe, Suffolk reflecting the existing SPA boundary. However the site extends down the River Bure to approximately Runham, and the River Blythe to encompass Blythburgh Water in the west. This area lies mainly within the 12 nautical mile (nm) zone, except for two small areas which extend slightly into the 12nm zone offshore from about Lowestoft, and a third area lying slightly further north and partly within 12nm, but also with a larger area extending well beyond the 12nm zone.

The landward boundary of the pSPA will mainly follow the existing Outer Thames Estuary SPA boundary which was drawn to Mean Low Water (MLW) or the seaward boundaries of existing SPAs, whichever is furthest seaward and based on red-throated diver survey data. The boundary is extending to Mean High Water (MHW) in places to encompass the foraging areas for little tern (*Sternula albifrons*) and common tern (*Sterna hirundo*) identified from qualifying SPAs.

The seaward boundary lies partly within the 20 m depth contour and marginally (along the outer eastern edge) within the 20-50 m depth contour.

Size of pSPA: The pSPA covers an area of 391,909.65 ha.

Site description:

The Outer Thames Estuary pSPA is located on the east coast of England between the counties of Norfolk (on the north side) and Kent (on the south side) and extends into the North Sea. The site comprises areas of shallow and deeper water, high tidal current streams and a range of mobile mud, sand, silt and gravely sediments extending into the marine environment, incorporating areas of sand banks often exposed at low tide. Intertidal mud and sand flats are found further towards the coast and within creeks and inlets inland down the River Yare, Bure, Blyth and Roach and Crouch estuaries. The diversity of marine habitats and associated species is reflected in existing statutory protected area designations, some of which overlap or abut the pSPA.

Qualifying species:

SPA site selection guidelines have been applied to the most up to date information for the site. Red-throated divers were a feature of the existing Outer Thames Estuary SPA and remain as part of the new pSPA.

The site qualifies under **article 4.1** of the Directive (2009/147/EC) as it is used regularly by 1% or more of the Great Britain populations of the following species listed in Annex I in any season:

Species	Season	Count (Period)	% of population
Red-throated diver *Gavia stellata*	Non-breeding	6,466 individuals (1989 – 2006/07)[5]	38.0% of GB population
Little tern *Sternula albifrons*	Breeding	746 individuals (2011 – 2015)	19.64% of GB population
Common tern *Sterna hirundo*	Breeding	532 individuals (2011 – 2015)	2.66% of GB population

Assemblage qualification:

The site does not qualify under SPA selection stage 1.3.

Principal bird data sources:

Colony counts from JNCC Seabird Monitoring Programme, Norfolk Bird & Mammal Reports, Foulness Area Bird Survey Group and contributed by colony managers from RSPB. Data on ringed common terns from national bird ringing scheme. Red-throated diver data from aerial surveys 1989 – 2006/07, as per Natural England (2010) and O'Brien *et al*. (2012).

[5] Value retained from original Outer Thames Estuary SPA standard data form (http://publications.naturalengland.org.uk/publication/3233957)

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 managers	Breeding seabird data for relevant colonies contributing to Outer Thames Estuary pSPA	2011-2014	Standard methodology	Verified by site manager and JNCC and published on website
Norfolk Bird & Mammal Report		Breeding seabird data for relevant colonies contributing to Outer Thames Estuary pSPA	2010 - 2013	Standard methodology	Published document undergoing editorial scrutiny
Data from RSPB	RSPB	Breeding seabird data for relevant colonies contributing to Outer Thames Estuary pSPA	2011 - 2015	Standard methodology	Data collected and agreed by site managers
Data from Foulness Area Bird Survey Group	FABSG	Breeding seabird data Foulness contributing to Outer Thames Estuary pSPA	2011 - 2015	Standard methodology	Data collected by group, scrutinised by group leader and published on website
National bird ringing scheme	BTO / Foulness ringing group	Counts of young common terns ringed at Foulness SPA	2011	Counts of ringed birds	Contributed to national ringing scheme
JNCC red-throated diver report	JNCC	Data on red-throated diver distribution and abundance from aerial surveys; summarised by Webb *et al*. (2009), Natural England (2010), O'Brien *et al*. (2012)	1989 – 2006/07	Visual aerial surveys, Kernel Density Estimation, Maximum Curvature analysis	Published in peer-reviewed journal (O'Brien *et al*. 2012)

Annex 4 Defining little tern foraging areas and seaward boundary

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[6] 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).

1. 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.

1.1. 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[7] index of

[6] '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.

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.

1.2. 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.

2. 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.

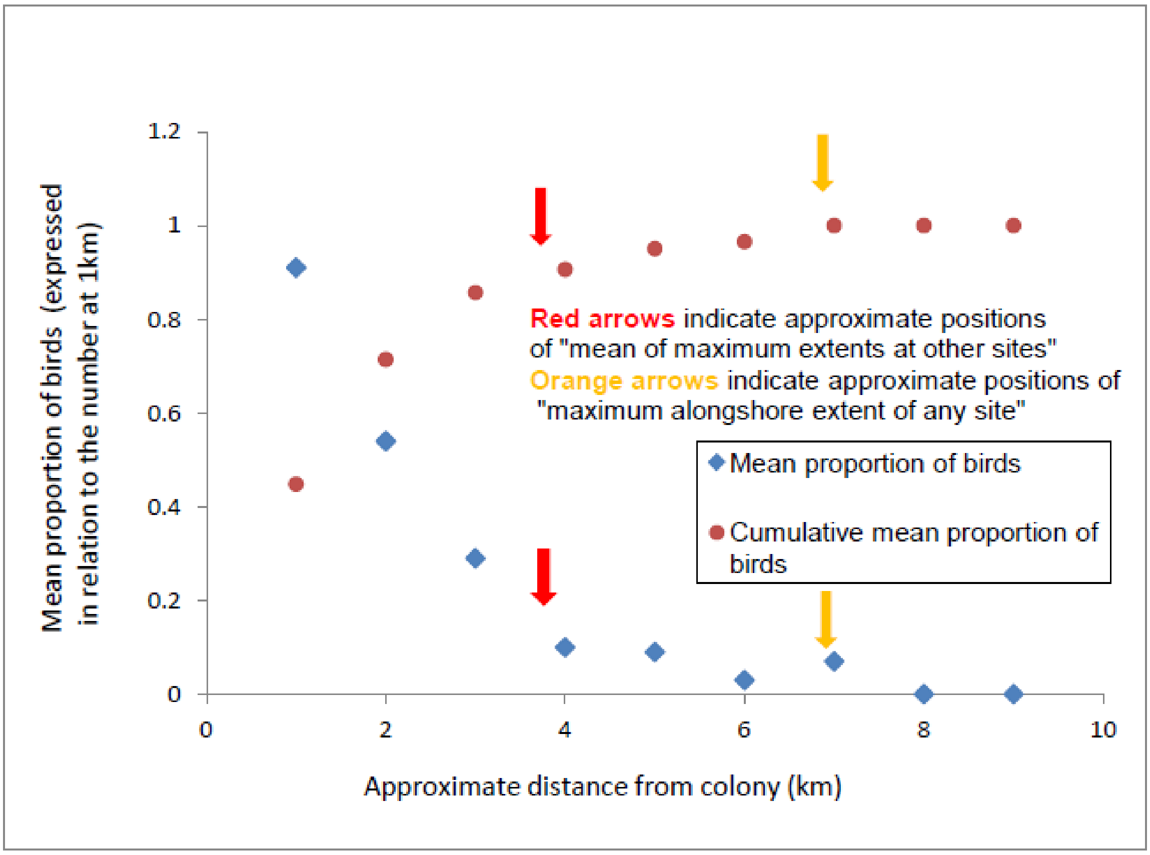

Figure 1: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing distances alongshore from the colony. Each blue point represents the mean proportional usage at each distance band from the colony averaged across colonies. The proportion at each distance (blue 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).

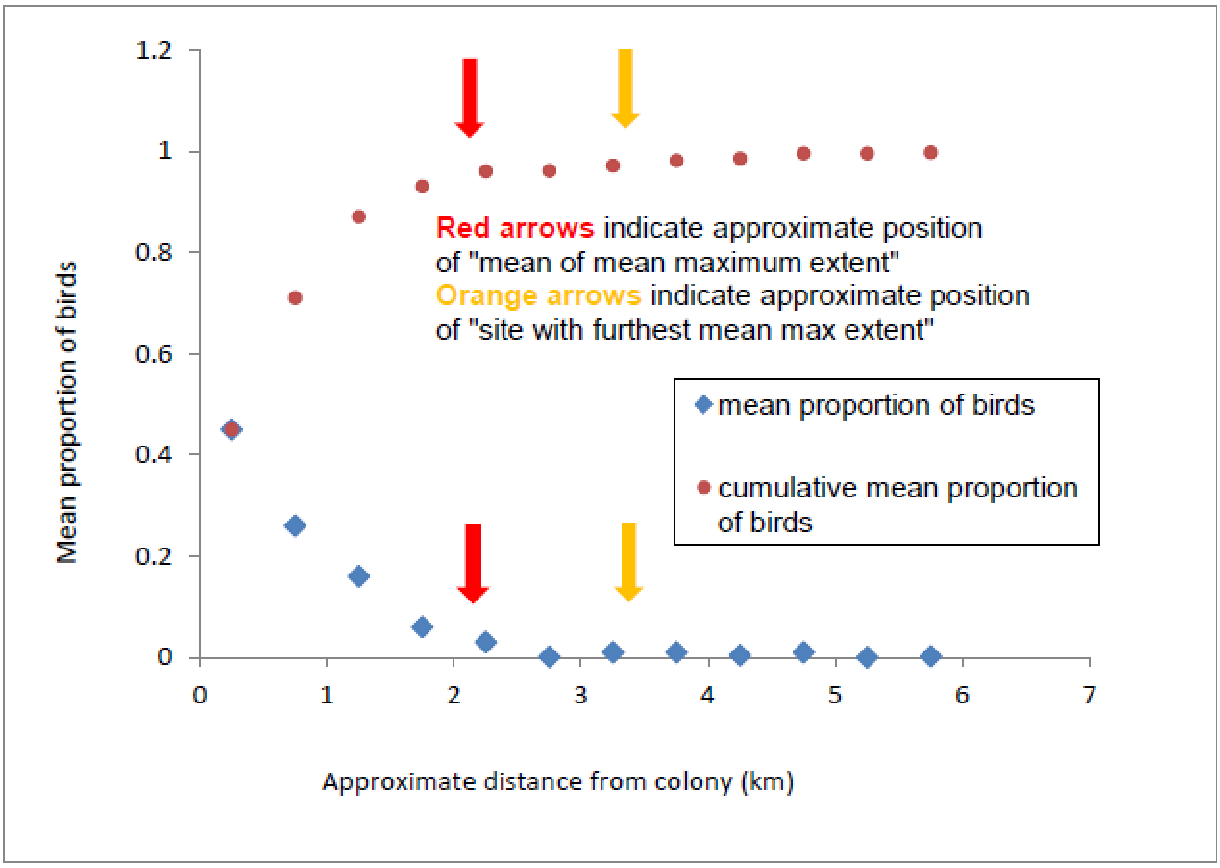

Figure 2: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing seaward distances from mean high water mark. Each blue point represents the mean proportional usage at each distance band from mean high water mark averaged across colonies. The red arrows indicate 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.

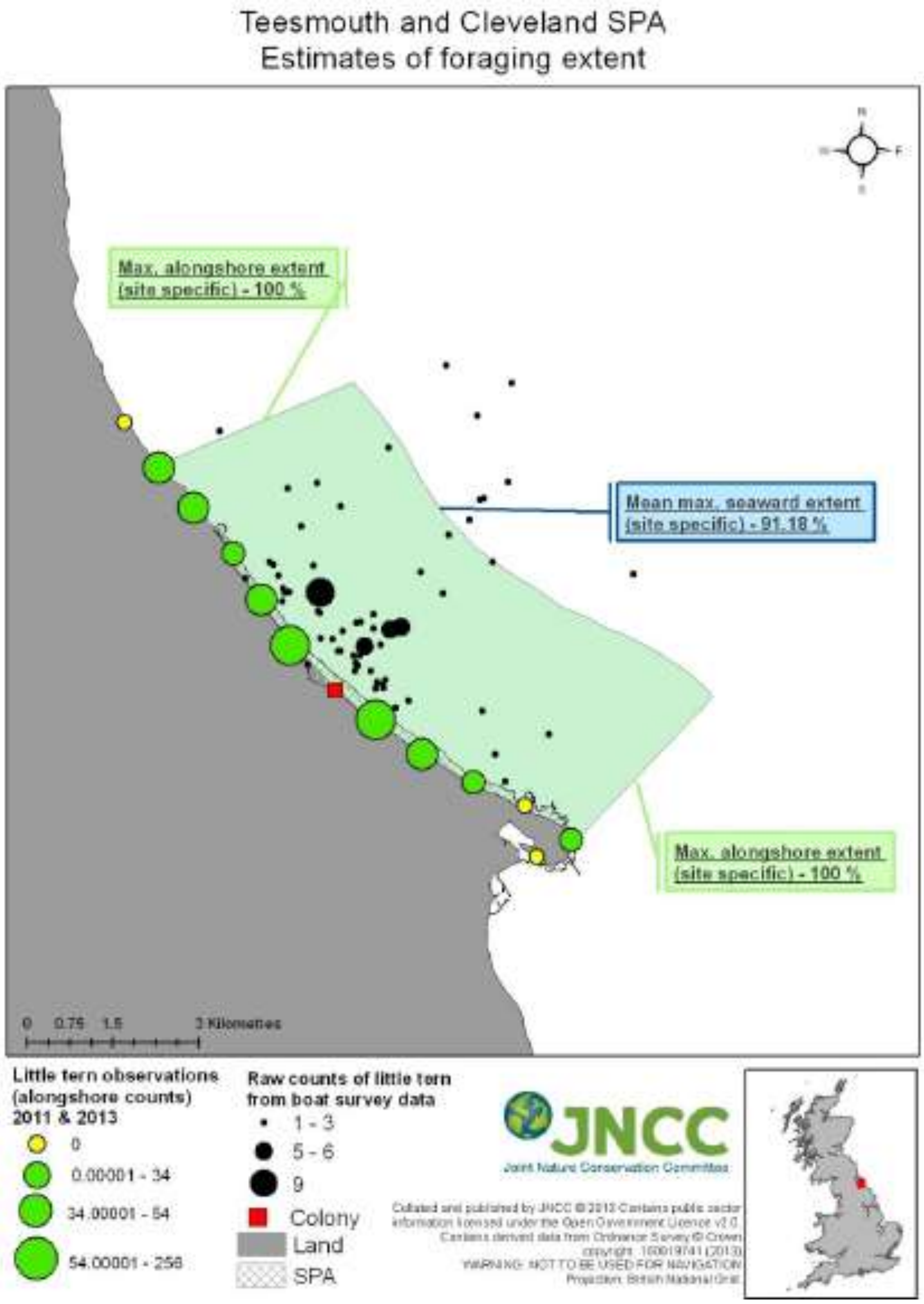

Figure 3. An example of the application of site specific alongshore and site specific seaward extents to define the boundaries to little tern foraging areas at the Teesmouth and Cleveland SPA. The % values given in the labels indicate the site specific % of little tern observations within the shore-based (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 5 Defining larger tern foraging areas and seaward boundary

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)[8].

1. 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[9] (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.

[8] 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.
[9] 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 environmental covariates (information on environmental conditions at an appropriate scale and extent). 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.

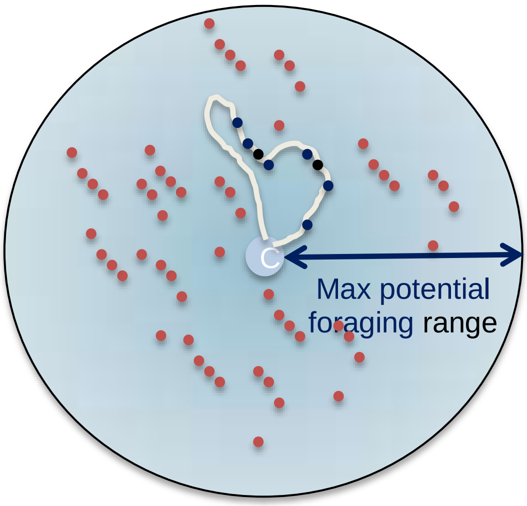
2. Data preparation and analysis

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 dilute the power of the analysis to identify important habitat relationships and therefore foraging areas. In addition, because terns are central place foragers (meaning they must travel to and from their nest site on each trip), it would 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[10] 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.

[10] 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[11], 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).

[11] 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.

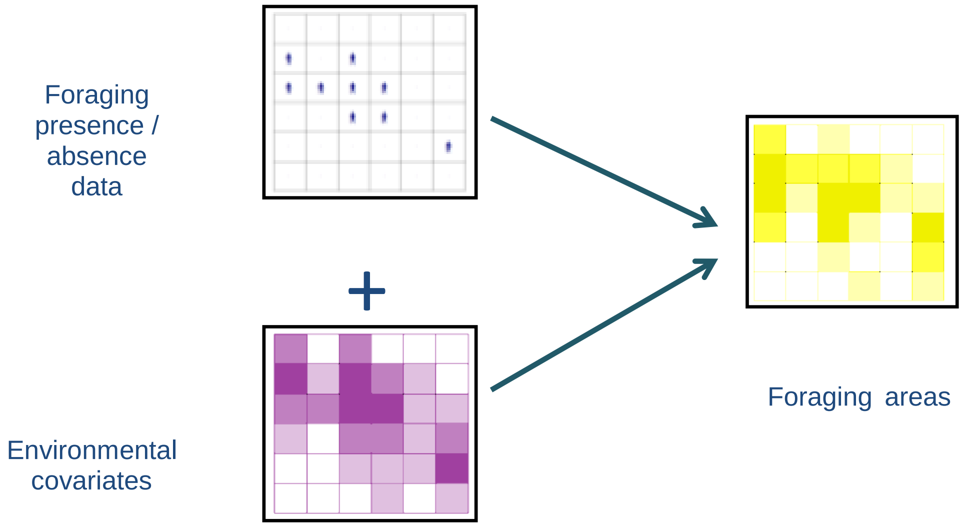


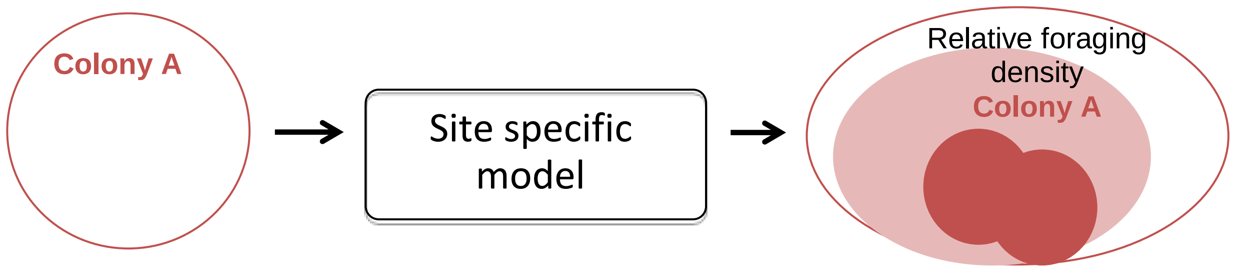
Figure 1. Simplified, schematic representation of the process of modelling distributions based on environmental information, using a single covariate distribution map in the example.

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.

PHASE 1: colony specific bird data



PHASE 2: no colony specific bird data

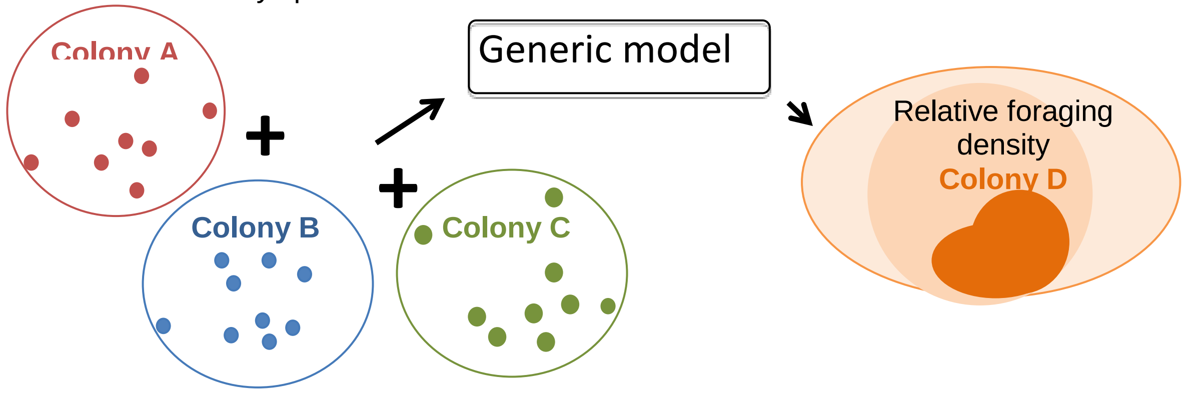


Figure 2. Simplified, schematic representation of the process whereby empirical observations of tern foraging locations around a colony were either: used to build predictive, site-specific models of tern usage that generated relative foraging density maps around that colony (phase 1 analyses); or combined 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 assessing the sufficiency of tracking sample size was used for the analysis (see Soanes *et al*. 2013). This method takes subsamples of the available data to examine how sample size influences estimates of the home range (the size of the area used) by the whole colony, based on the time spent in individual predefined grid cells. All of the cells within a home range represent the total area of use, whilst other fractions of the total area of use, determined by ranking the cells within the home range 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 by 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 68.1% 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 (including at other colonies where it was not possible to sample) where relatively high levels of usage by foraging terns might be expected based on their characteristics. In other words, the habitat models allow us to fill gaps in sampling effort, both at sampled colonies and at unsampled colonies.

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.

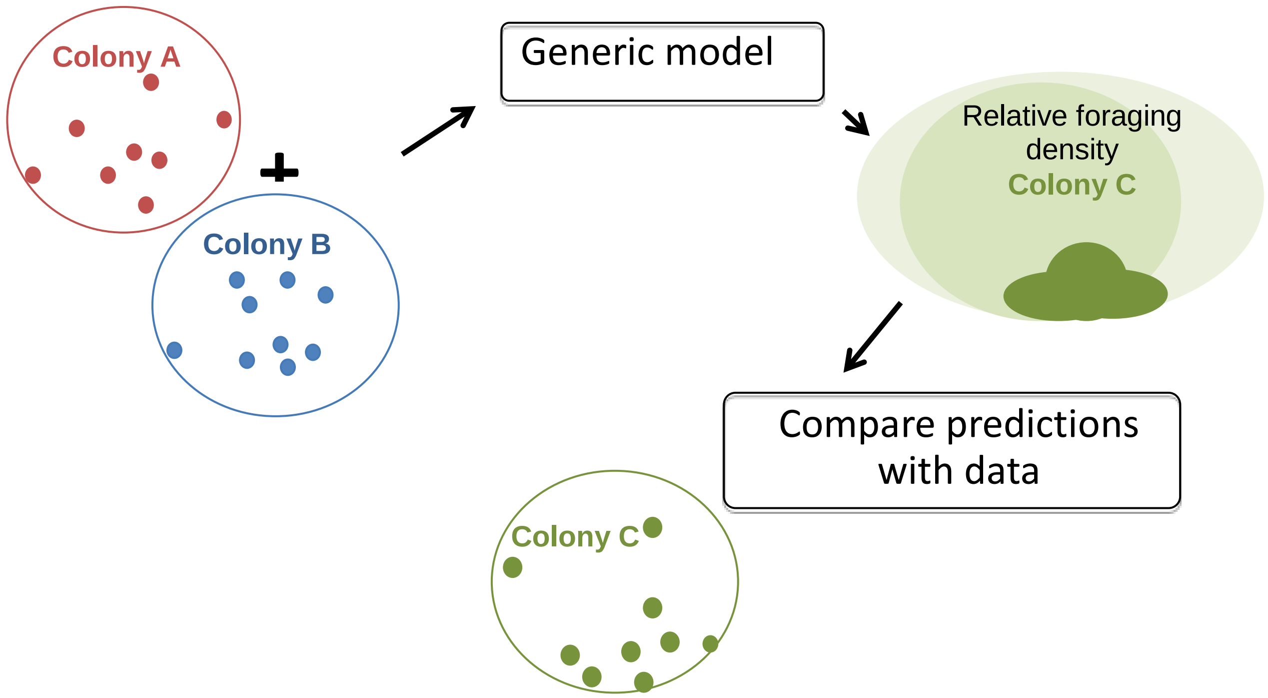


Figure 3. Schematic representation of the cross-validation process, using an example where we have data for three colonies A, B and C, of which data from two at a time (A and B in this diagram) are used to build a predictive model, the predictions of which are then tested by comparison with empirical 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 colony, bathymetry, distance to shore	**0.821**	**0.911**	**0.979**	**0.973**	**0.970**	**0.907**	**0.916**

3. 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 in 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[12], 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.

An example of a maximum curvature boundary drawn tightly around the modelled usage distribution of common terns from Foulness SPA is shown in Figure 4.

[12] 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.

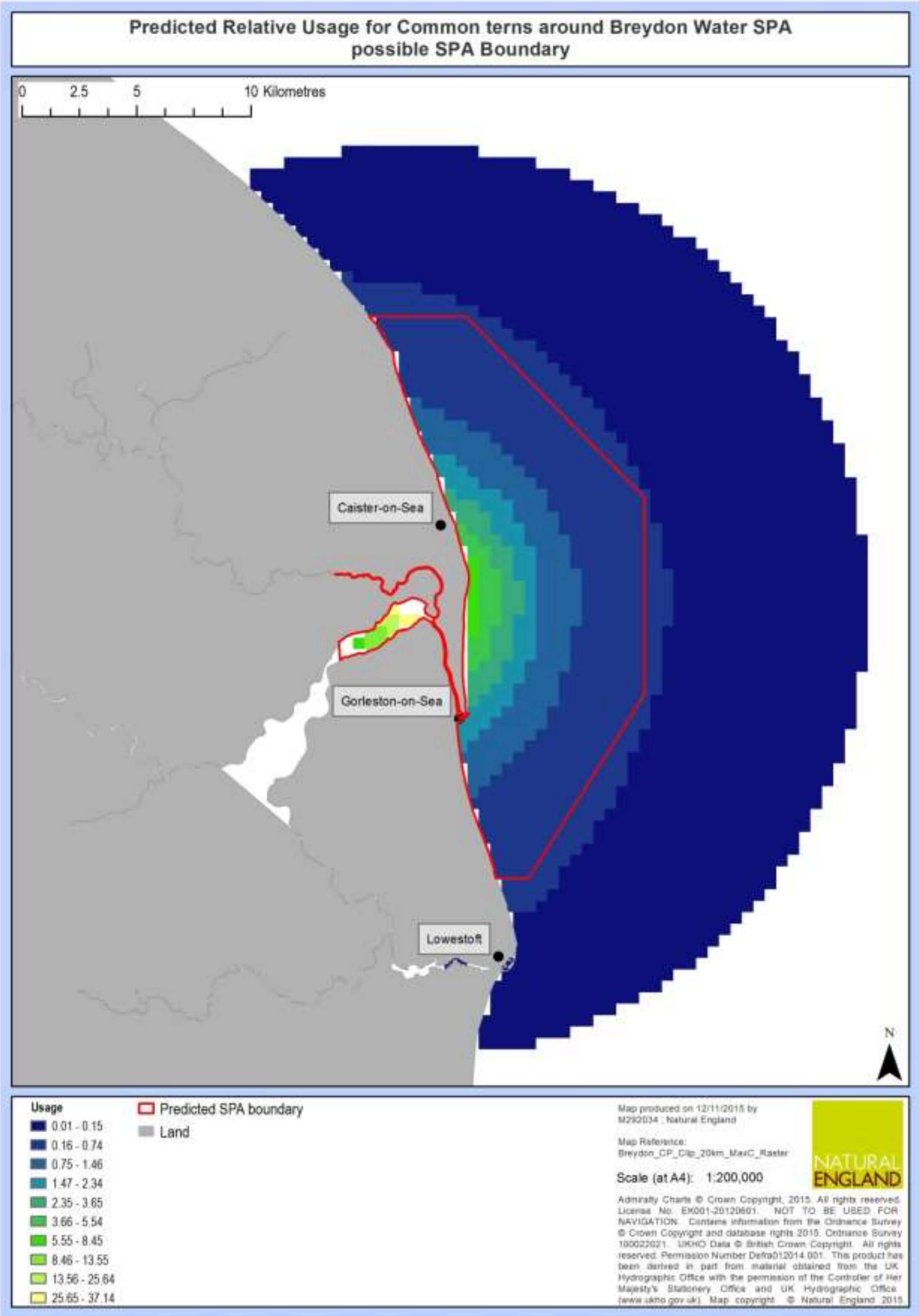


Figure 4 Maximum curvature derived boundary (red line) overlaid on map of model predictions of usage by common terns around Foulness SPA. The extent of the dark blue circle of model

predictions of usage is 20 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 (2015).

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 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 6 Implementation of Natural England Evidence Standards

Decision-making processes within Natural England 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 Outer Thames Estuary pSPA.

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

Quantification of qualifying feature population sizes

In order to determine the suite of species present within the pSPA which meet the SPA selection guidelines (JNCC 1999), most relevant bird count data were used, either pertaining to the current five year period (2011-2015 for breeding terns; 1989 - 2006/07 for non-breeding red-throated divers, as per the original SPA citation (Natural England 2010, O'Brien *et al*. 2012).

1. Data from JNCC's Seabird Monitoring Programme (SMP) (http://jncc.defra.gov.uk/smp/) Count data for breeding terns were taken from the national database wherever possible.

2. Data from colony managers and local expert groups (Foulness Area Bird Survey Group, Foulness ringing group) supplemented the SMP data where this was not available, for both little and common terns.

3. The Norfolk Bird & Mammal Report was used to provide data where neither SMP nor RSPB data were available.

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. Ringing data is submitted to the national ringing scheme, again providing most suitable available information.

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 Outer Thames Estuary 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 was 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) 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.	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 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.* 2015). 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 River Yare, Bure, and Crouch and Roach Estuaries, Benfleet and Southened SPA in the light of model predictions of the usage of such areas by foraging common terns from Foulness SPA. Additionally, the landward boundary of the pSPA has been drawn to MHW along the Blythe River in light of the model predictions of the usage of such areas by foraging little terns from Minsmere-Walberswick SPA.

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*". A combination of these approaches have been used in determining the seaward boundary of this pSPA; the former for parts of the boundary drawn for red-throated diver distribution, and the latter for areas added for foraging terns.

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.* 2015) 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*. 2015) 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 Outer Thames Estuary pSPA can be seen to be in accord with the guidelines set out in that document.

Establishment of the extent of Outer Thames Estuary pSPA

The extent of the pSPA boundary is determined almost entirely by the distribution of red-throated divers as per the classification of the Outer Thames Estuary SPA. The smaller new part of the extent is based on model-generated predictions of which areas of sea are most heavily used by foraging terns originating from two source colonies. The boundary of the pSPA is a composite of non-breeding feature distribution and breeding feature predicted foraging areas.

All 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 (e.g. 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.

The adequacy and relevance of these various models and of the modelling approach in general, was addressed by JNCC in three ways (Wilson *et al.* 2015):

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 5, 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

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 4).

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.

Analysis of non-breeding red-throated diver distribution has been published in a peer-reviewed journal (O'Brien *et al*. 2012)

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, or to count data originating from the time of original classification. As such the conclusions in this respect clearly relate to the best available evidence.

The conclusions regarding the drawing of parts of the landward boundary of the pSPA inland at MHW are based upon the evidence provided in the form of a model of predicted usage by foraging common tern. In this instance, the generic model was used which 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. That the use of such areas by larger tern species is also likely is supported by information in the scientific literature. A review of tern foraging ecology (Eglington 2013) notes that larger tern species including Sandwich tern routinely forage in areas of shallow water. There is no reason on the basis of that review to consider it likely that common terns 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 non-breeding divers through 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 5. 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. RSPB is a professional organisation with long-standing experience of seabird monitoring, and surveys are conducted by trained surveyors. There is therefore high confidence in RSPB survey data. Accordingly, such data referred to in this Departmental Brief can be considered to justify high confidence . Similarly, the Foulness Area Bird Survey Group are an organised collective with unrivalled local ornithological knowledge and experience. The data collected by the group also justify high confidence.

Ringing data (counts of numbers of birds ringed) are not subject to uncertainty. However, the method applied to estimate numbers of adult pairs will be. To account for this, several scenarios are presented, with selection of the scenario considered to be realistic (based on conversations with local site experts) informing the calculations of numbers of pairs of common terns breeding at Foulness SPA (Annex 7).

Any uncertainties with aerial survey data collected for red-throated divers are assumed to have been adequately addressed in classifying the original Outer Thames Estuary SPA.

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 parts of the coast has been made, is discussed in the previous section.

Seaward boundary

The position of the seaward boundary of the pSPA has largely been quality assured to the highest level (O'Brien *et al*. 2012). The position of the small additional extension to the seaward boundary has been determined on the basis of outputs of statistical models which are based on tern behaviour at colonies in other parts of the UK. 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 5.

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.

O'Brien *et al*. (2012) describes the process of boundary setting for red-throated divers, which determines the vast majority of the pSPA boundary. As a peer-reviewed publication in a scientific journal, this work was subject to the highest level of independent review.

The derivation of the alongshore extent and seaward boundary to the pSPA is based on a 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*. 2015) 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.* 2015) 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.

The first version of this Departmental Brief was drawn up by Alex Banks (Marine Ornithologist) and with input from Catherine Laverick.

Departmental Briefs are drafted by an ornithologist with support from the site lead who provides the local site specific detail. This 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 then circulated for external comments from Defra Marine Policy Officer, JNCC senior seabird ecologists, Marine Protected Area Technical Group (MPATG) and UK Marine Biodiversity Policy Steering Group (UKMBPSG). 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 in accordance with our Quality Management Standard. The brief is then signed off as required by our Non-Financial Scheme of Delegation by a representative of the Senior Leadership Team with delegated authority before being submitted to Defra.

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Annex 7 Common terns breeding at Foulness SPA

This annex presents relevant data for common terns breeding at Foulness SPA. Data are kindly supplied by Foulness Area Bird Survey Group and the Foulness ringing group. Treatment of data focuses on the main breeding area (New England Creek) within the SPA, and does not include the handful of pairs known to usually or occasionally breed at other scattered locations within the SPA, largely because of the patchy nature of available data on these locations.

Available data for the past six years are displayed in Table 1 below.

Table 1: count data for common terns at Foulness SPA in pairs / AON. Brackets show juveniles ringed. NC = No Count.

	New England raft	East Newlands
2010	Bred (72)	
2011	Bred (58)	
2012	25	
2013	NC	
2014	9	
2015	2	2-3

From Table 1, the past six years of data for Foulness (using data from New England Creek and ignoring small numbers of pairs elsewhere within the SPA) gives two years in which common terns 'bred', one with no count, and three years with counts of adults totalling 25, 9 and 2 pairs. Common tern numbers are thought to fluctuate partly in response to black-headed gull *Chroicocephalus ridibundus* abundance at the breeding location, with lower numbers likely reflecting lack of management intervention to discourage gull nesting. In years when this is possible (e.g. 2010, 2011), common terns numbers increase. We expect future management to lead to the same increases in commn tern nesting numbers.

Estimating adults from ringed young

In 2010 and 2011, the number of juvenile birds ringed suggests that numbers of adult common tern pairs were likely to have been greater than the value of 25 pairs used by JNCC to prioritise sites supporting regular breeding (as common terns produce two eggs per pair and numbers of young exceeded 50). In some other years, figures suggest that adult pairs may be underestimated (or that some years birds are extremely productive); for example, 134 pairs and 102 young in 2000; 33 pairs and 56 young in 2007.

No ringing data for 2012, 2014 or 2015 are available and so counts of adults are all that can be used, accepting that they may be undercounted. There are no data for 2013 of any type.

In order to estimate the number of adult pairs from juveniles, we can make some assumptions about productivity and thus calculate the number of pairs that are likely to have been present to produce the resulting number of young. Two ways to do this are to use national (UK) average productivity levels across time, or average productivity levels (for England) in the years in question (2010 and 2011) as a proxy for productivity at Foulness SPA. We assume that terns with fledged chicks do not make repeat attempts to breed within the same breeding season, likely a fair assumption based on tern ecology.

Horswill & Robinson (2015)[13] provide demographic rates for seabirds breeding in the UK. For common terns, 24 colonies in the UK (16 in England) are analysed and a mean is derived from these. This value is 0.764 chicks per pair (standard deviation = 0.470), assessed as a 'good' quality estimate (the highest category available). As the mean is provided with the standard deviation, it is possible to calculate an upper estimate of productivity, based on mean productivity plus two standard deviations. Within a normal distribution, 95% of individual colony productivity average values should lie within two standard deviations of the mean. The upper 95% value derived in this way equates to a productivity level that is seldom exceeded and so provides a very conservative estimate of the number of pairs that might produce a certain number of fledged young.

JNCC also provide information on annual seabird productivity, with plots summarising this by country within the UK. In England, estimated average common tern productivity in 2010 and 2011 was 0.57 and 0.45 chicks per pair respectively (JNCC 2014: http://jncc.defra.gov.uk/page-3201).

Table 2 displays the various estimated numbers of adult common terns. Five year means are shown relating to these estimated and counted totals of adult pairs. When using the most optimistic estimate of productivity (national average plus two standard deviations) to estimate the numbers of pairs present in 2010 and 2011, the five year mean 2010 – 2014 is 27.6 pairs and 17.5 pairs 2011 - 2015. Using alternative assumptions regarding productivity to estimate numbers of pairs in 2010 and 2011 gives greater five year means; 51.0 and 28.0 pairs (using national average productivity over the two five year periods) and 71.0 and 41.2 pairs (using average productivity in England 2010 and 2011).

In the opinion of the Foulness ringing group, based on casual observations of adult pairs at the time of ringing and observations of productivity, the most realistic estimates of adults are those based on the national average plus two standard deviations (42 pairs in 2010 and 34 in 2011). Foulness SPA is thus a very productive colony for common terns, when manangement intervention discourages black-headed gull nesting and allows the terns to breed.

Table 2: Five year mean population size for common terns at Foulness SPA based on estimated and actual counts of adult pairs. 2010 and 2011 values estimated according to: national average productivity, upper estimates of national productivity, and estimated average productivity in England in 2010 and 2011.

	National average	Upper national	England
2010	*94*	*42*	*126*
2011	*76*	*34*	*129*
2012		25	
2013		No data	
2014		9	
2015		2	
Five year mean (2010 – 2014)	51.0	27.6	71.0
Five year mean (2011 – 2015)	28.0	17.5	41.2

[13] Horswill, C. & Robinson R. A. 2015. Review of seabird demographic rates and density dependence. *JNCC Report No. 552*. Joint Nature Conservation Committee, Peterborough.