Departmental brief

Solent and Dorset Coast potential Special Protection Area (pSPA)

Natural England

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Contents

Summary

Solent and Dorset Coast potential Special Protection Area (pSPA) detailed in this Departmental Brief is proposed to protect important foraging areas at sea used by qualifying interest features from colonies within adjacent, already classified SPAs. These qualifying interest features are three species of tern: common tern, Sandwich tern and little tern. The site is located on the south coast within the English Channel. The site is approximately 255.2 nm2 and extends from the Isle of Purbeck in the West to Bognor Regis in the East, following the coastline on either side to the Isle of Wight and into Southampton Water. The boundary was established as a composite of the usage of the area within adjacent SPAs.

From west to east, the adjacent SPAs with these tern species as qualifying interest features (in parentheses) are: Poole Harbour (common tern) Solent and Southampton Water SPA (common, Sandwich and little tern) and Chichester & Langstone Harbours SPA (common, Sandwich and little tern). In addition to these species at these sites, Sandwich terns at the Poole Harbour SPA are included in determining the details of the pSPA. However, certain species at certain sites i.e. Roseate tern at Solent and Southampton Water SPA, and Sandwich, little and common tern at Pagham Harbour SPA are not included in determining the details of the pSPA. These exclusions are made on the basis of these birds either not being a qualifying feature at the source SPAs and/or being present in such low numbers either at classification or recently (or both) to merit influencing the size and shape of the pSPA.

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 aforementioned existing classified SPAs, this Departmental Brief does not: make any proposal to add or remove qualifying features, amend base-line population figures, or alter site boundaries.

This Departmental Brief sets out the scientific case for the classification of the Solent and Dorset Coast 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 Solent and Dorset Coast pSPA

Species[1]	Count[2,3] (period)	% of GB breeding population[4]	Stage of Selection Guidelines	Interest type
Sandwich tern *Sterna sandvicensis*	441 pairs (882 breeding adults)[5] (2008 - 2014)	4.01%	1.1	Annex 1
Common tern *Sterna hirundo*	492 pairs (984 breeding adults) (2009 - 2014)	4.77%	1.1	Annex 1
Little tern *Sternula albifrons*	63 pairs (126 breeding adults) (2009 - 2014)	3.31%	1.1	Annex 1

[1] Species of terns that depend on the Solent and Dorset Coast pSPA area of sea for foraging that derive from breeding colonies at the following existing SPAs: Poole Harbour SPA, Solent and Southampton Water SPA and Chichester & Langstone Harbours SPA.

[2] The pSPA population is derived from the sum of the most recent populations of each species within the existing SPAs identified above. These totals exclude: i) numbers of any terns that may undertake foraging within Solent and Dorset Coast pSPA, but derive from breeding colonies that are situated outside of existing SPAs; ii) numbers of any terns at existing SPAs which are not qualifying features of these sites and not currently present in numbers exceeding SPA selection criteria thresholds , iii) numbers of terns at existing SPAs which, although qualifying features of those sites, were not present at classification in numbers exceeding SPA selection criteria thresholds and/or are no longer present in such numbers, iv) numbers of terns at existing SPAs which, although qualifying features of those sites and present at classification in numbers exceeding SPA selection criteria thresholds are no longer present in such numbers at those

particular sites and when summed across all source SPAs that might contribute to the numbers supported by the pSPA.
[3] Sources of count data and recent mean figures for each contributing SPA are detailed later in this Department Brief.
[4] GB breeding populations taken to be that within Great Britain as presented in Musgrove *et al.* (2013).
[5] Pairs multiplied by 2 to arrive at breeding adult numbers; this rule applies to all species listed in the table

1. Assessment against SPA selection guidelines

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

1.1. Stage 1

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

Stage 1.1 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;

Stage 1.2 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;

Stage 1.3 an area is used regularly by over 20,000 waterbirds (waterbirds as defined by the Ramsar Convention) or 20,000 seabirds in any season

Stage 1.4 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.

Solent and Dorset Coast pSPA qualifies under Stage 1.1 by regularly supporting >1% of the GB population of Sandwich tern, common tern and little tern, species listed in Annex I of the Birds Directive. The pSPA does not quality under Stage 1.2 or 1.3.

1.2. Stage 2

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

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

Feature	Qualification	Assessment
1. Population size & density	✓	9th largest Sandwich tern population in the UK, 7th largest common tern population in the UK and 4th largest little tern population in the UK[1]
2. Species range	✓	The primary source SPA colonies are the furthest south SPAs in England which support all 3 breeding tern species, with the exception of little tern which is also a feature of Chesil Beach & The Fleet SPA.
3. Breeding success	✓	The primary source SPA colonies have a long history of breeding success. In recent years poor weather or anthropogenic reasons have led to poor or no breeding success at some sites. Many sites are now being managed to avoid anthropogenic disturbance or increase the availability of breeding habitat to increase breeding success.

[1] Note that this ranking should only be considered indicative of the relative importance of the pSPA as it is based on comparison of the sum of the most recent 5 year mean populations of each tern species at the source SPAs (as listed in Table 1) with the historical populations of each species at each SPA in the UK as listed in Stroud *et al.* (2001).

4. History of occupancy	✓	All primary source colonies have a long history of occupancy by breeding terns: Little terns (Chichester Harbour since 1960s, Langstone Harbour island since 1978, Solent & Southampton Water since 1932); common terns (Chichester Harbour since 1960s, Langstone Harbour island since 1980s, Solent & Southampton Water since 1948); Sandwich tern (Chichester Harbour since 1975, Langstone Harbour island since 1980s, Solent & Southampton water 1959-2004 at Beaulieu Estuary and since 1966 at Hurst-Pitt's Deep)[2]
5. Multi-species area	✓	Three qualifying Annex 1 features.
6. Naturalness		No longer applicable, following ruling from the SPA and Ramsar site Working Group
7. Severe weather refuge		Not relevant to breeding species

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

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

[2] Sources: Shrubb (1979) & Clark & Eyre (1993)

been implemented at 31 sites in Scotland and are under consideration at the Flamborough Head 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 Solent & Dorset Coast 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.

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 the fourth work strand as, given the likely extent of these areas, these cannot be addressed by application of the generic “maintenance” extensions approach and are not covered by the work on identifying inshore non-breeding aggregations or important offshore areas.

This Departmental Brief sets out information supporting the identification of the qualifying features of the Solent and Dorset Coast pSPA and definition of its proposed boundaries on the basis of 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. the populations of terns at the existing Poole Harbour, Solent and Southampton Water and Chichester & Langstone Harbours SPAs.

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

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

2.1. Data collection – defining the suite of species and numbers of those supported by the Solent and Dorset Coast pSPA

The size of each of the populations of terns supported by the Solent and Dorset Coast 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 Solent and Dorset Coast 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 have been taken to be the most recently available from the Seabird Monitoring Programme (SMP) website (http://jncc.defra.gov.uk/smp/) i.e. within the last 5 years, unless otherwise indicated. Where necessary and possible, this dataset has been augmented by information requested directly from colony managers.

The pSPA population calculation excluded: i) numbers of any terns that may undertake foraging within Solent and Dorset Coast pSPA, but derive from breeding colonies that are situated outside of existing SPAs; ii) numbers of any 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, were not present at classification in numbers exceeding SPA selection criteria thresholds and/or are no longer present in such numbers, iv) numbers of terns at existing SPAs which, although qualifying features of those sites and present at classification in numbers exceeding SPA selection criteria thresholds are no longer present in such numbers at those particular sites and when summed across all source SPAs that might contribute to the numbers supported by the pSPA. 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 Solent and Dorset Coast pSPA

The overall boundary of the Solent and Dorset Coast pSPA has been drawn to encompass the sea areas identified under the fourth strand of JNCCs work programme as being most important to support the terns which are already qualifying features of several source colony SPAs (or are proposed to be added as new qualifying features to them). The work done to identify the areas important to little terns and the larger tern species differed and was conducted separately. These separate pieces of work are described 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 2.2.3.

2.2.1. Identification of important marine areas for little tern

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 2.1km, mean of recorded maxima of 6.3km and maximum ever recorded in the literature being 11km (Thaxter *et al.* 2012). In the 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 areas most heavily used for foraging by breeding little terns would be to undertake a programme of shore based observations and of boat-based transects of offshore areas around colonies and to use the resultant distribution data directly in setting the alongshore and seaward boundaries respectively.

Accordingly, between 2009 and 2013 JNCC coordinated a programme of survey work to identify important foraging areas for little terns at a number of UK little tern colonies. These surveys were conducted during the chick rearing period in each year and comprised repeated shore-based counts of little terns seen at a series of observation stations at increasing distances from the colony locations, and repeated boat based surveys along transects across the waters offshore from the colonies. These surveys sought to establish the distances both alongshore and offshore that little terns were travelling to feed.

In total, 70 shore-based surveys were undertaken at 14 little tern colonies around the UK with a total of 7,006 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 Solent and Dorset Coast pSPA. Further general information on little tern survey programme is presented in Annex 4.

2.2.1.1. Solent & Southampton Water SPA

Three shore-based surveys were undertaken in 2013 which collected only 62 little tern observations. Three boat-based surveys were completed in 2012 (2) and 2013 (1) and recorded only 14 little tern observations. The total number of observations for both shore- and boat-based surveys was judged to be insufficient to justify a site-specific approach to boundary definition and therefore a generic approach was applied for boundary setting. The alongshore foraging extent for this colony was set to be the generic value derived from all of the surveys at all of the colonies i.e. 3,900m. The seaward foraging extent for this colony of little terns was set to be the generic seaward extent value derived from all of the surveys at all of the colonies, i.e. 2,176m (Figure 1a).

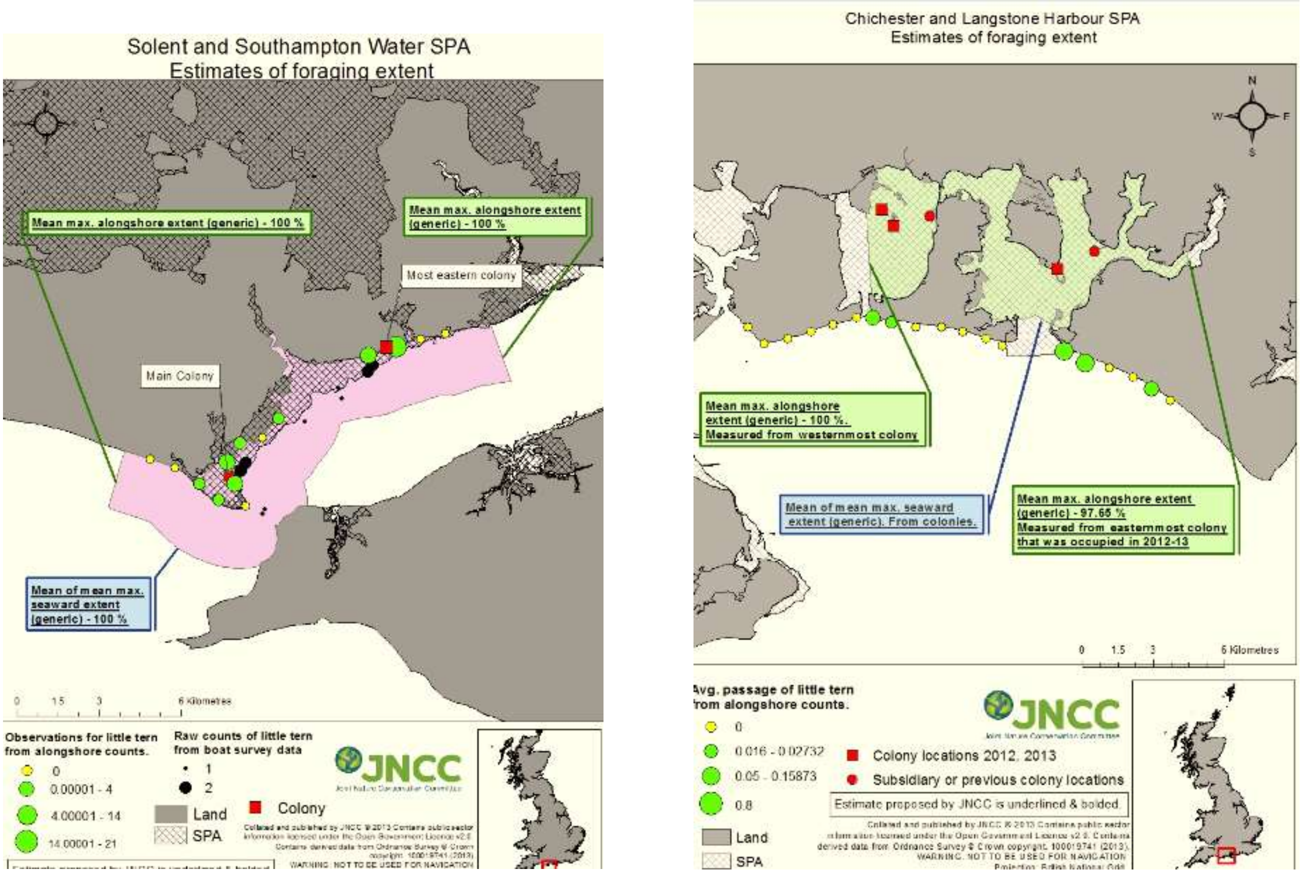
The little tern foraging area does not define the boundary of the proposed pSPA as it sits entirely within the boundary identified on the basis of the work on the larger tern species.

2.2.1.2. Chichester & Langstone Harbours SPA

Six shore-based surveys were undertaken in 2012 (3) and 2013 (3) along the shoreline between points to the west of the mouth of Portsmouth Harbour and to the east of the mouth of Chichester Harbour. In 2012 no little terns were recorded while in 2013 only 58 little terns were observed during survey. No boat-based surveys were completed. The low number of observations for shore-based surveys, coupled with the fact that these were derived in only one year, and the lack of boat-based surveys were judged to be insufficient to justify a site-specific approach to boundary definition and therefore a generic approach was applied for boundary setting. The alongshore foraging extent for this colony was set to be the generic value derived from all of the surveys at all of the colonies i.e. 3,900m. The seaward foraging extent for this colony of little terns was set to be the generic seaward extent value derived from all of the surveys at all of the colonies, i.e. 2,176m (Figure 1b). The location of the colonies within the inner parts of the harbours resulted in the seaward limit to the foraging areas being restricted to areas within the harbours and not extending outside the harbour mouths. This accords with the absence of little tern sightings along the coast in 2012 but may fail to capture the use of near shore areas in some years, which observations in 2013 confirmed. Note that the maximum possible extent of such areas would sit entirely within the boundary identified for the pSPA on the basis of the work on the larger tern species.

2.2.1.3. Pagham Harbour SPA

Although little tern is a qualifying feature of the Pagham Harbour SPA, only 7 pairs (5 year mean 1992-1996) (0.3% of the GB breeding population) are recorded in the Natura 2000 Standard Data Form. On the basis of the criteria used by JNCC to define recent occupation (i.e. a mean number of pairs over the most recent 5 years of data equalling or exceeding 1% of the national breeding population i.e. 19 pairs), Pagham Harbour was deemed in 2012 to be “not recently occupied”. On that basis, no surveys of little terns were carried out at Pagham Harbour. The most recent available 5 year mean (2009-2013) is 13 pairs which equates to 0.66% of the GB breeding population. Accordingly, the potential foraging areas of little terns from this SPA have not been considered in deriving the size and shape of the Solent and Dorset Coast pSPA. However, given the generic values for both alongshore and seaward extent to little tern foraging areas, the whole of the sea area that birds from this colony are most likely to use is included entirely within the areas of use of Sandwich terns originating from Chichester and Langstone Harbours SPA (section 1.2.3.2) and hence within the composite boundary of the pSPA (section 1.2.4).



a) b)

Figure 1. Application of generic alongshore and generic seaward extents to define boundaries to little tern foraging areas around colonies within a) the Solent and Southampton water SPA and b) Chichester & Langstone Harbour SPA.

2.2.2. Identification of important marine areas for larger terns

The four larger species of tern which breed regularly in Great Britain have recorded mean foraging ranges between 4.5km and 12.2km and maximum recorded foraging ranges between 15.2km and 49km (Thaxter *et al*. 2012). In the light of these larger areas of interest, JNCC, in agreement with all of the SNCBs, decided that the most effective method to determine the extent of the areas most heavily used by breeding terns of the four larger species would be different to that employed for little terns. In this case, the approach was to undertake a programme of boat-based visual tracking of foraging birds. 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. 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 for larger terns at a number of UK colonies. These surveys were conducted during the chick rearing period in each year and comprised repeated days of observations of individual terns whose tracks were followed by boat as they left the colony to forage.

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.

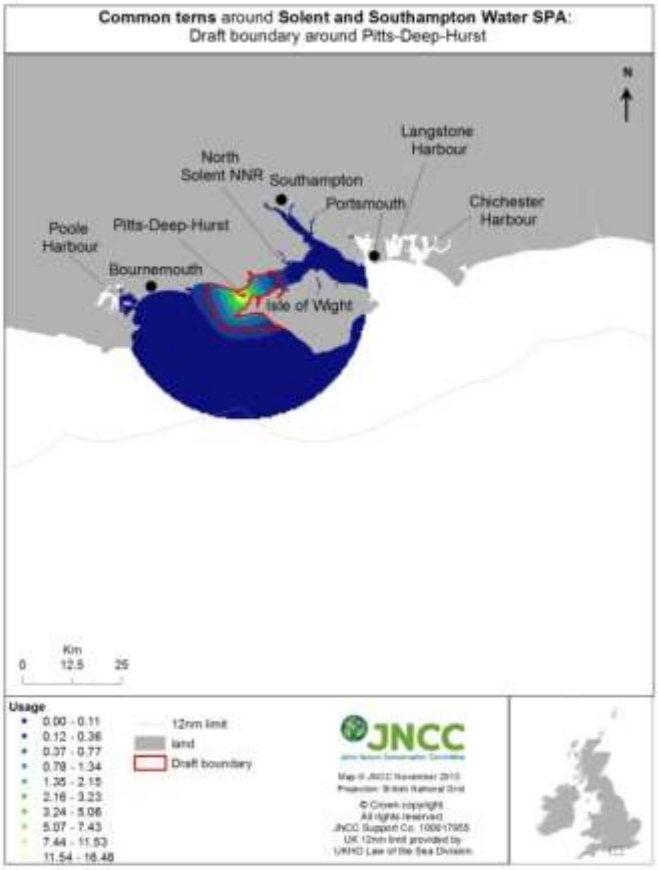
The total number of tracks obtained was 1004 including 55 tracks (6%) for roseate tern (2 SPAs), 184 tracks (18%) for Arctic tern (6 SPAs, 1 non-SPA), 381 tracks (38%) for common tern (7 SPAs, 1 non-SPA) and 384 tracks (38%) for Sandwich tern (5 SPAs, 1 non-SPA), with multiple years of data collected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

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 principal larger tern colonies within the Solent and Dorset Coast pSPA. Further general information on these surveys is presented in Annex 5.

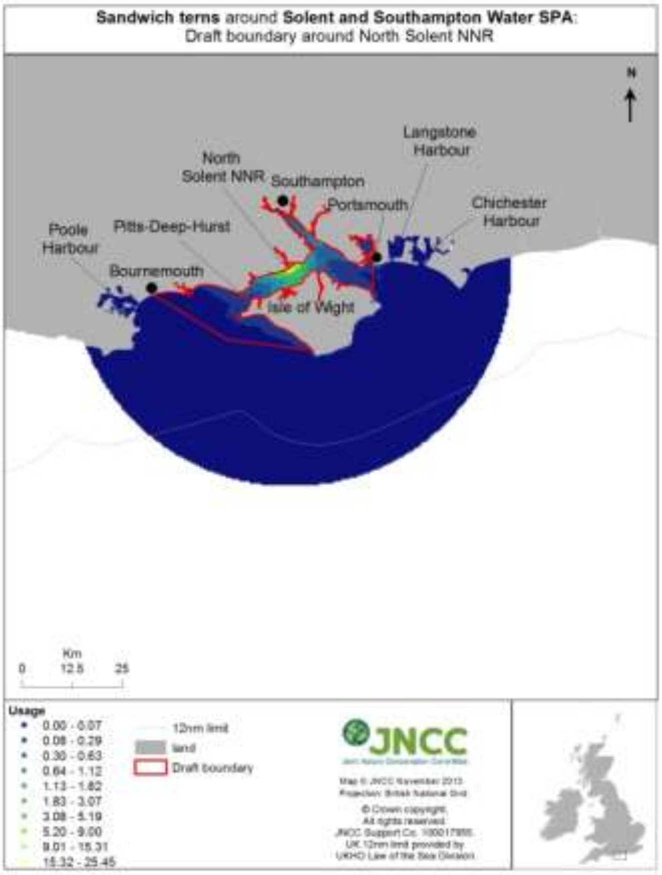
2.2.2.1. Solent & Southampton Water SPA

Breeding common, Sandwich, roseate (and little) terns are qualifying features for the Solent & Southampton Water SPA. However, roseate terns have not bred within the SPA since 2006 and so have not been considered in boundary definition. For the Solent and Southampton Water SPA the species of interest were common and Sandwich terns. Generic models for each species, generated from pooled data obtained from surveys of tern colonies across the UK as described in section 2.2.2, were used to generate foraging behaviour boundaries around source colonies within the Solent & Southampton Water SPA. Predictions of relative usage were made for breeding Sandwich terns originating from North Solent NNR and for Sandwich and common terns originating from Pitts-Deep-Hurst. 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 each colony, generally decreasing with increasing distance from each colony.

The model generated predictions of relative usage by each tern species, 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 Figure 2. In each of the images in Figure 2 the extent of the area of prediction was defined by the limit of the dark blue circles shown. 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.



a)



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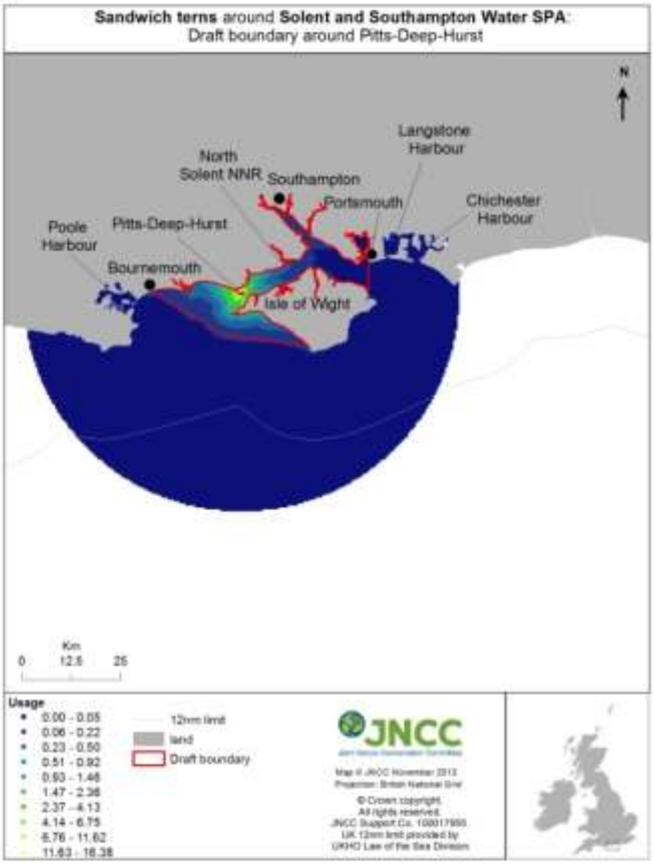


c)

Figure 2 Model predictions of tern usage overlaid with maximum curvature derived limits to areas of most importance around the North Solent NNR and Pitts Deep-Hurst colonies for (a) common tern and (b, c) Sandwich terns. Source: Win et al. (2013). In b) and c) the red line boundary defines the extent of a composite of the areas exceeding the threshold level of usage defined by the application of maximum curvature to the predicted usage maps around each location separately. It is only for this reason that it does not appear to match usage patterns around either colony on its own.

Based on generic model predictions of usage by both species from their source colonies within the SPA, and application of the maximum curvature technique to each species and colony specific predicted usage map, a composite boundary to the important foraging areas of birds of both species from both colonies within the SPA is shown in Figure 3. This indicates a potential SPA boundary for foraging terns from this SPA alone. This defines the seaward boundary of the central part of the Solent and Dorset Coast pSPA.

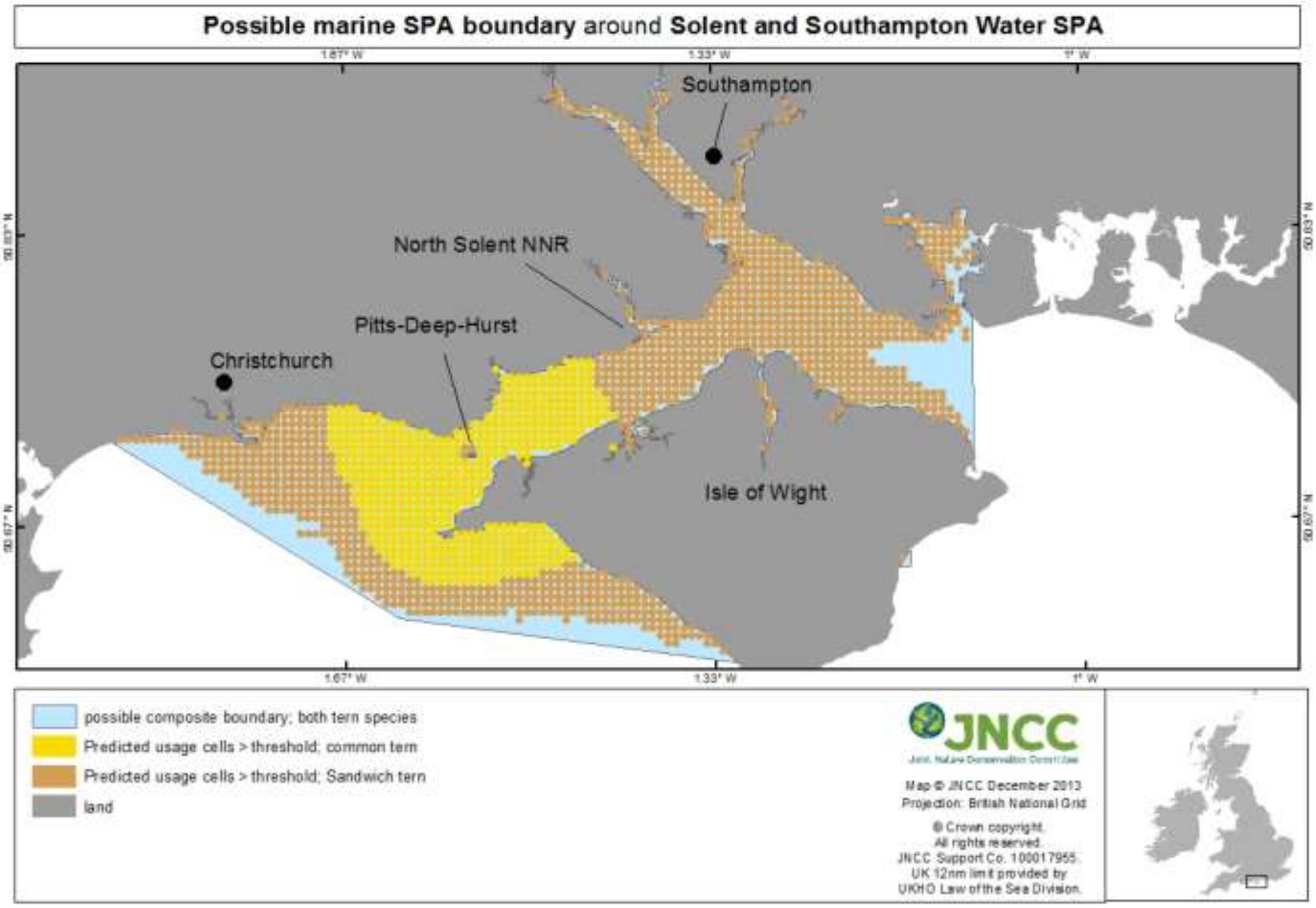


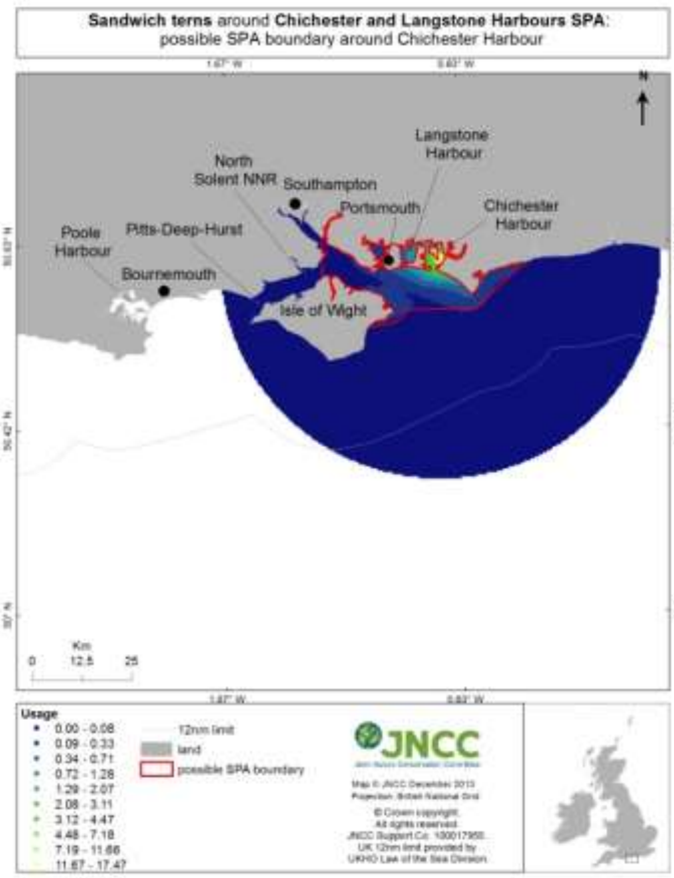
Figure 3. Proposed simple, composite boundary drawn around the cells within which predicted usage levels by either common terns or Sandwich terns, centred on each source colony, exceeded the threshold level identified by application of the maximum curvature methodology to the predicted usage surfaces (see Annex 5). Source: Win et al. (2013).

2.2.2.2. Chichester & Langstone Harbours SPA

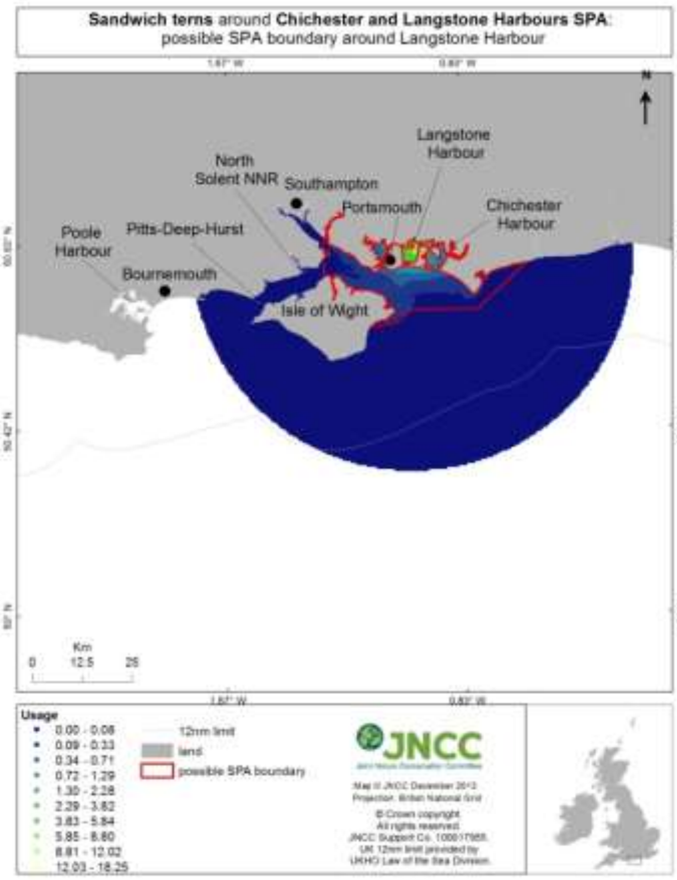
Breeding common, Sandwich (and little) terns are qualifying features for Chichester & Langstone Harbours SPA. For Chichester and Langstone Harbours SPA, the species of interest were common and Sandwich tern.

Generic models for each species, generated from pooled data obtained from surveys of tern colonies across the UK as described in section 2.2.2, were used to generate foraging behaviour boundaries around source colonies within the Chichester & Langstone Harbours SPA. Predictions of relative usage were made for both breeding Sandwich terns and common terns originating from both Chichester Harbour and Langstone Harbour. 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 each colony, generally decreasing with increasing distance from each colony.

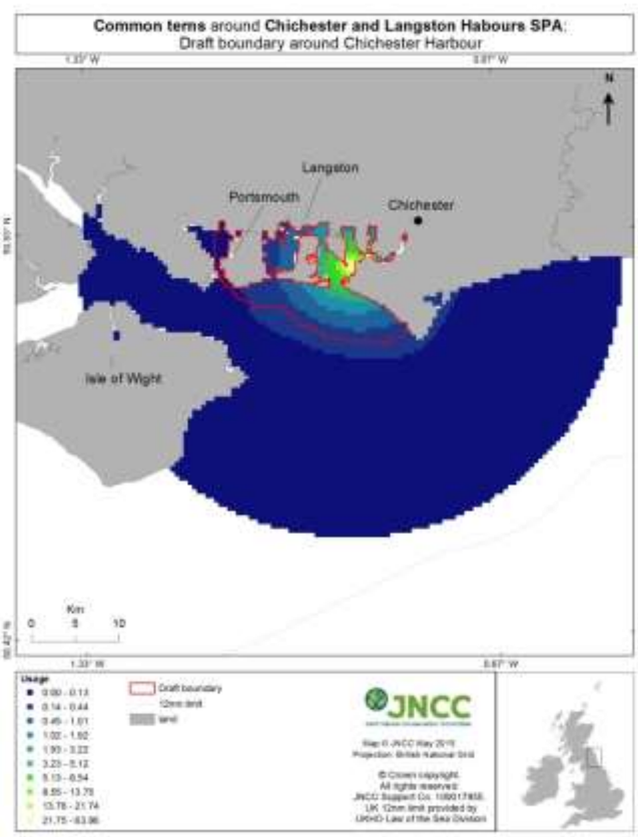
The model generated predictions of relative usage by each tern species, together with the boundary drawn around all of the areas in which predicted usage by that species exceeded the threshold identified by application of the maximum curvature approach (to define a limit to the extent of the most important areas) are shown in Figure 4. In each of the images in Figure 4 the extent of the area of prediction was defined by the limit of the dark blue circles shown. 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.



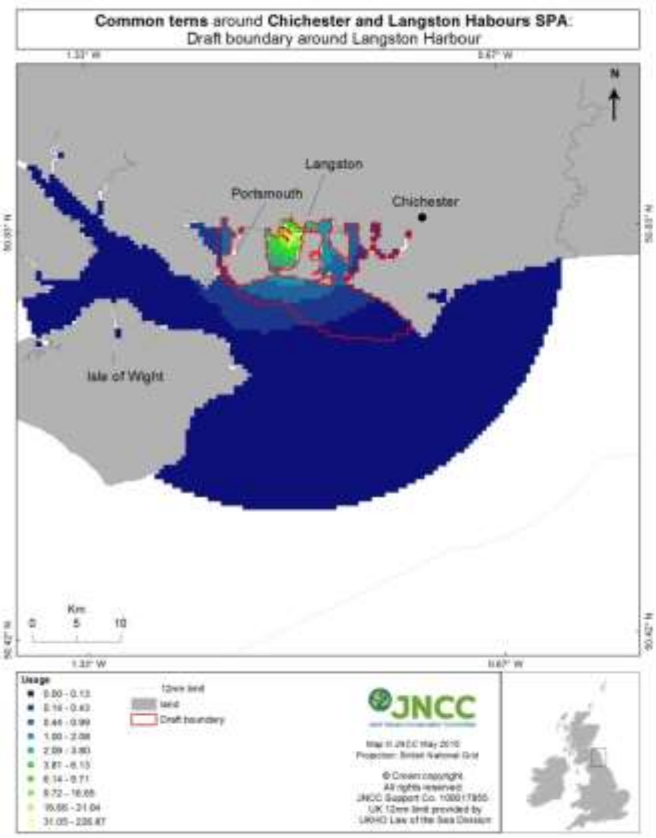
a)



b)



c)



d)

Figure 4. Model predictions of tern usage overlaid with maximum curvature derived limits to areas of most importance around the Chichester Harbour colonies (a) and c)) and Langstone Harbour colonies (b) and d)) for Sandwich terns (a) and b)) and common terns (c) and d)). Source: a) and b) Win et al. (2013), c) and d) JNCC unpublished information. The red line boundary for Sandwich tern shown in a) and b) defines the extent of a composite of the areas exceeding the threshold level of usage defined by the application of maximum curvature to the predicted usage maps for Sandwich terns around each location separately. It is only for this reason that it does not appear to match usage patterns around either colony on its own. The same applies to the red line boundary shown in c) and d) in the case of common tern.

Based on generic model predictions of usage by both species from their source colonies within the SPA, and application of the maximum curvature technique to each species and colony specific predicted usage map, a composite boundary to the important foraging areas of birds of both species from both colonies within the SPA is shown in Figure 5. This indicates a potential SPA boundary for foraging terns from this SPA alone. This defines the seaward boundary of the eastern part of the Solent and Dorset Coast pSPA.

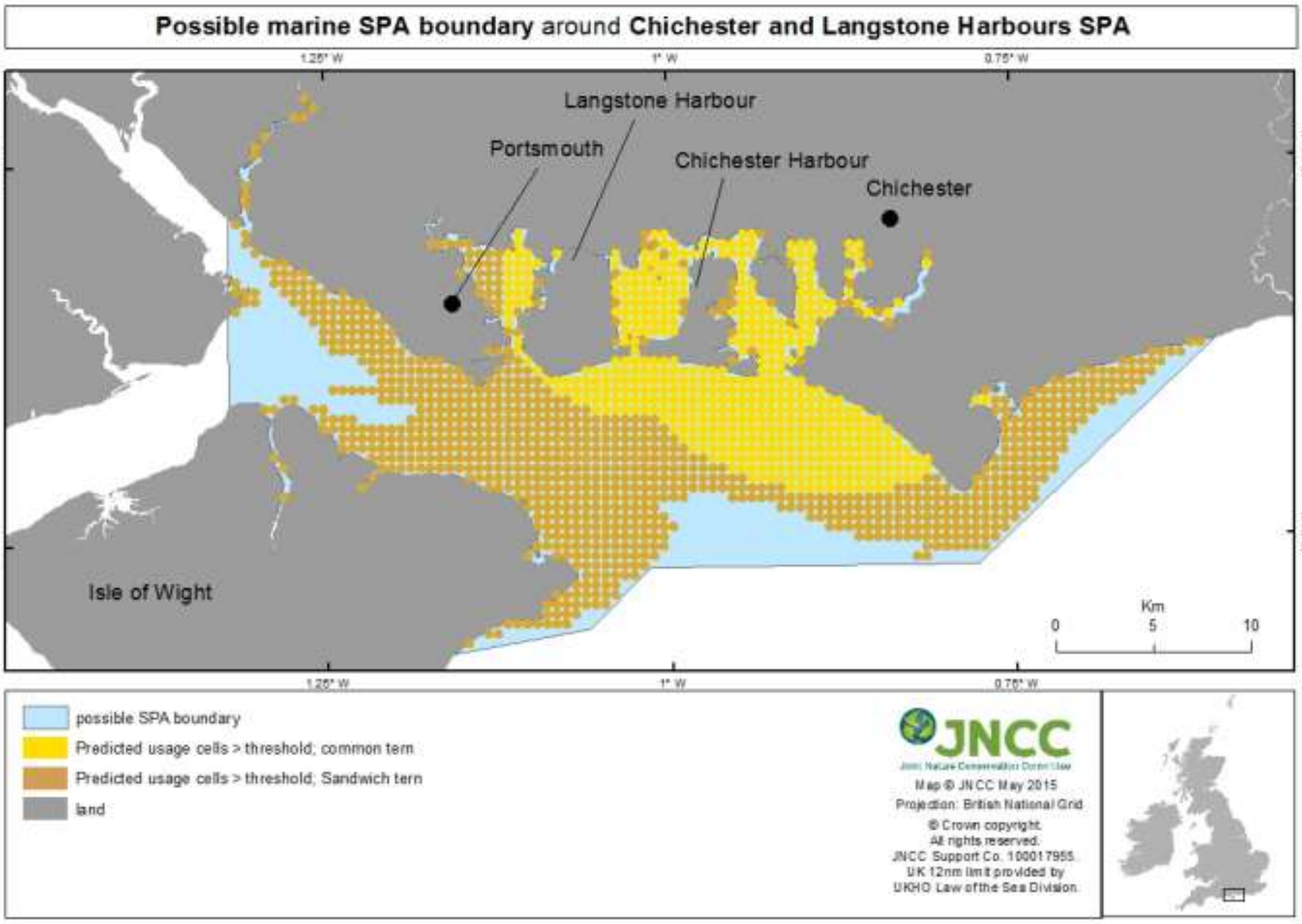


Figure 5. Proposed simple, composite boundary drawn around the cells within which predicted usage levels by either common terns or Sandwich terns, centred on each source colony, exceeded the threshold level identified by application of the maximum curvature methodology to the predicted usage surfaces (see Annex 5). Source: Win et al. (2013) (amended by inclusion of unpublished common tern data not presented in this source document).

2.2.2.3. Pagham Harbour SPA

Breeding common (and little) terns are a qualifying feature for Pagham Harbour SPA. However, numbers of common tern are recorded in the Natura 2000 Standard Data Form as being 0 pairs (as at 1996). Common terns have never bred in qualifying numbers in Pagham Harbour. Sandwich Terns have nested on Tern Island in Pagham Harbour but not in sufficient numbers to qualify as a feature of the SPA. Accordingly, no models of either Sandwich tern or common tern usage patterns were produced for this SPA and the numbers of both species supported by the pSPA do not include any numbers from Pagham Harbour. The extension of the eastern extremity of the pSPA to the east of the mouth of Pagham Harbour arises from the predicted alongshore usage pattern of Sandwich terns originating from Chichester Harbour. As Sandwich terns that are a qualifying feature of the Chichester and Langstone Harbours SPA are predicted to forage within Pagham Harbour, where the species is not a qualifying feature, the boundary of the pSPA will be drawn to Mean High Water within Pagham Harbour and so overlap the existing Pagham Harbour SPA. [NB. Where a tern species is already an existing feature of a contributing classified SPA, the pSPA boundary will abut its boundary – generally at mean low water]

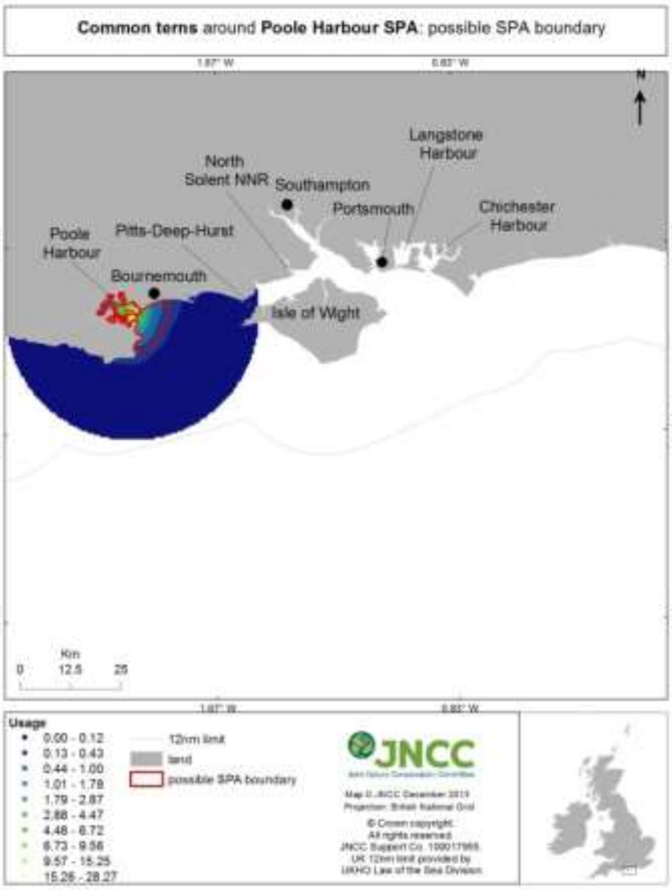
2.2.2.4. Portsmouth Harbour SPA

Historically, terns have attempted to nest on Pewit Island (NE *pers. comm.*). However, no species of tern qualifies as a feature for Portsmouth Harbour SPA. For this reason, no JNCC surveys were carried out at this SPA and no model predictions of usage patterns have been generated. As terns that are qualifying features of the nearby SPAs are predicted to forage within Portsmouth Harbour SPA, where they are not features, the new Solent and Dorset Coast pSPA boundary will be drawn to Mean High Water within Portsmouth Harbour and so overlap the existing Portsmouth Harbour SPA. [NB. Where a tern species is already an existing feature of a contributing classified SPA, the pSPA boundary will abut its boundary – generally at mean low water]

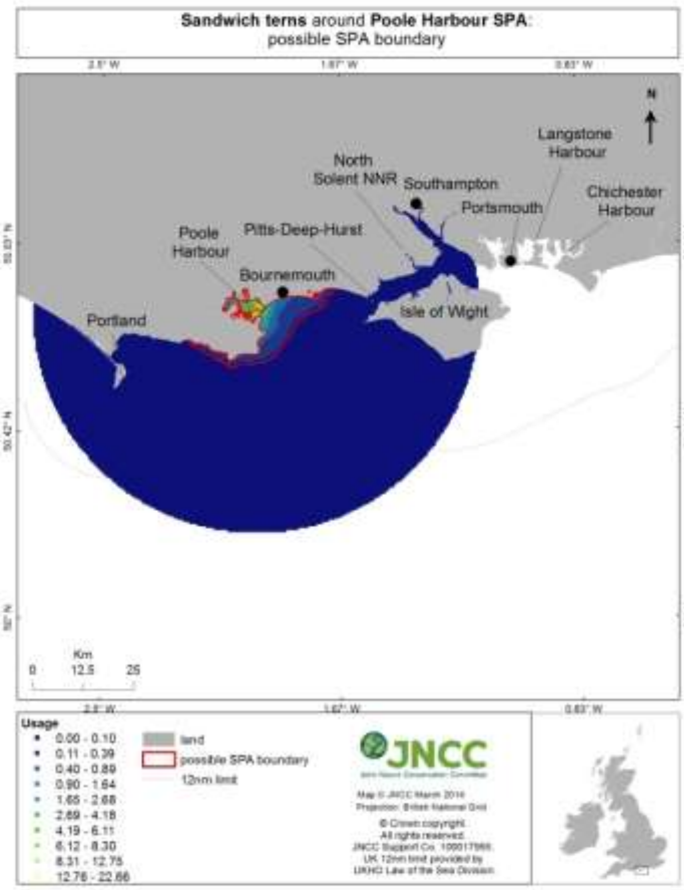
2.2.2.5. Poole Harbour SPA

Breeding common terns are a qualifying feature of the Poole Harbour SPA. Sandwich terns have bred in numbers exceeding the qualifying threshold of 110 pairs successively since 2003 and will be proposed as a new feature of this SPA in a forthcoming re-classification of the site (Poole Harbour pSPA Departmental Brief (Natural England 2015)). Thus, for Poole Harbour SPA, the species of interest were common and Sandwich terns. Generic models (phase 2), generated from pooled data obtained from surveys of tern colonies across the UK, were used to generate a foraging extent boundary around the source colony within the Poole Harbour SPA. The predictor variables used in the generic models to generate usage patterns of both common and Sandwich terns at this SPA were: i) distance to colony, ii) distance to shore, and iii) bathymetry. Predicted usage levels of common terns radiated out from the colony on Brownsea Island near the harbour mouth, generally declining with increasing distance from the colony both inside and outside the harbour. Predicted usage levels of Sandwich terns showed the same basic pattern but extended somewhat further out to sea and particularly further west and east along the shore than in the case of the common tern.

The model generated predictions of relative usage by each tern species, 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 Figure 6. In each of the images in Figure 6 the extent of the area of prediction was defined by the limit of the dark blue circles shown. This reflects the constraint imposed on the modelling by use of a radius the size of the global mean maximum distance to colony derived from tracking data held by JNCC, ECON Ecological Consultancy Ltd (for Scolt Head, Blakeney Point and Cemlyn Bay only) and Thaxter *et al.* (2012). It can be seen in both cases that very substantial areas of sea within that wider area which are distant to the colony and/or distant from the shore are predicted to have very little or no usage by foraging terns.



a)



b)

Figure 6. Model predictions of (a) common tern and (b) Sandwich tern usage overlaid with maximum curvature derived limits to areas of most importance around the Poole Harbour colony. Source: (of (a)) Win et al. (2013), (of (b)) JNCC unpublished.

Based on generic model predictions of usage by common and Sandwich terns originating from the colony within the SPA, and application of the maximum curvature technique to each species specific predicted usage map, a composite boundary to the important foraging areas of birds of both species within the SPA is shown in Figure 7. This indicates a potential SPA boundary for foraging terns from this SPA alone. This defines the seaward boundary of the western part of the Solent and Dorset Coast pSPA.

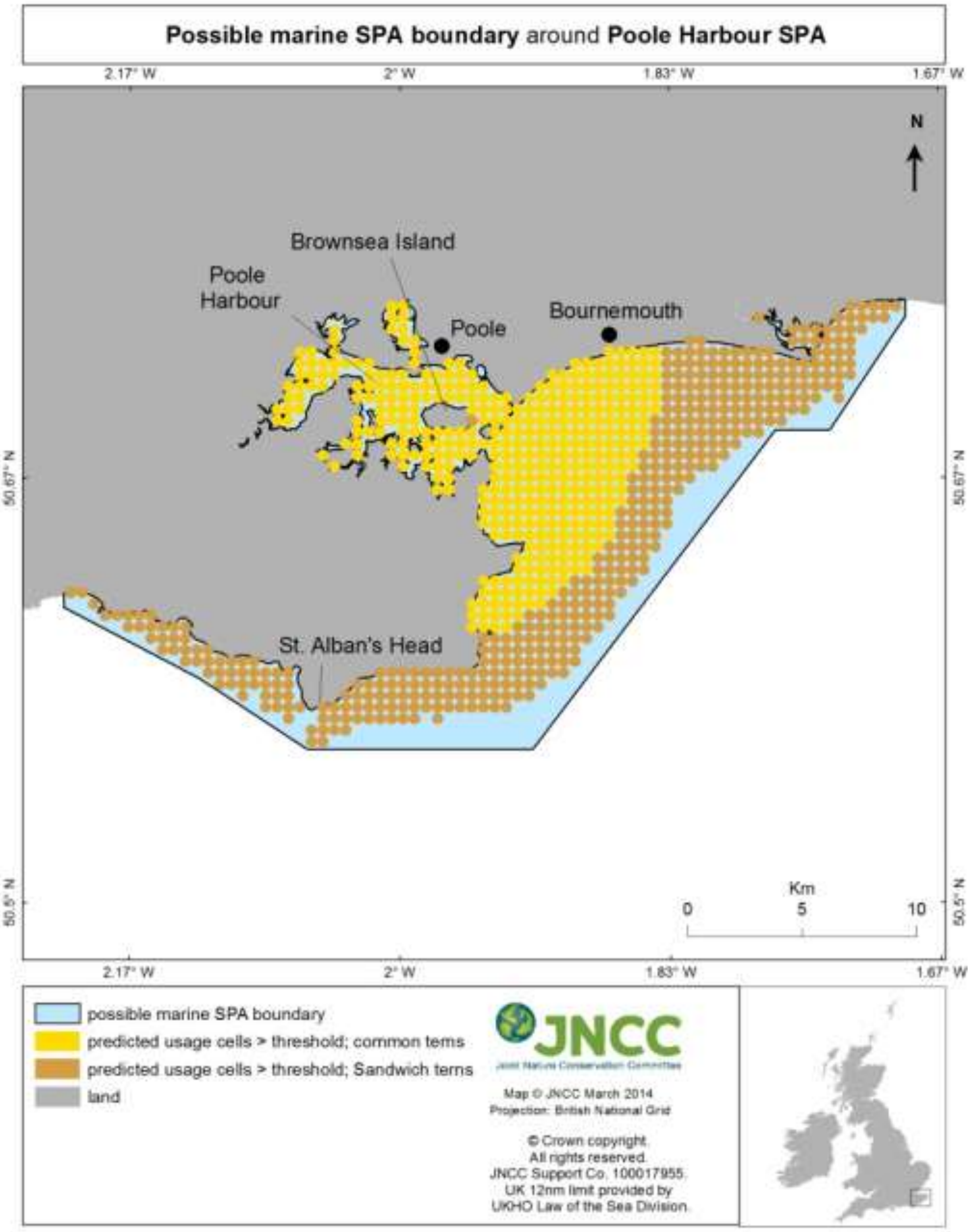


Figure 7. Proposed simple, composite boundary drawn around the cells within which predicted usage levels by common terns or Sandwich terns, centred on the source colony, exceeded the threshold level identified by application of the maximum curvature methodology to the predicted usage surfaces (see Annex 5). Source: Win et al. (2013) (amended by inclusion of unpublished Sandwich tern data not presented in this source document).

2.2.3. Composite boundary of Solent and Dorset Coast pSPA.

The seaward and alongshore extent of the Solent and Dorset Coast pSPA (Figure 8) is determined wholly by the modelled foraging distributions of Sandwich terns; from west to east by the distributions of birds originating from: Poole Harbour SPA, Solent and Southampton Water SPA (colony at Pitts-Deep-Hurst) and Chichester & Langstone Harbours SPA. It should be noted that the same generic model of Sandwich tern usage was used to generate relative density maps around each of these colonies. However, the precise size and shape of the areas defined by application of the maximum curvature method to the predicted density maps varies between colonies. This is due to the model containing bathymetry as a covariate and to there being differences in this respect between the sea areas around each colony. The overlapping nature of these foraging ranges is also evident, meaning that some sea areas will be supporting birds of the same species from different SPAs. The boundaries to the areas predicted to support most of the foraging activity by little terns originating from colonies within the Solent and Southampton Water SPA, and Chichester & Langstone Harbours SPA (and Pagham Harbour SPA) are contained entirely within the composite boundary of the pSPA, as are the areas predicted to support most of the foraging activity by common terns originating from colonies within the Chichester & Langstone Harbours SPA, the Solent and Southampton Water SPA and Poole Harbour SPA.

Given that in the case of this pSPA, its size and shape is determined purely on the basis of predictions of Sandwich tern usage patterns generated by a generic model, rather than a model based on observations of Sandwich terns on the south coast of England, 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 demonstrated that of the three generic models, the Sandwich tern model has the greatest reliability, as judged by its ability to predict the observed distribution of Sandwich terns at colonies which were (in the cross-validation process) excluded in turn from building the model. Overall, the average test statistic for this cross-validation process was classed as indicative of the model being “excellent”. This analysis indicated that there is great consistency between colonies around the UK in the characteristics of sea areas which hold the highest relative densities of foraging Sandwich terns. Accordingly, there is a correspondingly high degree of confidence that the boundary of this pSPA, being dependent upon the predicted usage patterns of Sandwich terns, is founded on a reliable evidence base, albeit not one derived directly from birds at the colonies in question.

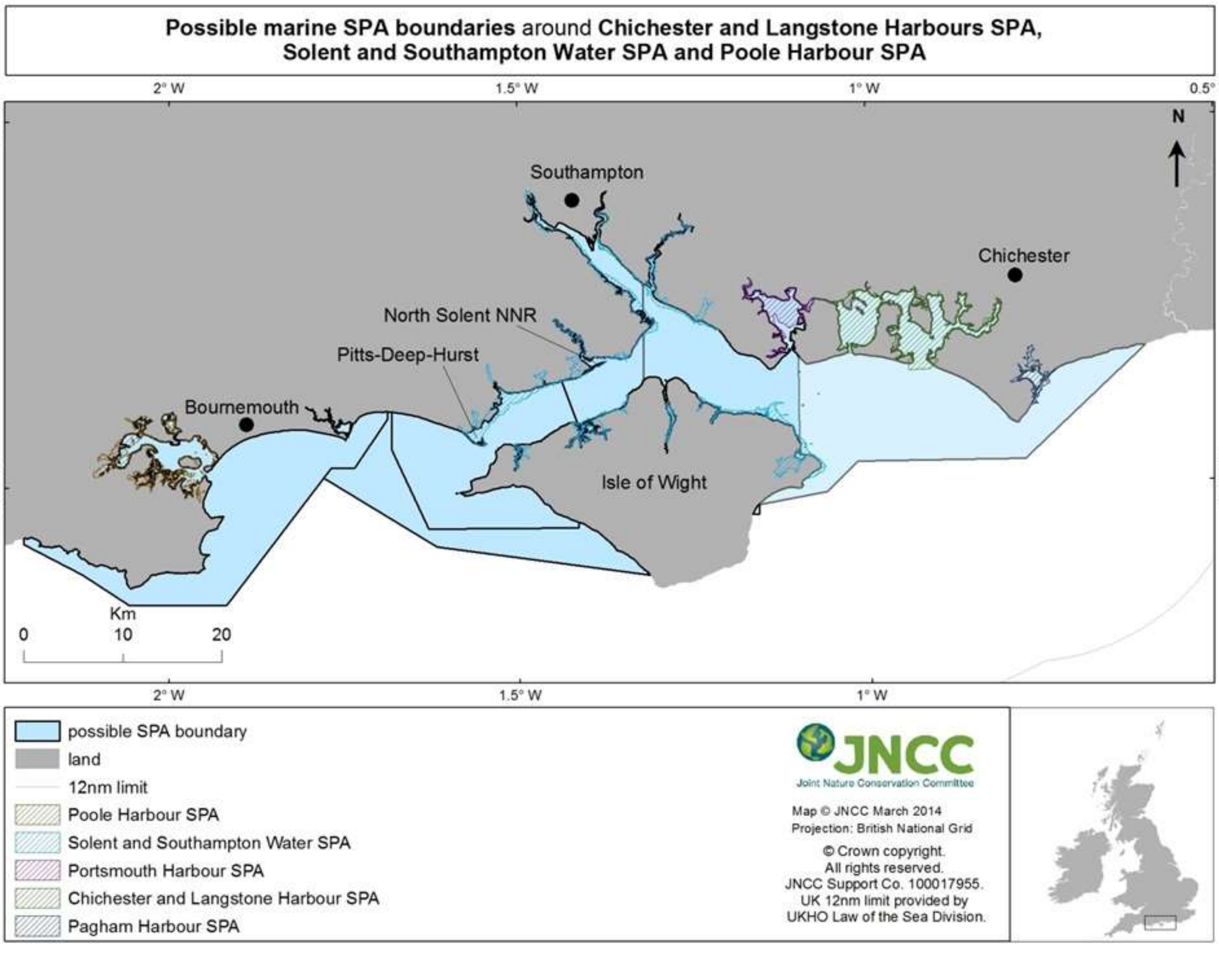


Figure 8. Proposed simple, composite boundary of the Solent & Dorset Coast pSPA drawn around the species and site specific boundaries presented in the preceding sections.

The overall site boundary was drawn as a composite of the separate species-specific boundaries as can be seen in sections 2.2.2.1 to 2.2.2.5.

3. Site status and boundary

By virtue of encompassing the principal sea areas of importance to support the foraging of populations of three Annex 1 species of tern (little tern, common tern and Sandwich tern) at three existing SPAs where the combined numbers of those species exceed SPA selection guidelines (JNCC 1999), the Solent and Dorset Coast pSPA itself qualifies by regularly supporting those nationally important numbers of breeding Annex 1 species.

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

3.1. Coastal boundary

The landward boundary is located on the coasts of Dorset, Hampshire, Isle of Wight and West Sussex. The westernmost extremity of the boundary is at Worbarrow Bay in Dorset and the easternmost extremity of the boundary lies approximately 88km to the east at Bognor Regis in West Sussex (Figure 8). The westernmost extremity is determined by the modelled usage of Sandwich terns foraging from the Poole Harbour SPA, whereas the easternmost extremity is determined by the modelled usage of Sandwich terns foraging from Chichester & Langstone Harbours SPA.

JNCC guidelines on selecting marine SPAs (Webb & Reid 2004: Annex B) states that where the distribution of birds is likely to meet land, landward boundaries should be set at Mean High Water (MHW) “unless there is evidence that the qualifying species make no use of the intertidal region at high water”. Observations indicated that little terns forage both in the intertidal zone and subtidal zone (Parsons *et al.* 2015). That the use of such areas by all larger tern species is also likely is supported by information in the scientific literature. A review of tern foraging ecology (Eglington 2013) notes that all three species of tern considered here routinely forage in areas of shallow water. There is no reason on the basis of that review to consider it likely that that these birds will not forage over intertidal areas. Thus, along this stretch of coast the boundary will come to Mean High Water, with the exception of those places where the tern species are already qualifying features of existing SPAs and the areas between Mean High Water and Mean Low Water are already protected for these features within the existing SPAs. In those places, the boundary of the pSPA will not overlap but will abut with that of the existing SPA (at the harbour mouth in the case of Poole Harbour SPA, and at MLW in the cases of Solent and Southampton Water and Chichester and Langstone Harbour. Pagham Harbour SPA and Portsmouth Harbour SPA lie within the predicted area of usage of Sandwich terns originating from Chichester & Langstone Harbours SPA. However, Pagham Harbour does not hold Sandwich tern in qualifying numbers and Portsmouth Harbour does not hold any of the three tern species in qualifying numbers. Thus, the landward boundary of the pSPA will extend to Mean High Water within both Pagham and Portsmouth Harbours and hence overlap with the existing SPA boundaries.

3.2. Seaward boundary

The pSPA wholly occupies The Solent with its seaward extent boundary beyond The Solent consisting, for simplicity, of angled straight lines. The seaward boundary consists broadly of three arcs running west to east (Figure 8). The furthermost extremes of these arcs lie at approximately 5km, 12km and 12km from the nearest points of the mainland shore and are determined by the modelled usage of Sandwich terns foraging from Poole Harbour SPA, Solent and Southampton Water SPA and Chichester & Langstone Harbour SPA.

3.3. Boundary around offshore islands

The seaward boundary of the Solent and Dorset Coast pSPA partially encompasses the coastline of the Isle of Wight. The boundary extends clockwise around the Isle of Wight coast from Blackgang Chine, near the southern tip of the island, around the whole of the northern shore of the island and round to the southeast side at Sandown. As stretches of the Solent and Southampton Water SPA already occupy sections of the Isle of Wight coastline, the pSPA boundary correspondingly extends to either Mean Low Water to abut this SPA or to Mean High Water where the coastline is not existing SPA.

Other islands to consider are Hayling Island, the estuarine islands of Langstone Harbour and Pewit Island, Portsmouth Harbour. The boundary will extend to Mean High Water along the south coast of Hayling Island where outside existing SPAs and around Pewit Island. However, the boundary will not reach the islands of Langstone Harbour (North Binness, South Binness, Long Island and Baker's Island) by virtue of these being well inside an existing SPA with terns as qualifying features, located away from Low Mean Water.

The Napoleonic forts within The Solent (Spitsand Fort and Horse Sand Fort) will be wholly surrounded by the pSPA boundary and since they are outside the Solent and Southampton Water SPA, the new boundary will extend up to Mean High Water and will exclude the actual structures above.

4. Location and habitats

The Solent and Dorset Coast pSPA is located along the coasts of Dorset, Hampshire, Isle of Wight and West Sussex and adjacent areas offshore. It will overlap, abut and be close to many designated areas, summarised below.

At its western point, South Dorset Coast SSSI, Townsend SSSI, Purbeck Ridge (West & East) SSSI, Studland Cliffs SSSI, Studland & Godlingson SSSI, Poole Bay Cliffs SSSI and Boscombe & Southbourne Overcliff LNR, Bournemouth fall just outside the pSPA boundary and have no tern interest.

A cluster of adjacent, overlapping and underpinning designations exist at Christchurch Harbour, Hengistbury Head and adjacent undeveloped land, which is surrounded by the Bournemouth and Christchurch sprawling conurbation on one side and open sea on the other. These designations include Dorset Heathlands SPA, Dorset Heaths SAC, Christchurch Harbour SSSI, Hengistbury Head LNR and Stanpit Marsh LNR, and are fed by the narrow Avon Valley (Bickton to Christchurch) Ramsar and SSSI, and River Avon System AONB and SSSI.

The new pSPA boundary will extend to Mean High Water and so become underpinned by Christchurch Harbour SSSI. Its citation describes, "The site comprises the drowned estuary of the rivers Stour and Avon and the peninsula of Hengistbury Head. The varied habitats include saltmarsh, wet meadows, drier grassland, heath, sand dune, woodland and scrub and the site is of great ornithological interest." Terns are not mentioned on the citation but, "Sandwich, common and little terns all occur in the harbour and have tried to nest in the past but they face a lot of pressure from predators, disturbance and tides and sadly have not been successful." (Christchurch Harbour Ornithological Group, 2007).

Along 9km of coastline, from Christchurch in Dorset to Milford on Sea in Hampshire, the cliffs and intertidal zone are designated Highcliffe to Milford Cliffs SSSI, which will underpin the pSPA.

Along the northwest coast of The Solent, the Solent and Southampton Water SPA and Ramsar overlaps with Solent Maritime SAC and these are underpinned by the Hurst Castle and Lymington River Estuary SSSI. East from River Lymington, the intertidal saltmarsh is also designated Boldre Foreshore LNR.

A series of intertidal SSSIs underpin the SPA and eastward are the North Solent SSSI (where also North Solent NNR), Dibden Bay SSSI, Hythe to Calshot Marshes SSSI, then east from Southampton is the Lee-on-Solent to Itchen Estuary SSSI (where also Hook and Warsash LNR), Titchfield Haven SSSI (also NNR and LNR) and Browndown SSSI near Gosport, Hampshire.

Portsmouth Harbour SPA is underpinned by Portsmouth Harbour SSSI, where since the pSPA will overlap this SPA, will itself become underpinned by this SSSI. Both estuaries of the Chichester and Langstone Harbours SPA are separately underpinned by SSSIs and Chichester Harbour is an AONB. Within or partly within this SPA's boundary are Farlington Marshes LNR, West Hayling LNR, Hayling Billy LNR, Gutler Point LNR, The Kench, Hayling Island LNR, Sandy Point LNR, Earnes Farm LNR, Nutborne Marshes LNR, Pilsey Island LNR, The Solent Maritime SAC overlaps this SPA and further extends nearly 2km out from the south coast of Hayling Island.

Since the southwest headland and offshore sandbank of Hayling Island are designated Sinah Common

SSSI and only abuts with the Solent Maritime SAC, the situation is thus created wherein the intertidal part of this SSSI up to Mean High Water will underpin the pSPA, but not the SAC.

In West Sussex, the shoreline for approximately 8.2km from near West Wittering, southeast to Selsey is designated the Bracklesham Bay SSSI. Pagham Harbour SPA is underpinned by Pagham Harbour SSSI and abutting this along the intertidal zone for approximately 4km adjacent to Bognor Regis is the Bognor Reef SSSI.

On the Isle of Wight, almost the entire coastal cliff-line and intertidal zone along the southwest side is designated SSSI, including Compton Chine to Steephill Cove SSSI, Compton Down SSSI and Headon Warren and West High Down SSSI, which occupies The Needles and west headland; all fall also within the Isle of Wight AONB. Colwell Bay SSSI occupies 1.7km of the northwest coastline. At Yarmouth is the Yar Estuary SSSI; either side and including Newtown is the Bouldnor and Hamstead Cliffs SSSI, Newtown Harbour SSSI (and NNR) and Thorness Bay SSSI.

The entire northeast Isle of Wight coastline, around Foreland (east headland of Isle of Wight) and to Yaverland, near Sandown along the southeast coastline is all SSSI. Clockwise these are King’s Quay Shore SSSI, Ryde Sands and Wootton Creek SSSI, Brading Marshes to St Helen’s Ledges SSSI, Whitecliff Bay and Bembridge Ledges SSSI and Bembridge Down SSSI.

Solent and Southampton Water SPA and Solent Maritime SAC also occupy intertidal and sub-tidal areas of the Isle of Wight. The South Wight Maritime SAC is relatively large, extending up to 4km out from the southwest coastline and out over 8km from the southeast coastline. Below Mean Low Water, these SACs are not underpinned by SSSIs.

Around the Isle of Wight, ‘The Needles’ Marine Conservation Zone was consulted on by Defra in spring 2015. Other MCZs have been recommended and may come forward in future MCZ tranches. The Needles falls wholly within the pSPA boundary.

5. Assessment of ornithological interest

5.1. Survey information and summary

Details of the site-specific survey work carried out to characterise the foraging areas used by breeding adult terns originating from existing SPAs around the coast of the UK are detailed in Appendices 4 and 5. The purpose of those surveys was to provide a robust empirical dataset on which to base predictive models of habitat usage around tern colonies, and not to derive estimates of the number of individuals of any species at sea within the boundaries of any marine pSPA at any point in time. Accordingly, the numbers detailed in Table 1 and in the following sections represent the numbers of breeding individuals of each species from contributing SPAs only which will make use of the resources provided by the supporting habitats within the Solent & Dorset Coast pSPA during the course of the breeding season rather than the numbers of birds from those sites that will be present within the site boundary at any particular point in time. Furthermore, breeding adult terns (and other terns) from colonies outside existing SPAs, but which may also forage within the pSPA, are excluded from the numbers presented in Table 1 and the following tables.

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

5.2. Annex 1 species

5.2.1. Common tern *Sterna hirundo*

The breeding population of common terns in Great Britain is estimated to be 10,000 pairs (Musgrove *et al.* 2013), representing at least 2% of the Northern & Eastern European breeding population (500,000 pairs derived by division by 3 of the upper estimate of 1,500,000 individuals: AEWA 2012). A significant proportion of the British population breeds in Scotland. Coastal colonies in England are concentrated in the north-east, East Anglia, at a few localities along the south coast, and in the north-west (Mitchell *et al*. 2004). Common terns breed not only around coasts but, unlike the other tern species which breed in the UK, also breed frequently beside inland freshwater bodies.

5.2.1.1. Poole Harbour SPA

Brownsea Island lagoon is the site of the principal and probably only nesting colony of common terns within the SPA. It is possible that occasional pairs have and still do nest elsewhere on islands within the harbour but there are no confirmed records.

The SPA citation states 155 pairs, at that time representing 1.3% of the GB breeding population (5-year peak mean, 1993-1997).

At Brownsea Lagoon, the number of pairs of common terns nesting during a recent 5-year period (2010-2014) were - **191** (2010), **222** (2011), **171** (2012), **163** (2013), **145** (2014). This provides a recent 5-year mean of **178** pairs (or 356 breeding adults). This represents 1.78% of the GB breeding population.

5.2.1.2. Solent & Southampton Water SPA

Common terns nest in loose colonies on extensive saltmarshes either side of the mouth to the River Lymington, sporadically across the site from Hurst Spit to Normandy Lagoon at scattered locations (Pylewell, Boiler, Cockleshell, Normandy Marsh) and sometimes several pairs attempt to nest on Normandy Lagoon. Elsewhere within the SPA, breeding occurs further east along the coast at North Solent NNR and on the Isle of Wight in Newton Harbour SSSI, but information on the breeding habits of birds from these sites is limited.

The SPA citation states 267 pairs, at that time representing 2.2% of the GB breeding population (5-year peak mean, 1993-1997).

From Hurst Spit to Normandy Lagoon, the number of pairs of common terns nesting during a recent 5-year period (2010-2014) were - **209** (2010), **145** (2011), **95** (2012), **210** (2013), **156** (2014).

At North Solent NNR, 10 nests were abandoned in 2013. As this site reportedly does not lend itself to counting terns, information from other years is scant and the 2013 count is not used.

At Newton Harbour SSSI, gulls reportedly took over the nesting site from 2008 and likely contributed to the lack of nesting attempts between 2009 and 2012. Nonetheless, 3 or 4 pairs of common terns nested in 2013 and 2014, and these counts are included in calculation of the pSPA population.

In total, the 5 year mean across these sites, and hence the SPA as a whole, is 164 pairs representing 328 breeding adults. This represents 1.64% of the GB breeding population.

5.2.1.3. Chichester & Langstone Harbours SPA

Common terns nest in mixed colonies on extensive shingle ridges and islands within Langstone harbour. At the North East side of the harbour abandoned oyster beds off of Hayling Island provide an artificial lagoon which provides foraging and nesting habitat for the terns. Within Chichester Harbour common terns nest on the shingle banks near to the harbour entrance. The Chichester Harbour Conservancy have experimented with creating shingle banks away from the entrance with no breeding success to date.

The SPA citation (1996) provides no figures for tern species, but the Natura 2000 Standard Data Form (JNCC, updated 1999) states 33 pairs as a 5-year mean (1992-1996) at the time representing 0.3% of the GB breeding population.

At Langstone Harbour, the annual maximum number[3] of pairs of common terns during a recent 5-year period (2009-2014[2]) were - **125** (2009†), **161** (2010†), **181** (2011‡†), **85** (2013‡†), **117** (2014‡).

At Chichester Harbour, the annual maximum number[1] of pairs of common terns during a recent 5-year period (2009-2014[4]) were - **18** (2009‡), **10** (2010‡), **35** (2011‡); flooded, no record (2012), **12** (2013‡), **1** (2014‡).

In total, the 5 year mean across these 2 sites, and hence the SPA as a whole, is **149** pairs, representing 298 breeding adults. This represents 1.49% of the GB breeding population.

5.2.1.4. Pagham Harbour SPA

Common tern is a qualifying feature of this SPA. However, numbers of common tern are recorded in the Natura 2000 Standard Data Form as being 0 pairs (as at 1996). Common terns have never bred in qualifying numbers in Pagham Harbour. The annual maximum number of pairs of common terns during a recent 5 year period (2010-2014) are - **5** (2010), **8** (2011), **14** (2012), **14** (2013), **13** (2014). This provides a recent 5 year mean of **11** pairs (or 22 breeding adults). This represents 0.11% of the GB breeding population. This population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging range from Pagham Harbour SPA is not considered in defining the boundary of the pSPA.

5.2.1.5. Solent & Dorset Coast pSPA

The principal common tern breeding colonies supported by the Solent & Dorset Coast pSPA during the breeding season are located at: Poole Harbour SPA, Solent & Southampton Water SPA and Chichester & Langstone Harbours SPA. The sum of the site-specific recent 5 year means across these SPAs yields a figure of 492 pairs or 984 breeding adults supported by the pSPA which constitutes 4.92% of the GB breeding population (Table 3).

Table 3. Summary of breeding populations of common tern within SPAs contributing to the foraging population of the Solent and Dorset Coast pSPA.

	Poole Harbour	**Solent and Southampton Water**	**Chichester and Langstone Harbours**
Citation or SDF (pairs)	155	267	33
Old % GB popln	1.3%	2.2%	0.3%
Data age	1993-1997	1993-1997	1992-1996
Recent mean (pairs)	178.4	164.2	149.0
Recent % GB popln	1.8%	1.6%	1.5%
Data age	2010-2014	2010-2014	2009-2011, 2013-2014
Solent and Dorset Coast pSPA population (sum max **mean** x2 for **individuals**):			983.2
Solent and Dorset Coast pSPA population: % of GB breeding population:			4.92%

5.2.2. Sandwich tern *Sterna sandvicensis*

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

5.2.2.1. Poole Harbour SPA

[3] The highest figure is used, sourced from either SMP† or the site manager‡, as indicated
[4] In 2012, flooding produced an under-representative figure for Chichester Harbour (92 pairs at Langstone recorded on the SMP online database) and this year for both estuaries is therefore not used in the 5-year mean calculation.

Brownsea Island lagoon is the site of the principal and probably only nesting colony of Sandwich terns within the SPA. It is possible that occasional pairs have and still do nest elsewhere on islands within the harbour, but there are no confirmed records.

Sandwich tern is not a named feature of the existing SPA as it did not occur in numbers exceeding qualifying thresholds at the time of classification. However, its numbers have increased in recent years.

At Brownsea Lagoon, the number of pairs of Sandwich terns nesting during a recent 5-year period (2010-2014) were - **232** (2010), **116** (2011), **167** (2012), **180** (2013), **210** (2014). This provides a recent 5-year mean of **181** pairs (or 362 breeding adults). This represents **1.65%** of the GB breeding population and accordingly it is proposed to add the species as a qualifying feature of the Poole Harbour pSPA in its reclassification (Poole Harbour pSPA Departmental Brief, Natural England 2015).

5.2.2.2. Solent & Southampton Water SPA

Sandwich terns tend to form large colonies that decline and increase in size locally in response to controlling factors such as predation. Within this SPA, individuals move between colonies on Hawkers Island near Keyhaven, Cockleshell and Pylewell Marsh.

The SPA citation states 231 pairs, at the time representing 1.7% of the breeding population in Great Britain (5 year peak mean, 1993-1997).

The annual maximum number[5] of pairs of Sandwich terns during a recent 5-year period (2010-2014) were – **60-70**[6] (2010[‡]), **96** (2011[†]), **106** (2012[‡]), **206** (2013[‡]), **45** (2014[‡]). This provides a recent 5-year mean of **104** pairs (or **208** breeding adults). This represents 0.94% of the GB breeding population.

5.2.2.3. Chichester & Langstone Harbours SPA

Sandwich terns nest on extensive shingle ridges and islands within Langstone Harbour. At the North East side of the harbour abandoned oyster beds off of Hayling Island provide an artificial lagoon which provides foraging and nesting habitat for the terns. Within Chichester Harbour, Sandwich tern nest on the shingle banks near to the harbour entrance. The Chichester Harbour Conservancy have experimented with creating shingle banks away from the entrance with no breeding success to date.

The SPA citation (1996) provides no figures for tern species, but the Natura 2000 Standard Data Form (JNCC, updated 1999) states 31 pairs as a 5-year mean (1993-1997) at the time representing 0.2% of the GB breeding population.

At Langstone Harbour, the annual maximum number[1] of pairs of Sandwich terns during a recent 5-year period (2008-2013[2]) are – **130** (2008), **153** (2009[†]), **205** (2010[†]), **175** (2011[‡†]), **6** (2013[‡†]).

At Chichester Harbour, the annual maximum number[7] of pairs of Sandwich terns during a recent 5-year period (2008-2013[8]) are – **0** (2008), **30** (2009[‡]), **0** (2010[‡]), **10** (2011[‡]); flooded, no record (2012), **71** (2013[†]).

In total, the 5 year mean across these 2 sites, and hence the SPA as a whole, is **156** pairs, representing 312 breeding adults. This represents 1.42% of the GB breeding population.

5.2.2.4. Pagham Harbour SPA

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[5] The highest figure is used, sourced from either Durnell (Hampshire County Council)[†] or the site manager[‡], as indicated.

[6] The median figure of 65 is used for the 5-year peak mean calculation.

[7] The highest figure is used, sourced from either SMP[†] or the site manager[‡], as indicated

[8] In 2012, flooding produced an under-representative figure for Chichester Harbour and this year for both estuaries is therefore not used in the 5-year mean calculation (46 pairs at Langstone recorded on the SMP online database). No count was available for Chichester Harbour in 2014 so to make use of the same set of years for both sites, the 2014 count from Langstone (66 pairs) was not used, and the counts from 2008 in both locations used to generate the 5 year mean up to and including 2013.

Sandwich tern is not a qualifying feature of this SPA. Unlike in the case of Poole Harbour SPA (section 4.2.2.1), the recent numbers of this species In Pagham Harbour remain very low. The annual maximum number of pairs of Sandwich terns during a recent 7- year period (2008-2014[9]) are - **0** (2008), **1** (2009), **20** (2012), **3** (2013), **1** (2014). This provides a recent 5 year mean of **5** pairs (or 10 breeding adults). This represents 0.05% of the GB breeding population. This population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging range from Pagham Harbour SPA is not considered in defining the boundary of the pSPA.

5.2.2.5. Solent & Dorset Coast pSPA

The principal Sandwich tern breeding colonies supported by the Solent & Dorset Coast pSPA during the breeding season are located at: Poole Harbour SPA, Solent & Southampton Water SPA and Chichester & Langstone Harbours SPA. The sum of the site-specific recent 5 year means across these three principal source colony SPAs yields a figure of 441 pairs or 882 breeding adults supported by the pSPA which constitutes 4.01% of the GB breeding population (Table 4).

Table 4. Summary of breeding populations of Sandwich tern within SPAs contributing to the population of the Solent and Dorset Coast pSPA

	Poole Harbour	**Solent and Southampton Water**	**Chichester and Langstone Harbours**
Citation or SDF (prs)	n/a	231	31
Old % GB popln	n/a	1.7%	0.2%
Data age	n/a	1993-1997	1993-1997
Recent mean max (prs)	181	104	156
Recent % GB popln	1.65%	0.94%	1.42%
Data age	2010-2014	2010-2014	2008-2011 and 2013
Solent and Dorset Coast pSPA foraging population (sum max **mean** x2 for **individuals**):			882
Solent and Dorset Coast pSPA foraging population (**prs**) % of GB breeding population:			4.01%

5.2.3. Little tern *Sternula albifrons*

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

5.2.3.1. Solent & Southampton Water SPA

Little tern nest sporadically across the site from Hurst Spit to Normandy Lagoon as scattered sub-colonies and often not as one discrete colony (Durnell *unpubl*).

The SPA citation states 49 pairs, at that time representing 2.0% of the GB breeding population (5-year peak mean, 1993-1997).

The annual maximum number[10] of pairs of little terns during a recent 5-year period (2010-2014) are - **28** (2010†), **6** (2011[11]), **30** (2012†), **c.23** (2013†), **10** (2014‡). This provides a recent 5-year mean of **19** pairs (or 38 breeding adults). This represents 1.02% of the GB breeding population.

[9] Figures supplied by the RSPB and SMP tally, but from neither source were figures for 2010 and 2011 available so data from 2008 and 2009 are considered to yield 5 recent, annual population figures.
[10] The highest figure is used, sourced from either SMP† or the site manager‡, as indicated
[11] In 2011, the site manager's figure of 6+ and the SMP's figure of 0 may be under-representative but given the vagaries of little tern colony occupation are nonetheless used in the 5-year mean calculation.

5.2.3.2. Chichester & Langstone Harbours SPA

Little tern nest in mixed colonies on extensive shingle ridges and islands within Langstone Harbour. At the North East side of the harbour abandoned oyster beds off of Hayling Island provide an artificial lagoon which provides foraging and nesting habitat for the terns. Within Chichester Harbour, little tern nest on the shingle banks near to the harbour entrance. The Chichester Harbour Conservancy have experimented with creating shingle banks away from the entrance with no breeding success to date.

The citation (1996) provides no figures for tern species, but the Natura 2000 Standard Data Form (JNCC, updated 1999) states 100 pairs as a 5-year mean (1992-1996) at the time representing 4.2% of the GB breeding population.

At Langstone Harbour, the annual maximum number[1] of pairs of little terns during a recent 5-year period (2010-2014) were - **55** (2010), **57** (2011), **40** (2012), **26** (2013), **31** (2014).

At Chichester Harbour, the annual maximum number[12] of pairs of little terns during a recent 5-year period (2010-2014) were - **0** (2010), **0** (2011), **0** (2012), **0** (2013), **8** (2014[‡]).

In total, the 5 year mean across these 2 sites, and hence the SPA as a whole, is **43** pairs, representing 86 breeding adults. This represents 2.28% of the GB breeding population.

5.2.3.3. Pagham Harbour SPA

The SPA citation (1996) provides no figures for tern species, but the Natura 2000 Standard Data Form (JNCC, updated 1999) states 7 pairs as the 5-year mean (1992-1996) at the time representing 0.3% of the GB breeding population.

The annual maximum number of pairs of little terns during a recent 5-year period (2009-2013[13]) were - **16** (2009), **6** (2010), **7** (2011), **23** (2012), **11** (2013). This provides a recent 5-year mean of **13** pairs (or 26 breeding adults). This represents 0.66% of the GB breeding population. This population is not included in calculation of the total population of this species supported by the pSPA and their potential foraging areas around Pagham Harbour SPA are not considered in defining the boundary of the pSPA (although by default are included within it).

5.2.3.4. Solent & Dorset Coast pSPA

The principal little tern breeding colonies supported by the Solent & Dorset Coast pSPA during the breeding season are located at: Solent & Southampton Water SPA and Chichester & Langstone Harbours SPA. The sum of the site-specific recent 5 year means across these SPAs yields a figure of 63 pairs or 126 breeding adults supported by the pSPA which constitutes 3.31% of the GB breeding population (Table 5).

Table 5. Summary of breeding populations of little tern within SPAs contributing to the population of the Solent and Dorset Coast pSPA

	Solent and Southampton Water	**Chichester and Langstone Harbours**
Citation or SDF (prs)	49	100
Old % GB popln	2.0%	4.2%
Data age	1993-1997	1992-1996
Recent mean max (prs)	19	43
Recent % GB popln	1.02%	2.28%
Data age	2010-2014	2010-2014
Solent and Dorset Coast pSPA foraging population (sum max **mean** x2 for **individuals**):		126

[12] The highest figure is used, sourced from either SMP or the site manager, but for little tern all the highest figures were supplied by the site manager (*i.e.* RSPB *pers comms*).
[13] 2014 was described as "a relatively good year" (*pers comms* RSPB), but no figure is available and so the mean is based on years 2009-2013.

Solent and Dorset Coast pSPA foraging population (**prs**) % of GB breeding population:	3.31%

5.2.4. Roseate tern *Sterna dougallii*

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

5.2.4.1. Solent & Southampton Water SPA

The SPA citation states 2 pairs, at that time representing 3.1% of the GB breeding population (5-year peak mean, 1993-1997). The Natura 2000 Standard Data Form (JNCC, updated 1999) states 2 pairs as the 5-year mean (1993-1997) at the time representing 3.1% of the GB breeding population.

Historically, one or two pairs of roseate terns nested in most years between 1967 and 1978. An average of two pairs nested between 1993 and 1997 (as reflected in the Natura 2000 Data Form for the SPA), and one or two pairs nested each year between 2002 and 2006. Since then, roseate terns have continued to be seen on passage, although it was subsequently suspected to be absent as a nesting species. However, "a pair of roseate terns was recorded in a colony of common terns on 21st June [2013]. They appeared well settled but were not seen on subsequent visits" (Durnell, *unpubl*). Thus, there are no confirmed records of breeding by this species within this SPA within the last 8 years.

5.2.4.2. Solent & Dorset Coast pSPA

With no records of breeding by roseate terns over the last 8 years within the only possible source SPA colony from which birds foraging in the pSPA might originate, no contribution to the total tern population supported by the pSPA has been included for this species from this colony. Any possible areas of usage within which birds from that colony may have foraged have not been modelled and so have not influenced the boundary of the pSPA. As the only available model of areas of usage by roseate terns is based on tracking data from Coquet Island and included a relatively large number of explanatory environmental covariates (Wilson *et al.* 2014) it is not possible to determine whether, if this model had been applied to the Solent, **all** of the predicted areas of significant usage by roseate terns would be encompassed within the currently proposed pSPA boundary. However, given that the mean of the recorded maximum foraging ranges for roseate terns (21.3km) is almost identical to that of common tern (20.4km) and smaller than that of sandwich tern (32.0km) (Win *et al.* 2013), and the maximum limit to the area of significant usage by roseate terns in any direction around Coquet Island is less than that of at least one of the other three large tern species at that site (Win *et al.* 2013), it is highly likely that the majority of the areas within the Solent & Dorset Coast pSPA that would be important as foraging areas for roseate terns is encompassed within the proposed pSPA boundary.

6. Comparison with other sites in the UK

A comparison is presented in Table 6 of the populations of each named qualifying feature of the Solent and Dorset Coast pSPA with the largest breeding populations supported by individual SPAs across Great Britain. The pSPA population is derived by summing the most recent 5 year means of the breeding populations of all its contributing SPAs. Unless otherwise stated, for the purposes of this comparison exercise, the breeding population from each of the other individual SPAs is that presented in the SPA review (Stroud *et al.* 2001), which in all cases are of course many years out of date. It is acknowledged that the ranking is therefore not based on like-for-like directly comparable information and instead merely indicates the pSPA's general level of relative importance in a national context. The ranking is based on the total number of the SPAs listed for each species in Stroud *et al.* (2001) plus the Solent and Dorset Coast pSPA. The original source SPAs are retained in this exercise as they continue to exist in their own right, so there is a degree of duplication of individual birds between the pSPA numbers and those at the source colonies listed in this table.

Table 6. Comparison of the numbers of individuals (and pairs) of each of the named features of the Solent & Dorset Coast pSPA (based on colony counts since 2009 at contributing source colony SPAs) with numbers at other SPAs/pSPAs for which figures are provided in Stroud *et al.* (2001).

Species	Site	Individuals (pairs) [14]	Rank[15]	Comments
Sandwich tern *Sterna sandvicensis* (breeding)	North Norfolk Coast	6,914 (3,457)	1st of 17	
	Farne Islands	4,140 (2,070)	2nd of 17	
	Coquet Island	3,180 (1,590)	3rd of 17	
	Ythan estuary, Sands of Forvie and Meikle Loch	1,200 (600)	4th of 17	
	Strangford Lough	1,186 (593)	5th of 17	
	Carlingford Lough	1,150 (575)	6th of 17	
	Loch of Strathbeg	1,060 (530)	7th of 17	
	Ynys Feurig, Cemlyn Bay and The Skerries	920 (460)	8th of 17	
	Solent and Dorset Coast	882 (441)	9th of 17	
Common tern *Sterna hirundo*	Firth of Forth Islands	1,600 (800)	1st of 23	
	Coquet Island	1,480 (740)	2nd of 23	
	Strangford Lough	1,206 (603)	3rd of 23	
	Glas Eileanan	1,060 (530)	4th of 23	
	Solent and Dorset Coast	984 (492)	5th of 23	
Little tern *Sternula albifrons*	North Norfolk Coast	754 (377)	1st of 28	
	Great Yarmouth North Denes	440 (220)	2nd of 28	
	Chichester and Langstone Harbours	200 (100)	3rd of 28	
	Solent and Dorset Coast	126 (63)	4th = of 28	
	Humber Flats, Marshes and Coast	126 (63)	4th = of 28	

[2] 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 Solent and Dorset Coast pSPA, and adding one site to account for the pSPA itself.. For brevity, only the top 5 ranked sites are tabulated for each species, except where the Solent and Dorset Coast pSPA position in the rank order is lower than this – in which case all sites down to that rank position are tabulated.

7. Conclusion

It can be seen from the evidence presented above that the Solent and Dorset Coast is in the top 10 sites for all the proposed features and that the site features meet the required selection criteria for classification as an SPA. We are confident that the proposed boundary offers sufficient protection for each species based on a composition of the key areas they use. We therefore feel that the evidence provided is sufficient to be able to take this site to consultation.

[14] 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 were substituted with subsequent census data for some sites provided by country agencies (especially in Scotland) and/or as a result of more recent surveys of particular species

[15] 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 Solent and Dorset Coast pSPA.

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Annex 1 Special Protection Area Map

(See website link)

Annex 2 Special Protection Area (SPA) Citation

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

Name: Solent and Dorset Coast pSPA

Counties/Unitary Authorities:

Dorset County Council, Purbeck District Council, Poole Borough Council, Christchurch Borough Council, Bournemouth Borough Council, New Forest District Council, Hampshire Country Council, Southampton City Council, Portsmouth City Council, Isle of Wight Council, Arun District Council, West Sussex Country Council, Gosport Borough Council, Fareham Borough Council, Havant Borough Council, Chichester District Council, Gosport Borough Council, Eastleigh Borough Council

Boundary of the SPA:

The landward boundary is located on the coasts of Dorset, Hampshire, Isle of Wight and West Sussex. The westernmost extremity of the boundary is at Worbarrow Bay in Dorset and the easternmost extremity of the boundary lies approximately 88km to the east at Bognor Regis in West Sussex. The westernmost extremity is determined by the modelled usage of Sandwich terns foraging from the Poole Harbour SPA, whereas the easternmost extremity is determined by the modelled usage of Sandwich terns foraging from Chichester & Langstone Harbours SPA.

The pSPA wholly occupies The Solent with its seaward extent boundary beyond The Solent consisting, for simplicity, of angled straight lines. The seaward boundary consists broadly of three arcs running west to east. The furthermost extremes of these arcs lie at approximately 5km, 12km and 12km from the nearest points of the mainland shore and are determined by the modelled usage of Sandwich terns foraging from Poole Harbour SPA, Solent and Southampton Water SPA and Chichester & Langstone Harbour SPA.

Size of SPA: The SPA covers an area of 87531.75 ha.

Site description:

The Solent and Dorset Coast pSPA is located along the coasts of Dorset, Hampshire, Isle of Wight and West Sussex and adjacent areas offshore. It will overlap, abut and be close to many designated areas, summarised below. At its western point, South Dorset Coast SSSI, Townsend SSSI, Purbeck Ridge (West & East) SSSI, Studland Cliffs SSSI, Studland & Godlingson SSSI, Poole Bay Cliffs SSSI and Boscombe & Southbourne Overcliff LNR, Bournemouth fall just outside the pSPA boundary and have no tern interest. A cluster of adjacent, overlapping and underpinning designations exist at Christchurch Harbour, Hengistbury Head and adjacent undeveloped land, which is surrounded by the Bournemouth and Christchurch sprawling conurbation on one side and open sea on the other. These designations include Dorset Heathlands SPA, Dorset Heaths SAC, Christchurch Harbour SSSI, Hengistbury Head LNR and Stanpit Marsh LNR, and are fed by the narrow Avon Valley (Bickton to Christchurch) Ramsar and SSSI, and River Avon System AONB and SSSI. The new pSPA boundary will extend to Mean High Water and so become underpinned by Christchurch Harbour SSSI. Its citation describes, “The site comprises the drowned estuary of the rivers Stour and Avon and the peninsula of Hengistbury Head. The varied habitats include saltmarsh, wet meadows, drier grassland, heath, sand dune, woodland and scrub and the site is of great ornithological interest.” Terns are not mentioned on the citation but, “Sandwich, common and little terns all occur in the harbour and have tried to nest in the past but they face a lot of pressure from predators, disturbance and tides and sadly have not been successful.” (Christchurch Harbour Ornithological Group, 2007). Along 9km of coastline, from Christchurch in Dorset to Milford on Sea in Hampshire, the cliffs and intertidal zone are designated Highcliffe to Milford Cliffs gSSSI, which will underpin the pSPA.

Along the northwest coast of The Solent, the Solent and Southampton Water SPA and Ramsar overlaps with Solent Maritime SAC and these are underpinned by the Hurst Castle and Lymington River Estuary SSSI. East from River Lymington, the intertidal saltmarsh is also designated Boldre Foreshore LNR. A series of intertidal SSSIs underpin the SPA and eastward are the North Solent SSSI (where also North Solent NNR), Dibden Bay SSSI, Hythe to Calshot Marshes SSSI, then east from Southampton is the Lee-on-Solent to Itchen Estuary SSSI (where also Hook and Warsash LNR), Titchfield Haven SSSI (also NNR and LNR) and Browndown SSSI near Gosport, Hampshire. Portsmouth Harbour SPA is underpinned by Portsmouth Harbour SSSI, where since the pSPA will overlap this SPA, will itself become underpinned by this SSSI. Both estuaries of the Chichester and Langstone Harbours SPA are separately underpinned by SSSIs and Chichester Harbour is an AONB. Within or partly within this SPA's boundary are Farlington Marshes LNR, West Hayling LNR, Hayling Billy LNR, Gutler Point LNR, The Kench, Hayling Island LNR, Sandy Point LNR, Earnes Farm LNR, Nutborne Marshes LNR, Pilsey Island LNR, The Solent Maritime SAC overlaps this SPA and further extends nearly 2km out from the south coast of Hayling Island.

Since the southwest headland and offshore sandbank of Hayling Island are designated Sinah Common SSSI and only abuts with the Solent Maritime SAC, the situation is thus created wherein the intertidal part of this SSSI up to Mean High Water will be underpin the pSPA, but not the SAC.

In West Sussex, the shoreline for approximately 8.2km from near West Wittering, southeast to Selsey is designated the Bracklesham Bay SSSI. Pagham Harbour SPA is underpinned by Pagham Harbour SSSI and abutting this along the intertidal zone for approximately 4km adjacent to Bognor Regis is the Bognor Reef SSSI. On the Isle of Wight, almost the entire coastal cliff-line and intertidal zone along the southwest side is designated SSSI, including Compton Chine to Steephill Cove SSSI, Compton Down SSSI and Headon Warren and West High Down SSSI, which occupies The Needles and west headland; all fall also within the Isle of Wight AONB. Colwell Bay SSSI occupies 1.7km of the northwest coastline. At Yarmouth is the Yar Estuary SSSI; either side and including Newtown is the Bouldnor and Hamstead Cliffs SSSI, Newtown Harbour SSSI (and NNR) and Thorness Bay SSSI. The entire northeast Isle of Wight coastline, around Foreland (east headland of Isle of Wight) and to Yaverland, near Sandown along the southeast coastline is all SSSI. Clockwise these are King's Quay Shore SSSI, Ryde Sands and Wootton Creek SSSI, Brading Marshes to St Helen's Ledges SSSI, Whitecliff Bay and Bembridge Ledges SSSI and Bembridge Down SSSI. Solent and Southampton Water SPA and Solent Maritime SAC also occupy intertidal and sub-tidal areas of the Isle of Wight. The South Wight Maritime SAC is relatively large, extending up to 4km out from the southwest coastline and out over 8km from the southeast coastline. Below Mean Low Water, these SACs are not underpinned by SSSIs. Around the Isle of Wight, the Tranche 2 rMCZ site 'The Needles' is under consultation, but three other possible sites are not being forwarded at this time, namely Norris to Ryde, Bembridge and Yarmouth to Cowes. The Needles falls wholly within the pSPA boundary.

Qualifying species:

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

- The site regularly supports more than 1% of the Great Britain breeding populations of 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 Solent and Dorset Coast pSPA

Species[1]	Count[2,3] (period)	% of GB breeding population[4]	Interest type
Sandwich tern *Sterna sandvicensis*	441 pairs (882 breeding adults)[5] (2008 - 2014)	4.01%	Annex 1
Common tern *Sterna hirundo*	492 pairs (984 breeding adults) (2009 - 2014)	4.77%	Annex 1
Little tern *Sternula albifrons*	63 pairs (126 breeding adults) (2009 - 2014)	3.31%	Annex 1

[1] Species of terns that depend on the Solent and Dorset Coast pSPA area of sea for foraging that derive from breeding colonies at the following existing SPAs: Poole Harbour SPA, Solent and Southampton Water SPA and Chichester & Langstone Harbours SPA.
[2] The pSPA population is derived from the sum of the most recent populations of each species within the existing SPAs identified above. These totals exclude: i) numbers of any terns that may undertake foraging within Solent and Dorset Coast pSPA, but derive from breeding colonies that are situated outside of existing SPAs; ii) numbers of any terns at existing SPAs which are not qualifying features of these sites and not currently present in numbers exceeding SPA selection criteria thresholds , iii) numbers of terns at existing SPAs which, although qualifying features of those sites, were not present at classification in numbers exceeding SPA selection criteria thresholds and/or are no longer present in such numbers, iv) numbers of terns at existing SPAs which, although qualifying features of those sites and present at classification in numbers exceeding SPA selection criteria thresholds are no longer present in such numbers at those particular sites and when summed across all source SPAs that might contribute to the numbers supported by the pSPA.
[3] Sources of count data and recent mean figures for each contributing SPA are detailed elsewhere in this Department Brief.
[4] GB breeding populations taken to be that within Great Britain as presented in Musgrove *et al.* (2013).
[5] Pairs multiplied by 2 to arrive at breeding adult numbers; this rule applies to all species listed in the table

Assemblage qualification:

N/A

Principal bird data sources:

JNCC Seabird Monitoring Project and site managers from the National Trust, RSPB and Hampshire Country Council

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 of the foraging locations of breeding terns tracked from several UK colonies and identification of important foraging areas using habitat association models	2009-2011	Visual tracking of individual terns from boat-based survey platform. Note. No visual tracking surveys occurred at sites relating to Solent and Dorset Coast pSPA. Generic modelled ranges were instead used.	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. Note. Surveys occurred at Chichester & Langstone Harbours SPA and at Solent and Southampton Water SPA, but results were not used to set site-specific foraging ranges due to insufficient data. Generic modelled ranges were instead used.	Verification by JNCC and external peer review of final report
Site managers	Hampshire County Council	Solent and Southampton Water SPA (Hampshire)	1966-2013	Standard methodology	Verified by site manager and JNCC (when entered to SMP)
Site managers	National Trust	Solent and Southampton Water SPA (Isle of Wight)	1998-2014	Standard methodology	

Site managers	RSPB	Chichester & Langstone Harbours SPA	1911-2014	Standard methodology	Verified by site manager and JNCC (when entered to SMP)
Site managers	RSPB	Pagham SPA	2013-2014	Standard methodology	Verified by site manager and JNCC (when entered to SMP)
Seabird Monitoring Programme	Site managers and JNCC	Annual counts of Sandwich tern colonies at Poole Harbour SPA	1989 To present	Standard methodology	Verified by JNCC

Annex 4 Detailed information on the definition of little tern foraging areas and seaward boundary definition.

1. Background and overview

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

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

2. Data collection

The study aimed to provide three years of colony specific data for all regularly occupied[16] 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. Shore-based surveys were conducted at all of the study colonies and boat-based surveys were conducted at 8 sites.

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

Boat-based surveys were carried out to assess how far out at sea foraging little terns would range (*i.e*. to confirm their maximum seaward foraging extent). Surveys involved the boats travelling along a series of parallel lines through a survey area around each colony. These surveys extended to 6km 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

[16] '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.

at pre-determined points (between 300m and 1800m apart). The instantaneous count area was an 180^{o} 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 300m) 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 index of relative abundance. Although observers recorded behaviour (foraging/flying), restricting the analysis to just foraging observations would have limited the sample size. Therefore, all records (foraging and not foraging) were included in the analyses.

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

Shore-based observations aimed to assess to what extent little terns forage away from their colony along the coastal strip. Observation points were chosen at 1km intervals to either side of the colony, up to a distance of 6km along the coast, according to the mean maximum foraging range indicated by the literature. If preliminary observations found birds going further than 6km, more observation points were added at successive 1km intervals. Birds were counted within a distance of 300m 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 1km) in either direction alongshore from the colony. Each side of the colony was analysed as a separate sample. This approach assumes that 100% of birds leaving the colony in a particular direction reach the first count point, and that all birds reaching subsequent count points have passed through (and had been counted at) point one on their way.

3. Data analysis

The density of little terns within each survey area was relatively small, leading to small numbers of observations within boat transects and shore based count points. This was particularly evident at the colonies with fewer breeding pairs. Given this, techniques successfully used for defining boundaries to areas of importance for other seabird and waterfowl species i.e. interpolation based on analyses of transect data to yield density maps (e.g. O'Brien *et al.* 2012) could not be used in this case. Furthermore, the small foraging range of the little terns precluded application of the habitat association modelling approach used in the case of the work on larger terns (Annex 5). 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 that 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 data.

B) Alongshore extent

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

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 with 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 all other sites	-	2176

Footnotes:
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 all other sites	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 2.2km offshore and 3.9km alongshore to define the boundaries in the case of colonies with insufficient or missing data.

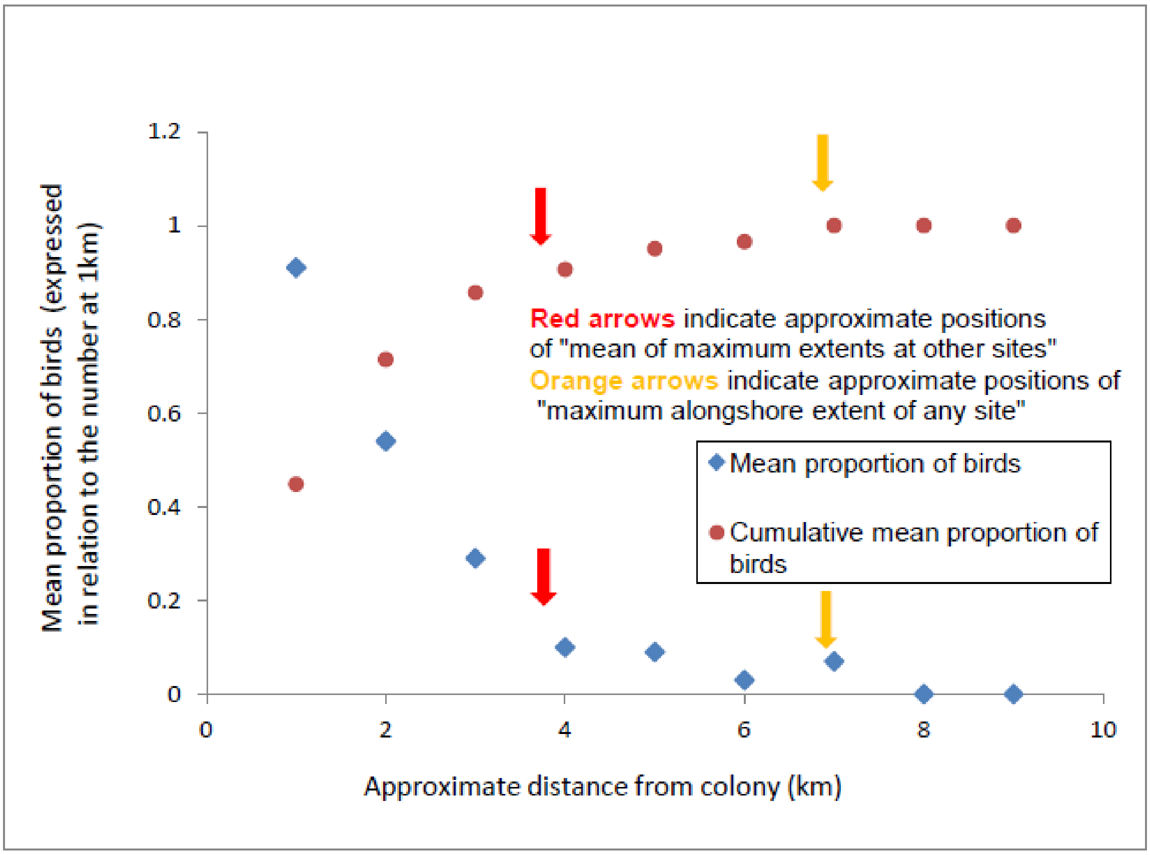

Figure 1: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing distances alongshore from the colony. Each blue point represents the mean proportional usage at each distance band from the colony averaged across colonies. The proportion at each distance (blue dots) is expressed relative to the number at the 1km 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 (3.9km) whereas the yellow arrows indicate the values at the greatest site-specific maximum alongshore extent recorded (7km at North Norfolk Coast and Morecambe Bay). Source: Parsons *et al.* (2015).

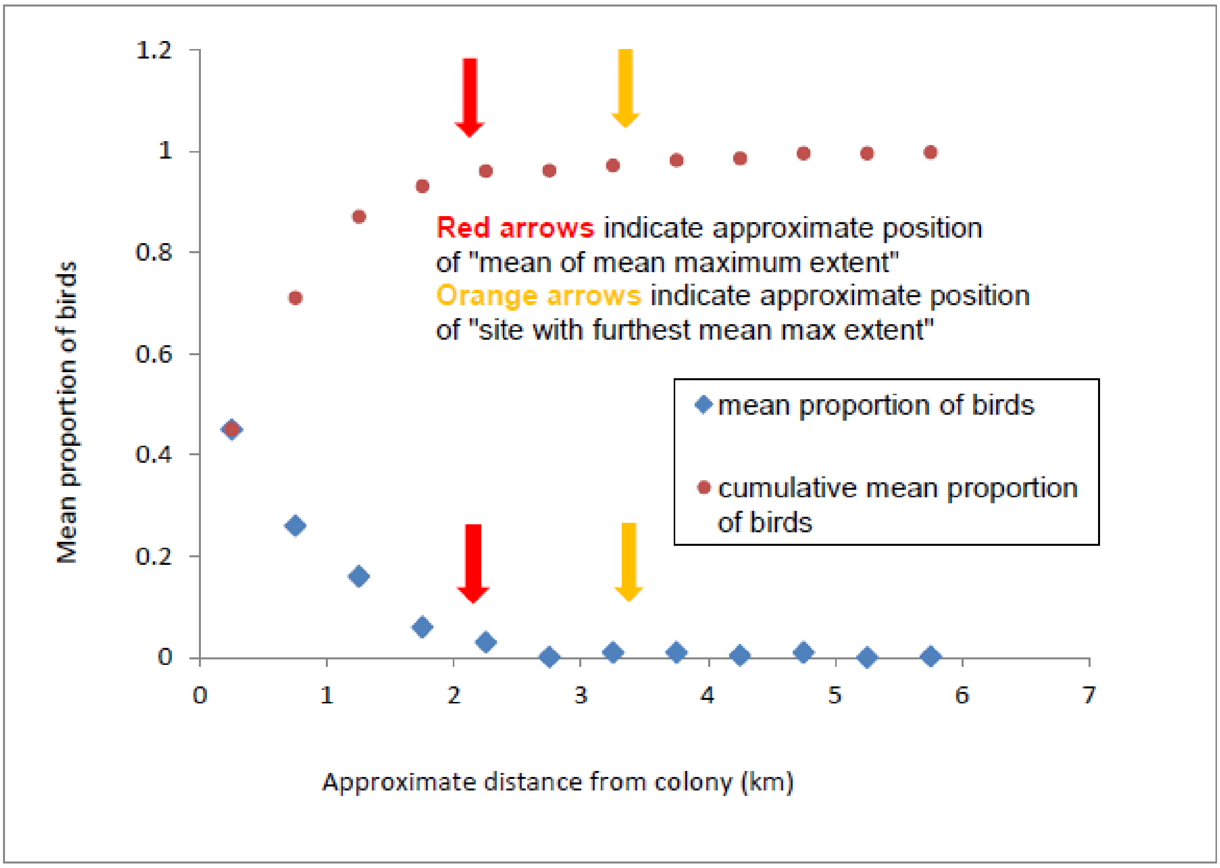

Figure 2: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing seaward distances from mean high water mark. Each blue point represents the mean proportional usage at each distance band from mean high water mark averaged across colonies. The red arrows indicate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

These figures demonstrate the nature of the relationship of decreasing cumulative usage with increasing distance from colony. For alongshore (Figure 1) approximately 0.86 of all recorded usage occurred within 3.9km 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 7km 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 2.18km 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 3.4km 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 7km) and almost all recorded usage in a seaward direction (0.99 at 3.4km) 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 3.1km alongshore in each direction and a further 1.2km 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 (3.9km) and the average of the site specific mean maximum seaward extents (2.2km) 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 while excluding areas which are likely to have very low usage.

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.

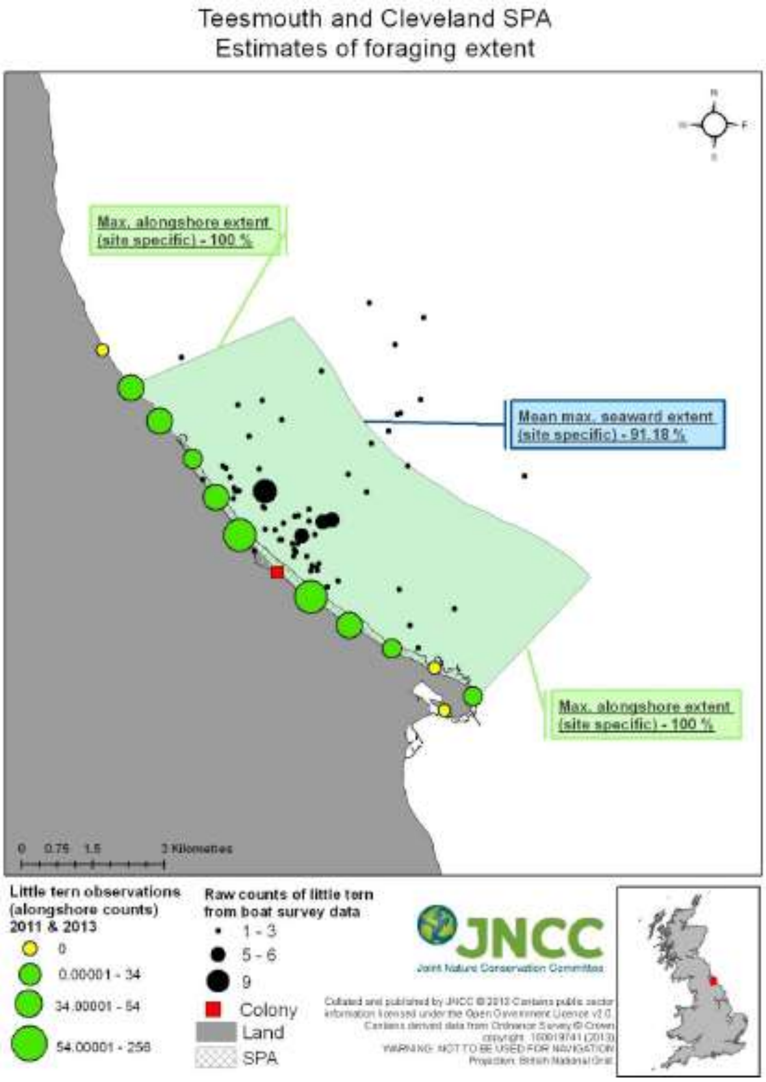

Figure 3. An example of the application of site specific alongshore and site specific seaward extents to define the boundaries to little tern foraging areas at the Teesmouth and Cleveland SPA. The % values given in the labels indicate the site specific % of little tern observations within the shore-based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

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 5) 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 for 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 2.1km, with the mean of maxima across studies as 6.3km. The work by JNCC, on a larger number of colonies, gave a mean maximum extent of 2.2km, with a range of 1.1-3.4km (for seaward extent) and a mean maximum of 3.9km, with a range of 0.5-7km (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 4km 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 Detailed information on the definition of larger tern foraging areas and seaward boundary definition.

1. Background and overview

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

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

This annex gives an overview of the survey and analytical work carried out by and on behalf of JNCC between 2009 and 2013 for the four larger tern species (*Sterna* species). A full and detailed description of the analysis can be found in the JNCC report on this work (http://jncc.defra.gov.uk/page-6644). A different approach was deemed appropriate for little terns as they search for food in a much more restricted area closer to the coast and to the breeding colony. An overview of that work is described in Annex 4 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)[1].

2. Data collection

To acquire information on the at-sea foraging distributions of breeding terns, three years of targeted data collection were carried out or commissioned by JNCC around selected tern colonies from 2009 to 2011, using the visual-tracking technique[2] (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.

[1] 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.
[2] 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.50-200m 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 30km (Arctic, common and roseate terns) or 54km (Sandwich terns) from their colonies. Accordingly, models were used to generate predicted distributions out to these maximum foraging ranges around the colonies of interest. To do so, information on habitat conditions across these areas was gathered from various sources to be fed into the habitat models as so-called 'environmental covariates'. Such environmental covariates were chosen for their potential to explain the observed tern distribution data. Due to a lack of information on actual prey distributions (e.g. sandeels, clupeids such as herring and sardine, zooplankton), environmental covariates which could relate to the occurrence or availability of these prey species such as water depth, temperature, salinity, current and wave energy, frontal features, chlorophyll concentrations, seabed slope and type of sediment as well as distance to colony (as a proxy for energetic costs) were used instead.

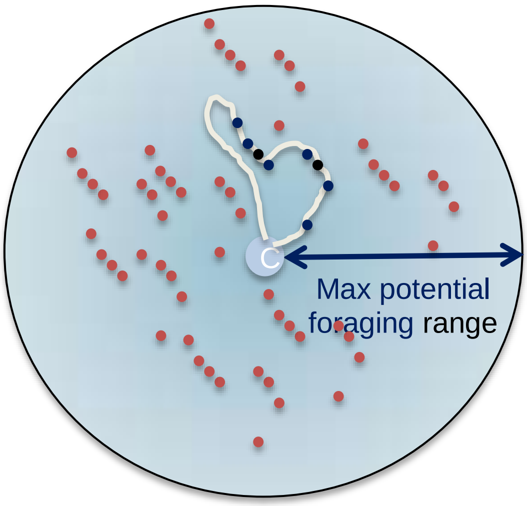
3. Data preparation and analysis

Prior to analysis within the habitat models, data had to be prepared and processed into a suitable format. Each track of a tern comprised periods of time when the bird was clearly not engaged in either actively searching for prey or in active foraging but appeared to be in transit to or from the colony or between areas of search at sea. As the aim of this work was to characterise important foraging areas and inclusion in the modelling of locations passed over in transit would, with terns being central place foragers (meaning they must travel to and from their nest site on each trip), almost certainly lead to a bias towards high usage of areas close to the colony, data from commuting periods (i.e. parts of the bird track where no foraging behaviour[3] 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.

[3] 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[4], 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).

[4] 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 500m by 500m 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.

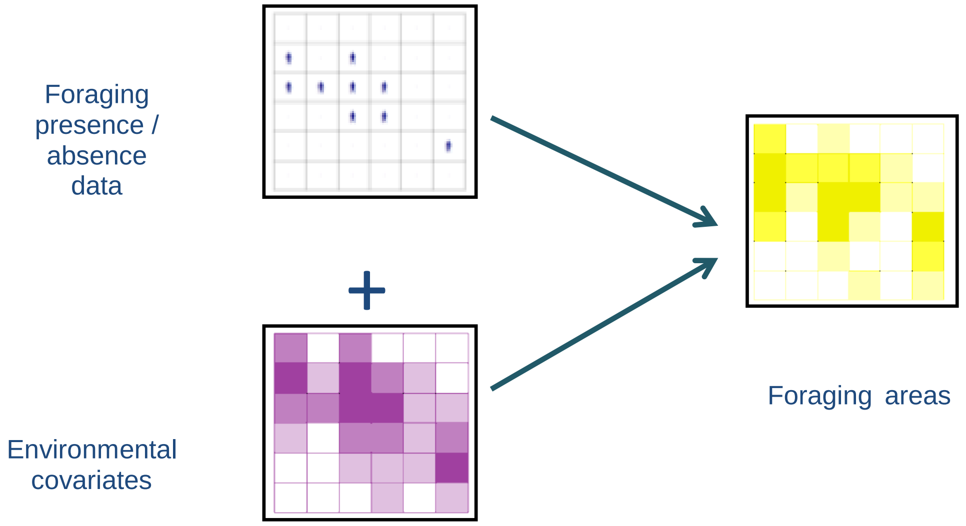


Figure 1. Process of modelling distributions based on environmental information, using a single covariate distribution map in the example.

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

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

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

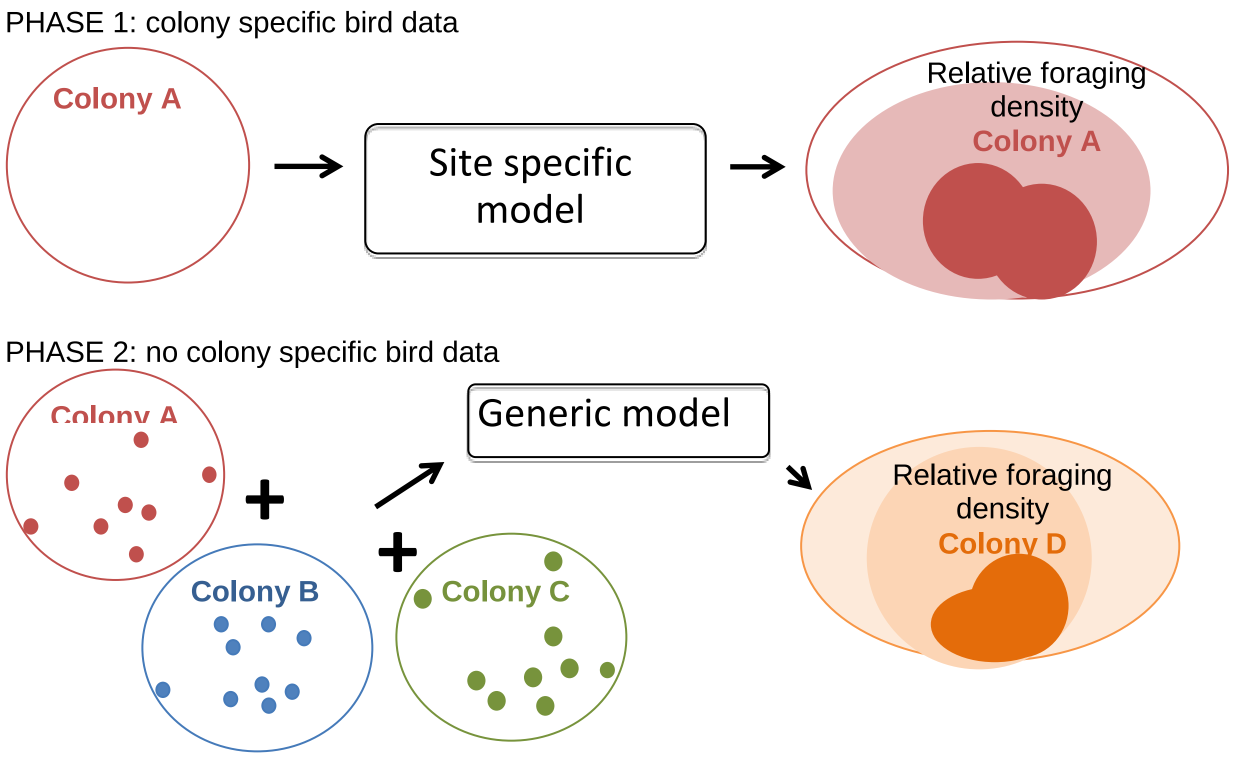


Figure 2. Simplified, schematic representation of the process whereby empirical observations of tern foraging locations around a colony were either: used to build predictive, site-specific models of tern usage that generated relative foraging density maps around that colony (phase 1 analyses); or combined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

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

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

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

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

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

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

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

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

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

Cross-validation involves omitting a sub-set of data (the validation set), and refitting the chosen model to the remaining data (the training set). Predictions, in this case of tern foraging locations, generated by models based on each training set are then compared with the validation set – which in this case comprises the actual tern foraging locations not used in building the model. Comparisons can be done by various scoring methods; three were used to avoid reliance on a single method, but for simplicity only one of these i.e. the 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.

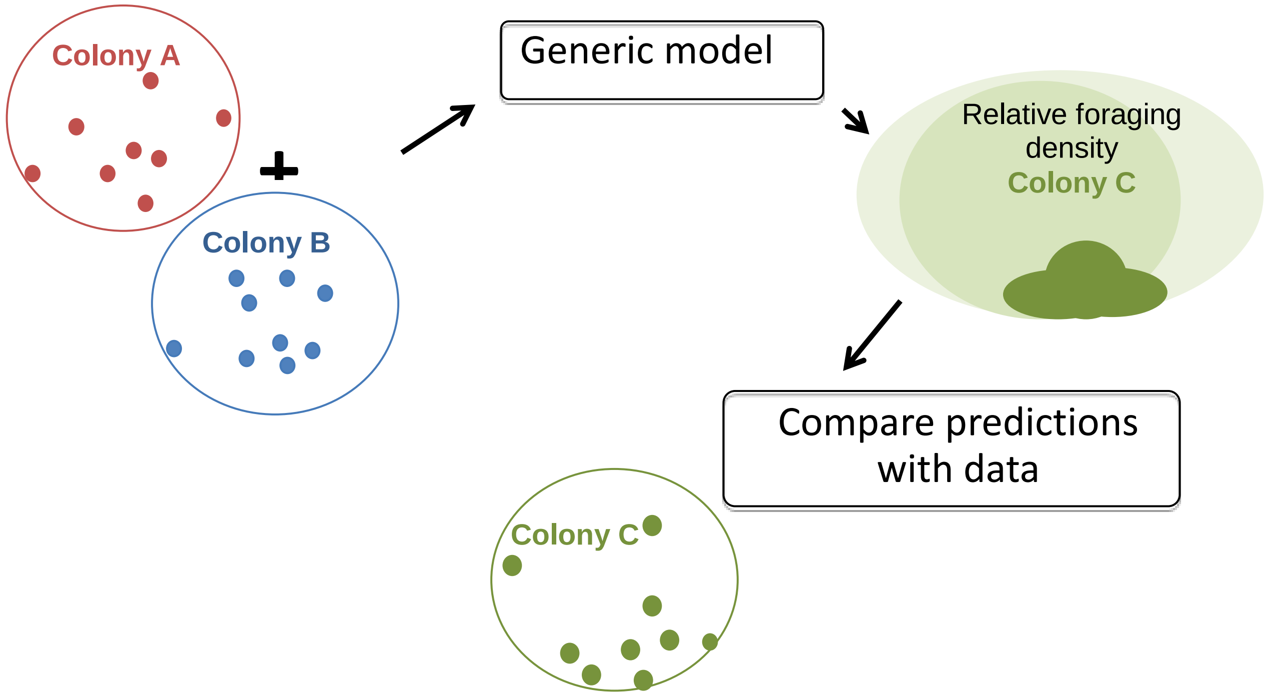


Figure 3. Schematic representation of the cross-validation process, using an example where we have data for three colonies A, B and C, of which data from two at a time (A and B in this diagram) are used to build a predictive model, the predictions of which are then tested by comparison with empirical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

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. For the 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**

4. Boundary Delineation

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

As the maximum curvature technique is sensitive to the size of the area to which it is applied, the analysis was based on a common area unit for each species. A species-specific mean maximum foraging range (*i.e*. the furthest that an average individual forages from a colony) was determined using all available data[5], resulting in 30km for Arctic, 20km for common, 32km for Sandwich and 21km 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 Artic terns from Coquet Island is shown in Figure 4.

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

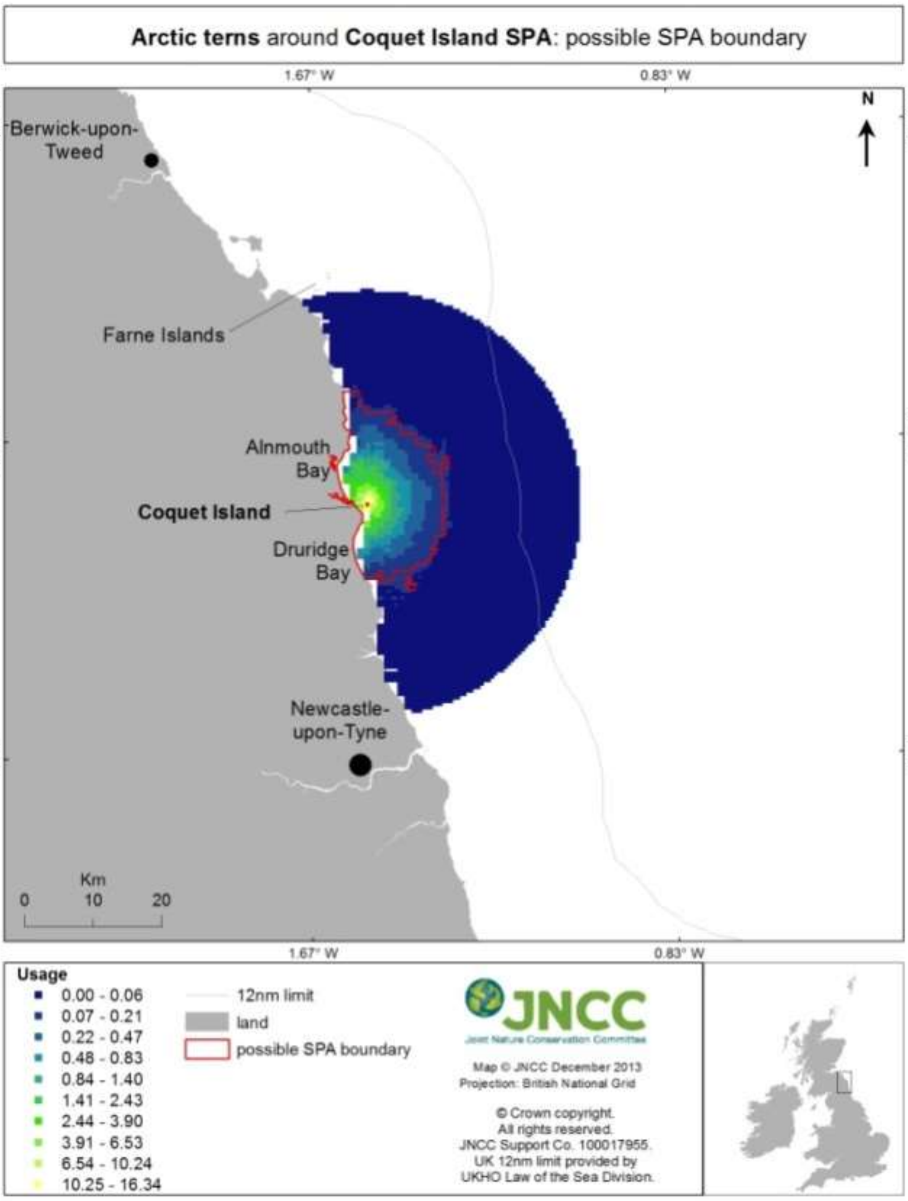


Figure 4 Maximum curvature derived boundary (red line) overlaid on map of model predictions of usage by Arctic terns around Coquet Island. The extent of the dark blue circle of model predictions of usage is 30km - the global mean maximum distance to colony, calculated using tracking data held by JNCC;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

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

In many 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 were not fit for purpose. For this approach to have been followed, a significant programme of bespoke, near-shore at sea transect surveys 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 and the inevitable limitations to survey effort that could be deployed, it was recognised 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 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 Evidence standards within Boundary Making decision process

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 Solent and Dorset Coast pSPA.

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

Quantification of Solent and Dorset Coast pSPA interest feature population sizes.

In order to determine the suite of species present within the Solent and Dorset Coast pSPA which meet the SPA selection guidelines, bird count data at each of the existing coastal SPAs from which birds within the marine pSPA may originate were used as the evidence base. These data were as follows:

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

   a) Poole Harbour: Sandwich tern - 2014

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

   a) Solent and Southampton Water: Hampshire County Council & National Trust, little, common and sandwich tern - 2014
   b) Chichester & Langstone Harbours: RSPB, little, common and sandwich tern - 2014
   c) Pagham Harbour: RSPB, little, common and sandwich tern - 2014

   These data were made available by site managers (RSPB, Hampshire County Council, and National Trust).

The count data taken from the SMP database is the best available information. 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.

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 Solent and Dorset Coast pSPA differs from those used in identifying sites for terrestrial birds and aggregations of non-breeding waterbirds, it is necessary to consider their adequacy and relevance.

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

Table 1 Criteria for inshore SPA data adequacy.

Criterion	Adequacy of JNCC led larger tern surveys	Adequacy of JNCC led little tern surveys
Experience of observers	All tracking of terns was undertaken either by JNCC staff or experienced contractors commissioned by JNCC to do the work.	All observations of terns were undertaken either by JNCC staff or experienced contractors commissioned by JNCC or volunteer counters who received training in the shore-based observation techniques.
Systematic surveys	Tern tracking was conducted in as systematic a way as possible. Tracking at each colony was carried out during well-defined periods of the breeding season (chick-rearing) in one or more years. Tracking was undertaken in accordance with a field protocol established by JNCC. In the context of tern tracking, the movements of birds is an essential component of the technique and not a source of systematic bias in the survey results as it may be in conventional transect surveys.	Boat-based survey work followed systematic transect survey designs that were appropriate to each colony and were followed on repeated surveys. Shore based survey work used systematic series of observation stations and a standard recording protocol which was used repeatedly at each colony.
Completeness	The aim of the tracking survey method was not to cover all of the areas of 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	At sea observations included instantaneous counts at predetermined

	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.	distances along transects at which all terns in flight within 300m in an 180^o 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 300m 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 1km observation station to standardise across sites.
Quality of sampling	Cross-validation tests of the models' predictions and analysis of sample size adequacy both suggest that the results of the models based on the samples of tracks are robust.	This was affected by the low numbers of birds at many colonies and the frequent breeding failures. At colonies with 5 or more shore-based surveys yielding records of 200 or more terns, this was deemed sufficient to derive site-specific along shore boundaries. At colonies with at least 2 boat-based surveys yielding at least 20 tern sightings this was deemed sufficient to derive site-specific seaward boundaries. At colonies where these criteria were not met, a generic approach was used by pooling sample data across sites to yield better-evidence based estimates of limits.
Robustness of population estimate	Not applicable as the tern tracking work was not used to generate a population estimate	Not applicable as the tern observation work was not used to generate a population estimate
External factors affecting the survey	Tracking was constrained by weather, e.g. tracking could not take place with sea state ≥3 and during rain. Thus, tracking data were gathered only under favourable weather conditions.	Although the aim was to collect data from most currently occupied SPAs, in many cases data on seaward or alongshore extent could not be collected due to colony failure (caused by tidal inundation, predation or disturbance) or simply too few breeding pairs for sufficient observations to be detected by surveys. Accessibility to count points in all parts of the possible extent of a foraging area limited the ability to provide site-specific alongshore extents in some cases.

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

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

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

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

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

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

The landward boundary of the pSPA has been drawn at MHW along the mainland coast in the light of: i) model predictions of the usage of such areas by foraging larger terns, ii) observations of tracked larger terns foraging in such areas, iii) observations that little terns forage in the intertidal zone and iv) to ensure protection of these areas as supporting habitat for tern species within the pSPA in locations where these species are not already features of the existing SPA with which the overlap will occur (eg common and Sandwich terns at Portsmouth Harbour SPA). Stretches of the Solent and Southampton Water SPA already occupy some sections of the Isle of Wight coastline down to Mean Low Water. As several tern species are already features of that SPA, along these stretches of coast, the pSPA boundary correspondingly extends to Mean Low Water to abut this SPA. Elsewhere on the Isle of Wight coastline, the boundary of the pSPA will extend to Mean High Water where the coastline is not existing 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*". This is the approach which has been used in the tern work which has determined the seaward boundary of this pSPA.

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

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

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

Establishment of the extent of the Solent and Dorset Coast pSPA

The extent of and boundary to the Solent and Dorset Coast pSPA is determined by the extent of the model generated predictions of which areas of sea are most heavily used by foraging Sandwich terns originating from five source colonies. The boundary of the pSPA is a composite of several discrete, but in some case overlapping, areas identified by the modelling as being most heavily used by terns from different colonies (Table 2). In all cases, the colony-specific areas of use have been derived from models based on at sea records of the foraging locations of Sandwich terns at other colonies around the UK i.e. generic models. The quality and relevance of this evidence is discussed in the following section.

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

Species	source colony	Model type	tern tracks contributing to model	number of sites contributing to model	number of tern site/years of data underlying model
Sandwich tern	Poole Harbour	generic	277	5	11
Sandwich tern	Pitts-Deep-Hurst	generic	277	5	11
Sandwich tern	North Solent NNR	generic	277	5	11
Sandwich tern	Chichester Harbour	generic	277	5	11
Sandwich tern	Langstone Harbour	generic	277	5	11

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

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

A summary of the results of the cross-validation of both site specific and generic models of larger tern usage is presented in Annex 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 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 iii) 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.

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

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

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

The conclusions regarding the drawing of the seaward boundary of the pSPA are based upon the evidence provided in the form of models of predicted usage by foraging larger tern species and the application of a standard analytical method, already well-established for use in marine SPA boundary setting i.e. maximum curvature (O'Brien et al 2012), to the models' outputs. The validity and robustness of the outputs of the 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. However, some of the more recent count data have not yet been verified by JNCC and indeed some have not yet been submitted to the SMP database. Those data have however been collected by the same organisations (and in some case by the same people) as the data already on the SMP database.

RSPB survey data are verified and quality assured by the RSPB count coordinator and site manager and will become part of the SMP, when next updated. The RSPB is a professional organisation and surveys are conducted by trained surveyors. There is therefore high confidence in RSPB survey data. Hampshire County Council and National Trust count data is collected and coordinated by an experienced and long-serving site warden, using a standardised method with some observations conducted via boat to achieve survey completeness.

Accordingly, even the most recent count data referred to in this Departmental Brief can be considered to justify high confidence.

Landward boundary

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

Seaward boundary

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

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

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

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

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

Internal peer review and quality assurance

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

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

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

The first version of this Departmental Brief was drawn up by Stephen Treby (Lead Adviser) of Natural England. This was edited by Ivan Lakin (Marine Ornithologist) and Dr Richard Caldow (with further input from the original author) to produce this version of the Departmental Brief.

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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