Departmental Brief:

Dungeness, Romney Marsh and Rye Bay potential Special Protection Area

Natural England and JNCC

May 2016

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Summary
The Dungeness to Pett Level SPA was classified on 2nd August 1999. Following revision of the qualifying interests, several potential extensions to the SPA were proposed in 2010. In addition it was proposed in 2010 that the SPA (incorporating the extensions) should be known as Dungeness, Romney Marsh and Rye Bay SPA. This site was classified as an SPA in March 2016 as it qualifies for the following reasons:

- The site regularly supports more than 1% of the GB populations of 12 species listed in Annex I to the European Commission (EC) Birds Directive (Marsh harrier *Circus aeruginosus,* Avocet *Recurvirostra avosetta,* Mediterranean gull *Larus melanocephalus,* Sandwich tern *Sterna sandvicensis,* Common tern *Sterna hirundo,* Little tern *Sternula albifrons,* Bewick's swan *Cygnus columbianus bewickii,* Bittern *Botaurus stellaris,* Hen harrier *Circus cyaneus,* Golden plover *Pluvialis apricaria,* Ruff *Philomachus pugnax* and Aquatic warbler *Acrocephalus paludicola)*. Therefore the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.1, 1.4).

- The site regularly supports more than 1% of the biogeographical population of one regularly occurring migratory species (shoveler *Anas clypeata*). Therefore the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.2).

- The site regularly supports more than 20,000 waterbirds during the non-breeding season. Therefore the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.3).

It is now proposed that the existing SPA be further extended to include important marine foraging areas used by the little terns *Sternula albifrons*, common terns *Sterna hirundo* and sandwich terns *Sterna sandvicensis* from the breeding colonies within the existing SPA. It is proposed that the westernmost boundary of the existing SPA at Cliff End is extended 21 km further west to Bexhill, that the stretch of foreshore around the point of Dungeness which currently separates the two coastal sections of the existing site are included within the new site and that the northernmost boundary of the existing site at the beach groyne at national grid reference TR08922669 is extended 9.6 km further north as far as West Hythe. Between these westernmost and northernmost limits it is proposed that the seaward boundary of the current site is extended up to a maximum of approximately 9 km further seaward to include subtidal waters that will be used for foraging by the breeding terns originating from the colonies within the existing SPA. The revised boundary reflects a composite of the marine areas used by birds of each of the three species of tern originating from each of the principal, recently occupied nesting locations within the SPA.

Sandwich tern and common tern occur in numbers (5 year mean, 2011-2015, of 420 pairs and 188 pairs respectively) greater than 1% of UK population. These species therefore qualify for protection under Stage 1.1 of the UK SPA Selection Guidelines (Stroud *et al*., 2001). At the time of the original classification of the Dungeness to Pett Level SPA, little tern occurred in numbers (35 pairs) greater than 1% of UK population. Although the size of this population has declined since then, the species is retained as a qualifying feature of the pSPA to reflect the level of ambition defined by its population size at the time of the original classification (i.e. 35 pairs). See Table 1 for the population figures of the 3 tern species.

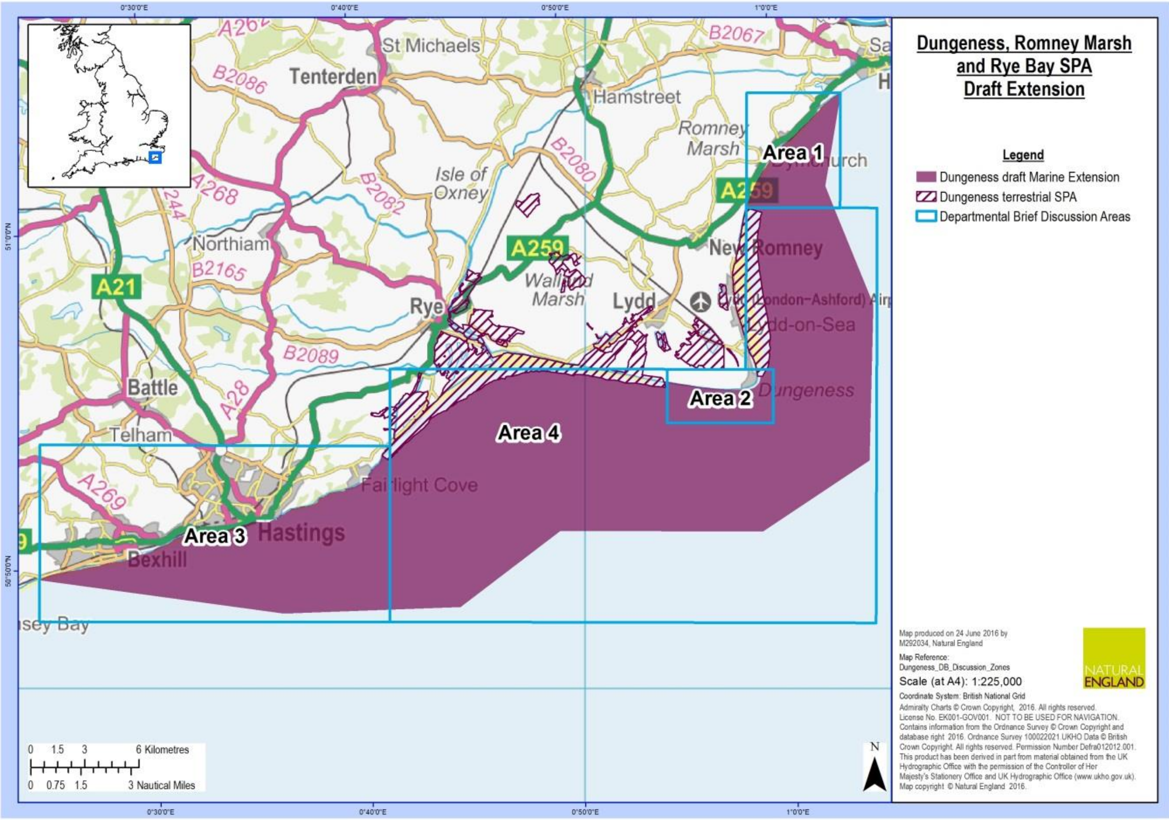


Figure 1. Each of the principal areas of the proposed boundary extensions to the Dungeness, Romney Marsh and Rye Bay pSPA.

This departmental brief sets out the scientific case for the extension to the boundary of the existing site and the estimated size of the breeding populations of each of the three species of tern which are qualifying features of the site. However, in respect of all of the other features of the existing SPA, this departmental brief does not make any proposal to amend baseline population figures, nor does it make any proposal to add or remove qualifying features of the site. The scientific case in support of the other features and all areas already included within the existing SPA remains the same as set out in the departmental brief published in 2010 (Natural England, 2010), which should be read in conjunction with this document if required. This departmental brief also makes no reference to the classification, features or boundaries of the Dungeness, Romney Marsh and Rye Bay Ramsar site, the evidence in support of which is presented in full in the departmental brief published in 2010 (Natural England, 2010).

Table 1: Summary of tern interest in the Dungeness, Romney Marsh and Rye Bay pSPA (including proposed extensions). Figures for all 3 tern species and the sources of those figures are identical to those in the 2010 departmental brief, with the two exceptions that are indicated([1]).

Species	Count (period)	% of subspecies or population	Interest type
Sandwich tern *Sterna sandvicensis*	420 pairs – breeding (2011 – 2015) [1]	3.8% GB[2]	Annex I
Common tern *Sterna hirundo*	188 pairs – breeding (2011 – 2015) [1]	1.9% GB[2]	Annex I
Little tern *Sternula albifrons*	35 pairs – breeding (1992 – 1996)	1.5% GB[3]	Annex I

[1] Qualifying population size for this species updated to be the 5 year mean based on the most recent 5 year period.

[2] GB breeding population derived from Musgrove *et al.* (2013)
[3] Data from: Dungeness to Pett Level SPA citation (English Nature 1999)

1. Assessment against SPA selection guidelines

The UK SPA selection guidelines require that SPA identification should be determined in two stages (Stroud *et al*., 2001). The first stage is intended to identify areas that are important for a significant proportion of birds on a regular basis (Stage 1.1 – 1.3), or which are of otherwise outstanding ecological importance for the birds (Stage 1.4). The second stage further considers these areas using one or more judgements to select the most suitable areas in number and size for SPA classification (Stroud *et al*., 2001).

As no changes to the list of qualifying features are proposed as part of this marine extension to the Dungeness, Romney Marsh and Rye Bay SPA, this document does not provide justification for the features of the existing SPA, and hence of the pSPA (with the exception of the three species of breeding tern that are the focus of this departmental brief), against the JNCC selection guidelines for Special Protection Areas. Full justification for inclusion of all of those other features is given in the departmental brief which was the basis of the classification of the existing SPA in 2010 (Natural England 2010) and those features are only considered in brief in this document.

1.1. Stage 1

Under stage 1 of the SPA selection guidelines (Stroud *et al*., 2001), sites eligible for selection as a potential SPA must demonstrate one or more of the following:

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

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

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

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

In relation to the breeding tern species, the Dungeness, Romney Marsh and Rye Bay pSPA qualifies under stage 1(1) because it regularly supports greater than 1% of the GB populations of Sandwich tern, common tern and little tern.

1.2. Stage 2

In relation to the breeding tern species, the Dungeness, Romney Marsh and Rye Bay pSPA is assessed against Stage 2 of the SPA selection guidelines in Table 2. It should be noted that in applying the SPA selection guidelines, Stroud *et al.* (2001), note that a site which meets only one of these Stage 2 judgments is not considered any less preferable than a site which meets several of them, as the factors operate independently as indicators of the various different kinds of importance that a site may have. The pSPA meets most of the Stage 2 criteria indicating the different kinds of importance the site holds.

Table 2: Assessment of the bird interest against stage 2 of the SPA selection guidelines[2]. Note in the first row the rank order for common terns has been updated since this table was presented in the departmental brief published in 2010 to reflect the most recent (lower) population size in the pSPA which is used in this document.

Feature	Qualification	Assessment
1. Population size & density	✔	In England, the site is the 4th most important SPA for Sandwich terns, 7th for common terns and 17th for little tern.
2. Species range	✔	The site is at the core of the breeding ranges of Sandwich tern, common tern and little tern.
3. Breeding success	✔	At Rye Harbour LNR during the period 2001-5, breeding Sandwich terns 0.13-1.5 young per pair (this species only experienced one “bad” year with <1 young per pair) and common terns 1.39-1.67 young per pair. Little terns at Rye Harbour LNR have variable breeding success depending on weather and level of predation. During 2001-5 success varied from 0.28-1.24 fledged young per pair.
4. History of occupancy	✔	Common and little tern were recorded using the site in 1970 (Sussex Ornithological Society, 1971; Kent Ornithological Society, 1972). 1970 has been chosen as the reference year because it coincides with the survey period of the first national breeding bird atlas (Sharrock, 1976).
5. Multi-species area	✔	Overall the pSPA supports one migratory species, twelve species listed in Annex I and a non-breeding waterbird assemblage.
6. Naturalness	N/A	No longer applicable, following ruling from the SPA & Ramsar Scientific Working Group.
7. Severe weather refuge	✔	Ridgill and Fox (1990) found that during periods of abnormally cold weather, waterbirds are displaced from the Waddensee coast to refuge areas to the south and west, including the wetlands of eastern Britain. It is likely that similar patterns of displacement are common to many of the waterbird species using the SPA (including proposed extensions), especially given its close proximity to Continental Europe.

[2] Comparisons with other SPAs are based on Stroud *et al.* (2001), the most recent comprehensive review of SPA bird populations available.

2. Rationale and data underpinning site classification

In 1979 the European Community (EC) adopted Council Directive 79/409/EC on the conservation of wild birds (EEC, 1979), known as the ‘Birds Directive’. This has been amended subsequently as Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 on the conservation of wild birds. This provides for protection, management and control of naturally occurring wild birds within the European Union through a range of mechanisms. One of the key provisions is the establishment of an ecologically coherent network of protected areas. Member States are required to identify and classify the most suitable territories in size and number for rare or vulnerable species listed in Annex I to the Directive (Article 4.1) and for regularly occurring migratory species (under Article 4.2). These sites are known as Special Protection Areas (SPAs). Guidelines for selecting SPAs in the UK are derived from knowledge of common international practice and based on scientific criteria (Stroud *et al*., 2001).

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

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

Under all three of these strands, the Joint Nature Conservation Committee (JNCC) has recommended the classification of new sites in the marine environment and produced generic guidance to implement this measure (Webb and Reid, 2004). To define SPAs at sea, the JNCC has developed specific statistical methods to formulate site boundaries for wintering waterbirds, such as divers (O'Brien *et al.,* 2012).

Since then, a fourth strand was added to the work conducted by the JNCC to address the need for:

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

Under Strand i), 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 the essential ecological requirements of the breeding seabird populations (e.g. preening, bathing, displaying and potentially local foraging). The distance of the extension is dependent upon the qualifying species breeding within the SPA. However, these generic boundary extensions are not influenced by or meant to encompass the principal foraging areas used by the species for which they are identified or any other species at the colonies concerned. Generic seaward extensions to the boundaries of existing SPAs have been implemented at 31 sites in Scotland and are under consideration at the Flamborough and Filey Coast pSPA (Natural England 2014). However, in line with the recommendations of Reid & Webb (2005), generic extensions have only been implemented at sites holding certain seabird species, none of which occur as breeding birds within the Dungeness, Romney Marsh and Rye Bay SPA. 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 classified as marine SPAs to complement the existing terrestrial suite.

Terns are seabirds and as such were analysed as part of the ESAS analysis under strand iii). However insufficient data was available in the ESAS database because these birds are small and difficult to identify to species level when surveyed by aircraft or from boat. JNCC therefore, under strand iv), worked with the four Statutory Nature Conservation Bodies (SNCBs) and collected visual tracking data as a means to identify the most important at-sea foraging areas around important tern breeding colonies.

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 which have recently been put out to formal consultation, or are in the process of being prepared, sites are referred to as SPAs when referring to existing classified sites. Where reference is made to an entirely new site, or to an extended site (as at Dungeness, Romney Marsh and Rye Bay), 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.

This departmental brief sets out information supporting a proposed extension to the sea area included within the boundaries of the Dungeness, Romney Marsh and Rye Bay pSPA in order to include the areas of sea identified as being most important to the populations of little tern, common tern and Sandwich tern that are qualifying features of the existing SPA and hence of the pSPA.

It is known that common terns at this site feed up to 8km inland and that freshwater fish can make up an important component of their diet (Lewis Yates 2014). However, this departmental brief is concerned solely with the identification and protection within the pSPA of the most important marine foraging areas of this and the other species of tern.

2.1. Data collection – defining the suite of species and numbers of those supported by the Dungeness, Romney Marsh and Rye Bay pSPA

The data which were used as the basis for defining the suite of species and the numbers of those species supported by the existing SPA are set out in detail in the departmental brief published in 2010 and formed the basis of the classification of the SPA in March 2016 (Natural England 2010). With the exception of the tern species, no changes to the list of qualifying features or to the numbers of these are proposed in this re-classification. Accordingly, for the sake of brevity, no further information is provided in this departmental brief relating to any feature apart from the three tern species, as only these species are the reason for the proposed boundary extension. For information relating to all other species and their supporting habitats, readers should consult the departmental brief published in 2010 (Natural England 2010) if necessary.

2.1.1. Little tern

In the departmental brief published in 2010 (Natural England 2010), the populations of all of the qualifying features of the site, with the sole exception of little tern, were updated on the basis of count data gathered between 2002 and 2008. The population for little tern was not amended because this was the only species for which the SPA was first classified in 1999 where the entire population remained within the classified boundary (as at 1999). Furthermore, the population of little tern had declined since classification (i.e. 35 pairs between 1992 and 1996) to an average breeding population of 14 pairs between 2004 and 2008. Lewis Yates (2014) provides an overview of the populations of each of the three tern species which breed within the existing colonies within the SPA. These data indicate that the breeding population of little terns has declined further since 2004-2008 with an average breeding population of 10 pairs between 2011 and 2015. These contemporary data reveal that this species is no longer present in qualifying numbers within the SPA (i.e. 1% of the GB population is 19 pairs).

Little tern's choice of nesting habitat and the pressures from disturbance and predation are considered to be the factors leading to very low productivity, and hence perhaps also to the population decline (Lewis Yates 2014). Ongoing monitoring and research into both feeding and breeding success around Rye Harbour, coupled with active management measures targeted at predation control, are being employed to try and address the reasons behind possible little tern declines. Given the possible role of anthropogenic influences (disturbance) in affecting the nesting locations of the species and the size of the population at the site, Department for Environment, Food and Rural Affairs (Defra) policy requires that the feature should be retained until such time as the reasons for the reduction in population can be clearly established. Natural England have applied Defra's policy here so that this species is retained on the citation of the pSPA, and the level of ambition set out in the conservation objectives for the species maintained, until there is evidence to support the conclusion that declines are a result of natural processes and that the SPA is no longer suitable for this species. In the meantime, the data source regarding the size of the population of little terns therefore remains the same as used in the departmental brief published in 2010. While Defra guidance suggests the use of contemporary bird data for features which are the basis for boundary extensions/amendments, it can be argued that this is not appropriate in the case of little tern, as although this species is a focus of the marine extension, the distribution of this population when at sea does not dictate the overall size or shape of the extension to this pSPA (see sections 2.3 - 2.5). Thus, this departmental brief, like that published in 2010, does not propose any amendments to the citation in relation to the size of the population of this qualifying feature.

2.1.2. Common tern

Recent count data (Lewis Yates 2014) indicate that the number of pairs of common terns has declined somewhat from that in 2004-2008 (i.e. 273 pairs) to a 5 year mean between 2011 and 2015 of 188 pairs. Predation appears to be an ongoing issue for this species although efforts to reduce this impact are ongoing alongside monitoring. This most recent population still exceeds the qualifying value of 1% of the GB breeding population i.e. 100 pairs, so still merits inclusion as a qualifying feature based on the most recent count data. On that basis, unlike in the case of little tern, it is proposed to make use of the most recent colony count data to amend the size of this qualifying feature. This is in line with Defra guidance which, as noted above, is to use contemporary bird data for features which are the basis for boundary extensions or amendments. Given that the modelled distribution of common tern foraging activity does influence the size and shape of the marine extension to the pSPA this is considered appropriate. However, as in the case of little tern, Natural England will apply Defra's policy steer for this species in relation to the associated conservation objectives. The conservation objectives of the pSPA will be maintained at that dictated by the historical size of this population in 2004-2008. It should be noted that the boundary of the pSPA is not influenced by the size of the population present. Rather it is influenced by factors such as the location of the colony, distance from the colony and water depth.

2.1.3. Sandwich tern

In contrast to the other two tern species, recent count data indicate that the population of breeding Sandwich terns at Rye Harbour has increased in recent years (Lewis Yates 2014); a 5 year mean of 420 pairs having been recorded between 2011 and 2015. This increase in population may be due to relocation of Sandwich terns back to the south coast of England from the French coast at the start of the century and management measures reducing the impact of predation (Lewis Yates 2014). In this case, Natural England consider it appropriate to amend the size of the notified population of this qualifying feature in line with Defra's advice to use contemporary bird data for features which are the basis for boundary extensions or amendments. This position reflects the fact that of the three species of tern, it is the extent of the foraging areas of importance to Sandwich tern that largely dictate the size of the marine extensions to the pSPA. Thus, in this case, it is proposed to make use of the most recent colony count data to amend the size of this qualifying feature. It should be noted that the boundary of the pSPA is not influenced by the size of the population present. Rather it is influenced by factors such as the location of the colony, distance from the colony and water depth.

2.2. Data collection – defining the boundary of the Dungeness, Romney Marsh and Rye Bay pSPA

The proposed extension to the Dungeness, Romney Marsh and Rye Bay pSPA has been drawn to encompass the sea areas identified under the fourth strand of JNCC's work programme as being most important to support the foraging activity of each of the three tern species which are already qualifying features of the existing SPA. The work done to identify the areas important to little terns and to the two larger species of terns 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 2.5 and section 3.

2.3. 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 2.1 km, mean of recorded maxima of 6.3 km and maximum ever recorded in the literature being 11 km (Thaxter *et al.,* 2012). In light of this evidence, JNCC, in agreement with all of the SNCBs, decided that the most effective method to determine the extent of the areas used most heavily for foraging by breeding little terns would be to undertake a programme of shore-based observations and boat-based transects around colonies and to use the resultant distribution data directly in setting the alongshore and seaward boundaries, respectively.

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

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

Where sufficient colony-specific data were available from the above surveys, these were used to determine the alongshore and seaward extents of important foraging areas. Where colony-specific data were not available, generic distances were applied.

The following text summarises the reasons for and results of application of generic distances in the case of little terns originating from colonies within the Dungeness, Romney Marsh and Rye Bay pSPA. Further information on the little tern survey programme is presented in Parsons *et al*. (2015a) and on the JNCC website[1].Further general information on the little tern survey programme is presented in Appendix 4

JNCC's programme of work (described in brief in Appendix 4) sought to identify foraging areas adjacent to 'recently occupied' terrestrial little tern colony SPAs. Recent occupation was defined where the mean of the most recent five years of data equalled or exceeded the UK SPA selection guideline of 1% of GB population (19 pairs). Although little tern is a qualifying feature of the Dungeness, Romney Marsh and Rye Bay SPA, the size of this population has declined in recent years and the mean population between 2008 and 2012 was just 5 pairs. Accordingly, the colony within the existing SPA was deemed in 2012 to be "not recently occupied" for the purpose of prioritisation of work on little terns by JNCC. On that basis, no surveys of little terns were carried out in Rye Bay.

Accordingly, in this case, it was necessary to apply the generic alongshore (3.9km) and seaward extents (2.18km) derived from analysis of the at sea distribution of little terns

[1] http://jncc.defra.gov.uk/pdf/SAS_Identification_of_important_marine_areas_for_little_terns

recorded as part of JNCC's work programme at other study colonies (Parsons *et al.* 2015). As virtually all little terns have nested only at Rye Harbour over several recent decades (Lewis Yates 2014), these generic distances were centred on the colony location at Rye Harbour (Figure 2). It should be noted that the extent of the area identified as being important for little terns on the basis of these generic values (Figure 2) does not dictate the alongshore or seaward extent of the proposed extensions to the pSPA boundary. These are dictated by the foraging needs of the larger terns which are described in the following section.

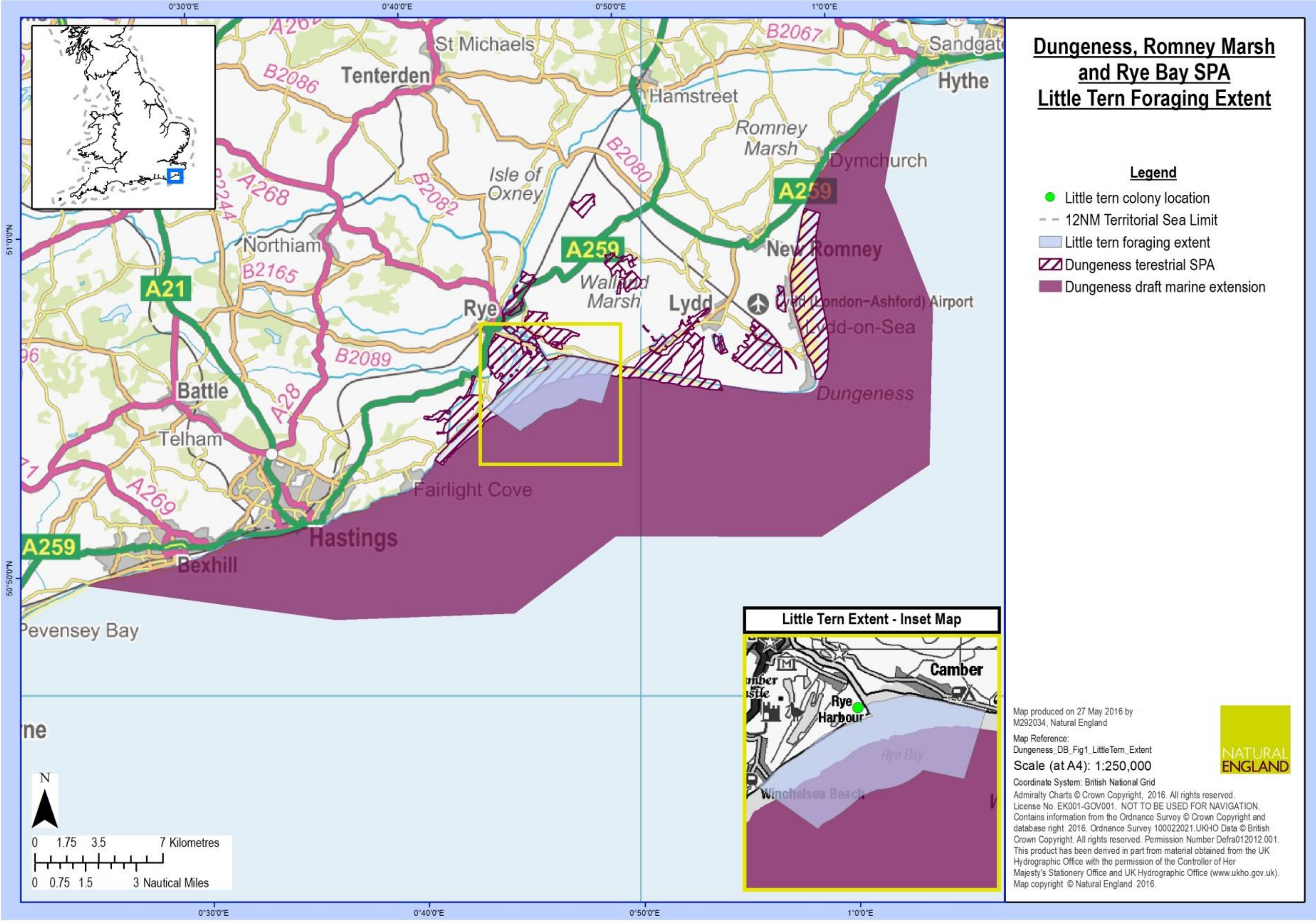


Figure 2. Application of generic alongshore and generic seaward extents to define boundaries to little tern foraging areas around the Rye Harbour colony within the Dungeness, Romney Marsh and Rye Bay pSPA.

2.4. Identification of important marine areas for larger terns

The four larger species of tern which breed regularly in Great Britain have recorded mean foraging ranges between 4.5 km and 12.2 km and maximum recorded foraging ranges between 30 km and 54 km (Thaxter *et al.,* 2012). In the light of these larger areas of interest, JNCC, in agreement with all of the SNCBs, decided that the most effective method to determine the extent of the areas most heavily used by breeding terns of the four larger species would be different to that employed for little terns. In this case the approach was to take the results from a programme of boat-based visual tracking of foraging birds which identified the foraging locations chosen by the birds. These results were then used in conjunction with information on the habitat characteristics of those locations, relative to other areas available to the birds, to construct habitat association models of tern usage. These models were used to predict tern usage patterns around breeding colony SPAs. Usage predictions were made out to the maximum recorded foraging range from each colony. The process of producing usage predictions around colonies for which tracking data had been gathered relied upon colony (and species) specific analysis which produced a smoothed map of foraging usage around the colony in question (Phase 1). For colonies for which no (or insufficient) data were available, analysis of pooled data across colonies (species-specific) produced generic models which also allowed production of maps of smoothed foraging usage in these cases (Phase 2). Further information on the larger tern survey programme is presented in Wilson *et al.* (2014) and on the JNCC website[2].Further general information on the larger tern survey programme is presented in Appendix 5.

To gather the empirical data necessary for the modelling, JNCC coordinated a programme of visual tracking work between 2009 and 2011 to identify important foraging areas at a number of UK colonies. These surveys were conducted during the chick rearing period in each year and comprised repeated days of observations of individual terns whose tracks were followed by boat as they left the colony to forage.

Visual tracking was carried out or commissioned by JNCC at 10 of 32 UK 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 common terns and Sandwich terns in Rye Bay.

The total number of tracks obtained was 1,004 including 55 tracks (6%) for roseate tern (2 SPAs), 184 tracks (18%) for Arctic tern (6 SPAs, 1 non-SPA), 381 tracks (38%) for common tern (7 SPAs, 1 non-SPA) and 384 tracks (38%) for Sandwich tern (5 SPAs, 1 non-SPA), with multiple years of data collected at five of the ten JNCC study colony SPAs. In addition, visual tracking data were obtained through a data-sharing agreement with ECON Ecological Consultancy Ltd for two SPAs: Ynys Feurig, Cemlyn Bay and The Skerries SPA (136 Sandwich, 2 common and 1 Arctic tern tracks, all collected in 2009) and North Norfolk Coast SPA (108 Sandwich and 24 common tern tracks collected 2006-2008). This gave a total of 1,275 tracks available to the project, although not all data were used in the modelling; incomplete tracks or those which recorded no foraging behaviour were excluded.

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

[2] http://jncc.defra.gov.uk/pdf/JNCC_Report_500_web.pdf

the boundary to accommodate any more increase in, in this case, foraging tern usage[3]. Further information on the boundary setting process is presented in Win *et al.* (2013) and on the JNCC website[4].Further general information is presented in Appendix 5.

The species of interest were common tern and Sandwich tern. No visual tracking data were available from this colony so generic models of usage distribution were applied using phase 2 of the analysis described above. Details of the modelling process including the phase 2 common tern and sandwich tern models used here are given in Wilson *et al.* (2014), and summarised in a document available from the JNCC website[5]. The following sections summarise the results of application of generic models to predict the areas of greatest importance to foraging common and Sandwich terns originating from the colonies within the Dungeness, Romney Marsh and Rye Bay pSPA.

2.4.1. Common tern

The principal locations within the pSPA which have held the bulk of the common tern nests in recent years have been: Pett Level, Rye Harbour and on Dungeness (Lewis Yates 2014). Predictions of usage of marine areas by foraging common terns around Rye Bay were made in relation to common terns originating from each of these locations by using a generic model, generated from pooled data obtained from surveys at common tern colonies across the UK. The predictor variables used in the generic model to generate usage patterns were: i) distance to colony, ii) distance to shore, and iii) bathymetry. Predicted usage levels were highest around each of the colony locations, generally decreasing with increasing distance from these and from the shore. The model generated predictions of relative usage by common tern originating from each of the 3 source locations are shown in Figure 3, together with the boundary drawn around the areas in which predicted usage around each location exceeded the threshold identified by application of the maximum curvature approach (to define a limit to the extent of the most important areas). The extent of the area of prediction was defined by the limit of the dark blue area 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 that very substantial areas of sea within that wider area which are distant to the colony locations and/or distant from the shore are predicted to have very little or no usage by foraging common terns.

[3] http://jncc.defra.gov.uk/pdf/SAS_Defining_SPA_boundaries_at_sea
[4] http://jncc.defra.gov.uk/pdf/JNCC_Report_500_web.pdf
[5] http://jncc.defra.gov.uk/pdf/SAS_Identification_of_important_marine_areas_for_larger_terns

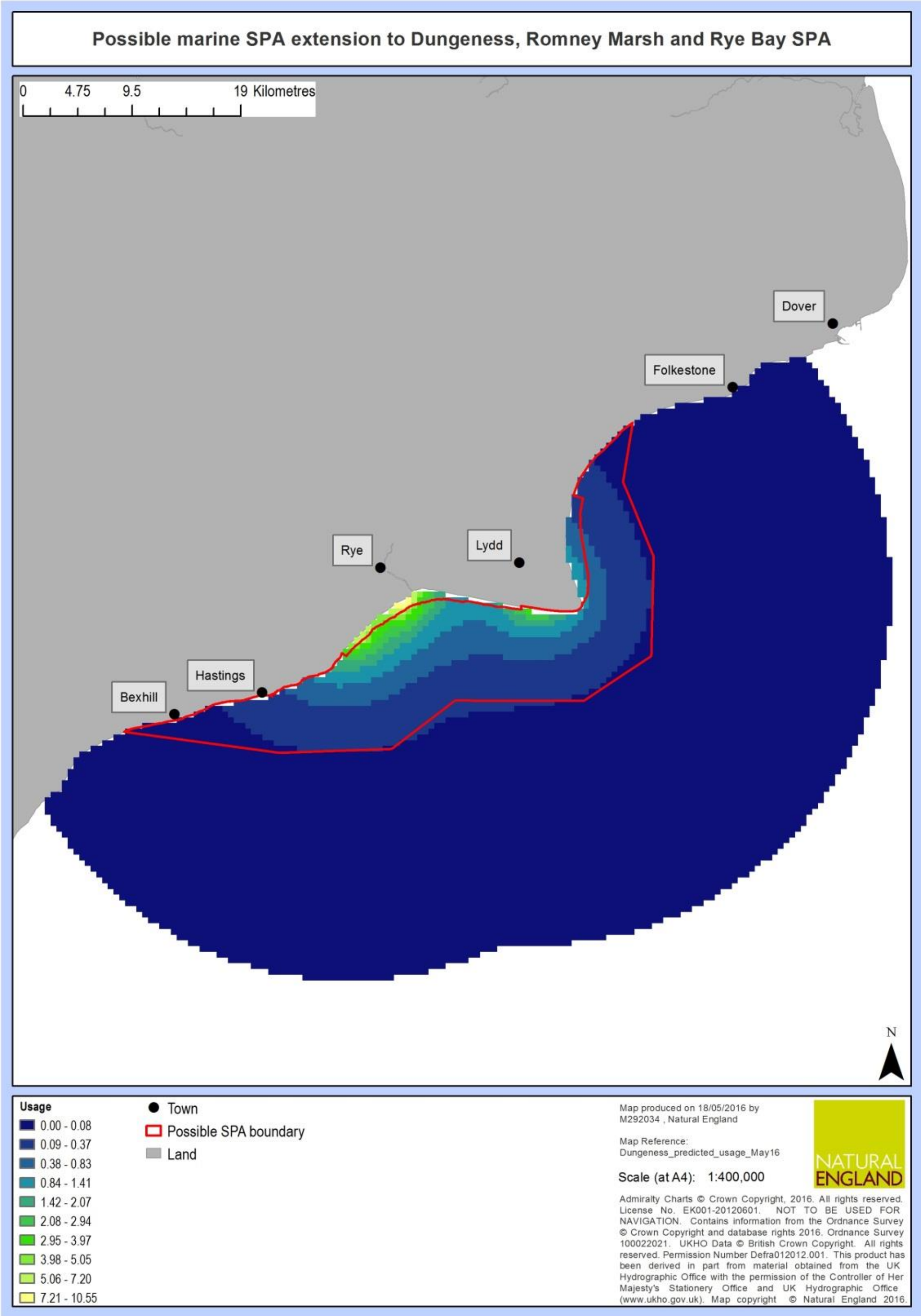


Figure 3: Model prediction of common tern usage overlaid with maximum curvature derived limits (red lines) to areas of most importance around the Rye Harbour colony, the Pett Level colony and the Dungeness colony. The red line boundary shown defines the extent of the areas exceeding the threshold level of usage defined by the application of maximum curvature to the predicted usage maps for common terns around each location separately. Note that the landward red line boundary shown in this figure does not include some of the area of highest usage along the shore because those areas are already included within the boundary of the existing SPA which extends down to Lowest Astronomical Tide (LAT) in some sections. All areas of usage exceeding the maximum curvature threshold are nonetheless included within the overall pSPA boundary.

2.4.2. Sandwich tern

The principal location within the pSPA which has held all of the Sandwich tern nests in the last two decades has been Rye Harbour (Lewis Yates 2014). Predictions of relative usage of marine areas by foraging Sandwich terns around Rye Bay were made in relation to Sandwich terns originating from this location using a generic model, generated from pooled data obtained from surveys at Sandwich tern colonies across the UK. The predictor variables used in the generic model to generate usage patterns were: i) distance to colony, ii) distance to shore, and iii) bathymetry. Predicted usage levels were highest around the colony, generally decreasing with increasing distance from the colony alongshore and with increasing distance from shore (Figure 4).

The model generated predictions of relative usage by Sandwich tern originating from Rye Harbour are shown in Figure 4, together with the boundary drawn around 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). The extent of the area of prediction was defined by the limit of the dark blue circle shown. This reflects the constraint imposed on the maximum curvature analysis 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 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 Sandwich terns.

Possible marine SPA extension to Dungeness, Romney Marsh and Rye Bay SPA

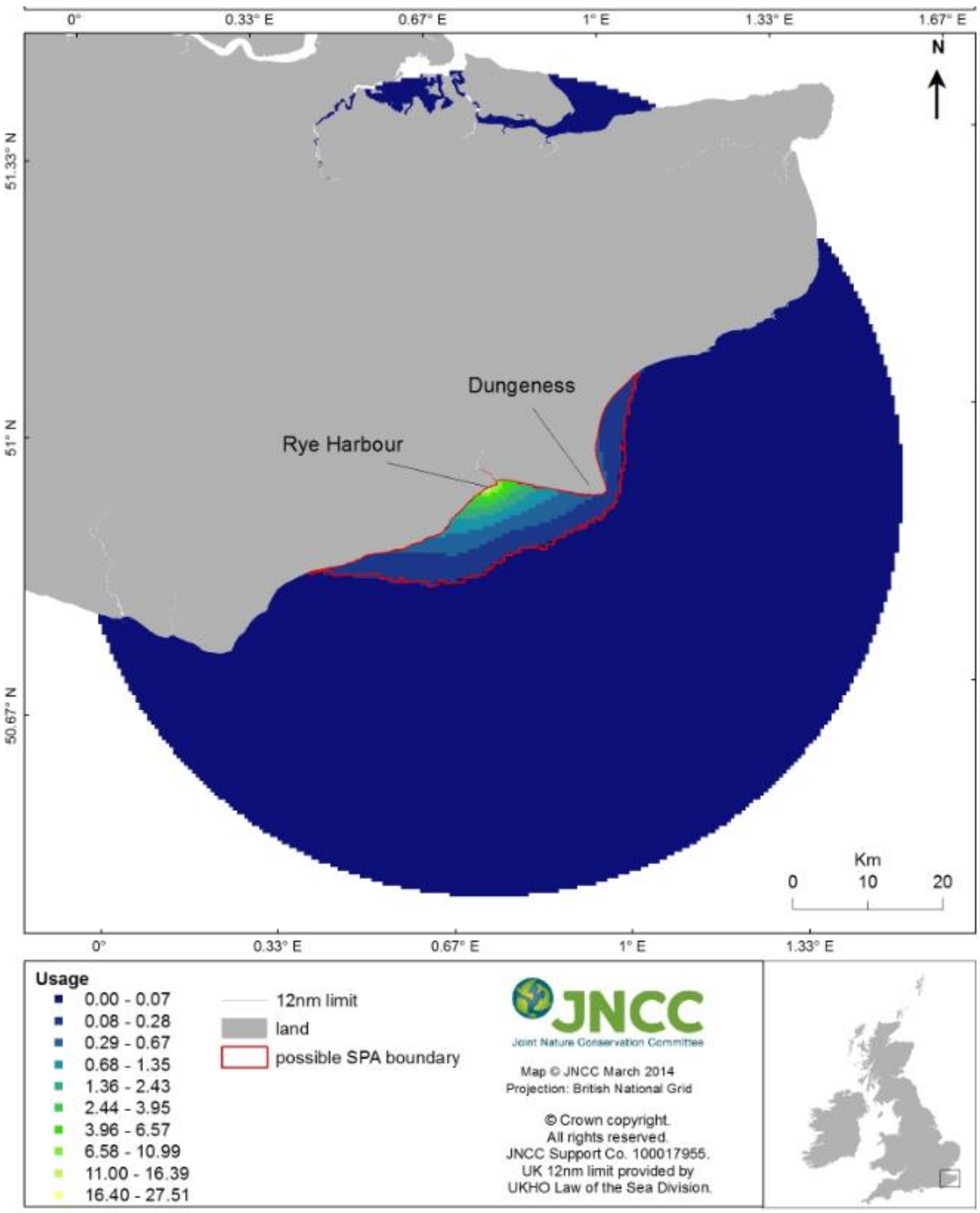
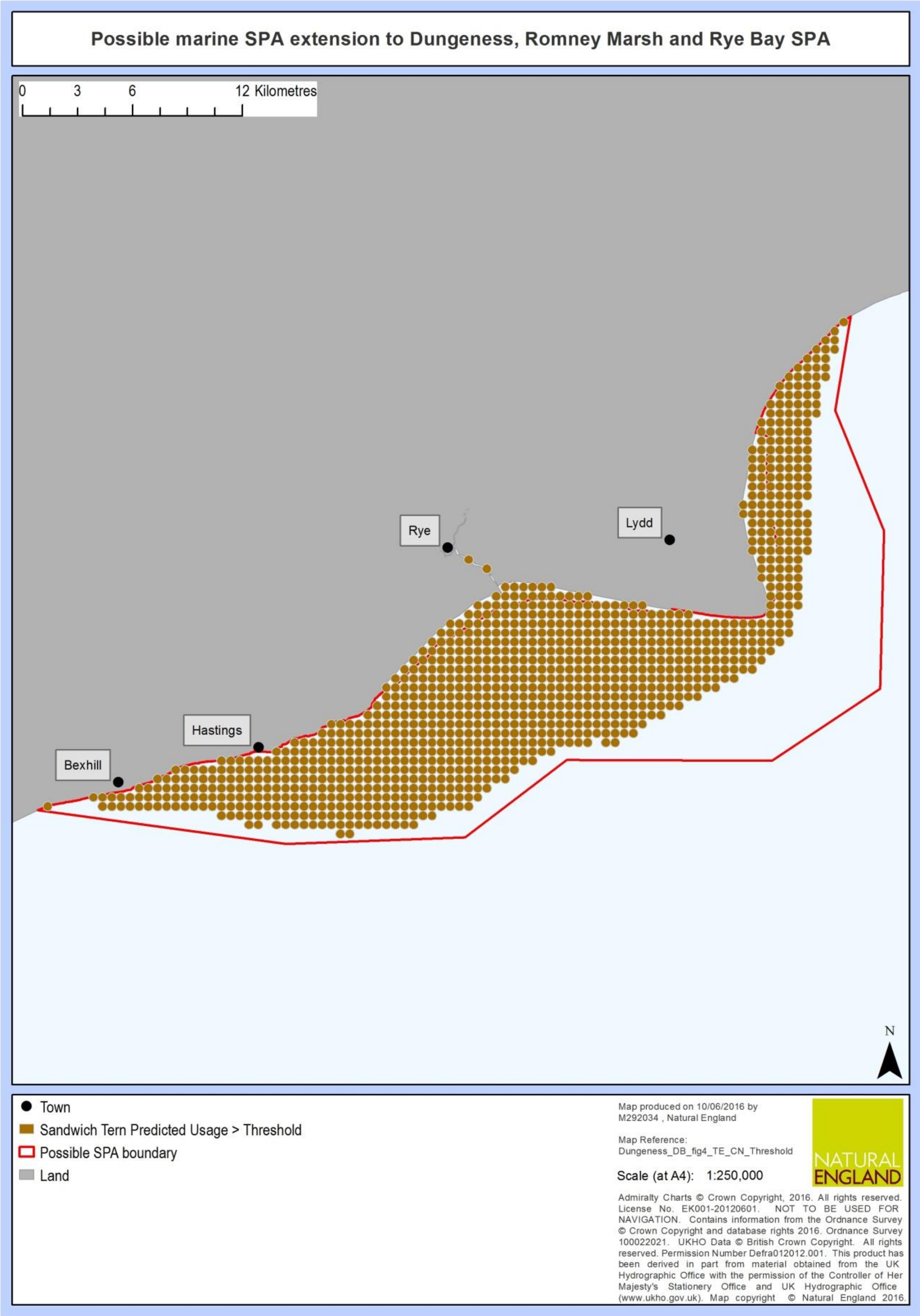


Figure 4: Model prediction of Sandwich tern usage overlaid with maximum curvature derived limits (red lines) to areas of most importance around the Rye Harbour colony. Note in this figure, the landward red line boundary has not yet been adjusted to account for the fact that some of the areas of highest usage along the shore are already included within the boundary of the existing SPA which extends down to Lowest Astronomical Tide (LAT) in some sections. All areas of usage exceeding the maximum curvature threshold are nonetheless included within the overall pSPA boundary.

2.4.3. Composite larger tern boundary

Based on generic model predictions of usage by both species from their source colonies within the pSPA, 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 all of the principal sources within the SPA is shown in Figure 4. This indicates a potential pSPA boundary for foraging larger terns.

4a)



Possible marine SPA extension to Dungeness, Romney Marsh and Rye Bay SPA
0
3
6
12 Kilometres
Rye
Lydd
Hastings
Bexhill
N
Town
Sandwich Tern Predicted Usage > Threshold
Possible SPA boundary
Land
Map produced on 10/06/2016 by M292034 , Natural England
Map Reference: Dungeness_DB_fig4_TE_CN_Threshold
Scale (at A4): 1:250,000
NATURAL ENGLAND
Admiralty Charts © Crown Copyright, 2016. All rights reserved. License No. EK001-20120601. NOT TO BE USED FOR NAVIGATION. Contains information from the Ordnance Survey © Crown Copyright and database rights 2016. Ordnance Survey 100022021. UKHO Data © British Crown Copyright. All rights reserved. Permission Number Defra012012.001. This product has been derived in part from material obtained from the UK Hydrographic Office with the permission of the Controller of Her Majesty's Stationery Office and UK Hydrographic Office (www.ukho.gov.uk). Map copyright © Natural England 2016.

4b)

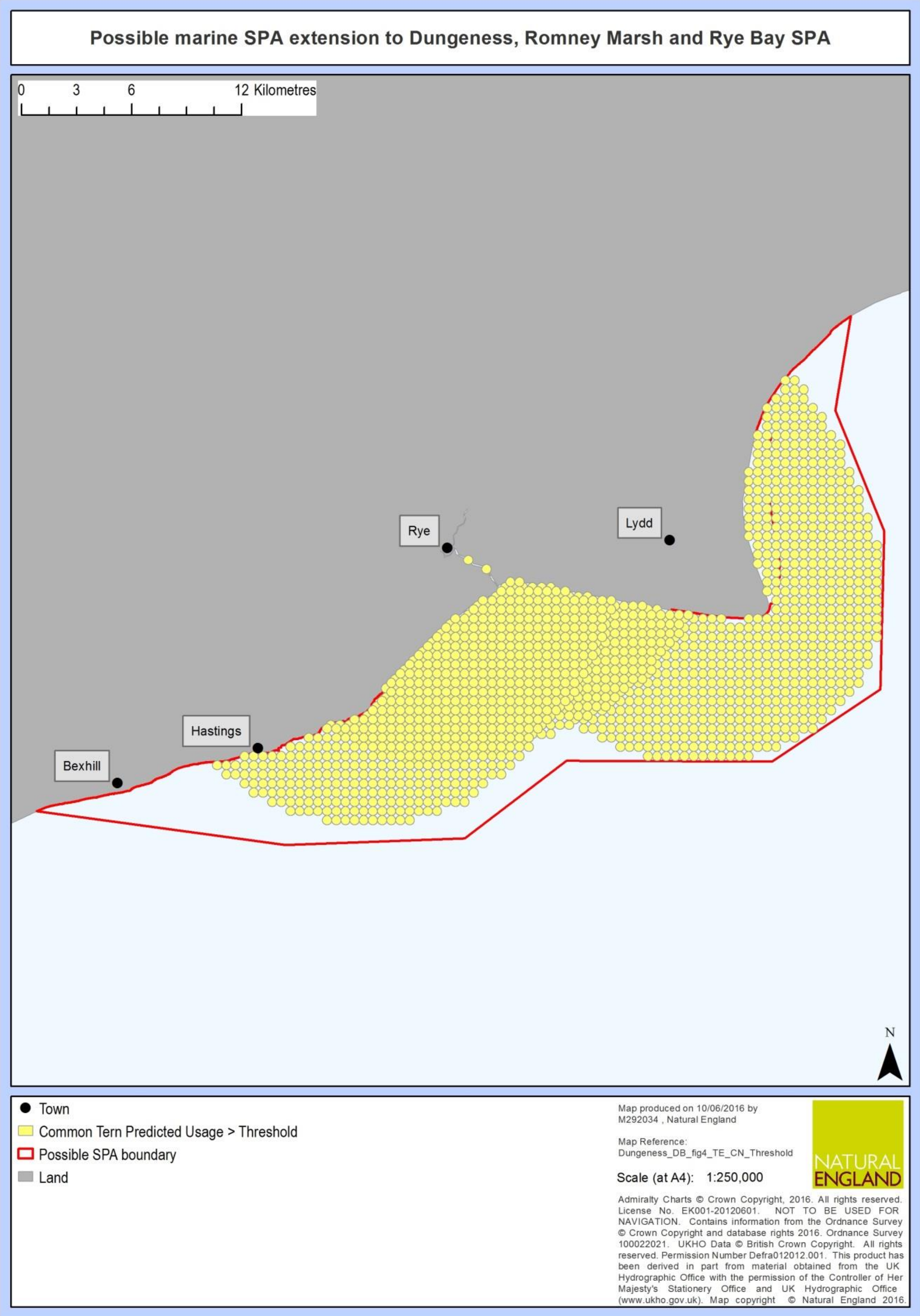


Figure 4b: Proposed simple, composite boundary drawn around the cells within which predicted usage levels by Sandwich terns (4a) or common terns (4b), originating from the source colonies, exceeded the threshold level identified by application of the maximum curvature methodology to the predicted usage surfaces (see Appendix 5). Source: Win *et al.* (2013) (amended by inclusion of: i) usage patterns of Sandwich terns originating from Rye Harbour and ii) usage patterns of common terns originating from Pett Level and Dungeness nesting locations, neither of which are presented in this source document).

2.5. Composite boundary of Dungeness, Romney Marsh and Rye Bay pSPA

The seaward and alongshore extent of the Dungeness, Romney Marsh and Rye Bay pSPA (Figure 5) is determined wholly by the modelled foraging distribution of Sandwich terns and common terns; from west to east by the distributions of birds originating from: Pett Level, Rye Harbour and Dungeness. It should be noted that the same generic model of common tern usage was used to generate relative usage maps around each of these nesting locations. However, the precise size and shape of the areas defined by application of the maximum curvature method to the predicted usage maps varies between colonies. This is due to there being differences in the sea areas around each nesting location in regard of: i) bathymetry and ii) the relative distribution of sea areas at different distances from the colony and the shore. 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 nesting locations within the existing SPA. The boundaries to the areas predicted to support most of the foraging activity by little terns originating from the colony at Rye Harbour are contained entirely within the composite boundary of the pSPA dictated by the distributions of the larger tern species.

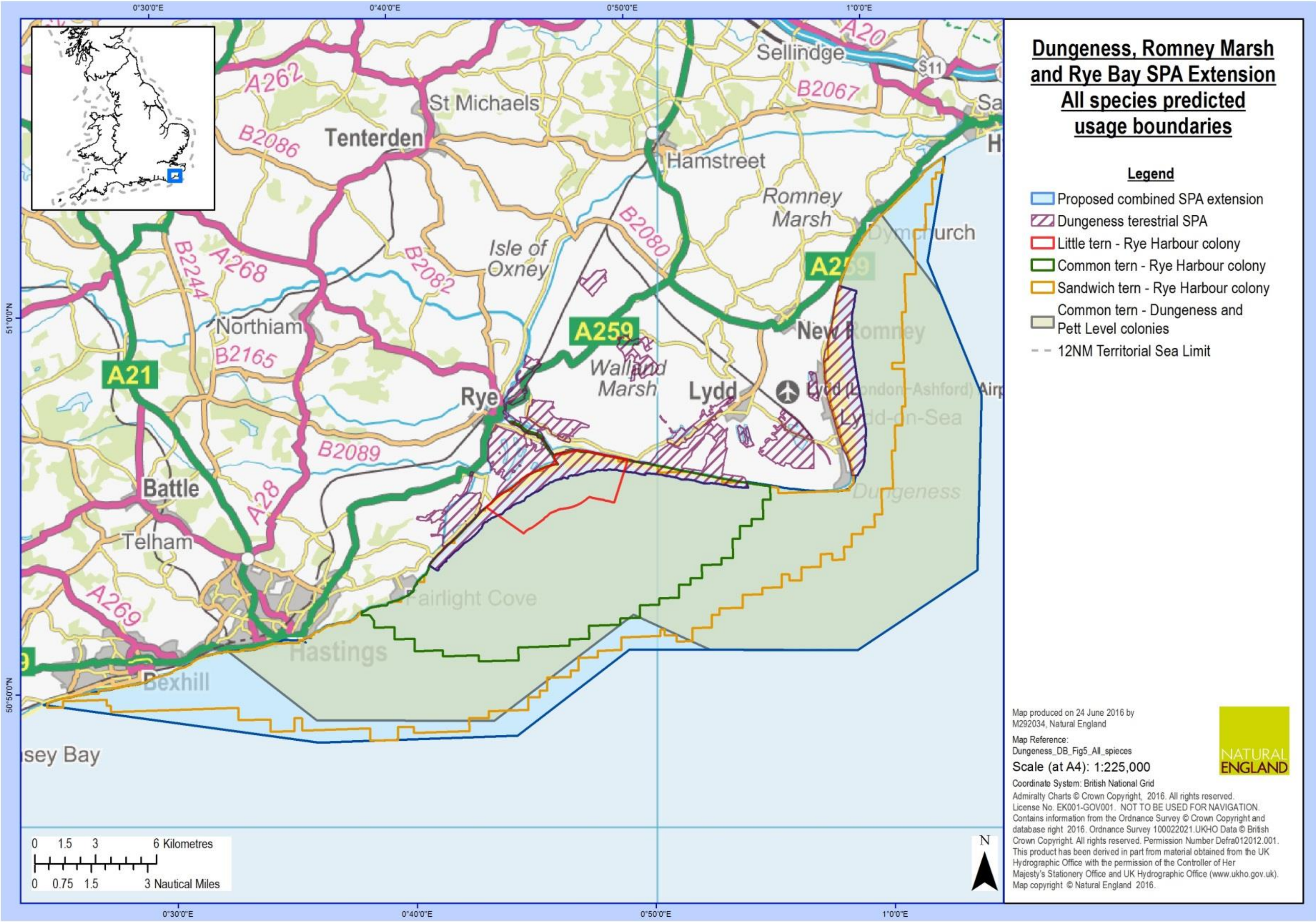


Figure 5. Predicted usage boundary of the Dungeness, Romney Marsh and Rye Bay pSPA drawn around the species and nesting location specific boundaries to areas of greatest relative usage at sea presented in the preceding sections.

2.6. Verification of predictions of generic modelled boundaries

Given that in the case of this pSPA, its size and shape is determined purely on the basis of predictions of Sandwich tern and common tern usage patterns generated by generic models, rather than on models based on observations of these species in Rye Bay, it is appropriate to consider the reliability of that evidence base.

There are three sources of information which can be used in considering the reliability of the use of generic approaches to define the areas of importance to each tern species in the case of this pSPA. Each of these is described in brief below, in various degrees of detail in Appendices 5-7 to this document and in full detail in source documents, the details of which are given in these appendices.

Appendix 5 describes the process of cross-validation by which the robustness of each generic model was assessed using standard statistical criteria during the modelling analysis. This demonstrated that both the Sandwich tern model and common tern model were considered reliable, as judged by their ability to predict the observed distribution of birds of the same species at colonies which were (in the cross-validation process) excluded in turn from building the models used for each species. Overall, the average test statistic for this cross-validation process was classed as indicative of the model being “excellent” in the case of the Sandwich tern model and “good” in the case of the common tern model (see Appendix 5). 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 and common 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 and common terns generated by generic models, is founded on a reliable evidence base, albeit not one derived directly from birds at the colony in question.

In addition to this cross-validation work undertaken as part of the modelling analysis, there are two further sources of information which can be used in considering the reliability of the use of generic approaches to define the areas of importance to each tern species in the case of this pSPA.

In 2014, Lewis Yates (2014) carried out a systematic programme of observations of tern activity in and around Rye Bay. Appendix 6 presents a summary of the findings of that work. In brief, this work involved systematic and repeated observations of tern activity along the coast between Hastings and Hythe. This confirmed the very limited alongshore distribution of little tern activity on either side of the colony at Rye Harbour in comparison with the distribution of the two larger tern species. In the case of the more wide ranging of the two larger tern species, i.e. Sandwich tern, this work confirmed close alignment between the modelled predictions of the occurrence of Sandwich terns all along the coastline between Fairlight-on-Sea and Greatstone-on-Sea and the systematic programme of observations. The boundaries of the pSPA do extend further to the west and north-east than the observations in 2014 give support for. That is likely to be a reflection of the difficulty in any empirical programme of data collection of confirming levels of activity that become steadily lower further from the colony without conducting disproportionate levels of survey effort in such places. Nonetheless, the validity of the full alongshore limit to boundaries of other pSPAs predicted by the same generic Sandwich tern model as used in the case of this pSPA has been demonstrated in 2014 during verification work in Northern Ireland (Allen & Mellon Environmental Ltd 2015) and in further verification work commissioned by Natural England in 2015 in e.g. the Solent and Dorset Coast pSPA (ECON 2015) as described in Appendix 7.

In 2015, Natural England commissioned a programme of survey work at a number of locations around England which had been identified as being of importance to foraging terns and hence included within the boundaries of various pSPAs or sites that may in due course become pSPAs. The results of this national programme of work showed that in no case was an area that was identified by modelling as being likely to support levels of significant usage not found to support foraging terns. More information on this work is presented in Appendix 7 where a link is provided to the report on that work.

3. Boundary description

3.1. Existing site boundary

The full details of the extent and landward and seaward boundaries of the existing SPA are presented in the departmental brief published in 2010 (Natural England, 2010).

In summary, the total area of the existing Dungeness, Romney Marsh and Rye Bay SPA is 4,010.29 ha. It is comprised on numerous blocks of land and two lengthy stretches of coastline. The first of these stretches of coastline extends from Dungeness Point to just south of St Mary's Bay (Figure 6a). The second extends from Cliff End in the west to Lydd Ranges in the east (figures 6a and b).

Generally, along the stretches of open coast, the existing landward boundary follows existing coastal defence structures e.g. green wall, other sea walls, seaward boundaries of properties and coast roads and includes areas of foreshore and coastal shingle. The boundary has been drawn to exclude major roads, railways and permanent buildings. However, where bridges and other structures cross intertidal, subtidal and other wetland habitats, the boundary has not been drawn to exclude these man-made structures. Annotations that appear on the existing boundary maps as shown in the departmental brief published in 2010 (Natural England, 2010), confirm that the site excludes permanent structures such as buildings, roads, bridges, culverts, slipways, jetties and houseboats. However, the site does include any exposed bank side, intertidal, subtidal or other wetland habitats beneath the aforementioned structures. Along the two stretches of open coastline, the seaward boundary of the existing site follows the line of Lowest Astronomical Tide (LAT).

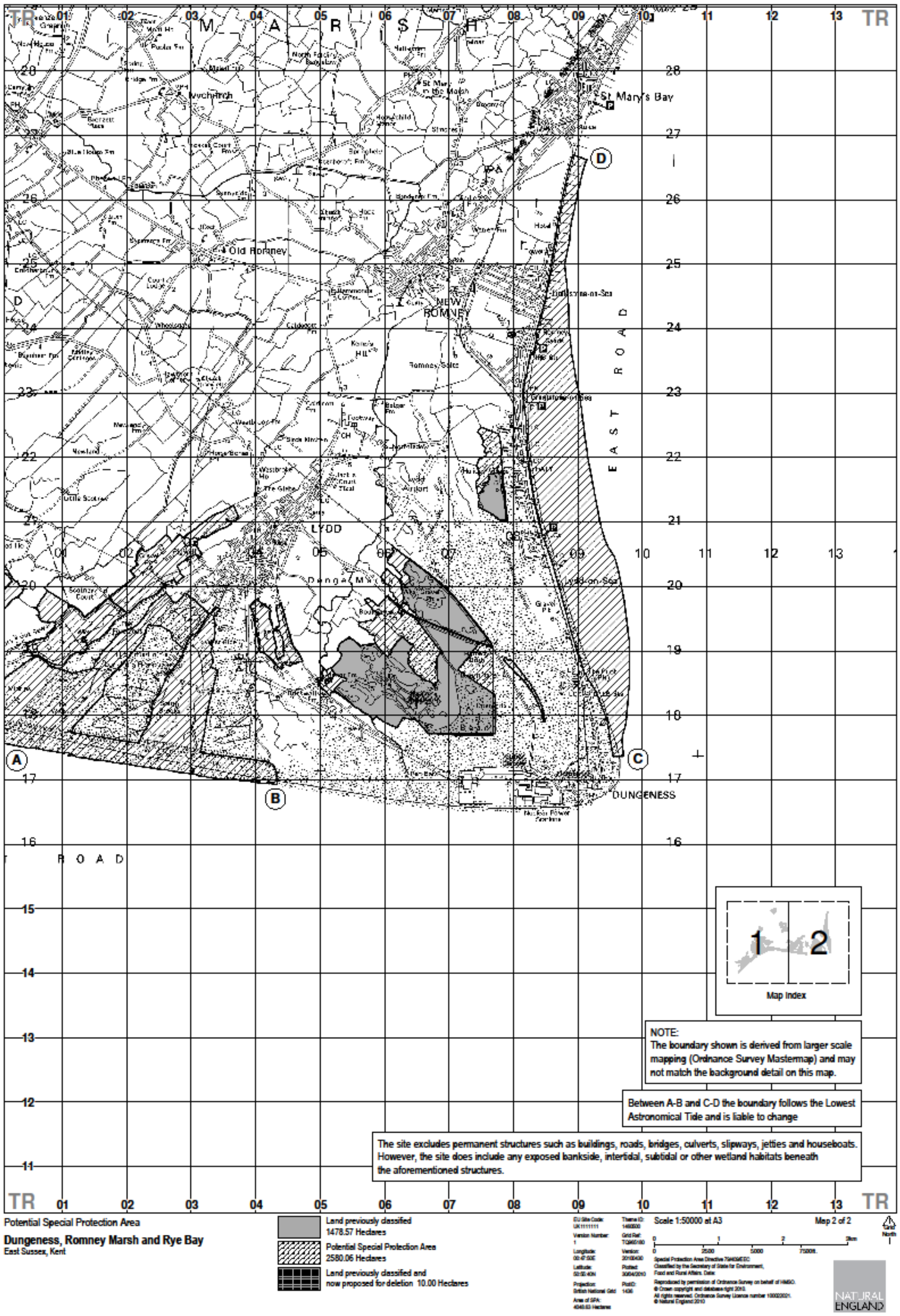

St Mary's Bay
Old Romney
NEW ROMNEY
LYDD
Lydd-on-Sea
DUNGENESS
EAST ROAD
ROAD
NOTE:
The boundary shown is derived from larger scale mapping (Ordnance Survey Mastermap) and may not match the background detail on this map.
Between A-B and C-D the boundary follows the Lowest Astronomical Tide and is liable to change
The site excludes permanent structures such as buildings, roads, bridges, culverts, slipways, jetties and houseboats. However, the site does include any exposed bankside, intertidal, subtidal or other wetland habitats beneath the aforementioned structures.
Map Index
Potential Special Protection Area
Dungeness, Romney Marsh and Rye Bay
East Sussex, Kent
Land previously classified 1478.57 Hectares
Potential Special Protection Area 2580.06 Hectares
Land previously classified and now proposed for deletion 10.00 Hectares
Scale 1:50000 at A3
Map 2 of 2
NATURAL ENGLAND

Figure 6a.

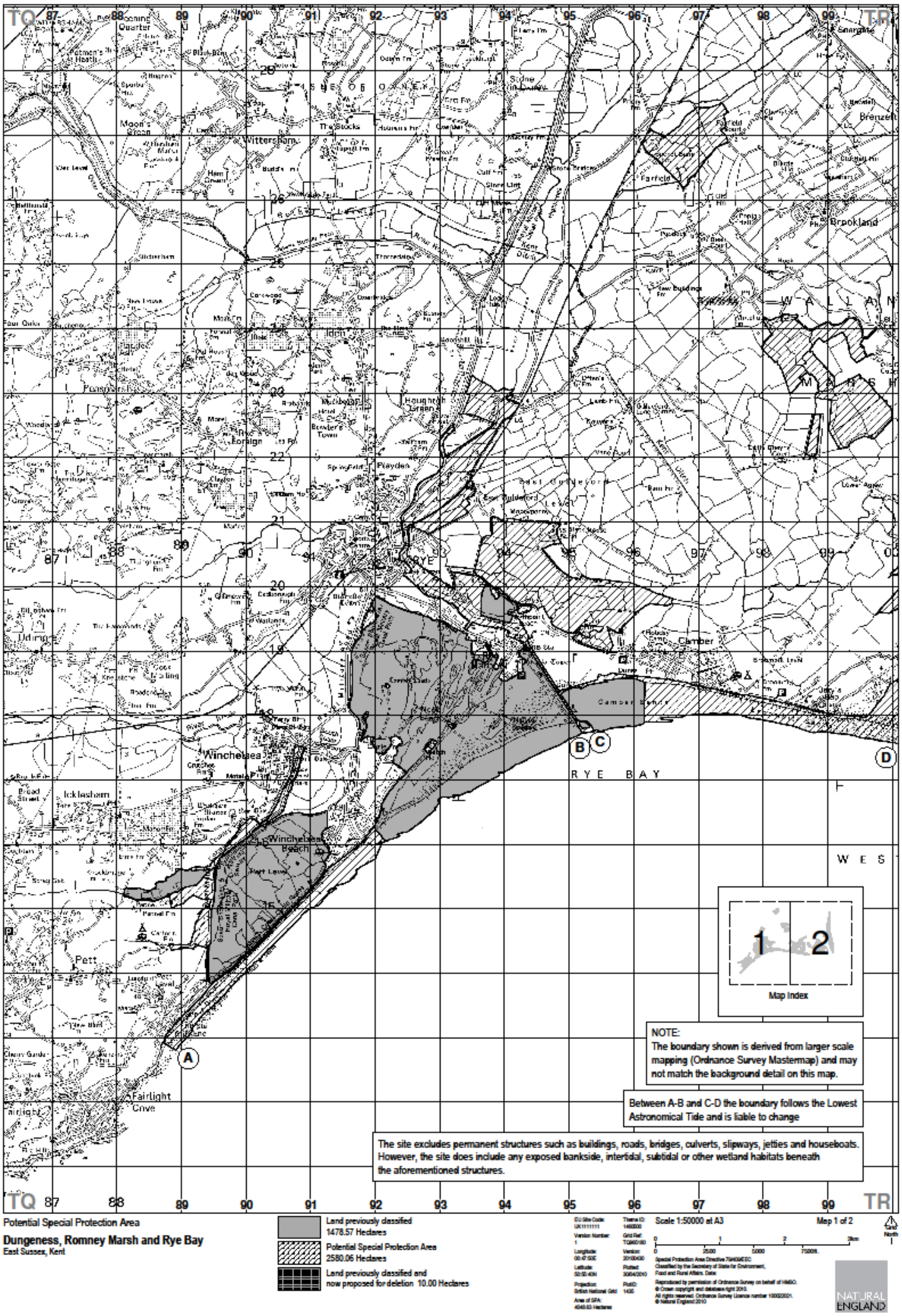


Figure 6b.

Figures 6a and 6b – Dungeness, Romney Marsh and Rye Bay terrestrial SPA boundary: South of St Mary’s Bay in the north and along the coast to Cliff End in the east (Departmental Brief, 2010).

3.2. Dungeness, Romney Marsh and Rye Bay pSPA boundary

The existing Dungeness, Romney Marsh and Rye Bay pSPA is approx. 4,010ha and the proposed extension is an additional 30,364.13 ha of marine habitat. The total area of the proposed pSPA is therefore 34,374.42 ha.

The proposed boundary changes to the existing Dungeness, Romney Marsh and Rye Bay SPA are based upon projected foraging areas of little terns, common terns and Sandwich terns breeding within the existing SPA.

3.2.1. Landward boundary of the pSPA.

In the guidelines for the selection of marine SPAs for aggregations of inshore non-breeding waterbirds published by JNCC (Webb and Reid, 2004) it is stated that where the distribution of birds is likely to meet land, landward boundaries should be set at MHW “unless there is evidence that the qualifying species make no use of the intertidal region at high water”. There are no equivalent guidelines for the identification of landward boundaries to marine SPAs for breeding seabirds, and in their absence it was considered appropriate to adhere to these existing guidelines where appropriate to do so. Observations indicated that little tern forage both in the intertidal zone and subtidal zone (Parsons *et al.,* 2015). Lewis Yates (2014) confirms this to be the case at Rye Bay as it is noted that “They could often be seen at high tide very close to the shoreline but otherwise exploited the small pools that formed as the tide receded”. The coastline within the proposed extension has similar characteristics to that at Rye Bay and there is no evidence that terns would not use the intertidal areas in a similar way throughout the pSPA. Use of such areas by all larger tern species is also likely, as all species of tern considered routinely forage in areas of shallow water (Eglington, 2013). There is therefore no reason to conclude that these species will not forage close to shore and over intertidal areas. Accordingly, in line with the guidance from JNCC concerning inshore non-breeding waterbirds, the landward boundary of the areas of importance to foraging terns has been taken to be MHW. This means that within the two principal blocks of the existing SPA which are connected to the coast, including the one which extends up the River Rother, no change is required to the landward boundary to accommodate the foraging needs of the terns because they are already qualifying features of the existing SPA and the existing site landward boundary already lies at or above MHW. However, where the existing site is being extended to include additional stretches of coastline, a new landward boundary has to be defined and will, in line with guidance, follow the MHW mark. This will be the new landward boundary along three stretches of coast now included within the pSPA i.e. from Cliff End to Bexhill, between Galloways lookout (national grid reference TR04331705) and Dungeness Point and from the beach groyne on Romney sands (national grid reference TR08922669) to Hythe (Figure 7).

3.2.2. Seaward boundary of the pSPA

The seaward boundary is defined by the areas of importance to Sandwich terns originating from Rye Harbour and common terns originating from each of 3 principal nesting locations within the existing SPA. The seaward limit to the areas considered to be of greatest use by little terns from Rye Harbour is contained well within the areas of use by the two larger tern species. Due to the negative relationship between the distance from the colony and usage levels of the larger tern species, the seaward limits to the areas of importance mostly extend furthest offshore relatively close to the colonies and approach the shore increasingly closely at increasing distances along the coast from the colony (Figure 7). At its furthest, the seaward limit to the pSPA extends 9km offshore from the existing SPA boundary at Rye Harbour (Figure 7).

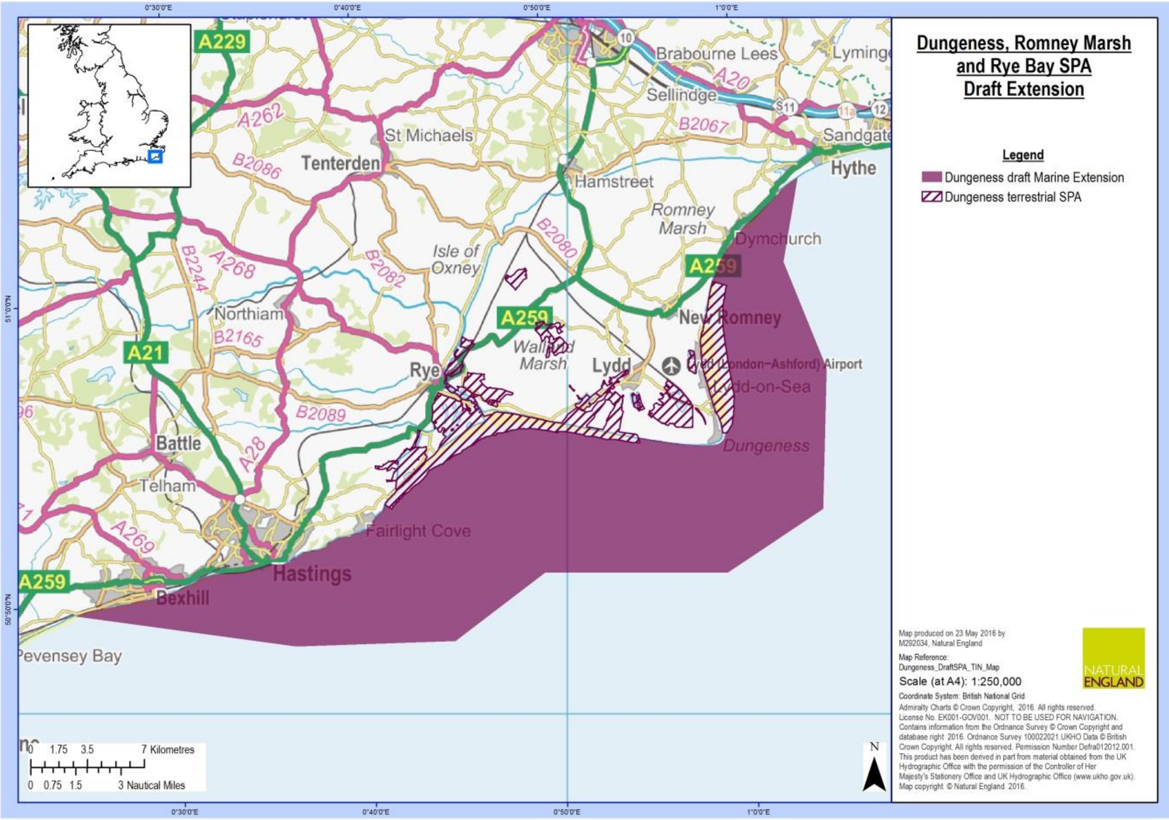


Figure 7 – Dungeness, Romney Marsh and Rye Bay SPA extension

4. Location and Habitats

4.1. Existing Dungeness, Romney Marsh and Rye Bay SPA

The existing Dungeness, Romney Marsh and Rye Bay SPA is located on the east Sussex and Kent coast, between the towns of Pett Level in the south and St Mary's Bay in the north. There are areas inland at Denge Marsh which do not join up with the coastal sections but that are part of the SPA. The areas of the SPA extension will extend the site location between Norman's Bay in the south and south of Hythe in the north.

The existing SPA site contains a diverse coastal landscape comprising a number of habitats all of which exist today because of coastal processes that have formed and continue to shape a barrier of extensive shingle beaches and sand dunes across an area of intertidal mud, sandflats and saltmarsh. Extensive areas of open water, ditch systems and dykes are also present. Details of these habitats and the importance of them in supporting various communities and as important feeding and roosting resources for birds and fish are described in full within the departmental brief published in 2010 (Natural England 2010).

Large areas of the existing SPA are owned and managed as nature reserves by a variety of organisations. These are detailed below;

- Dungeness National Nature Reserve (NNR) situated in and around Dungeness Point is owned and leased by EDF Energy, Natural England, the Royal Society for the Protection of Birds (RSPB) and Shepway District Council.
- The Romney Marsh Countryside Project managed land within the NNR and Romney Warren Local Nature Reserve (LNR).
- The RSPB manage nearly 1000ha east of Lydd Ranges as a nature reserves for birds and other wildlife.
- Rye Harbour LNR is managed on behalf of a committee representing a range of bodies including county councils, Environment Agency, Sussex Wildlife Trust and private landowners.
- Cheyne Court private nature reserve is owned by the Elmley Conservation Trust
- Pett Level and East Guldeford Levels private nature reserves are owned and managed by the Wetland Trust.

The entire area of the existing SPA is designated at an international level as a Ramsar site. There are additional areas of Pett Level SSSI, Rye Harbour SSSI, Camber Sands and Rye Saltings SSSI, Walland Marsh SSSI and Dungeness SSSI which are also within the Ramsar site designation boundary. The features listed on the Ramsar site citation were;

- rare and nationally scarce plants;
- Red Data Book wetland invertebrates; and
- an internationally important wintering population of Bewick's swan.

The entire area of the existing SPA is also designated at a national level as a Site of Special Scientific Interest (SSSI). As well as aggregations of overwintering waterfowl, breeding numbers of 16 birds, the SSSI is notified for its important coastal habitats and assemblage of nationally scarce vascular plants and wetland invertebrates.

Dungeness is also a Special Area of Conservation (SAC) designated for great crested newt (*Triturus cristatus*) and the following qualifying habitats;

- Annual vegetation of drift lines
- Perennial vegetation of stony banks (coastal shingle vegetation outside the reach of waves)

4.2. Proposed Dungeness, Romney Marsh and Rye Bay pSPA

The proposed extensions to the pSPA are entirely within the marine environment and are the important areas used for foraging by the three species of terns which are qualifying features of the existing SPA. Populations of these species and their productivity have been variable and often very low and concerns have been raised over the supply of fish in the local waters (Lewis Yates 2014). Each of the principal areas of extension is further detailed below.

4.2.1. Romney Sands (area 1 on Figure 1)

This alongshore extension stretches from the point which marks the current northern most boundary of the existing SPA (beach groyne at TR08922669) past St Mary's Bay and Dymchurch to West Hythe. The landward boundary follows the MHW Mark and so the area included within the extension consists of any intertidal habitats along this stretch of coast below that shore level and areas of sea beyond that out to the seaward limit. Heading south along this stretch of the proposed extension, the seaward boundary moves progressively further offshore and merges into the now more seaward boundary to the coastal part of the existing SPA which runs from Littlestone-on-Sea to Dungeness Point.

The following EUNIS Level 3 broadscale habitats are present within this area;

- Sub-tidal sand (A5.2)
- Sub-tidal coarse sediment (A5.1)
- Inter-tidal sand and muddy sand (A2.2)
- Moderate energy infralittoral rock (A3.2)

4.2.2. Dungeness Point to Galloways lookout (area 2 on Figure 1)

This alongshore extension stretches between the points which mark the southernmost limit of the northern coastal part of the existing SPA that already extends from Littlestone-on-Sea to Dungeness Point, and the easternmost limit of the coastal part of the existing SPA which already extends along Camber Sands and Broomhill Sands i.e. Galloways Lookout. The proposed extension fills in this gap. The landward boundary along this stretch of coast follows the Mean High Water (MHW) Mark and so the area included within the extension consists of any intertidal habitats along this stretch of coast below that shore level and areas of sea beyond that out to the seaward limit. The seaward boundary off this stretch of coast simply merges with those of the seaward extensions to the two blocks of coast included on either side in the existing SPA.

The following EUNIS Level 3 broadscale habitats are present within this area;

- Sub-tidal sand (A5.2)
- Sub-tidal coarse sediment (A5.1)
- Sub-tidal mud (A5.3)
- Inter-tidal sand and muddy sand (A2.2)
- Moderate energy infralittoral rock (A3.2)

4.2.3. Cliff End to Norman's Bay (Bexhill) (area 3 on Figure 1)

This alongshore extension stretches from the point which marks the current southernmost boundary of the existing SPA at Cliff End, past Fairlight, Hastings and St Leonards to a new westernmost boundary near Bexhill. This stretch of coast includes the rockier foreshore between Cliff End and Hastings and the more mixed rocky/sandy shores between Hastings and Bexhill. Along this stretch of coastline, the landward boundary follows the MHW Mark and so the area included within the extension consists of any intertidal habitats along this stretch of coast below that shore level and areas of sea beyond that out to the seaward limit. Heading north and east along this stretch of the proposed extension from its westernmost point, the seaward boundary moves progressively further offshore and merges into the now more seaward boundary to the coastal part of the existing SPA which runs from Cliff End to Broomhill Sands.

The following EUNIS Level 3 broadscale habitats present within this area are;

- Sub-tidal sand (A5.2)
- Sub-tidal coarse sediment (A5.1)
- Sub-tidal mud (A5.3)
- Sub-tidal mixed sediments (A5.4)
- Sub-tidal biogenic reefs (A5.6)
- Moderate energy infralittoral rock (A3.2)
- Moderate energy circalittoral rock (A4.2)

4.2.4. Rye Bay (area 4 on Figure 1)

The proposed extension of the existing SPA into the marine environment and the joining together of the two parts of the existing SPA that front the shore means that the seaward boundary of the pSPA now encompasses all of Rye Bay out to a maximum distance of 9km offshore.

The following EUNIS Level 3 broadscale habitats present within this area are;

- Sub-tidal sand (A5.2)
- Sub-tidal coarse sediment (A5.1)
- Inter-tidal sand and muddy sand (A2.2)
- Moderate energy infralittoral rock (A3.2)

5. Assessment of ornithological interest

The Dungeness, Romney Marsh and Rye Bay SPA (including proposed extensions that comprise the pSPA) supports over 1% of the GB populations of 12 species listed in Annex I to the EC Birds Directive (2009/147/EC) and over 1% of the biogeographical population of one regularly occurring migratory species (shoveler). It also supports a waterbird assemblage of European/international importance during the non-breeding season.

Counts of breeding tern populations have been derived from Rye Harbour Local Nature Reserve records, the Wetland Trust (Pett Level), and Dungeness RSPB Reserve records. All of the tern data sources are summarised with details of their method of data collection and verification process in Appendix 3 to this document.

5.1. Sandwich tern *Sterna sandvicensis*

The breeding population of Sandwich tern in GB 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: African-Eurasian Waterbird Agreement (AEWA), 2012). In the UK, the species is restricted to relatively few large colonies, most of which are on the east coast of Britain with a few smaller ones on the south and north-west coasts of England and in Northern Ireland. Colonies are mostly confined to coastal shingle beaches, sand dunes and offshore islets (Mitchell *et al*., 2004).

Sandwich tern was not a qualifying feature of the original Dungeness to Pett Level SPA when it was classified in 1999. However, by 2004-2008 the site supported an average of 350 breeding pairs of Sandwich terns, which represented 3.3% of the GB breeding population at that time. The species was included as a qualifying feature of the Dungeness, Romney Marsh and Rye Bay SPA when this was classified in 2016. Since 2004-2008, the Sandwich tern population has increased further. The number of pairs of Sandwich terns at Rye Harbour during a recent 5-year period (2011-2015) were; 850 (2011), 600 (2012), 120 (2013), 280 (2014) and 250 (2015). This provides a recent 5-year mean of 420 pairs (or 840 breeding adults). This represents 3.8% of the GB breeding population.

The principal feeding grounds of Sandwich terns that nest at Rye Harbour are considered to lie predominantly in marine areas of Rye Bay, extending up to 28km along the shore to the west of Rye Harbour and 30km following the coast eastwards and northwards from Rye Harbour around Dungeness Point to the East Road area.

5.2. Common tern *Sterna hirundo*

The breeding population of common tern in GB is estimated to be 10,000 pairs (Musgrove *et al*., 2013), representing at least 15% of the Southern and Western European breeding population (67,000 pairs derived by division by 3 of the upper estimate of 200,000 individuals and rounded to nearest 1,000: AEWA, 2012). A significant proportion of the British population breeds in Scotland. Coastal colonies in England are concentrated in the north-east, East Anglia, at a few localities along the south coast, and in the north-west (Mitchell *et al*., 2004). Common tern not only breeds around coasts but, unlike the other tern species which breed in the UK, also frequently beside inland freshwater bodies.

Common tern was a qualifying feature of the original Dungeness to Pett Level SPA when it was classified in 1999. The SPA citation states 266 pairs, at that time representing 2.2% of the GB breeding population (5-year peak mean, 1992-1996). By 2004-2008 the site supported an average of 273 breeding pairs of common terns, which represented 2.7% of the GB breeding population at that time. The species was included as a qualifying feature of the Dungeness, Romney Marsh and Rye Bay SPA when this was classified in 2016. Since 2004-2008, the common tern population has declined. The number of pairs of common terns at Rye Harbour during a recent 5-year period (2011-2015) were - 235 (2011), 155 (2012), 79 (2013), 90 (2014) and 135 (2015). The number of pairs of common terns at Pett Level during a recent 5-year period (2011-2015) were; 2 (2011), 25 (2012), 70 (2013), 57 (2014) and 0 (2015). The number of pairs of common terns on Dungeness during a recent 5-year period (2011-2015) were; 3 (2011), 23 (2012), 19 (2013), 36 (2014) and 9 (2015). This provides a recent 5-year mean across all three of these nesting locations of **188** pairs (or 376 breeding adults). This represents 1.9% of the GB breeding population.

The principal marine feeding grounds of common terns that nest at each of the three principal nesting locations overlap with one another. In combination, the sea areas of importance are considered to extend from St Leonards in the west to Dymchurch in the north.

5.3. Little tern *Sternula albifrons*

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

Little tern was a qualifying feature of the original Dungeness to Pett Level SPA when it was classified in 1999. The SPA citation states 35 pairs, at that time representing 1.5% of the GB breeding population (5-year peak mean, 1992-1996). At the time of the start of the re-classification process for the Dungeness, Romney Marsh and Rye Bay SPA, the numbers using the SPA had declined and between 2004 and 2008 the SPA, including the proposed extensions at that time, supported an average of 14 breeding pairs of little terns, which represents 0.7% of the GB breeding population. However, it was considered appropriate to retain the original baseline population of 35 pairs. This decision was partly a reflection of the fact that little tern was the only species for which the Dungeness to Pett Level SPA was classified in 1999 where the entire population remained within the re-classified boundary of the Dungeness, Romney Marsh and Rye Bay SPA as set out in Natural England (2010).

All breeding little terns during the period 2004 to 2008 nested at Rye Harbour LNR. However, in 2008 and 2009 no little terns nested at Rye Harbour (Lewis Yates 2014). Lewis

Yates (2014) reports that since then, management at Rye Harbour has become more active with decoys and sound recordings prompting the re-colonisation in 2010 after two years of no activity. This promoted colony formation within permanent electric fences, away from the foreshore which is exposed to high tides, the full force of sea winds, predation (by fox and badger) and disturbance, eg. by the public and dogs. The number of pairs of little terns at Rye Harbour during a recent 5-year period (2011-2015) were; 7 (2011), 13 (2012), 11 (2013), 10 (2014) and 11 (2015). This provides a recent 5-year mean of 10 pairs (or 20 breeding adults). This represents 0.53% of the GB breeding population. The current population no longer meets the threshold for inclusion as a qualifying feature. Nonetheless, given the likely significant role of disturbance and predation in the decline of this colony, and the scope to manage such activities on a site-specific basis, it is appropriate to retain the species as a feature of the pSPA and to retain the level of ambition for it, as defined by the size of the population for which the Dungeness to Pett Level SPA was first classified in 1999 i.e. 35 pairs (5 year mean breeding population between 1992 and 1996).

The principal feeding grounds of little terns that nest at Rye Harbour are considered to lie entirely in marine areas of Rye Bay, extending up to 3.9 km to the west and east of Rye Harbour and up to 2.2km offshore.

6. Comparison with other sites in the UK

A comparison is presented in Table 2 of the populations of each of the breeding tern populations which are qualifying features of the Dungeness, Romney Marsh and Rye Bay pSPA with the largest breeding populations supported by individual SPAs across Great Britain. 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).

The equivalent comparisons, albeit against other SPAs in England alone, for all of the other features of the existing Dungeness, Romney Marsh and Rye Bay SPA, and hence of the pSPA, are presented in Table 11 of the departmental brief published in 2010 (Natural England 2010) and are for the sake of brevity, not repeated here.

Table 2. Comparison of the numbers of individuals (and pairs) of each of the named tern features of the Dungeness, Romney Marsh and Rye Bay pSPA (based on colony counts since 2011 at contributing source colonies in the case of Sandwich tern and Common tern and between 1992 and 1996 in the case of Little tern) with numbers at other SPAs for which figures are provided in Stroud *et al.* (2001).

Species	Site	Individuals (pairs) [6]	Rank[7,8]	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	
	Dungeness, Romney Marsh and Rye Bay	840 (420) (2011-2015)	9th of 17	
Common tern *Sterna hirundo*	Firth of Forth Islands	1,600 (800)	1st of 22	
	Coquet Island	1,480 (740)	2nd of 22	
	Strangford Lough	1,206 (603)	3rd of 22	
	Glas Eileanan	1,060 (530)	4th of 22	
	North Norfolk Coast	920 (460)	5th of 22	
	Carlingford Lough	678 (339)	6th of 22	
	Inner Moray Firth	620 (310)	7th of 22	
	Cromarty Firth	588 (294)	8th of 22	
	The Dee Estuary	554 (277)	9th of 22	
	Solent and Southampton Water	534 (267)	10th of 22	
	Ythan Estuary, Sands of Forvie and Meikle Loch	530 (265)	11th of 22	
	Farne Islands	460 (230)	12th of 22	
	Foulness	440 (220)	13th of 22	
	Monach Isles	388 (194)	14th of 22	
	Ynys Feurig, Cemlyn Bay and The Skerries	378 (189)	15th of 22	
	Dungeness, Romney Marsh and Rye Bay	376 (188) (2011 – 2015)	16th of 22	
Little tern *Sternula albifrons*	North Norfolk Coast	754 (377)	1st of 27	
	Great Yarmouth North Denes	440 (220)	2nd of 27	
	Chichester and Langstone Harbours	200 (100)	3rd of 27	
	Humber Flats, Marshes and Coast	126 (63)	4th of 27	
	The Dee Estuary	112 (56)	5th of 27	
	Chesil Beach and The Fleet	110 (55)	6th of 27	

[6] 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

[7] Note that these rankings should only be considered indicative of the relative importance of the pSPA as they are based on comparison of more recent 5 year mean populations of both Sandwich and common tern at the Dungeness, Romney Marsh and Rye Bay pSPA (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). In the case of little terns, the numbers within the pSPA and other sites listed are comparable as the figure used for the pSPA is the same as that used in Stroud *et* al. (2001) i.e. 35 pairs. The number of sites ranked is based on the number of sites listed for each species in Stroud *et al.* (2001). Where Dungeness to Pett Level SPA was included in that list the number of sites is as tabulated in Stroud *et al.* (2001), otherwise one has been added to allow for inclusion of the new pSPA in the rank order.

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

	Hamford Water	110 (55)	7th of 27	
	Benacre to Easton Bavents	106 (53)	7th of 27	
	Solent and Southampton Water	98 (49)	9th of 27	
	Alde-Ore Estuary	96 (48)	10th of 27	
	Firth of Tay and Eden	88 (44)	11th of 27	
	Ythan estuary, Sands of Forvie and Meikle Loch	82 (41)	12th of 27	
	Northumbria Coast	80 (40)	13th of 27	
	Colne Estuary	76 (38)	14th of 27	
	Lindisfarne	76 (38)	14th of 27	
	Teesmouth and Cleveland Coast	74 (37)	16th of 27	
	Blackwater Estuary	72 (36)	17th of 27	
	Dungeness, Romney Marsh and Rye Bay	70 (35) (1992-1996)	18th of 27	

7. Conclusion

The extension to the Dungeness, Romney Marsh and Rye Bay SPA to include additional marine areas within the Dungeness, Romney Marsh and Rye Bay pSPA is proposed to protect the important areas used for feeding by the breeding little tern, common tern and Sandwich tern populations which are qualifying features of the existing SPA.

An assessment has been made of the evidence used against Natural England's Evidence Standards (Appendix 7), and Natural England is confident that the evidence is sufficient to support the pSPA being taken forward to formal consultation.

7.1. Qualification

The numbers of one of the three populations of breeding terns which are qualifying features found within the pSPA rank within the top ten sites proposed for this species across the UK SPA suite.

Sandwich tern and common tern occur in numbers (420 pairs and 188 pairs respectively) greater than 1% of UK population. These species therefore qualify for protection under Stage 1.1 of the UK SPA Selection Guidelines (Stroud *et al*., 2001). At the time of the original classification of the Dungeness to Pett Level SPA, little tern occurred in numbers (35 pairs) greater than 1% of UK population. This species therefore qualified for protection under Stage 1.1 of the UK SPA Selection Guidelines (Stroud *et al*., 2001) at that time. Although the size of this population has declined since then and it no longer exceeds 1% of the UK population, the species is retained as a qualifying feature of the pSPA as is the level of ambition defined by its population size at the time of the classification of the original Dungeness to Pett Level SPA i.e. 35 pairs.

7.2. Population estimation and identification of areas

Populations of the tern species are taken from breeding colony data provided by site managers and/or obtained from the JNCC Seabird Monitoring Programme (SMP).

No site-specific survey data regarding the distribution at sea of any of the tern species originating from the Dungeness, Romney Marsh and Rye Bay SPA were collected as part of the JNCC programme of work on little terns or larger terns. However, tracking data gathered over a period of up to three years and from numerous tern colonies around the UK were used to construct generic models that generated predictions of the most important foraging areas for Sandwich and common tern from the Dungeness, Romney Marsh and Rye Bay SPA. Alongshore and seaward limits to important foraging areas for little terns originating from the colony at Rye Harbour were based on average values derived from field studies at numerous little tern colonies around the UK. The general reliability of boundaries based on the predictions of such generic models has been provided by a programme of verification surveys around England in 2015 (ECON 2015) and, in the case of Rye Bay in particular, by the results of the tern survey work conducted in Rye Bay in 2014 (Lewis Yates 2014).

7.3. Boundary

The proposed boundary of the pSPA is a composite of all of the areas identified as important to each species through, in the case of the two larger tern species, application of Maximum Curvature Analysis (MCA) to modelled usage maps and, in the case of little terns, the use of alongshore and seaward limits to little tern activity observed at other colonies.

The pSPA boundary encompasses the important areas identified for Sandwich tern, common tern and little tern. Where the pSPA overlaps the existing SPA, the landward boundary of the pSPA remains the same as that of the existing SPA. Where new stretches of coast have been included in the pSPA, the landward boundary follows MHW. The seaward boundary of the pSPA extends offshore beyond that of the current SPA and is determined by the distribution of Sandwich tern and common tern. The at sea distribution of little tern is contained within the composite seaward boundary determined by usage patterns of the larger tern species.

Care has been taken to establish a boundary around these areas to provide sufficient protection for each qualifying feature, whilst ensuring that where practical areas that do not meet the MCA usage threshold for any qualifying feature were excluded.

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Appendix 1

Please refer to accompanying site maps

Appendix 2 Site Citation

Directive 2009/147/EC on the Conservation of Wild Birds Special Protection Area (SPA)

Name: Dungeness, Romney Marsh and Rye Bay SPA

Counties/Unitary Authorities: Kent County Council, East Sussex County Council

Boundary of the SPA:

The current seaward boundary of the SPA is mostly drawn to Lowest Astronomical Tide (LAT) to include all areas of intertidal habitat that occur between its westernmost and northernmost coastal points. The proposed marine extension will extend the SPA out to sea to protect important foraging areas for little, common and Sandwich terns. The new seaward boundary of the site after the proposed marine extension will reach, at its furthest, approximately 9km out to sea at Rye Harbour from the seaward boundary of the existing SPA. The alongshore extent of the proposed marine extension extends further to the west and north than the limits to the current SPA. The westerly most point of the marine extension is Norman's Bay just west of Bexhill. The northern most point of the marine extension lies just south of Hythe. The landward boundary of the existing SPA follows the SSSI boundary and the proposed marine extension will make no change to this in those places where the SPA already exists. Along stretches of coast not included within the existing SPA boundary but now encompassed within the proposed marine extension the landward boundary will follow Mean High Water (MHW).

Size of SPA: The area already classified within Dungeness, Romney Marsh and Rye Bay SPA is 4,010.29 ha and the proposed SPA marine extension area is 30,364.13 ha. Therefore, the total area of the SPA with the proposed marine extension is 34374.42 ha.

Site description: Dungeness, Romney Marsh and Rye Bay SPA is located on the south coast of England between Hythe in Kent crossing the county border of East Sussex to Norman's Bay. This is a large area with a diverse coastal and marine landscape comprising a number of habitats, which appear to be unrelated to each other. However, all of them exist today because coastal process have formed and continue to shape a barrier of extensive shingle beaches and sand dunes across an area of intertidal mud and sand flats and broadscale marine habitats.

The site includes the largest and most diverse area of shingle beach in Britain, with low-lying hollows in the shingle providing nationally important saline lagoons, natural freshwater pits and basin fens. Rivers draining the Weald to the north were diverted by the barrier beaches, creating a sheltered saltmarsh and mudflat environment, which was gradually infilled by sedimentation, and then reclaimed on a piecemeal basis by man. Today this area is still fringed by important intertidal habitats, and contains relict areas of saltmarsh, extensive grazing marshes and reedbeds. Human activities have further modified the site, Dungeness, Romney marsh and Rye Bay is important for breeding and wintering waterbirds, birds of prey, passage warblers and breeding seabirds.

The site includes a diverse range of broadscale habitats within the marine environment which support a variety of prey species for the foraging seabirds. These habitats include the following; sub-tidal and intertidal sand and muddy sand, sub-tidal biogenic reeds, coarse and mixed sediments and moderate energy infralittoral and circalittoral rock.

Qualifying species:

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

Species	**Count (period)**	**% of subspecies or population**	**Interest type**
Qualifying features with revised counts			
Common tern *Sterna hirundo*	188 pairs - breeding (5 year mean 2011-2015)	1.9 % of GB population	Annex 1
Sandwich tern *Sterna albifrons*	420 pairs - breeding (5 year mean 2011-2015)	3.8 % of GB population	Annex 1
Qualifying features with counts remaining as at 2016 classification using data in Departmental Brief published in 2010			
Avocet *Recurvirostra avosetta*	31 pairs – breeding (5 year mean 2004-2008)	3.5% of GB population	Annex 1
Bewick's swan Cygnus columbianus bewickii	155 individuals – wintering (5 year peak mean 2002/3 – 2006/7)	1.9% of GB population	Annex 1
Bittern Botaurus stellaris	5 individuals – wintering (5 year peak mean 2002/3 – 2006/7)	5.0% of GB population	Annex 1
Hen Harrier *Circus cyaneus*	11 individuals – wintering (5 year peak mean 2002/3 – 2006/7)	1.5% of GB population	Annex 1
Golden Plover *Pluvialis apricaria*	4,050 individuals – wintering (5 year peak mean 2002/3 – 2006/7)	1.6% of GB population	Annex 1
Little tern *Sternula albifrons*	35 pairs – breeding (5 year mean 1992-1996)	1.5% of GB population	Annex 1
Ruff *Philomachus pugnax*	51 individuals – wintering (5 year peak mean 2000/01-2004/5)	7.3% of GB population	Annex 1
Aquatic warbler *Acrocephalus paludicola*	2 individuals – passage (5 year mean 2004-2008)	6.1% of GB population	Annex 1
Marsh harrier *Circus aeruginosus*	4 females – breeding (5 year mean 2004-2008)	2% of GB population	Annex 1

The site qualifies under **article 4.2** of the Directive (79/409/EEC) as it is used regularly by 1% or more of the biogeographical populations of the following regularly occurring migratory species (other than those listed in Annex I) in any season:

Species	**Count (period)**	**% of subspecies or population**	**Interest type**
Shoveler *Anas clypeata*	485 individuals – wintering (5 year peak mean 2002/3 – 2006/7)	1.2% NW & C Europe (non-breeding)	Migratory

Assemblage qualification:

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

During the period 2002/03 – 2006/07, Dungeness, Romney Marsh and Rye Bay SPA (including proposed extensions) supported an average peak of 34,625 individual waterbirds in the non-breeding season, comprised of almost 16,000 wildfowl and over 19,000 waders. This assemblage is of both European and international importance. In the context of SPA qualification the assemblage includes the wintering and passage species of European importance described above (i.e. Bewick's swan, bittern, Hen harrier, golden plover, ruff, aquatic warbler and shoveler), as well as species whose numbers exceed 1% of the GB wintering or passage populations i.e.: European white-fronted goose *Anser albifrons albifrons*, wigeon *Anas penelope*, gadwall *A. strepera*, pochard *Aythya ferina*, little grebe *Tachybaptus ruficollis*, great crested grebe *Podiceps cristatus*, cormorant *Phalacrocorax carbo*, coot *Fulica atra*, sanderling *Calidris alba*, whimbrel *Numenius phaeopus* and common sandpiper *Actitis hypoleucos.* Lapwings *Vanellus vanellus* are also present in sufficient numbers to warrant their being listed as a major component species of the assemblage, since their numbers exceed 2,000 individuals (10% of the minimum qualifying assemblage of 20,000 individuals).

Principal bird data sources:

1) Dungeness Bird Observatory Annual Reports
2) Dungeness RSPB Reserve Records
3) Innogy. 2004. Little Cheyne Court Wind Farm – Ornithological Assessment: update on wintering birds. Report to Npower Renewables Ltd, Kent
4) Kent Bird Reports
5) Marsh Environmental. 2003, 2004 & 2008. Breeding and Wintering Bird Survey of Proposed Wind Farm Area at Little Cheyne Court
6) MoD Lydd Ranges Conservation Group
7) Wetland Trust Records (Pett Level)
8) Romney Marsh Harrier Recording Group
9) Rye Harbour Local Nature Reserve Records
10) Sussex Bird Reports
11) Wetland Bird Survey (WeBS database)
12) A Survey of the Feeding Activity of the Breeding Terns of Rye Bay. Lewis Yates October 2014.

Appendix 3 Sources of bird data[1]

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 Dungeness, Romney Marsh and Rye Bay 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. No visual surveys occurred at Dungeness, Romney Marsh and Rye Bay pSPA. Generic ranges were instead used.	Verification by JNCC and external peer review of final report
Site managers	RSPB, Rye Harbour Local Nature Reserve, Wetland Trust	Colony count data at Dungeness, Romney Marsh and Rye Bay for Sandwich terns, Common terns and Little terns.	Various dates since 1970. 2008-2015 full data sets.	Standard methodology	Verified by site manager and JNCC (when entered to SMP)
Seabird Monitoring Programme	Site managers and JNCC	Annual counts of tern colonies		Standard methodology	Verified by JNCC

[1] The data sources listed in this table relate only to the numbers of terns and the identification of the foraging areas. All other sources of data that support the inclusion of all of the other qualifying features of the Dungeness, Romney Marsh and Rye Bay SPA (and which are retained as features of the pSPA (and as listed in Table 1, and Appendix 2) are presented in Annex 1 to the departmental brief published in 2010 (Natural England 2010) and, for the sake of brevity, are not repeated here.

Appendix 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 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 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[8] 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 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

[8] '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.

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.

2.1. 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) <u>Seaward extent</u>

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) <u>Alongshore extent</u>

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) <u>Seaward extent</u>

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

B) <u>Alongshore extent</u>

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	Mean of maximum seaward

	per survey (m)	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

[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.

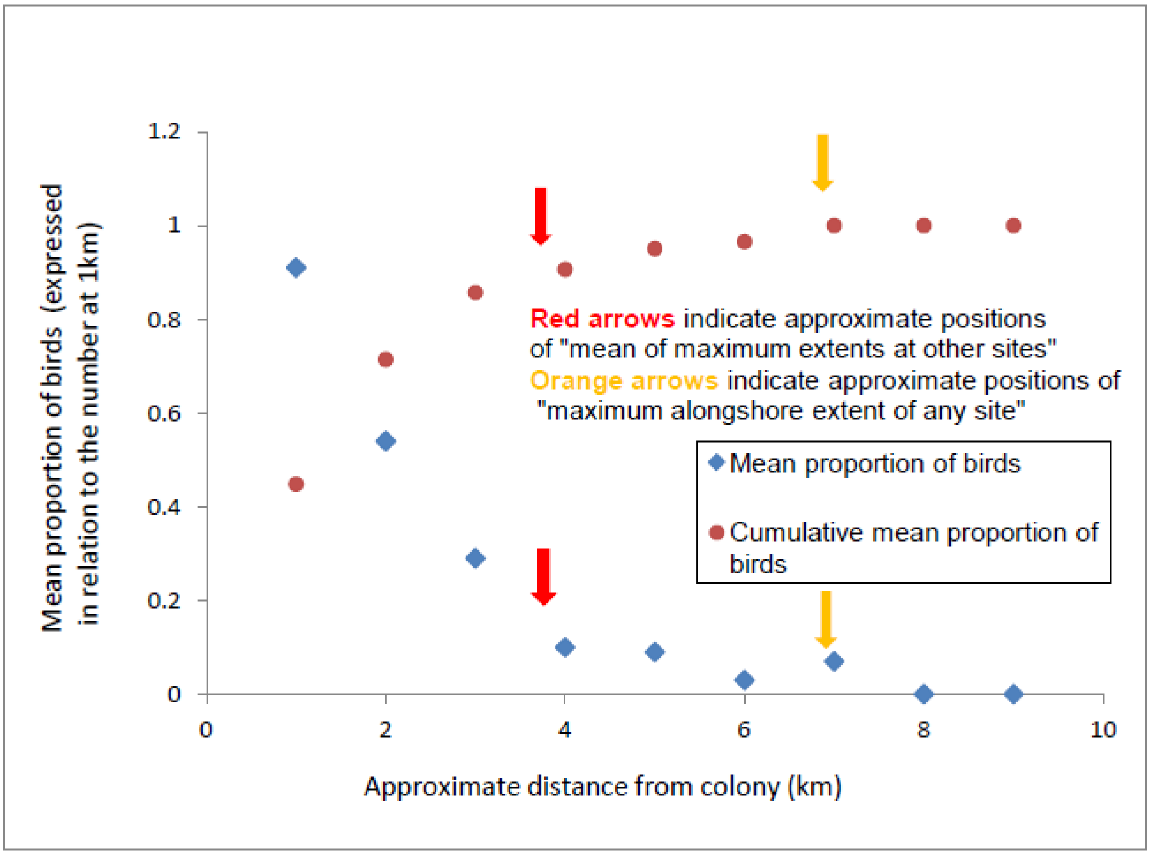

Figure 1: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing distances alongshore from the colony. Each blue point represents the mean proportional usage at each distance band from the colony averaged across colonies. The proportion at each distance (blue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

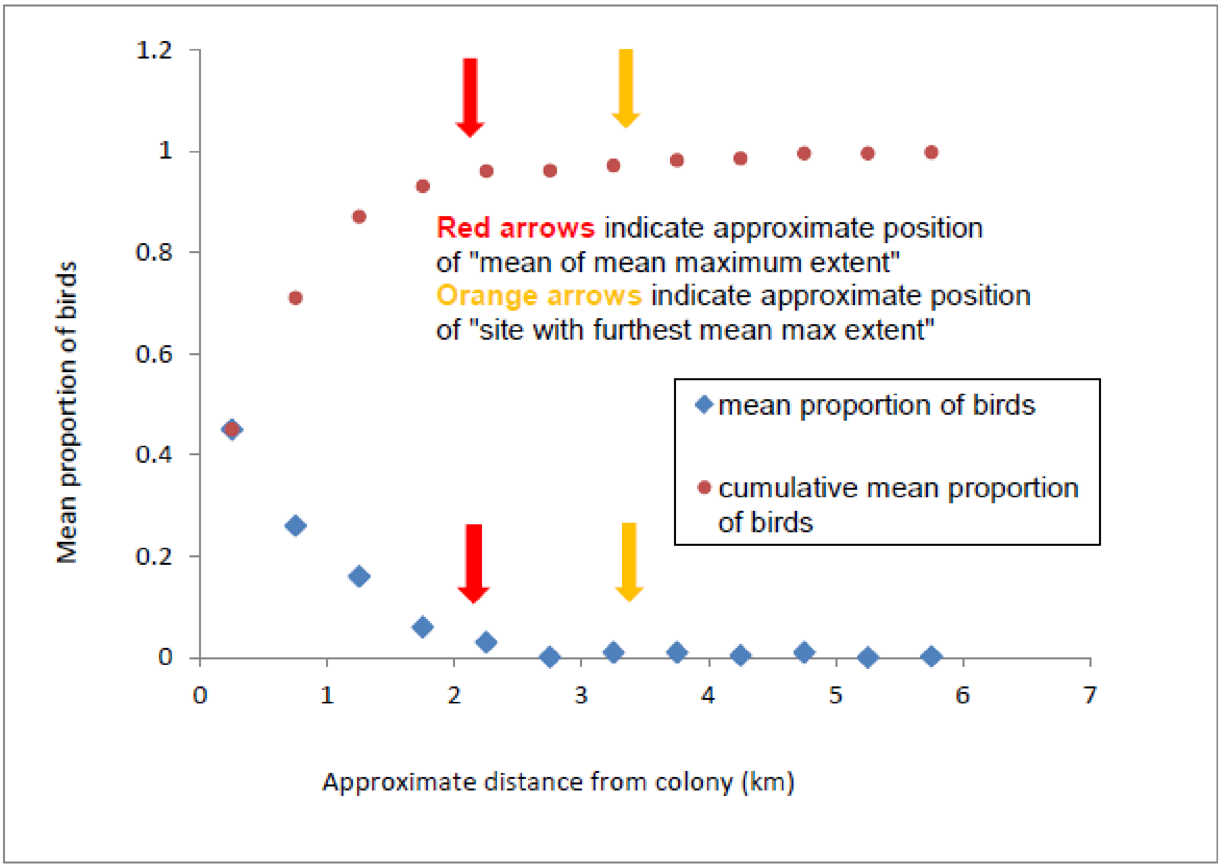

Figure 2: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing seaward distances from MHW mark. Each blue point represents the mean proportional usage at each distance band from MHW mark averaged across colonies. The red arrows indicate the values at the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

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

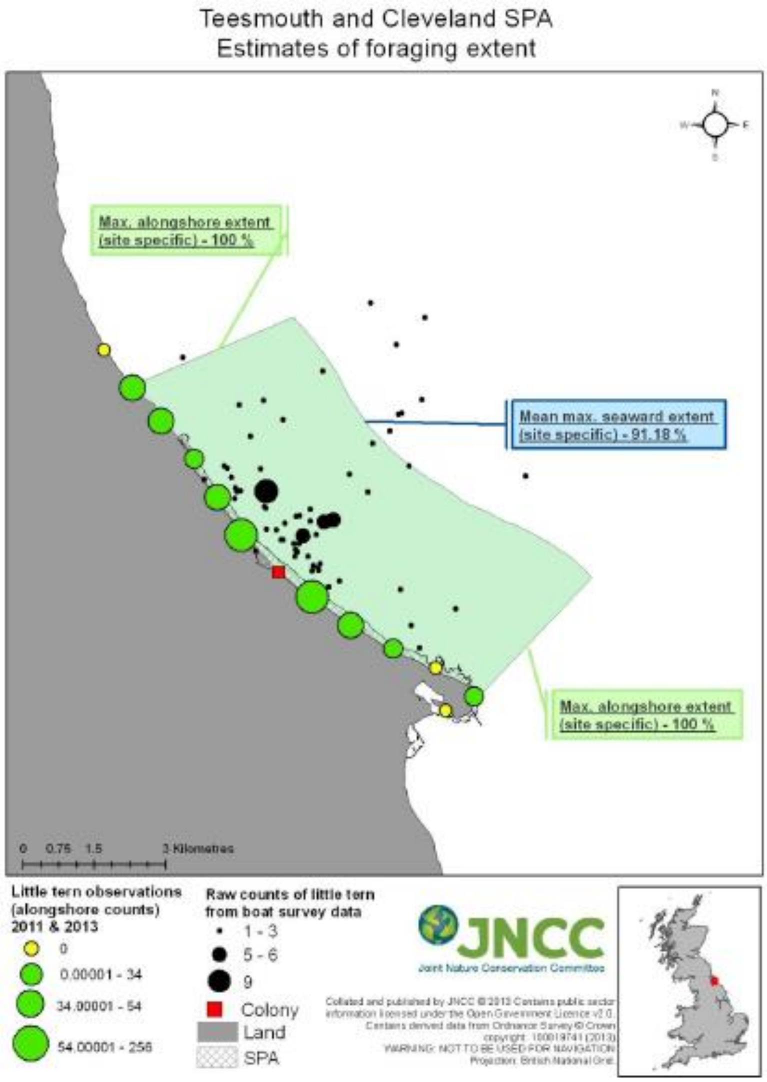

Figure 3. An example of the application of site specific alongshore and site specific seaward extents to define the boundaries to little tern foraging areas at the Teesmouth and Cleveland SPA. The % values given in the labels indicate the site specific % of little tern observations within the shore-based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

6. References

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Appendix 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 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)[9].

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[10] (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.

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[9] 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.

[10] 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.

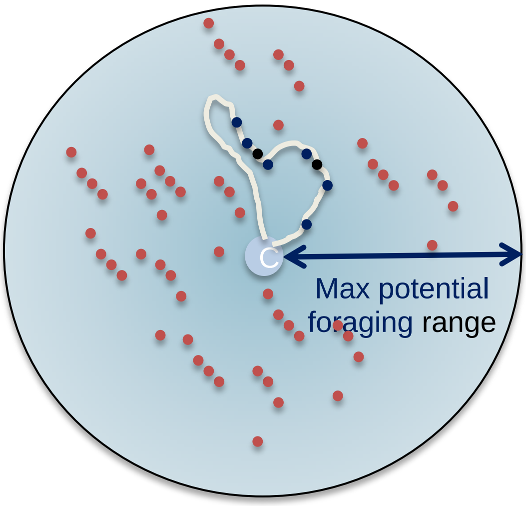
3. Data preparation and analysis

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[11] 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.

[11] 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[12], 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).

[12] 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.

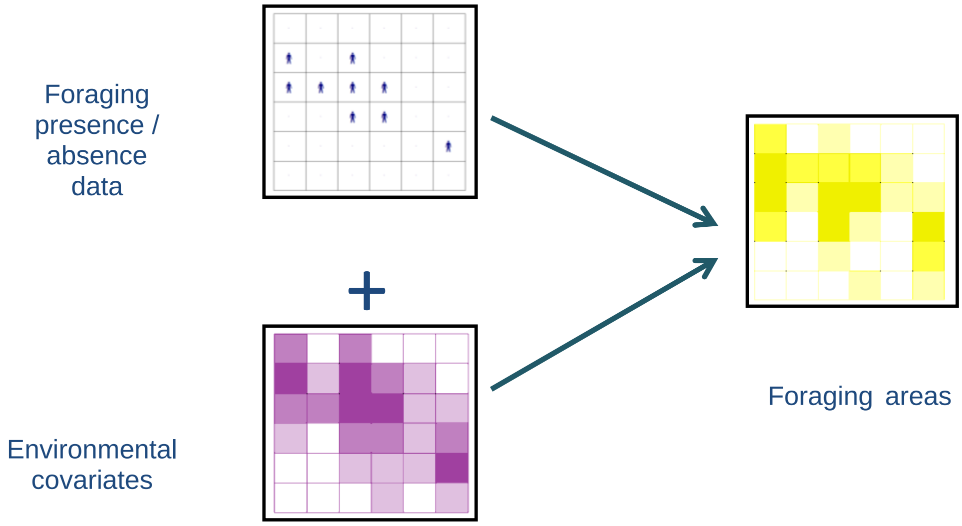


Figure 1. Process of modelling distributions based on environmental information, using a single covariate distribution map in the example.

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.

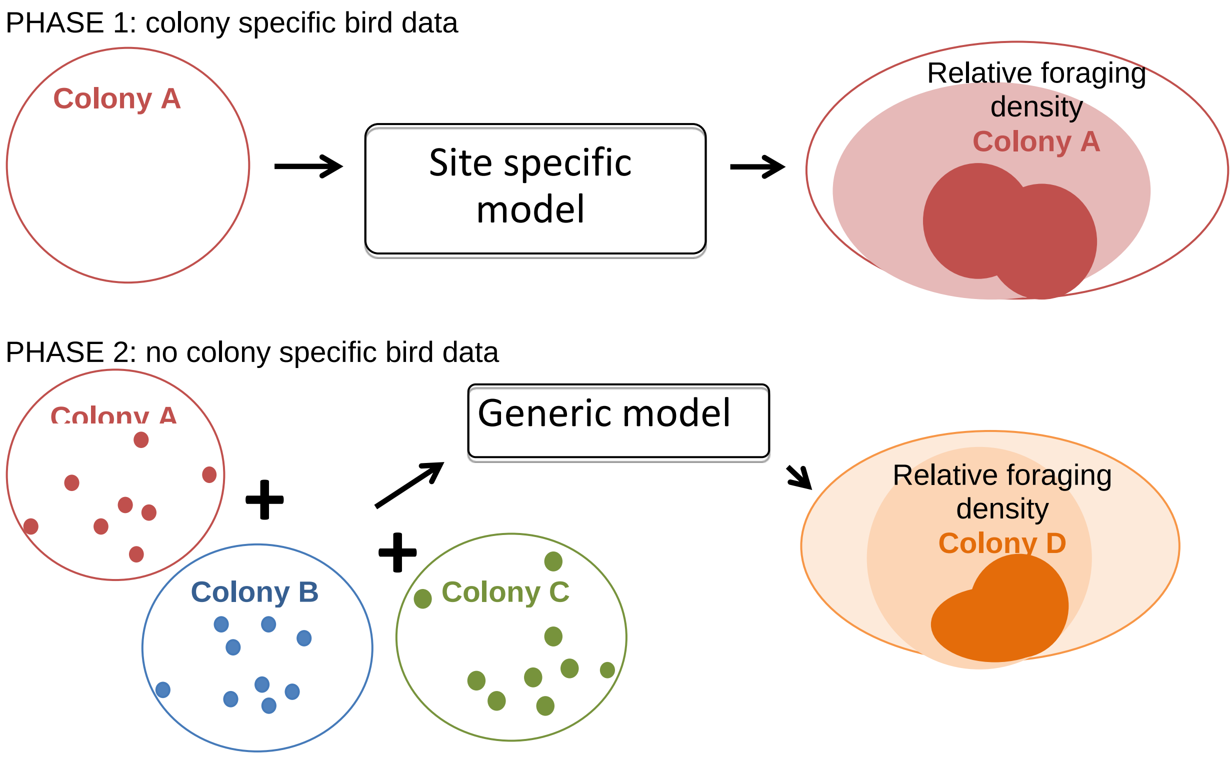


Figure 2. Simplified, schematic representation of the process whereby empirical observations of tern foraging locations around a colony were either: used to build predictive, site-specific models of tern usage that generated relative foraging density maps around that colony (phase 1 analyses); or combined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

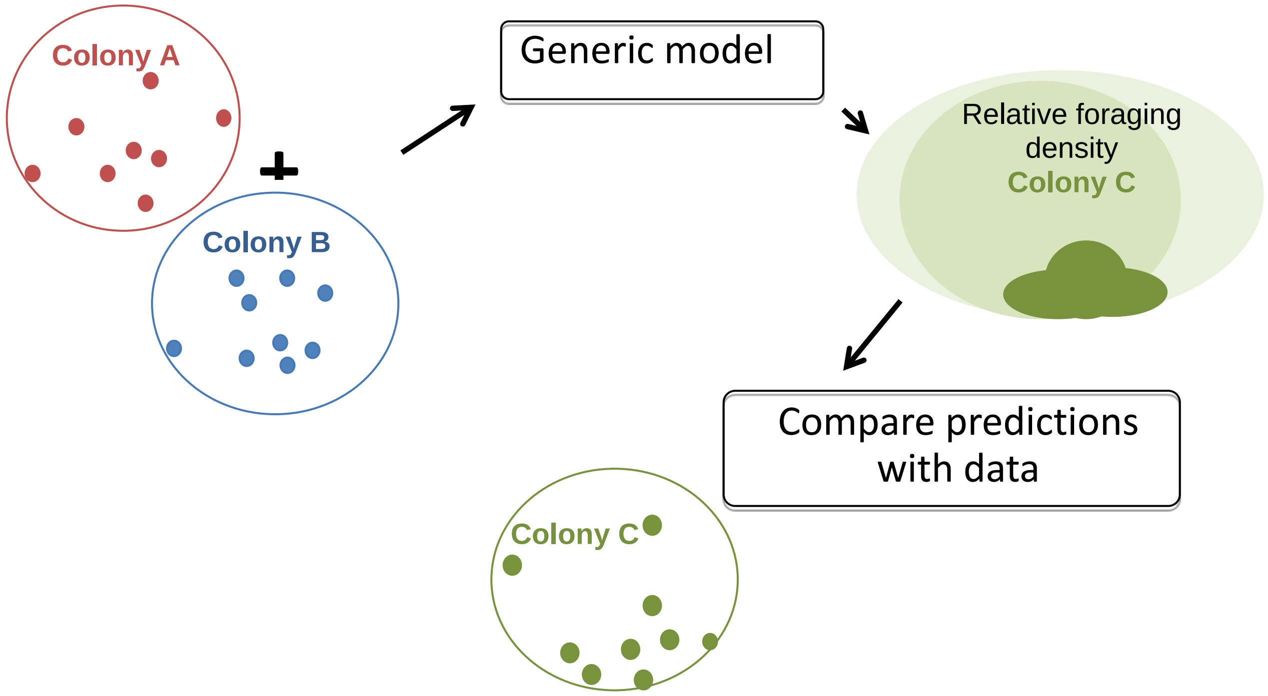


Figure 3. Schematic representation of the cross-validation process, using an example where we have data for three colonies A, B and C, of which data from two at a time (A and B in this diagram) are used to build a predictive model, the predictions of which are then tested by comparison with empirical 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[13], 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.

[13] 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.

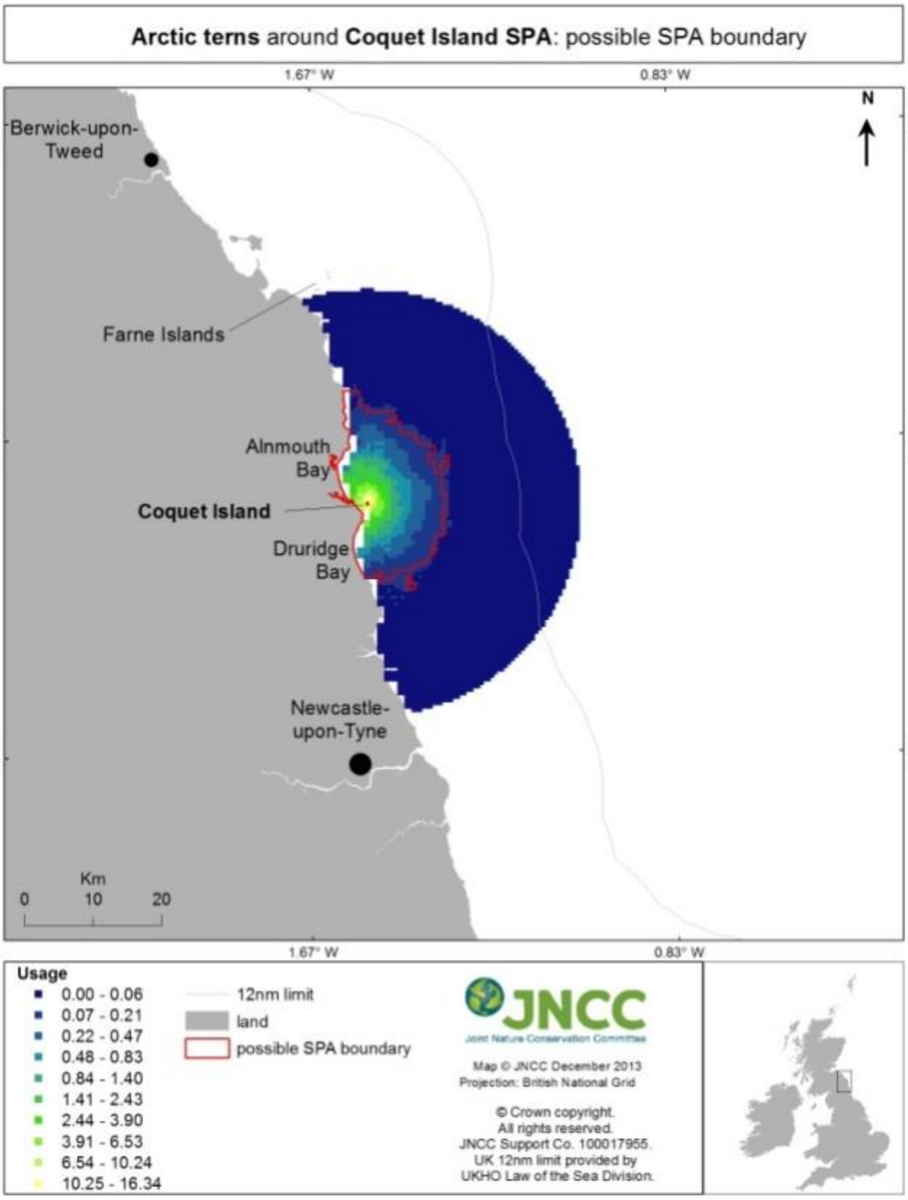


Figure 4 Maximum curvature derived boundary (red line) overlaid on map of model predictions of usage by Arctic terns around Coquet Island. The extent of the dark blue circle of model predictions of usage is 30km - the global mean maximum distance to colony, calculated using tracking data held by JNCC; 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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Appendix 6. Summary of the at-sea distribution of tern activity recorded in Rye Bay in 2014.

1. Introduction

Lewis Yates (2014) carried out visual observations of tern foraging activity and passage rates from a series of shore-based observation locations between Fairlight (TQ 8786 1160) and Greatstone (TR 084 244), just north of Dungeness (Figure 1). These surveys were conducted between 20th March and 22nd August 2014. Inland waters were also observed in this study but the results of that element of the work are not discussed here as the focus of this departmental brief is on the tern usage of the marine environment.

Over 52 separate days of observations, a total of 1056 observations of tern numbers were made. Each observation comprised a 10 minute period during which one of the three tern species was the focus of attention. The 10 minute observation periods were split between 373 Sandwich Tern observations, 575 Common Tern observations and 121 Little Tern observations. These observations included a total of 6179 Sandwich Terns (non-unique individuals) where 247 were recorded as fishing and 1127 were recorded as passing through (the remainder being at roost or at the nest), 4332 Common Terns (1295 fishing and 1004 passing through) and 436 Little Terns (124 fishing and 143 passing through). Note that in the following figures, the low values of tern activity recorded on the y axis reflect the units in which activity has been recorded i.e. birds per hectare per observation. Therefore, to a degree, the low absolute values reflect the short duration of each observation period i.e. 10 minutes. It is not these absolute values which should be considered when comparing the distribution of tern activity recorded in 2014 with that predicted by the models developed by JNCC because: i) the models predict values of relative usage, not absolute tern abundance and ii) what is more of interest is the west-east range of stations at which terns were seen in 2014 and the **relative** levels of activity across that range.

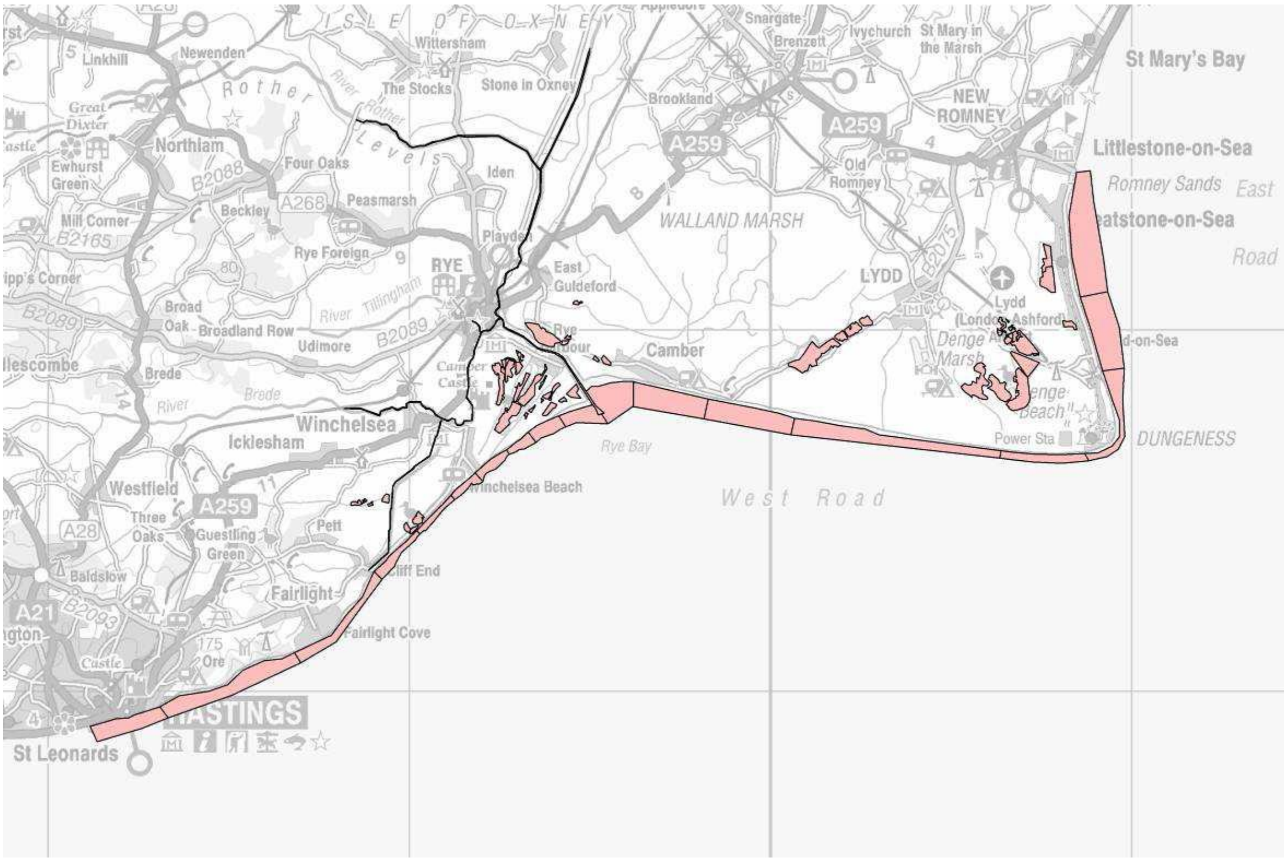

Figure 1. Map of the plots in which observations of tern activity were undertaken in 2014. S1 – S18 are the sectors along the shoreline running from St Leonards in the west to Greatstone-on-Sea in

the east. The River Rother was split into five sections, with ascending numbers R1-R5 indicating increased distance from the sea. (Source: Lewis Yates 2014).

2. Little tern

Little tern foraging activity and passage was recorded at the nesting colony in 2014 i.e. Flat Beach (FB), in the lowest reaches of the River Rother (R1), on the coast immediately on either side of the mouth of the River Rother (S9 and S10) and in the coastal sectors to the west (S6-S8) and east (S11 & S12) (Figure 2). The centre points of sectors S6 and S11 lie approximately 4km to the west and east of the mouth of the River Rother. This restricted range along the coastline is in close agreement with the generic alongshore boundaries derived on the basis of the JNCC little tern work programme i.e. 3.9km.

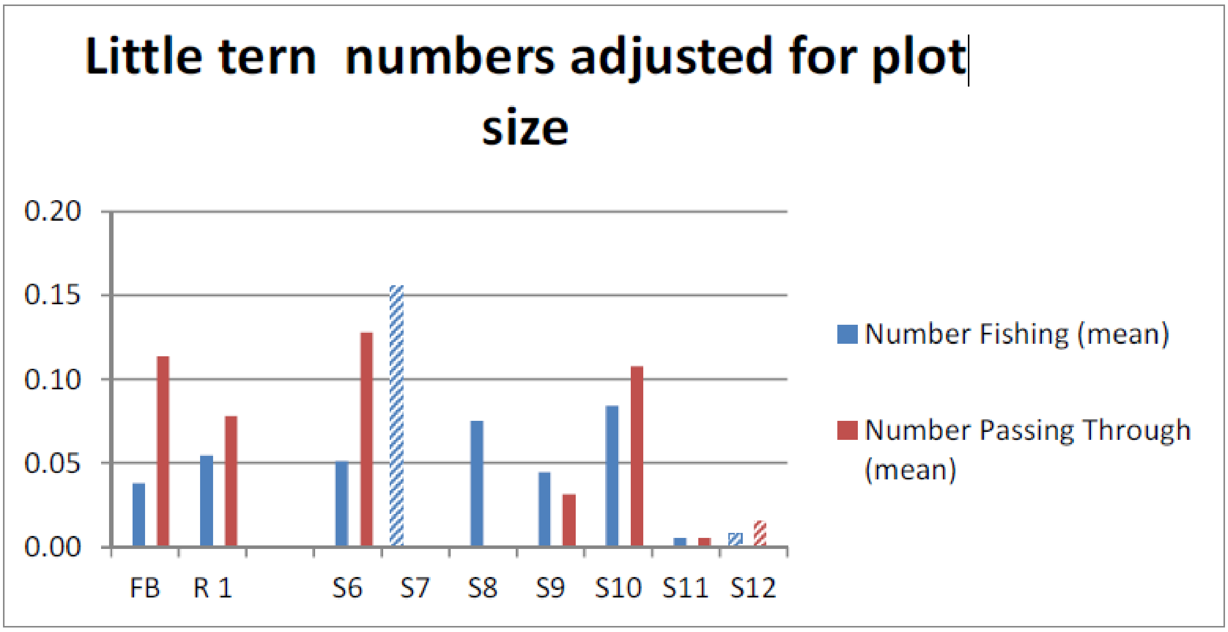

Figure 2. Little tern numbers adjusted for plot size- showing the mean number of little terns fishing and passing through each observation plot (birds per hectare per observation). Plots S7 and S12 had a small number of observations so these averages are unreliable and given for illustrative purposes only (hatched columns). Source: Lewis Yates (2014).

3. Common tern

Common tern foraging activity was recorded along large sections of the coast between S5 in the west (near Winchelsea beach) and S11 to the east ((Camber Sands) and between "The Patch" (S15) and around Dungeness Point (S16) (Figure 3). There was a hotspot of foraging activity at "*The Patch*" i.e. at the Dungeness Power Station outfall (S15-S16). A very few common terns were seen feeding in the western coastal sections of S4 (near Cliff End) and S2 (west of Fairlight Cove). Common terns in transit were also seen along the entire stretch of coast from S5 to S16 with a pronounced peak in birds passing Dungeness Point (S16), but also with a few being seen further north in section S17 (near Lydd-on-Sea) (Figure 3).

The frequent observations of common terns along the coastline between S5 (Winchelsea beach) and S16 (Dungeness Point) are broadly in line with the predicted usage generated by the generic model of common tern distribution based on birds originating from Rye Harbour, Pett Level and from Dungeness itself, as too are the less frequent observations of birds along the frontage from Fairlight Cove to Hastings (see Figure 2). However, it must be acknowledged that the modelled boundary to areas of importance for common tern extends further west than the 2014 observations would suggest. On the other hand, the model correctly predicts the slightly lower levels of tern activity heading east along Camber Sands before an increase towards Dungeness Point. Generic models of the sort used for common tern cannot always be relied upon to identify site-specific hotspots of activity such as at “The Patch” although the boundaries generated by such models invariably include such hotspots within them (Allen & Mellon Environmental Ltd 2015, ECON 2015). In this particular instance however, the model does identify the hotspot of tern feeding activity near “The Patch” and the peak in birds seen in transit in S16 which was probably associated with birds’ usage of “The Patch”. It is the model’s predictions of usage by birds originating from Dungeness that generate this agreement with the empirical sightings in 2014, although in reality birds from both Rye Harbour and Pett Level probably also fly to this particular hotspot too and contribute to the high levels of activity seen in 2014.

It is also clear that the areas predicted by the model to be of relatively high importance to birds nesting on Dungeness (Figure 2 of main report) are predicted to extend further north along the coast from Dungeness Point towards Hythe than observations in 2014 would suggest. However, it must be borne in mind that, although the number of pairs that nested on Dungeness in 2014 was higher than in other recent years (36 pairs); this was still only about one sixth of the total across all three locations in that year (183 pairs). While this uneven distribution of nesting activity in 2014 is probably reflected in the distribution of common tern activity recorded by Lewis Yates (2014) it is not reflected in the 3 maps of predicted relative usage centred on each of the 3 nesting locations. These areas have been combined to yield a composite boundary to areas of relatively high importance to common terns around each of these three nesting locations irrespective of the current levels of relative usage associated with the three separate nesting locations. This is considered to be an appropriate approach given that numbers of common terns at each of the 3 principal nesting locations in use today (Pett Level, Rye Harbour and Dungeness) have changed dramatically over the last 40 or so years, and continue to do so (Lewis Yates 2014). The composite boundary drawn around areas of importance to common terns nesting on all three locations future proofs the boundary to further shifts in distribution of nesting common terns between these three locations.

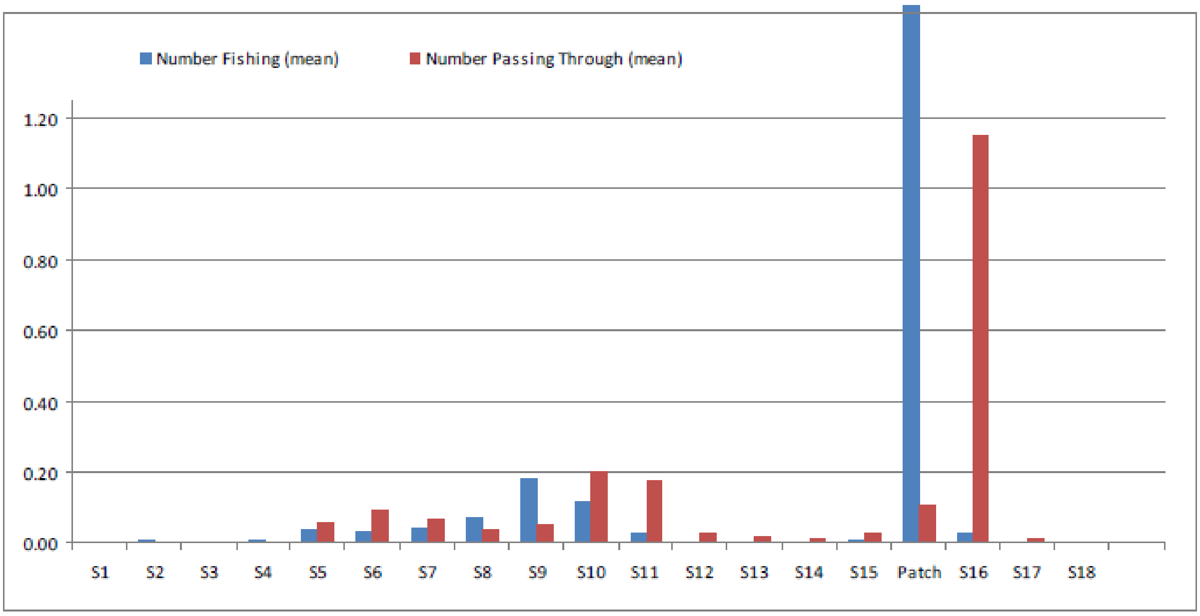

Figure 3. Number of common terns adjusted for plot size, showing number of birds fishing and passing through each marine observational plot (birds per hectare per observation). The Patch (44.5) has a high value due to its small size and regular presence of feeding birds, it is cut off to give a better view of the smaller values. Source: Lewis Yates (2014).

4. Sandwich tern

Sandwich tern foraging activity was recorded at almost all coastal observation sections between S4 in the west (near Cliff End) and S17 in the north-east (near Lydd on Sea) in 2014 (Figure 4).

There was a hotspot of foraging activity at "*The Patch*" i.e. at the Dungeness Power Station outfall (S15-S16). Sandwich terns in transit were also seen along this entire stretch of coast but also further afield in low numbers further to the west (section S3 near Fairlight on Sea) and to the north-east (section S18 near Greatstone-on-Sea) (Figure 4). Sandwich terns in transit were, as expected for a central place forager, most abundant close to the nesting colony location i.e. in the lower reaches of the River Rother (section R1) and at the Rye Harbour Nature Reserve (Quarry). These observations of Sandwich terns all along the coastline between Fairlight-on-Sea and Greatsone-on-Sea (Figure 4) are in close agreement with the boundaries of the proposed pSPA derived on the basis of the JNCC generic model of Sandwich tern foraging distribution (Figures 3 and 4 of main report). It must be acknowledged however, that the boundaries of the pSPA do extend further to the west and to the north-east than sightings of Sandwich terns in 2014. Lewis Yates (2014) state that the survey work was carried out over 52 days of observation. This is a significant survey effort. However, it is unclear from this report whether every section was surveyed on each of these 52 days. If that were not the case and individual sections were sampled on far fewer days, then it may be that in sections at the extreme western and north-eastern limits to the survey area in which Sandwich tern activity might (as predicted by the model) be relatively lower than closer to the colony, their occurrence was simply missed. Indeed, Lewis Yates (2014) note that "occasional observations were made at the easterly end of Hastings seafront but were not maintained over the whole survey as very few terns were seen". It is considered likely that had more observations been made in such areas that Sandwich tern activity would have been detected, confirming the full extent of the model generated proposed boundary. This view is informed by the fact that this proved to be the case when additional field surveys of Sandwich tern foraging activity were undertaken on the Purbeck coast in Dorset in 2015 to verify the westernmost extent of the boundary of the Solent and Dorset Coast pSPA (summary and link to report provided in Appendix 7).

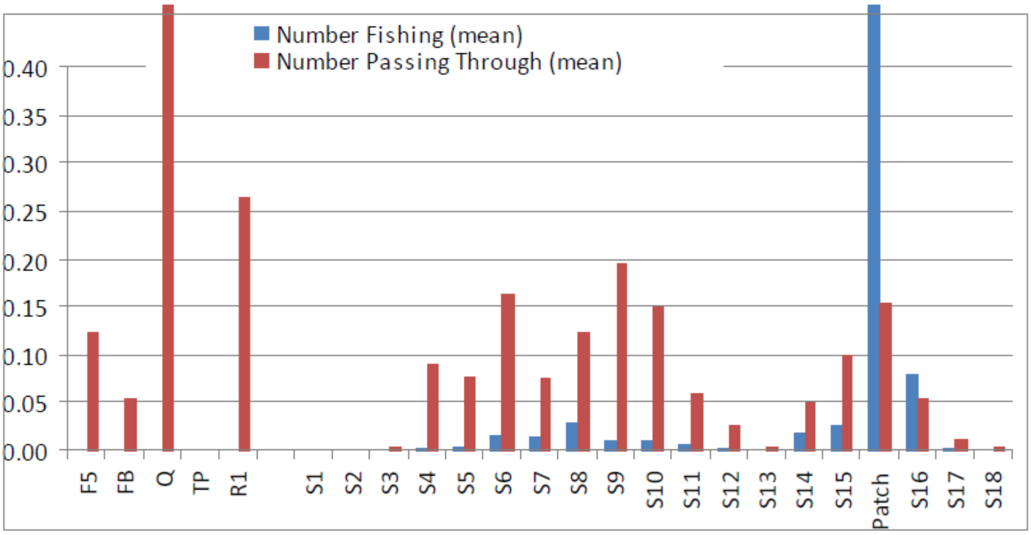

Figure 4: Numbers of Sandwich terns adjusted for plot size, showing numbers of Sandwich terns fishing and passing through each observation plot (birds per hectare per observation). Quarry (1.08) and Patch (2.77) bars are cut off to show the lower values more clearly. These two plots had much higher values due to their small size and regular presence of Sandwich Terns. Source: Lewis Yates (2014).

5. Conclusions

In summary, this work confirmed that little terns are very restricted in their foraging grounds, in line with the generic model proposed on the basis of JNCC's work programme on little terns. Common Tern were found to be much more dispersed and were seen fishing along the whole stretch of coastline, broadly in line with the predictions of the generic model for this species. Lewis Yates (2014) notes that Sandwich Tern were observed to fish much less often, probably due to them fishing far offshore and out of sight from land based observers. Nonetheless, the distribution of sightings of Sandwich terns along the whole stretch of coastline is again broadly in line with the predictions of the generic model for this species.

6. References

Allen & Mellon Environmental Ltd. 2015. *Validation of selected tern foraging areas associated with breeding colony SPAs. Reference Number CT2 (4).* Unpublished report to Department of the Environment Northern Ireland. 53pp.

ECON Ecological Consultancy Ltd (2015) Tern verification surveys for marine sites. Ref: ECM 18246. Report to Natural England. 109pp. *Available at:* https://nepubprod.appspot.com/review/agtzfm5lcHVicHJvZHIWCxIJTkVQdWJQcm9kGICAwPWC4fALDA

Lewis Yates (2014). A Survey of the Feeding Activity of the Breeding Terns of Rye Bay. 43pp.

Appendix 7 Implementation of Evidence standards within Boundary Making decision process

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

Strategic Evidence Standard:

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

Analysis of Evidence Standard:

http://publications.naturalengland.org.uk/publication/7850003?category=3769710

An explanation follows as to how the principles within the *Analysis of Evidence* standard have been applied in defining the set of qualifying features and boundary of the Dungeness, Romney Marsh and Rye Bay pSPA.

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

Quantification of Dungeness, Romney Marsh and Rye Bay pSPA interest feature population sizes.

This is a proposed extension of the existing Dungeness, Romney Marsh and Rye Bay SPA, and with only two exceptions, which are set out in the departmental brief (and covered in the following section), no changes are being proposed to the suite of qualifying features or the notified populations sizes of those features. Accordingly, the evidence on which those original features were identified and populations quantified is not re-considered in this Annex and readers are referred to the departmental brief published in 2010 (Natural England 2010) for that information. Rather, this Annex focuses on the only species for which this departmental brief describes a new notified population size i.e. breeding Sandwich tern and common tern.

The evidence base underpinning the identification of the current population of breeding Sandwich tern and common tern within the pSPA is provided by bird count data from two main sources. These data sources are as follows (see also **Appendix 3):**

1. Data from JNCC's Seabird Monitoring Programme (SMP) (http://jncc.defra.gov.uk/smp/) for the Rye Harbour, Pett Level and Dungeness Sandwich tern and common tern colonies between 2011 and 2015.

2. Data from the Sussex Wildlife Trust and other Rye Harbour site wardens supplemented the SMP data where this was not available.

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 Dungeness, Romney Marsh and Rye Bay 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 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	At sea observations included instantaneous counts at predetermined 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.

	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.	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 the existing SPA.

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*.”

Along stretches of coast which are already contained within the existing SPA, the landward boundary remains unchanged as it lies at or above MHW already. Along additional stretches of coast which are now included within the pSPA, 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 elsewhere in the UK, iii) observations that little terns forage in the intertidal zone elsewhere in the UK and in Rye Bay in particular and iv) to ensure protection of these areas as supporting habitat for tern species within the pSPA.

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 larger 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 (O'Brien *et al.* 2012) since the publication of Webb *et al*. (2003).

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 Dungeness, Romney Marsh and Rye Bay pSPA can be seen to be in accord with the guidelines set out in that document.

Establishment of the extent of the Dungeness, Romney Marsh and Rye Bay pSPA

The extent of and boundary to the Dungeness, Romney Marsh and Rye Bay pSPA is determined by the extent of the model generated predictions of which areas of sea are most heavily used by foraging Sandwich terns and common terns originating from various nesting colonies within the existing SPA. 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 Sandwich and common terns from different nesting locations (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 and common 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 Dungeness, Romney Marsh and Rye Bay 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	Rye Harbour	generic	277	5	11
common tern	Rye Harbour	generic	297	6	11
common tern	Pett level	generic	297	6	11
common tern	Dungeness	generic	297	6	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 can be found at http://jncc.defra.gov.uk/pdf/SAS_Identification_of_important_marine_areas_for_larger_terns, as is a summary of the analysis addressing the adequacy of the sample sizes. Another summary of both of these issues can be found in Appendix 5 of this document.

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 Appendix 4).

The habitat association modelling approach is a novel one which has not been used in defining the extent or boundaries of any classified 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 several pSPAs ie Outer Thames Estuary pSPA and Greater Wash pSPA, 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

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

The conclusions regarding the qualifying tern 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 count data from the principal nesting locations within the existing SPA (SMP data & data from site managers). 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 new coastal stretches included within 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) and Lewis Yates (2014). 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 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 Appendix 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.

In addition to the verification surveys commissioned by the DoENI, Natural England commissioned and participated in a programme of shore-based and boat-based verification work at six sites around England in 2015 (ECON 2015). This work was undertaken to verify the occurrence and level of tern foraging activity at various locations within: Northumberland Marine pSPA, Teesmouth and Cleveland Coast SPA, Hamford Water pSPA, Morecambe & Duddon Estuaries pSPA, Liverpool Bay pSPA and Solent & Dorset Coast pSPA. In doing so, it was hoped that the work would give insights into the reliability of the predictions of generic models of tern usage in general rather than just at the particular sites studied. In summary, the surveys confirmed the presence of foraging larger terns in every area of interest, in many cases birds being seen as far as and in some cases beyond proposed site boundaries. Furthermore, foraging terns were seen far upriver in very small relatively rural settings such as the Rivers Alne, Coquet, Wansbeck and Blyth, as well as in heavily industrialised settings such as the Rivers Tees and Mersey, all of which had been predicted to have levels of usage that were sufficiently great to merit inclusion within proposed site boundaries. In no case was an area that was identified as supporting levels of significant usage not found to support foraging terns. Thus, these surveys, like those conducted in Northern Ireland in the previous year provide confirmation that hotspots of usage near colonies are contained within modelled boundaries and provide no evidence that their alongshore, seaward or upriver extents of the boundaries generated on the basis of generic models of tern usage are in any way excessive.

2.) ***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 breeding populations of each of the species of tern 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. Count data from other colony locations at Rye Harbour and Pett Level are collected and coordinated by an experienced and long-serving site warden at Rye Harbour Local Nature Reserve and experienced Wetland Trust reserve staff at Pett Level, using a standardised method. There is therefore high confidence in the survey data.

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 was 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 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 and verification with independent data, 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, sample-size sufficiency analyses and subsequent collection of independent verification survey data around the UK, and comparison with site-specific survey data gathered in 2014 (Lewis Yates 2014) indicate that proposed alongshore and seaward 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 Appendix 5.

3.) 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 preliminary draft of this departmental brief was drawn up by Andrew Simpkin of Natural England. This was then significantly modified and added to by Dr Richard Caldow (Senior Environmental Specialist in Marine Ornithology) and Catherine Laverick (Lead Adviser with Sussex and Kent Team) 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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