Departmental Brief:

Teesmouth and Cleveland Coast potential Special Protection Area (pSPA) and Ramsar

Natural England

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Table of Contents

Summary

The Teesmouth and Cleveland Coast Special Protection Area (SPA) was first classified in 1995 for its numbers of European importance of breeding little tern *Sternula albifrons*, passage Sandwich tern *Thalasseus sandvicensis*, wintering red knot *Calidris canutus* and passage common redshank *Tringa totanus*, as well as an assemblage of over 20,000 waterbirds. The SPA was updated in 2000 to include additional areas of coastal and wetland habitats important for waterbirds. The SPA maintains its original interest and it is now proposed to further extend the SPA to include at sea foraging areas for breeding little tern and breeding and foraging areas for common tern *Sterna hirundo,* the latter being proposed as a new qualifying feature in the light of recent increases in the size of the breeding population within the site. The inclusion of additional terrestrial habitats within the SPA extension is also proposed and includes wet grassland, saltmarsh, deep and shallow pools and intertidal areas important for other foraging and roosting waterbirds which are features of the existing SPA. In the light of discussions of the UK Special Protection Area and Ramsar Scientific Working Group regarding application of the 'minimum of 50' guideline[1] to the selection of SPAs for non-breeding ruff *Calidris pugnax,* and the relaxation of that guideline in the case of the recently classified Morecambe & Duddon Estuary SPA, non-breeding ruff is proposed as a new qualifying feature of the pSPA. Furthermore, recent population increases within the existing SPA of (breeding) pied avocet *Recurvirostra avosetta*, lead to this species also being proposed as a new qualifying feature of the pSPA.

The boundary of the proposed SPA extension covers an area from Castle Eden Denemouth in the north to Marske-by-the Sea in the south and includes the River Tees up to the Tees Barrage resulting in a revised SPA area of 12,226.28 ha. This increases the area of the existing SPA (1,251.50 ha) by 10,974.78 ha (Annex 1, map 1). The seaward boundary has been drawn to include waters out to around 3.5km from Crimdon Dene, to include the areas of greatest importance to the little terns at that colony, and out to around 6km offshore further south to include the areas of greatest importance to the common terns at the Saltholme colony. The seaward boundary has been drawn as simply as possible to include all of the most important foraging areas for these terns while ensuring only very limited areas of sea considered less important are also included within the seaward boundary.

The existing Teesmouth and Cleveland Coast Ramsar boundary will also be extended to include the additional terrestrial wet grassland, saltmarsh, deep and shallow pools and intertidal areas for breeding and non-breeding waterbirds as proposed for the pSPA. The Ramsar extension will not extend outside of the pSPA extension and will only cover those terrestrial extension areas of the pSPA down to Mean Low Water. The area of the original Ramsar site was 942.56 ha. The extension in 2000 added an area of 304.75 ha, giving a revised Ramsar site area of 1,247.31 ha.

[1] This guideline is described in Stroud *et al* (2001). It has been long-standing practice in the UK to apply this guideline in the context of wintering waterbirds in order to prevent sites being considered for selection as an SPA when, although the numbers of a species listed on Annex 1 of the Birds Directive that are supported by a site exceed the required 1% of GB population threshold usually applied to such species, the numbers in the site are in absolute terms very small i.e. less than 50 individuals and so of no major significance for sustaining viable biogeographical populations of the species. Nonetheless, this guideline has not been applied in the case of several species on the basis that its application would constrain selection of an appropriate SPA suite. The UK SPAR Scientific Working Group considered the applicability of this guideline to non-breeding ruff in 2015 ((UK SPAR Scientific Working Group, Minutes of 3 November 2015 meeting (http://jncc.defra.gov.uk/page-1770)), and the subsequent inclusion of non-breeding ruff as a qualifying feature of the Morecambe Bay & Duddon Estuary SPA in 2017 saw the guideline relaxed in that case. Inclusion of non-breeding ruff as a qualifying feature of that SPA was justified on the grounds of the conclusion of the 3rd SPA review (Stroud *et* al 2016) that the existing SPA suite for the species was considered insufficient in terms of both the population numbers within it, and its geographical range coverage, especially in western Britain. The numerical and range insufficiency of the existing SPA suite for non-breeding ruff also justifies relaxation of the guideline in the present case and inclusion of this species as a qualifying feature of the Teesmouth and Cleveland Coast pSPA. Between 2011/12 and 2015/16 the pSPA, including proposed extensions, supported an average of 19 individuals, which represents 2.4% of the GB non-breeding population, more than twice the number (8 individuals) supported by the recently classified Morecambe Bay & Duddon Estuary SPA and makes the site the 7th most important for the species in the UK. This pSPA also lies to the north of all other sites within the existing suite of SPAs for this species and so extends the range coverage of the species' SPA suite. Accordingly it is proposed to add the species as a feature of the pSPA. Ruff occur at shallow waterbodies across the site, in particular on the pools at RSPB Saltholme.

This latter figure has been re-calculated due to positional accuracy improvements (PAI) applied in 2017 and determined to be 1,253.76 ha. The extension proposal would add an area of 840.24 ha to the site giving a total site area of 2,094.00 ha (Annex 1, map 2).

The Teesmouth and Cleveland Coast potential Special Protected Area (pSPA) including the proposed extensions (terrestrial and seaward) detailed in this Departmental Brief qualifies under Article 4 of the Birds Directive (2009/147/EC) for the following reasons (tabulated in Table 1):

- The site regularly supports more than 1% of the GB breeding populations of three species which are listed in Annex 1 of the EC Birds Directive i.e. pied avocet *Recurvirostra avosetta*, little tern *Sternula albifrons* and common tern *Sterna hirundo.*
- The site regularly supports more than 1% of the GB non-breeding population of one species listed in Annex 1 of the EC Birds Directive i.e. ruff *Calidiris pugnax*,
- The site regularly supports (during passage) more than 1% of the GB population of one species listed in Annex 1 of the EC Birds Directive i.e. Sandwich tern *Thalasseus sandvicensis*.
- The site regularly supports more than 1% of the biogeographical population of two regularly occurring migratory species not listed in Annex 1 of the EC Birds Directive: red knot *Calidris canutus* and common redshank *Tringa totanus*.
- The site regularly supports more than 20,000 waterbirds.

The proposed terrestrial extension to the Teesmouth and Cleveland Coast Ramsar site qualifies for the following reasons:

- The site historically supported 1% of the biogeographical populations of three waterbird species (Sandwich tern, red knot and common redshank)
- The site regularly supports more than 20,000 waterbirds.

Table 1: Summary of qualifying ornithological interest of the Teesmouth and Cleveland Coast pSPA and Ramsar site.

Entries in bold indicate changes from the figures used in the original classification of the SPA in March 2000.

Feature	Count (period)	% of population	Interest type	Selection Criteria	New feature? (Y/N)
Sandwich Tern *Thalasseus sandvicensis*	1,900 individuals[2] (1988-1992)	4.3% GB[3], 1.3% Western Europe/Western Africa[4]	Annex I (non-breeding)	Stage 1.1 (SPA), Criterion 6 (Ramsar)	N
Little tern *Sternula albifrons*	**81 pairs[5] (2010-2014)**	**4.3% GB[6]**	**Annex I (breeding)**	**Stage 1.1**	**N**
Common tern *Sterna hirundo*	**399 pairs[7] (2010-2014)**	**4.0% GB[8]**	**Annex I (breeding)**	**Stage 1.1**	**Y**

[2] Data from: Carter 1993, SPA Departmental Brief; recent average of 134 individuals (WeBS: 2011/12-2015/16) representing 0.3% of GB
[3] Data from: Carter 1993, SPA Departmental Brief. Note: this passage population of 1,900 individuals was expressed as equating to 6.8% of the GB breeding population of Sandwich terns (14,000 pairs) in The Natura 2000 Standard Data Form for this site.
[4] Data from: Ramsar citation March 2000 version 0.4
[5] Data from: Cleveland INCA little tern monitoring reports
[6] Data from: Musgrove *et al.* 2013: 1,900 pairs (2000)
[7] Data from: Cleveland Bird Reports
[8] Data from: Musgrove *et al.* 2013; 10,000 pairs (2000)

Pied avocet *Recurvirostra avosetta*	**18 pairs[9] (2010-2014)**	**1.2% GB[10]**	**Annex I (breeding)**	**Stage 1.1**	**Y**
Ruff *Calidris pugnax*	**19 individuals (2011/12-2015/16)[11]**	**2.4% GB[12]**	**Annex I (non-breeding)**	**Stage 1.1**	**Y**
Red knot *Calidris canutus*	5,509 individuals[13] (1991/92-1995/96)	1.6% NE Canada/ Greenland/Iceland/ UK population[14]	Migratory (winter)	Stage 1.2 (SPA), Criterion 6 (Ramsar)	N
Common redshank *Tringa totanus*	1,648 individuals[15] (1987-1991)	1.1% East Atlantic population[16]	Migratory (passage)	Stage 1.2 (SPA), Criterion 6 (Ramsar)	N

Feature	Count (period)	Average number of individuals	Selection Criteria
Waterbird assemblage	2011/12-2015/16	26,014 individuals (SPA assemblage), 26,786 individuals (Ramsar assemblage)[17]	Stage 1.3 (SPA), Criterion 5 (Ramsar)

1. Assessment of SPA against Selection Guidelines

1.1. Stage 1

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

1) an area used regularly by 1% or more of the Great Britain population of a species listed in Annex I of the Birds Directive (79/409/EEC as amended) in any season;

2) an area 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 used regularly by over 20,000 waterfowl (waterfowl 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.

The Conference of the Contracting Parties to the Ramsar Convention has defined the term 'regularly' as used in the Ramsar site selection criteria, and this definition also applies to the SPA selection guidelines (Stroud *et al.* 2001). A wetland regularly supports a population of a given size if:

- the requisite number of birds is known to have occurred in two-thirds of the seasons for which adequate data are available, the total number of seasons being not less than three; or

[9] Data from: Cleveland Bird Reports. Note: this figure refers to the 'breeding pairs of avocet' over a 5 year mean (2010-2014) which equates to an average of 18 breeding pairs
[10] Data from: Musgrove *et al.* 2013; 1,500 pairs (2006-10)
[11] Data from; WeBs 2011/12-2015/16
[12] Data from: Musgrove *et al*. 2013: 800 individuals
[13] Data from: SPA citation March 2000 version 0.4; recent average of 876 individuals (WeBS: 2011/12-2015/16) representing 0.2% of NE Canada & Greenland/Western Europe population (AEWA 2012)
[14] Data from: Wetlands International 2012; 345,000 individuals 1982-1992
[15] Data from: Carter 1993, SPA Departmental Brief; recent average of 881 individuals (WeBS: 2011/12-2015/16) representing 0.3% of the Iceland & Faroes/Western Europe population (AEWA 2012).
[16] Data from: Carter 1993, SPA Departmental Brief
[17] Greylag goose and mute swan are not included in the SPA assemblage because they are not migratory populations

- the mean of the maxima of those seasons in which the site is internationally important, taken over at least five years, amounts to the required level (means based on three or four years may be based on provisional assessments only).

Teesmouth and Cleveland Coast potential SPA (pSPA) (including the proposed extensions) qualifies under stage 1.1 because it regularly supports greater than 1% of the GB population of five species listed in Annex 1 of the Wild Birds Directive: pied avocet (1.2%), ruff (2.4%), common tern (4%), Sandwich tern (4.3% at original classification) and little tern (4.3%), and under stage 1(2) as it regularly supports more than 1% of the biogeographical populations of two regularly occurring migratory species: red knot (1.6% at original classification) and common redshank (1.1% at original classification). The site also qualifies under stage 1(3) by regularly supporting more than 20,000 waterbirds. The site has not been selected for any species under stage 1(4).

1.2. Stage 2

Under Stage 2 of the SPA selection guidelines, the site is assessed as follows:

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

Feature	Qualification	Assessment
1. Population size & density	✓	The site is 8th most important SPA in the UK for breeding pied avocet, 5th for breeding common tern, 4th for breeding little tern, 7th for non-breeding ruff and, based on historical figures when classified in 2000, 1st for non-breeding Sandwich tern, and 15th and 26th for non-breeding red knot and common redshank respectively. See Section 8 for derivation of ranking scores.
2. Species range	✓	The site is one of the 4 most northerly locations for breeding pied avocet and within the core of the breeding range of common and little tern and the non-breeding ranges of red knot, common redshank, ruff and Sandwich tern.
3. Breeding success	✓	Within the site: Pied, pied avocet has low breeding success with an average productivity of 0.2 fledged juveniles per pair 2010-2014, Common terns have an average productivity (where recorded) of one fledged juvenile per pair 2010-2014 and little terns have variable breeding success ranging from zero to 1.75 fledged juveniles per pair during 2010-2014.
4. History of occupancy	✓	Pied avocets started breeding on the site in 2008 and common and little terns have been breeding regularly on the site since 1986 and 1969 respectively. Red knot, common redshank and ruff have been regularly recorded on the site since at least 1970 and use of the site by passage Sandwich tern has been known of since the 1980s.
5. Multi-species area	✓	Five species listed in Annex I, two regularly occurring migratory species and a waterbird assemblage
6. Severe weather refuge		There is some evidence that the site acts as a cold weather refuge for knot when larger and more exposed intertidal sites in the North Sea basin (e.g. The Wash, The Waddenzee) are subject to prolonged periods of freezing weather.

2. Assessment of Ramsar site interests

The site qualifies as a Wetland of International Importance under the Ramsar Convention because it meets the following criteria[18] :

2.1 Criterion 5

'A wetland should be considered internationally important if it regularly supports 20,000 or more waterbirds.'

During the period 2011/12-2015/16, the Ramsar site supported an average peak of 26,786 individual waterbirds. This total is slightly different from the SPA figure because it includes mute swan *Cygnus olor* and greylag goose *Anser anser*. These species are not included within the SPA total because their populations using the site are not migratory.

The proposed extension to the Ramsar site includes additional terrestrial, wetland and intertidal areas and mirrors the proposed SPA terrestrial extension areas above Mean Low Water.

2.2 Criterion 6

'A wetland should be considered internationally important if it regularly supports 1% of the individuals in a population of one species or subspecies of waterbird.'

Between 1988 and 1992 the Teesmouth and Cleveland Coast Ramsar site supported 1.3% of the Western Europe/Western Africa non-breeding population of Sandwich terns. Between 1991/92 and 1995/96 the Ramsar site supported 1.6% of the NE Canada/Greenland/ Iceland/NW Europe population of non-breeding red knot. Between 1987 and 1991 the Ramsar site supported 1.1% of the East Atlantic population of non-breeding common redshank.

3. Rationale and data underpinning site classification

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

According to Stroud *et al.* (2001), the task of identifying a coherent network of terrestrial sites in the UK was largely complete, comprising at that time some 243 sites of which some include areas used by inshore non-breeding waterbirds, for example in estuaries. However, the JNCC's SPA Selection Guidelines do not review requirements of birds using the wholly marine environment in which many birds access resources that are critical for their survival and reproduction. Johnston *et al*. (2002) described a process consisting of three strands by which SPAs might be identified for marine birds under the Birds Directive *i.e*. the identification of:

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

[18] http://www.ramsar.org/sites/default/files/documents/librarby/manual6-2013-e.pdf

Strand 3: offshore areas used probably for feeding but also for other purposes.

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

Strand 4: other types of SPA http://jncc.defra.gov.uk/page-4184 that would identify some important areas for marine birds that may not be included within the above three categories and will be considered individually

In the intervening years between the publication of the second and third terrestrial SPA reviews (Stroud *et al.* 2001, Stroud *et al*. 2016), many new SPAs have been classified and existing SPAs extended. The total number of classified SPAs in the UK reviewed by Stroud *et al*. (2016) was 270. This total includes many existing SPAs that have been extended into the marine environment to implement conservation measures under Strand 1, and the first three fully marine SPAs identified under Strand 2 i.e. Bae Caerfyrddin/ Carmarthen Bay, Liverpool Bay / Bae Lerpwl and Outer Thames Estuary. Stroud *et al.* (2016) also noted the existence at that time of a further fourteen potential SPAs that were either entirely new potential marine SPAs or potential extensions of existing SPAs into the marine environment – all of which have been identified under one or other of the four work strands listed above. Since the publication of Stroud *et al*. (2016) several of these 14 pSPAs have been formally classified.

To implement conservation measures under Strand 1, the JNCC produced guidance (McSorley *et al.* 2003, 2005, 2006; Reid & Webb 2005) to extend the seaward extent of SPA boundaries from seabird colonies. The seaward extensions of existing boundaries in these cases include waters vital for ensuring that some of the essential ecological requirements of the breeding seabird populations are met (e.g. preening, bathing, displaying and potentially some local foraging). The distance of the extension in these cases is dependent upon the qualifying species breeding within the SPA. However, these generic maintenance 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. Maintenance seaward extensions to the boundaries of existing SPAs have been implemented at 31 sites in Scotland and are under consideration at the Flamborough Head and Filey Coast pSPA (Natural England 2014). However, in line with the recommendations of Reid & Webb (2005) generic maintenance extensions have only been implemented at sites holding certain seabird species, none of which occur as breeding birds within the Teesmouth and Cleveland Coast SPA. Reid & Webb (2005) also 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) as defined by the generic maintenance guidance and conclude that 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 *Thalasseus 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 59 breeding colony SPAs for which at least one species of tern is protected. However, only very recently, have additional important areas for terns foraging at sea been identified and classified as marine SPAs to complement the existing terrestrial suite. Since 2007, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of such areas under the fourth work strand as, given the likely extent of these areas, these cannot be addressed by application of the generic "maintenance" extensions approach and are not covered by the work on identifying inshore non-breeding aggregations or important offshore areas. Since the publication of Stroud *et al.* (2016), three new SPAs have been classified (Northumberland Marine, Morecambe Bay & Duddon Estuary and Anglesey Terns) and a further five (Hamford Water, Dungeness, Romney Marsh & Rye Bay, Outer Thames Estuary, Poole Harbour and Liverpool Bay) extended into the marine environment to protect additional important areas for terns foraging at sea.

This Departmental Brief sets out information supporting the identification of the qualifying features of the Teesmouth and Cleveland Coast pSPA and the definition of its proposed boundaries. Additional marine areas that have been included within the revised boundary are those considered to be most important as the foraging habitat of the breeding tern populations that comprise the qualifying features of this pSPA i.e. the breeding populations of little and common terns. Additional terrestrial areas that have been included within the revised boundary are those known to be important as feeding, roosting or breeding habitat of several of the sites qualifying species or species within the waterbird assemblage.

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). For the purpose of clarity in this and other Departmental Briefs, sites are referred to as “SPA” when referring to the existing classified site. Where reference is made to an entirely new site, or to a proposed extended site, or to a site including new features being proposed, it will be referred to as pSPA since the site (if new), or any additional extent or feature, is not yet fully classified.

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

4. Site status and Boundary

4.1. Description of the existing SPA/Ramsar and boundary

The Teesmouth and Cleveland Coast SPA was first classified in 1995 with an area of 942.56 ha. The site was re-classified in 2000 with a total area of 1,247.31 ha following the inclusion of an extension of 304.75 ha. As part of this current extension proposal, the existing SPA area has been re-calculated due to positional accuracy improvements (PAI) applied in 2017 to be 1,251.50ha. This extension proposal increases the area of existing SPA (1,251.50 ha) by 10,974.78 ha to 12, 226.28 ha (Annex 1, map 1). The original SPA included all or parts of Seal Sands SSSI; Seaton Dunes and Common SSSI; Cowpen Marsh SSSI; Redcar Rocks SSSI; and South Gare and Coatham Sands SSSI. The extended area added in 2000 is within or coincident with the above SSSI boundaries and also included parts of Durham Coast SSSI and all of Tees and Hartlepool Foreshore and Wetlands SSSI.

The Teesmouth and Cleveland Coast Ramsar site was first classified in 1995 with an area of 942.56 ha and extended in 2000 with an area of 1,247.31 ha to include additional wetland areas in line with the existing Teesmouth and Cleveland Coast SPA which was also extended in 2000. As part of this current extension proposal to the pSPA, the area has been increased to be 2,094.00 ha.

The Teesmouth and Cleveland Coast SPA is of European importance because it supports:

Internationally important populations of regularly occurring Annex 1 species.

Species	Population (5 year peak mean)*	% GB population
Little tern *Sternula albifrons*	40 pairs - breeding (1995-1998)	1.7% Great Britain
Sandwich tern	1,900 individuals – passage (1988-	6.8% Great

Thalasseus sandvicensis	1992)	Britain[19]

Internationally important populations of regularly occurring migratory bird species.

Species	Population (5 year peak mean)*	% of population
Red knot *Calidris canutus islandica*	5,509 individuals – wintering (1991/92 – 1995/96)	1.6% NE Canada/ Greenland/ Iceland/UK
Common redshank *Tringa totanus totanus*	1,648 individuals – passage (1987-1991)	1.1% Eastern Atlantic (non-breeding)

An internationally important assemblage of waterfowl.

Importance	Population (5 year peak mean)*
Teesmouth and Cleveland Coast SPA supports large populations of waterfowl.	21,312 individual birds (1991/92 – 1995/96)

*SPA citation (March 2000) held on Register of European marine sites for Great Britain.

Recent count data shows that the site regularly supports qualifying numbers of breeding pied avocet *Recurvirostra avosetta* and common tern *Sterna hirundo* and non-breeding ruff *Calidris pugnax* which are now proposed to be added as new qualifying features of the pSPA.

The seaward boundary of the existing SPA and Ramsar was drawn at the Mean Low Water mark. Consequently, areas of marine habitat lying below this shore-level and which are exploited for resting, roosting or feeding by many of the site's qualifying features and by many component species within the waterbird assemblage, lie outside the current protected site. In particular, the foraging areas of the already qualifying, and proposed new breeding tern features which extend considerably out to sea are not included within the existing site boundary. Thus, a potential seaward extension to the SPA is now proposed to include within the site the sea areas identified as being most important to support foraging terns during the breeding season. The seaward boundary of the seaward extension lies several kilometres out to sea. The landward boundary of the seaward extension has been drawn to the Mean High Water mark as defined by the foraging behaviour of breeding common tern. Furthermore, it is proposed to extend the landward boundary of the SPA and Ramsar to include certain terrestrial areas which are also considered to be of importance to breeding common terns, breeding pied avocet and existing features of the SPA. The existing Ramsar boundary is the same as the existing Teesmouth and Cleveland Coast SPA boundary and the proposed Ramsar extension areas are included within the proposed extension to the pSPA (Annex I, Map 3).

4.2. Description of the pSPA, Ramsar extension and boundaries

4.2.1 Seaward boundary of the pSPA

The proposed extension to the area of the existing Teesmouth and Cleveland Coast SPA is approximately 10,974.78 ha. The overall boundary of the Teesmouth and Cleveland Coast 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 breeding terns which are already qualifying features of the existing SPA (or are proposed to be added as new qualifying features to it). The

[19] The value of 6.8% given here is as used in the Standard Data Form for the SPA and expresses the 1,900 individuals as a % of the GB breeding population of 14,000 pairs. This differs from the value of 4.3% presented in Table 1 and elsewhere in this document (and as presented in Carter (1993) which is calculated as a % of the estimate of the GB passage population of Sandwich terns (44,300 individuals)).

work done to identify the areas important to little terns and common 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 simple composite of the separate species-specific boundaries. It was drawn in such a way as to ensure only limited areas of sea where usage levels did not meet the necessary threshold to qualify for inclusion were incorporated. The existing SPA and proposed pSPA extension boundary map can be found in Annex 1, Map 1.

4.2.1.1 Identification of important marine areas for little tern

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

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

In total, 70 shore-based surveys were undertaken at 14 little tern colonies around the UK with a total of 7,006 registrations of little terns at various points along the shore. 23 boat-based transect surveys were undertaken across waters near eight colonies around the UK with a total of 781 registrations of the little terns at various distances offshore.

The following section outlines in brief the survey work and boundaries identified at the little tern colony at Crimdon Dene which falls within the area of the Teesmouth and Cleveland Coast pSPA. Further general information on the little tern survey programme is presented in Annex 4.

Six shore-based surveys were undertaken at the colony at Crimdon Dene in 2011 (3) and 2013 (3) and recorded a total of 656 tern passes. Three boat-based surveys were completed in 2012 and two were completed in 2013 on which a total of 102 little terns were sighted.

Based on site-specific survey data the maximum alongshore extent of little tern observations from the Crimdon Dene colony was 5,000m to both the north and south of the colony, and the mean maximum seaward extent was 3,448m (Figure 1).

The little tern foraging area defines the more northern half of the seaward boundary of the proposed extension of the pSPA, the southern half being defined on the basis of the work on the larger tern species.

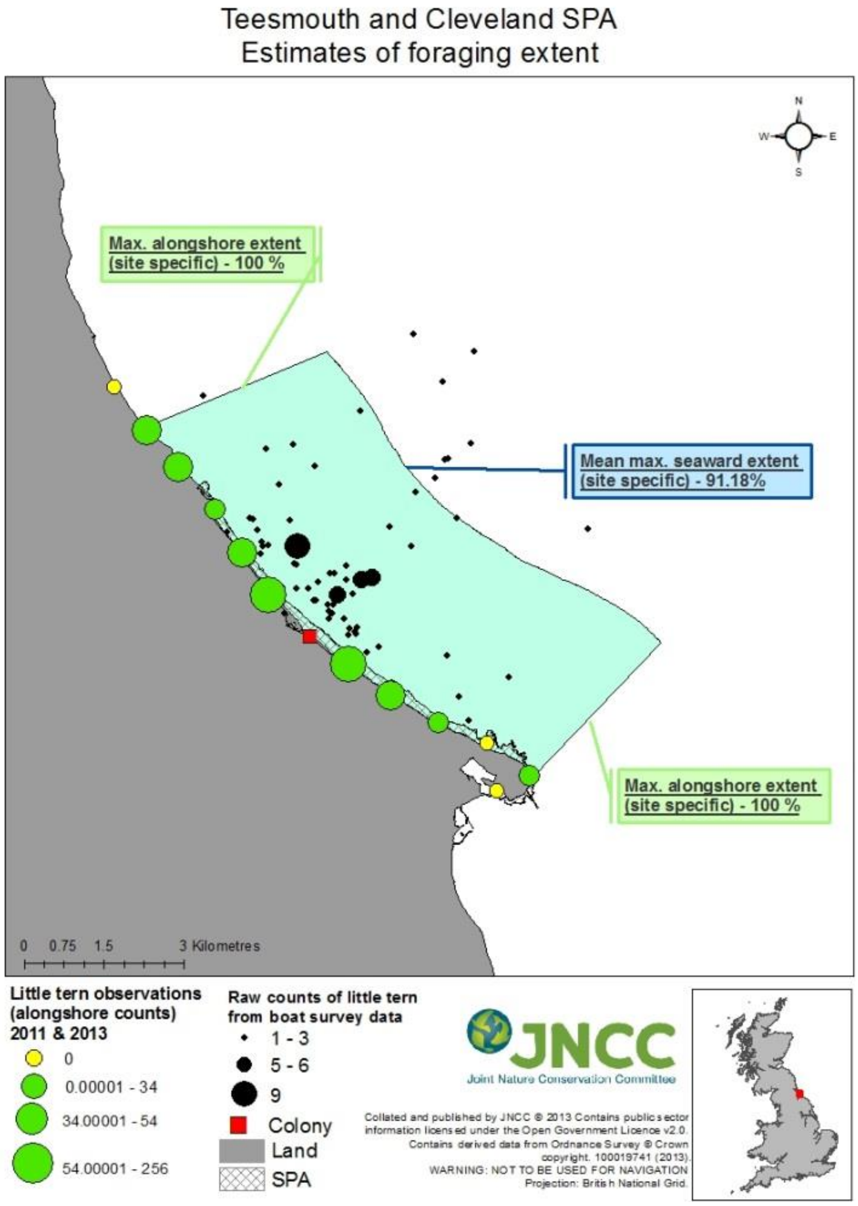

Figure 1: Application of site specific alongshore and site specific seaward extents to define the boundaries of little tern foraging areas at the Crimdon Dene colony. The percentage values given in the labels indicate the site specific percentage of little tern observations within the shore-based (alongshore) dataset and boat-based (seaward) dataset captured within the proposed alongshore and seaward boundaries. Note that the 91.18% of sightings recorded within the seaward boundary is not the measure used to define where that boundary should be, but is simply a consequence of setting the boundary at the mean of the survey specific maxima and thereby excluding sightings of a few individuals seen furthest offshore.

4.2.1.2 Identification of important marine areas for larger terns

The four larger species of tern which breed regularly in Great Britain have recorded mean foraging ranges between 4.5 km and 12.2 km and maximum recorded foraging ranges between 15.2 km and 49 km (Thaxter *et al.* 2012). In the light of these larger areas of interest, JNCC, in agreement with all of the SNCBs, decided that the most effective method to determine the extent of the areas most heavily used by breeding terns of the four larger species would be different to that employed for little terns. In this case the approach was to undertake a programme of boat-based visual tracking of foraging birds and to use the resultant information on foraging locations chosen by the birds, in conjunction with information on the habitat characteristics of those locations relative to other areas available to the birds, to construct habitat association models of tern usage. These models were used to predict tern usage patterns around breeding colony SPAs. Usage predictions were made out to the maximum recorded foraging range from each colony. In order to draw a boundary around the most important foraging areas for terns from each colony of interest, a minimum cut-off or threshold value of usage has to be found and only those areas in which usage exceeds that value are included within a possible SPA boundary. An objective and repeatable method to identifying a threshold value, based on the law of diminishing returns, is maximum curvature (O'Brien *et al.* 2012). This method identifies a threshold value below which disproportionately large areas would have to be included within the boundary to accommodate any more increase in, in this case, foraging tern usage. Further details of this work are given in Annex 5.

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

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

Visual tracking was carried out or commissioned by JNCC at ten of 32 colony SPAs which were deemed to be recently regularly occupied (Wilson *et al.* 2014). Survey effort was prioritised at these ten 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). Due to these considerations, survey effort was focussed at other parts of the coast of the United Kingdom, and no boat-based tracking work was undertaken on the common terns within the Teesmouth & Cleveland Coast SPA.

Accordingly, the following section summarises the application of generic boundaries from the modelling of tracking data at other UK tern colonies, to the common tern colony within the Teesmouth and Cleveland Coast pSPA. These generic models are referred to as 'Phase 2' models, which were amenable to maximum curvature analysis. Further general information on these surveys and the application of generic boundaries using the models is presented in Annex 5.

The only larger tern population which is a qualifying feature of the existing Teesmouth and Cleveland Coast SPA is the passage population of Sandwich tern. JNCC's research programme was not targeted at identifying the limits to the most important areas of usage by terns at this stage in the annual cycle and so Sandwich terns have not been considered in boundary definition for this pSPA. For the Teesmouth and Cleveland Coast pSPA the larger tern population of interest was the breeding population of common tern which now exceeds the SPA qualifying threshold, and is proposed as a new feature of the site. Generic models, generated from pooled data obtained from surveys of tern colonies across the UK, were used to generate predictions of relative usage in relation to common terns originating from Saltholme. The predictor variables used in the generic models 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.

The model generated predictions of relative usage by common tern, together with the boundary drawn around all of the areas in which predicted usage exceeded the threshold identified by application of the maximum curvature approach (to define a limit to the extent of the most important areas) are shown in Figure 2. 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 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 and/or distant from the shore are predicted to have very little or no usage by foraging terns.

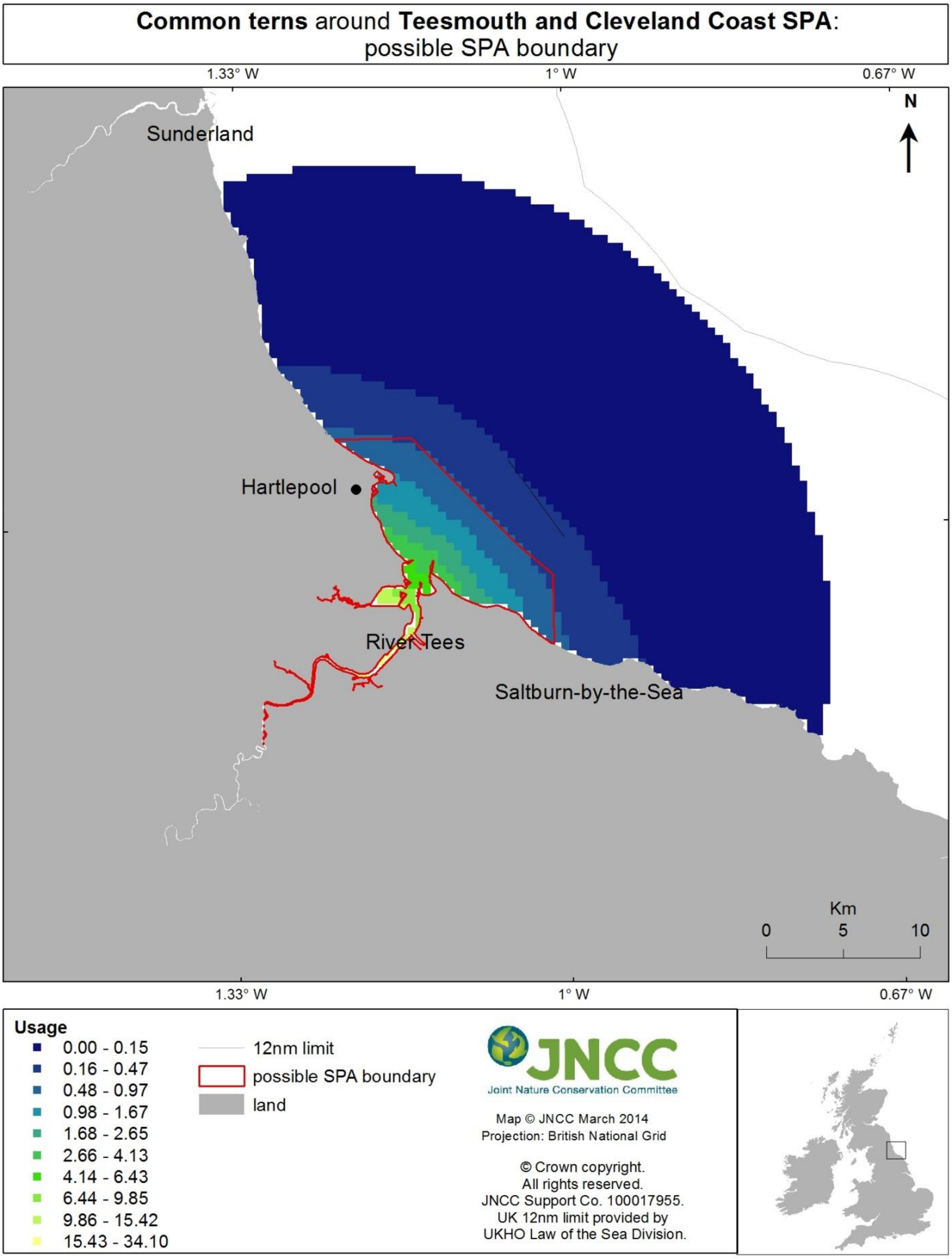

Figure 2: Model predictions of common tern usage overlaid with maximum curvature derived limits to areas of most importance around the Teesmouth and Cleveland Coast pSPA colony at Saltholme.

4.2.1.3 Composite seaward boundary of the Teesmouth and Cleveland Coast pSPA

Based on site-specific data on the distribution of little terns around Crimdon Dene and generic model predictions of usage by common terns from their source colony at Saltholme, and application of the maximum curvature technique to that predicted usage map, a composite of the seaward limit of the foraging ranges of terns is shown in Figure 3, indicating a potential SPA boundary.

The seaward boundary of the site comprises the outer limits of the common tern and little tern marine foraging areas. The marine extension for the little tern foraging area extends up to a maximum of 3.5 km offshore from the colony on the coast at Crimdon Dene. The marine extension for the common tern colony on the Tees centred at Saltholme extends up to a maximum of circa 6 km offshore from the mouth of the River Tees. The boundary on the open coast extends from Castle Eden Denemouth in the north to Marske-by-the Sea in the south.

4.2.1.4 Verification of predictions of generic modelled boundaries

Given that to a large degree the revised boundary of the Teesmouth & Cleveland Coast pSPA is based upon the predictions of a generic model of common tern usage rather than a model based on observations of the species at this site, 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 a generic modelling approach to define the areas of importance to common tern in the case of this pSPA. Each of these is described in brief below, in various degrees of detail in Annexes 5 and 6 to this document and in full detail in source documents, the details of which are given in these annexes.

As part of the model building and testing process carried out by JNCC and their statistical consultants, each generic model (one for each species of tern studied) was subjected to a process of cross-validation. In this process tern tracking data from each one of the colonies which contributed data to the development of the generic model was in turn excluded from the dataset and the model re-built on the basis of the data from the remaining colonies. The agreement between the new predictions of the distribution of tern usage around the excluded colony (generated by the reduced model) and the observed distribution at that colony (now in the excluded data) was then established using standard statistical criteria. This process is described in more detail in Annex 5. This process demonstrated that the common tern generic model performed well (performance classed as “good”) when judged by its ability to predict the observed distribution of common terns at colonies which were (in the cross-validation process) excluded in turn from building the model. This analysis indicated that there is reasonable consistency between common tern colonies around the UK in the characteristics of sea areas which hold the highest relative densities of foraging breeding birds. Accordingly, there is a corresponding degree of confidence that the boundary of this pSPA, being dependent upon the predicted usage patterns of common terns, is founded on a reliable evidence base, albeit not one derived directly from birds at the colony in question.

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 common tern in the case of this pSPA.

In 2015 Natural England commissioned ECON Ecological Consultancy Ltd to carry out additional surveys at a number of pSPA sites in England to verify the predicted distribution of patterns of tern usage generated by the modelling work on terns carried out by the JNCC (Natural England 2016). The additional survey work involved shore-based and boat-based surveys within 6 different pSPAs including the Teesmouth and Cleveland Coast pSPA. In this case, the issue of particular interest was to gather site-specific empirical data on tern abundance and distribution to verify the inclusion of the River Tees within the pSPA boundary. Three boat-based surveys were carried out between Hartlepool Marina and Victoria Harbour in Hartlepool, across Hartlepool bay and up the River Tees to the Tees Barrage. *En route,* three different types of record of tern activity were made: timed counts of all terns seen within 300m of the boat over a 30 minute period at a series of fixed observation stations, a series of instantaneous snapshot counts taken every minute within the 30 minute timed count periods, and a full transect survey on which instantaneous snapshot counts of all terns within 300m of the boat were recorded every 300m along the transect. The results of these verification surveys in 2015 confirmed common terns were present in both Hartlepool Marina and Victoria Harbour at Hartlepool and throughout the length of the River Tees up to the Tees Barrage, and that many “off-river” locations such as Tees dock, Middlesbrough dock, Dabholme Gut etc. were seemingly used by the foraging terns as part of the riverine environment. More details of the survey methods and results are presented in Annex 7.

In 2016, Industry Nature Conservation Association (INCA) was commissioned by Natural England to repeat the 2015 tern verification surveys for the stretch of the River Tees between Tees Barrage and Seaton Channel in order to obtain additional information to verify the JNCC modelling work. Another three surveys were carried out. The survey method and type of data recorded were essentially identical to those in 2015. In addition to repeating the 2015 survey, which focused on tern numbers and activity, the INCA surveys also intended to gather data about whether existing anthropogenic activities on the river were causing disturbance to terns. The results of the 2016 confirmed the results of the 2015 survey in that common terns were found throughout the entire length of the River Tees between Seaton Channel and Tees Barrage and in each of the “off-river” locations surveyed. The results for the disturbance work were inconclusive and further survey work is required to determine to what extent foraging common terns are sensitive to specific activities along the River Tees. More details of the survey methods and results are presented in Annex 7.

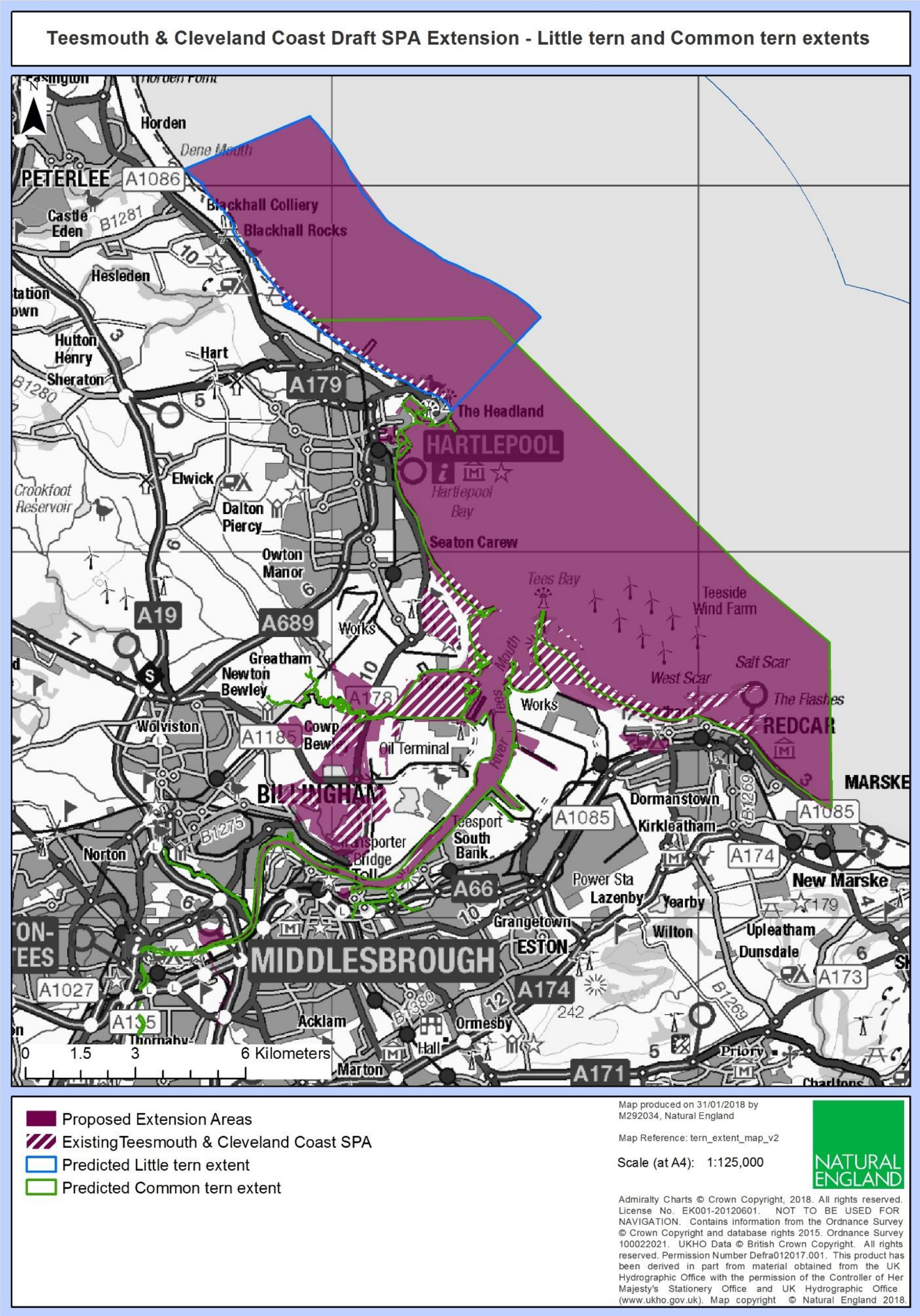

Figure 3: Proposed simple, composite boundary of the Teesmouth and Cleveland Coast pSPA drawn around the little tern and common tern boundaries presented in the preceding sections

4.2.2 Landward boundary of the marine extent of the pSPA

The landward boundary to the marine elements of the pSPA is located on the Teesmouth and Cleveland Coast and includes the main River Tees channel below the barrage, estuary waters, and runs along the open coastline between Marske-by-the-Sea in the south (NZ634222) and Castle Eden Denemouth in the north (NZ460404). The northern limit to the alongshore boundary at Castle Eden Denemouth is determined by the little tern foraging areas around the colony at Crimdon Dene. This reaches 5km in both directions along the coast from the colony between Hartlepool Headland and Castle Eden Denemouth. The southern limit to the alongshore boundary of the pSPA is at Marske-by-the Sea (NZ634222), and is determined by the limit to the area of shoreline along which usage by common tern exceeded the threshold for inclusion determined by maximum curvature analysis. Due to the location of the principal common tern colony (at Saltholme) some distance inland, a stretch of the River Tees has been included in the boundary up to the Tees Barrage. Along the main river Tees channel, near the mouth of the Tees estuary, and at Hartlepool there are a number of harbours, docks, marinas, basins etc. that fall within the common tern foraging areas. The use of such places by foraging common terns was confirmed during the surveys in 2015 and 2016 (Natural England 2016, INCA 2016 (see Annex 7)). The common terns also travel a direct route from the colony at Saltholme towards Seal Sands and the estuary mouth as well as using the River Tees to forage.

JNCC guidelines on selecting marine SPAs (Webb & Reid 2004: Annex B) states that where the distribution of birds is likely to meet land, landward boundaries should be set at Mean High Water (MHW) “unless there is evidence that the qualifying species make no use of the intertidal region at high water”. Observations indicated that little terns forage both in the intertidal zone and subtidal zone (Parsons *et al.* 2015). That the use of such areas by all larger tern species is also likely is supported by information in the scientific literature. A review of tern foraging ecology (Eglington 2013) notes that all five species of tern considered routinely forage in areas of shallow water. There is no reason on the basis of that review to consider it likely that that these birds will not forage over intertidal areas. Thus, the landward boundary of the marine areas of the pSPA will extend to Mean High Water, with the exception of those places where the tern species are already qualifying features of other existing SPAs and the areas between Mean High Water and Mean Low Water are already protected for these features within these other existing SPAs. In those places, the boundary of the pSPA will not overlap but abut with that of any existing SPA.

The little tern foraging areas for the Crimdon Dene colony overlaps with a short stretch of the Northumbria Coast SPA, approximately 2.2 km. As the Northumbria Coast SPA already protects little terns, albeit from a different colony, the landward boundary of the pSPA has been drawn to clip this area of potential overlap between MLW and MHW out of the pSPA extension and abut at MLW the boundary of the existing Northumbria Coast SPA along this stretch of coast.

The proposed extension for foraging terns includes all areas on the intertidal foreshore below Mean High Water that lie between the boundaries of existing component parts of the SPA. The main stretch of currently undesignated intertidal foreshore is between the pumping station at Seaton Sands (NZ534291) and Middleton jetty on the south side of Hartlepool’s Victoria Harbour. The intertidal areas between these points will now be included within the pSPA boundary.

4.2.3 Terrestrial extension of the pSPA and Ramsar

The terrestrial extension of the pSPA includes a number of land parcels, intertidal areas and estuarine waters. These areas include important open waters, grazing marshes and intertidal mudflats. A carefully targeted programme of wintering bird surveys was carried out in 2014/15 to improve understanding of the use of wet and brownfield grassland, open water and intertidal mudflats supporting wintering waterbirds currently outside the existing SPA. The extended pSPA terrestrial areas include wet grassland, brownfield grassland developed on recently reclaimed land, deep and shallow pools and saltmarsh in several locations (Coatham Marsh, Bran Sands Lagoon, Cowpen Bewley, Dabholme Gut, Greatham North / Saltern Wetlands, the North Tees Marshes, RSPB Saltholme, and Portrack Marsh). These habitats are of great importance to a diverse assemblage of bird species. Areas of saltmarsh provide significant feeding and roosting opportunities for many species of waterbird, in particular common redshank, shelduck *Tadorna tadorna* and teal *Anas crecca*. The latter two species are known to feed upon the seeds of saltmarsh vegetation such as common glasswort *Salicornia europaea* and orache *Atriplex* species as a secondary food source. The intertidal areas support densities of benthic invertebrates which provide an important food source for the majority of the assemblage bird species such as common redshank, red knot and Eurasian oystercatchers *Haematopus ostralegus*.

The proposed Ramsar boundary extension includes the additional terrestrial habitats proposed for the pSPA including saltmarsh, wet grassland, intertidal areas and open pools which support the waterbird assemblage. The Ramsar boundary extension follows the northern and southern extent of the pSPA down to Mean Low Water to include important intertidal areas which support large numbers of waterbirds and existing features of the SPA. Due to different selection criteria for Ramsar sites (Ramsar Convention) and SPAs (Birds Directive), the proposed Ramsar extension will not follow the marine extension of the pSPA. Instead, the Ramsar extension will extend as far seaward into the marine environment as MLW and therefore will not include marine tern foraging areas or the main Tees River channel as the Ramsar Convention does not apply to these areas[20]. Intertidal muddy areas at the mouth of the Tees Estuary (to MLW) have been included in the Ramsar extension as they provide suitable habitat for the non-breeding waterbird assemblage. A map of the proposed extension to the Ramsar site is shown in Annex 1, Map 2.

5. Location and Habitats

The estuary of the River Tees is situated on the coast of north-east England, close to the built up areas of Middlesbrough and Hartlepool. It changed rapidly during the last century due to progressive land-claim for industrial development and as a result of increasing levels of pollution. Despite much loss of habitat, significant areas of intertidal sand and mudflat, salt and freshwater marsh, grazing marsh, sand dunes and rocky shore remain, supporting numbers of waterbirds of European importance.

The Seal Sands part of the proposed SPA and Ramsar site is the only extensive area of intertidal mudflats on the east of England between Lindisfarne to the north and the Humber Estuary to the south, a distance of over 300 kilometres. The flats support high densities of invertebrates important as prey for overwintering waterbirds, particularly shelduck, red knot and common redshank. Adjacent areas of reclaimed land are used for feeding and roosting when the mudflats are inundated at high tide. Smaller areas of intertidal mudflats important for waterbirds occur on both sides of the mouth of the estuary and at Greatham Creek adjacent to Cowpen Marsh.

The sandy beaches of Seaton Sands, North Gare Sands, Seaton Snook and Coatham Sands are important feeding and roosting areas for non-breeding waders, notably red knot and sanderling *Calidris alba*. The pools behind Coatham Dunes support roosting redshank as well as a range of foraging waterbirds. Seaton Snook, a large sandy spit developed on top of a slag training wall on the north side of the estuary mouth, is a particularly important roost for large numbers of gulls, terns and waders as it remains uncovered at high tide.

Greatham Creek and Cowpen Marsh include the largest area of saltmarsh between Lindisfarne and the Humber Estuary. The majority of the marsh is dominated by common saltmarsh-grass *Puccinella maritima* with much sea aster *Aster tripolium*. Higher levels of the marsh on the southern side of Greatham Creek support species-rich associations of red fescue *Festuca rubra*, sea plantain *Plantago maritima*, sea arrowgrass *Triglochin maritima*, greater sea-spurry *Spergularia media* and sea milkwort, *Glaux maritima*, and there is an unusual community of common sea-lavender *Limonium vulgare* with thrift *Armeria maritima* which occurs here close to the northern edge of its range in eastern Britain. Saltmarsh has developed at South Gare since the construction of the breakwater there in the 1860s, and more recently on Seaton Snook. Notable plants within the developing saltmarsh include sea wormwood *Artemisia maritima,* common saltwort and orache species, as well as common glasswort and annual seablite *Suaeda maritima.*

[20] http://www.ramsar.org/sites/default/files/documents/library/manual6-2013-e.pdf

These areas of saltmarsh provide significant feeding and roosting opportunities for many species of waterbird, in particular, common redshank, shelduck and teal, and for notable passerines such as snow bunting *Plectrophenax nivalis* and twite *Carduelis flavirostris*.

The subtidal habitat across the site includes a wide range of habitats. North of Crimdon, towards Castle Eden Denemouth, there are magnesian limestone wave-cut platforms and boulders interspersed with sandy substrates. The rocky shores provide a hard substrate for a different range of prey species including dense beds of mussels *Mytilus edulis.* North and South Gare Sands consist predominantly of well graded sand exposed to wave action and supporting sparse fauna. In contrast, Seal Sands consist of soft mud supporting an abundant fauna. The coastline between Redcar and Marske which lies within the pSPA extension comprises a gently shelving sandy foreshore.

Offshore at a depth of approximately 25 m the seabed is muddy with boulders and mixed sediments dominated by animal communities, particularly dead's man fingers *Alcyonium digitatum.* Owing to the vast amount of industry associated with the Tees Estuary and a number of sewage discharges, water quality has historically been very poor. However, following improvements to discharges, monitoring has shown that since the 1970s water quality in the Tees Estuary has gradually improved and there is evidence that the benthic communities have become more diverse and abundant. The water depth at the mouth of the Tees at low tide is approximately 15m.

The location and description of habitats for proposed terrestrial extensions to the pSPA and Ramsar are further detailed below and in Figure 4. These areas are important for non-breeding birds and are used for foraging and roosting. They include important open water, grazing marsh and intertidal mudflats. The use of each area by the species and assemblages of European and international importance is described in Section 6.

5.1. Portrack Marsh

Portrack Marsh lies on the north bank of the River Tees between the Tees Barrage and the A19 flyover. Originally an artificial oxbow lake resulting from the straightening of the river, the area incorporates freshwater pools (of varying depths) fringed by reedbed and swamps, grassland and scrub.

5.2. South end of Cowpen Marsh

This proposed extension encompasses the wet grassland and pools on Cowpen Marsh between the existing SPA/Ramsar site and the A1185 to the south (excluding the brine reservoirs).

5.3. Number 4 Brinefield

This proposed extension south of Greatham Creek includes grasslands and pools between the A178 Tees Road and the north-south aligned Long Drag embankment. It includes a saline lagoon and tidal pool (formerly the course of Greatham Creek prior to its canalisation) and adjacent areas in the north of the site, an area of pools and adjacent grassland immediately to the west of Long Drag in the south east, and a strip of wet grassland immediately east of the A178 Tees Road in the west.

5.4. Greenabella Marsh

It is proposed to extend the SPA/Ramsar site at Greenabella Marsh to include a saline lagoon currently outside the designated site boundary, as well as grassland and pools adjacent to the lagoon. These habitats support similar species to those using the designated land at Greenabella Marsh. The extension will provide protection across a broader management unit area including land with probable hydrological connectivity to the saline lagoon.

5.5. Greatham Tank Farm

The proposed extension at Greatham Tank Farm lies north of Greatham Creek and west of the A178 Tees Road. The extension comprises wet grassland and pools.

5.6. Hartlepool Bay and foreshore

The proposed offshore extension for foraging terns includes all areas of intertidal foreshore between the existing SPA boundary at Seaton Sands pumping station north to Middleton jetty on the south side of Hartlepool's Victoria Harbour.

5.7. Seal Sands brownfield extension

This proposed extension includes a raised track and short, rabbit-grazed grassland on made ground (comprising slag) immediately south of the Seal Sands intertidal area. The intertidal area north of Seal Sands mudflat has also been included in the proposed extension as this provides important habitat for wintering waders.

5.8. Vopak foreshore

This area of intertidal mud is located on the north bank of the River Tees a short distance upstream of Seal Sands.

5.9. Bran Sands South

The proposed extension at Bran Sands South, on the south side of the River Tees, includes the large Bran Sands lagoon south of the steelworks and the tidal channel of Dabholm Gut immediately to the south.

5.10. Coatham Marsh

The proposed extension at Coatham Marsh includes the freshwater pools and grassland at Coatham Marsh Nature Reserve north and south of the railway line south of Warrenby.

5.11. Coatham Lagoons

The proposed extension includes pools located towards the landward edge of Coatham Dunes, south of South Gare.

5.12. Greatham North / Saltern Wetlands

The proposed extension at Greatham North includes a large area of intertidal mud and saltmarsh as well as some freshwater pools. These habitats were created in a managed realignment of the flood defence on the north bank of Greatham Creek. The old sea wall was breached in May 2014. The freshwater pools were developed from borrow pits that supplied material for the managed realignment.

5.13. Cowpen Bewley

Natural England advise that Reclamation Pond is excluded from the SPA extension on the basis that the effects[21] on it were properly considered and taken into account in accordance with the assessment and decision-making provisions of the Habitats Directive prior to the grant of an extant planning permission for this area.[22] Natural England recommend the land referred to in the

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[21] The effects on the SPA specifically relate to noise disturbance during construction and operation of the proposed facility, the loss of open water habitat as a result of the partial infill of Rec Pond, and any subsequent impacts on wider SSSI habitats, such as freshwater grazing, saltmarsh and dune grasslands, which support SPA species as a result of any emissions to air that could affect the species composition of these habitats (e.g. oxides of nitrogen and sulphur).

[22] Planning permission was issued in January 2004 to reclaim Reclamation Pond for industrial use. That planning permission has now been partly implemented. In granting approval in 2004 the Council accepted that the provision of Port Clarence Pool and the Cowpen wetland areas were adequate compensation measures. Stockton Council were consulted in 2007 as the Local Planning Authority on an application to the Secretary of State for Trade & Industry for deemed consent to construct and operate a natural gas fired combined cycle gas turbine power station and associated substation on land forming part of Reclamation Pond. Stockton's Planning Committee agreed they would not object in 2007 and planning permission was finally granted on the 2nd May 2011 by the Secretary of State.

planning documents as mitigation land (Port Clarence and Cowpen Bewley) are included in the SPA extension as they are designed and predicted to be suitable compensatory habitat for Reclamation Pond. Recent ornithological records indicate that birds potentially displaced from Reclamation Pond are starting to use the mitigation areas and there is a reasonable expectation the areas will provide supporting habitat for SPA features and be part of the most suitable territory for the SPA species in the long term.

5.14. RSPB Saltholme

Some of RSPB Saltholme reserve is already with the SPA/Ramsar. The proposed extension adds the remainder of the important wetland features of the core reserve, which comprise an extensive area of wet grassland, scrapes and pools, to the SPA/Ramsar.

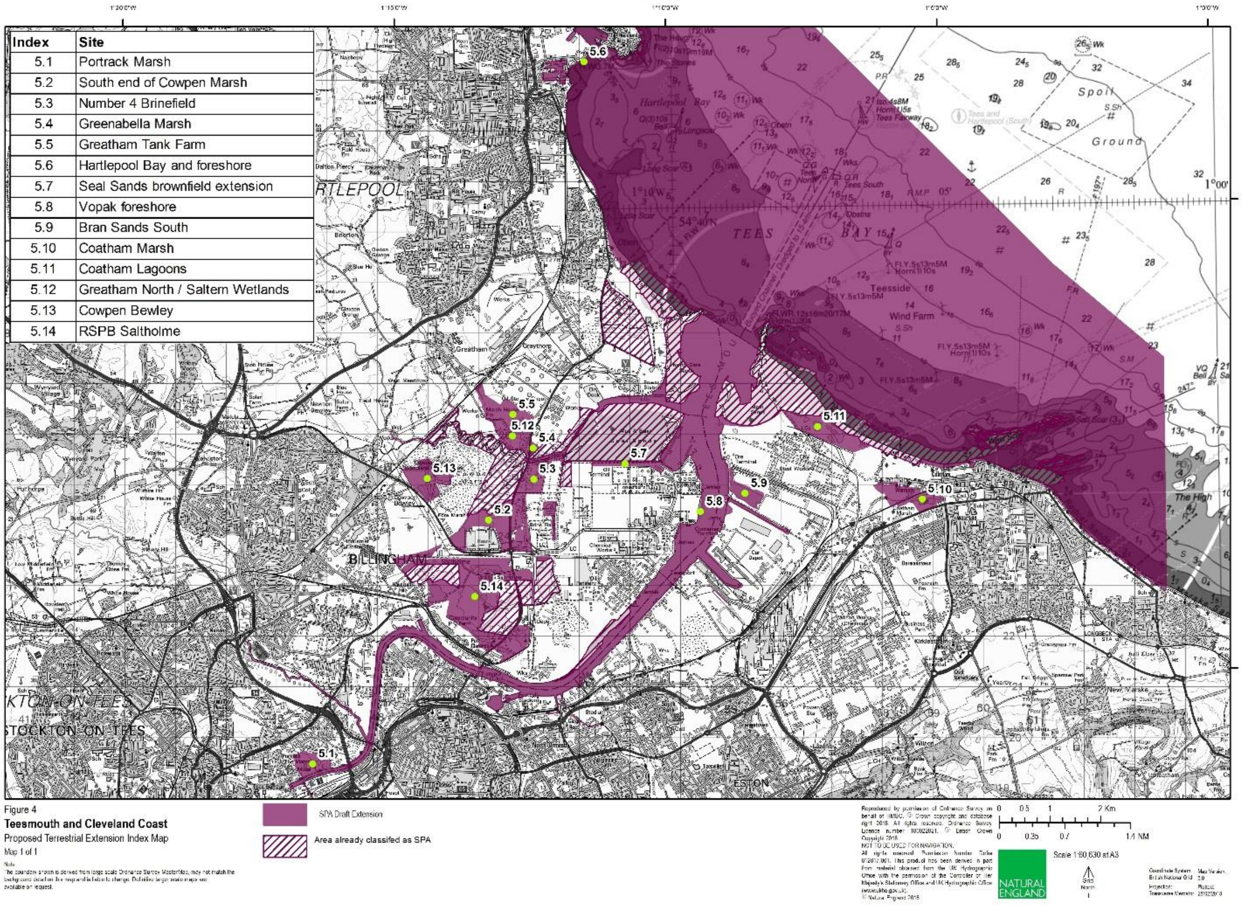

Figure 4: The proposed terrestrial extensions to the Teesmouth and Cleveland Coast pSPA and Ramsar.

6. Assessment of Ornithological Interest

6.1. Survey information and summary

The counts of most breeding, wintering and passage birds have been derived from Cleveland Bird Reports 2009-2013 and Wetland Bird Survey (WeBS) core counts for the years 2011/12 to 2015/16. The Teesmouth and Cleveland Coast SPA and Ramsar Site (including extensions) includes a section of the coast outside of the Tees Estuary WeBS site. This area is covered by two WeBS core count sectors – Durham Coast - Sector 1a WeBS core count sector 53408 and Durham Coast - Sector 1b WeBS core count sector 53409. Unfortunately no recent data are available for Durham Coast - Sector 1b WeBS core count sector 53409 and so population estimates for 2011/12 - 2015/16 have been derived by combining counts from Tees Estuary core count WeBS sectors and the Durham Coast – Sector 1a 53408. All Tees Estuary WeBS site core count sectors have been used except the Reclamation Pond WeBS core count sector 52421. This is because Reclamation Pond is excluded from the pSPA extension on the basis that the effects on it were properly considered and taken into account in accordance with the assessment and decision-making provisions of the Habitats Directive prior to the grant of an extant planning permission for this area. Additional data sources to those just described are cited in the species accounts below. All of the bird data sources are summarised with details of their method of data collection and verification process in Annex 4 to this document. The Teesmouth and Cleveland Coast pSPA and Ramsar site (including proposed extensions) supports over 1% of the GB populations of five species listed in Annex I of the EC Birds Directive (2009/147/EC) and over 1% of the biogeographical population of two regularly occurring migratory species. It also supports a waterbird assemblage of European/international importance during the non-breeding season. The species and assemblages of European and international importance are described below.

SPA site selection guidelines have been applied to the most up to date, comprehensive information for the site. However, these recent data reveal that Sandwich tern, red knot and common redshank are no longer present in qualifying numbers. It is not clear whether anthropogenic influences have affected the numbers of birds at this site. Defra policy indicates that in these circumstances the features should be retained until such time as the reasons for their reduction in population can be established. It is therefore considered that Sandwich tern, red knot and common redshank should be retained on the citation, and the level of ambition set out in the conservation objectives for the species maintained, unless and until evidence is available to support the conclusion that declines are a result of natural processes and that the SPA is no longer suitable for these species.

6.2. Annex 1 species

Pied avocet *Recurvirostra avosetta*

The breeding population of pied avocets in Great Britain is estimated to be 1,500 pairs (Musgrove *et al.* 2013), representing about 6.2% of the West Europe and North-west Africa breeding population (24,300 pairs derived by division by three of the estimate of 73,000 individuals: AEWA 2012). In the UK, the main breeding areas are in East Anglia and Kent, with particular concentrations around the Norfolk, Suffolk and north Kent coasts. The preferred nesting habitat of pied avocets is shallow, brackish coastal lagoons with bare or sparsely vegetated low islands (Gibbons *et al.* 1993), although the species also breeds at several inland colonies.

Pied avocet is not currently a named feature of the existing Teesmouth and Cleveland Coast SPA or Ramsar as it did not occur in numbers exceeding qualifying thresholds at the time of the initial classification. Between 2010 and 2014 the pSPA/Ramsar, including the proposed extensions, supported an average of 18 breeding pairs (2010 - 15 pairs, 2011 - 18 pairs, 2012 – 27 pairs, 2013 -17 pairs and 2014 - 12 pairs), which represents 1.2% of the GB breeding population and accordingly it is proposed to add the species as a named feature of the pSPA. Avocet does not qualify as a Ramsar feature as it does not meet the 1% biogeographic threshold i.e. c 243 pairs. The majority of birds breed on Number 4 Brinefield, mainly on the saline lagoon south of Greatham Creek, with smaller numbers on Greenabella Marsh.

Ruff *Calidris pugnax*

The non-breeding population of ruff in Great Britain is estimated to be 800 individuals (Musgrove *et al*. 2013), representing about 0.05% – 0.08% of the Northern Europe & Western Siberia/West Africa population (1,000,000-1,500,000 individuals during the period 1950-2000, Wetlands International 2012).

Ruff is not a feature of the existing Teesmouth and Cleveland Coast SPA. This is because SPA selection guidelines (Stroud *et al*. 2001) acknowledged that for Annex 1 species with large biogeographical populations but only very small populations in Great Britain (such as non-breeding ruff), the small numbers required for a site to support in excess of the usual 1% national population threshold (8 individuals in the case of non-breeding ruff) would not be of major significance for sustaining viable biogeographical populations of these species. Accordingly, when considering wintering waterbirds it has been the statutory agencies' long-standing practice to require at least 50 individuals to be regularly present on a site before that area is considered for site selection (Stroud *et al.* 2001). Nonetheless, this guideline has not been applied in the case of several species on the basis that its application would constrain selection of an appropriate SPA suite. The UK SPA Scientific Working Group (2002) recommended that there was a need to "**Discuss and agree** those circumstances (perhaps in terms of conservation benefits) where exemptions from the greater than 50 rule might be considered." In 2015 the UK SPAR SWG considered the applicability of this guideline to non-breeding ruff (UK SPAR Scientific Working Group, Minutes of 3 November 2015 meeting (http://jncc.defra.gov.uk/page-1770)). The subsequent inclusion of non-breeding ruff as a qualifying feature of the Morecambe Bay & Duddon Estuary SPA in 2017 (8 individuals) saw the guideline relaxed in that case. Inclusion of non-breeding ruff as a qualifying feature of that SPA was justified on the grounds of the conclusion of the 3rd SPA review (Stroud *et al.* 2016) that the existing SPA suite for the species was considered insufficient in terms of both the population numbers within it, and its geographical range coverage, especially in western Britain. The numerical and range insufficiency of the existing SPA suite for non-breeding ruff also justifies relaxation of the guideline in the present case and inclusion of this species as a qualifying feature of the Teesmouth and Cleveland Coast pSPA. Between 2011/12 and 2015/16 the pSPA, including proposed extensions, supported an average of 19 individuals which: represents 2.4% of the GB non-breeding population; is more than twice the number (8 individuals) supported by the recently classified Morecambe Bay & Duddon Estuary SPA, and makes the site the 7th most important for the species in the UK (see Table 6 in Section 8). This pSPA also lies to the north of all other sites within the existing suite of SPAs for this species and so extends the range coverage of the species' SPA suite. Accordingly it is proposed to add the species as a feature of the pSPA. Ruff occur at shallow waterbodies across the site, in particular on the pools at RSPB Saltholme.

Common tern *Sterna hirundo*

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

Common tern is not a named feature of the existing Teesmouth and Cleveland Coast SPA as it did not occur in numbers exceeding qualifying thresholds at the time of classification. Between 2010 and 2014 the pSPA, including the proposed extensions, supported an average of 399 breeding pairs of common terns, which represent about 4% of the GB breeding population. Accordingly it is proposed to add the species as a named feature of the pSPA. Nesting birds are typically concentrated on islands within the various waterbodies at Saltholme, with variable and smaller numbers of nests on the saline lagoon in No. 4 Brinefield south of Greatham Creek, and on rafts at Cowpen Marsh. Two pairs also bred on Portrack Marsh in 2014.

Sandwich tern *Thalasseus sandvicensis*

The passage population of Sandwich terns in Great Britain is estimated to be 44,300 individuals (Carter 1993), representing about 26% of the Western Europe/West Africa population (upper estimate 171,000 individuals: AEWA 2012). After breeding, adult birds and juveniles often gather in large numbers at sites away from the main colonies before migrating to wintering grounds on the south and west coasts of Africa.

The passage population of Sandwich tern is a qualifying feature of the existing SPA and Ramsar site. The SPA citation (dated 2000) lists 1,900 individuals. The Natura 2000 Standard Data Form (JNCC, updated 2000) also states 1,900 individuals as the 5-year mean (1988-1992) at the time representing 6.8% of the GB breeding population or 4.3% of the GB passage population. Numbers on the site have since declined and between 2011/12 and 2015/16 the pSPA/Ramsar site, including the proposed extensions, supported an average of 134 individuals, representing around 0.3% of the GB passage population estimate of Carter (1993). For the reasons set out in Section 5.1, this Departmental Brief does not propose any amendment to the notified population of 1,900 individuals. Highest numbers occur from mid-July to September when adults and juveniles disperse from breeding colonies. The majority use roosts at Coatham Sands, Seal Sands, North Gare Sands/Seaton Snook and Bran Sands. They feed in shallow inshore waters in and around the estuary mouth.

Little tern *Sternula albifrons*

The breeding population of little terns in Great Britain is estimated to be 1,900 pairs (Musgrove *et al.* 2013), representing about 10% of the Eastern Atlantic breeding population (18,500 pairs derived by division by three of the upper estimate of 55,500 individuals: AEWA 2012). Breeding occurs in scattered colonies around much of the east and west coasts of Britain, from the north of Scotland to 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, using sand and shingle beaches and spits, as well as tiny islets of sand or rock close inshore (Mitchell *et al.* 2004).

Breeding little tern is a qualifying feature of the existing SPA and a non-qualifying species of interest for the existing Ramsar site. The SPA citation (dated 2000) lists 40 pairs. The Natura 2000 Standard Data Form (JNCC, updated 2000) also states 40 pairs as the 4-year mean (1995-1998) at the time representing 1.7% of the GB breeding population. Between 2010 and 2014 the pSPA, including the proposed extensions, supported an average of 81 breeding pairs of little terns, which represent 4.3% of the GB breeding population. Because the little tern population data should, ideally be contemporary with the foraging tern distribution data used to inform the proposed revision to the site boundary (2011-2013), it is proposed that this new, updated population estimate should replace the earlier SPA population estimate. Virtually all breeding birds are currently located at Crimdon Dene, north of Hartlepool. The feeding grounds of the little terns that nest at Crimdon Dene lie predominantly in marine areas within 5 km alongshore of the colony and within 3.5 km offshore.

6.3. Regularly occurring migratory species

Red knot *Calidris canutus*

At the time leading up to the first extension of the SPA/Ramsar site in 2000 the non-breeding population of red knot in Great Britain was estimated to be 290,000 individuals, representing about 84% of the NE Canada & Greenland/Iceland/UK population (345,000 individuals during the period 1982-1992: Wetlands International 2012). Red knot are widely distributed in coastal areas and occupy a variety of habitats, principally estuaries, mudflats and sandflats.

Non-breeding Red knot is a qualifying feature of the existing SPA and Ramsar site. Between

1991/92 and 1995/96 the SPA/Ramsar site supported an average of 5,509 individuals which, at that time, represented 1.6% of the NE Canada/Greenland/Iceland/UK population. Numbers in the site have since declined and between 2011/12 and 2015/16 the pSPA/Ramsar site, including the proposed extensions, supported an average of 876 individuals, representing around 0.2% of the NE Canada & Greenland/Western Europe population (450,000 individuals during the period 1997 - 2007: AEWA 2012). For the reasons set out in Section 5.1, this Departmental Brief does not propose any amendment to the notified population of 5,509 individuals. Birds feed at low tide on intertidal mudflats, mussel beds and rocky shores on both sides of the estuary. Formerly present in large numbers in the estuary on Seal Sands, particularly when the rising tide covered other foraging habitats, the birds are now increasingly located outside the estuary, on Coatham Sands, Redcar Rocks and around Hartlepool Headland.

Common redshank *Tringa totanus*

At the time of the original classification of the SPA/Ramsar site in 1995 the non-breeding population of common redshank in Great Britain was estimated to be 75,400 individuals, representing about 69% of the north-west European component of the East Atlantic flyway population (Carter 1993). Non-breeding common redshank are widely distributed, mainly in coastal areas of Britain, where flocks are often a mixture of individuals largely from the *robusta* and *brittanica* races (Holt *et al*. 2012).

Non-breeding common redshank is a qualifying feature of the existing SPA and Ramsar site. Between 1987 and 1991 the SPA/Ramsar site supported an average of 1,648 individuals which, at that time, represented 1.1% of the East Atlantic population (SPA Citation, 2000). Numbers on the site have since declined and between 2011/12 and 2015/16 the pSPA/Ramsar site, including the proposed extensions, supported an average of 881 individuals representing around 0.3% of the Iceland & Faroes/Western Europe population (275,000 individuals derived from the midpoint of 150,000-400,000 individuals during the period 1998 - 2008: AEWA 2012). For the reasons set out in Section 5.1 this Departmental Brief does not propose any amendments to the notified population of 1,648 individuals. Within the site birds feed on intertidal mudflats including Seal Sands, North Tees Mudflat, Bran Sands and Hartlepool Bay, saltmarsh areas at Greatham Creek and intertidal rocky shores at Hartlepool Headland, Redcar and Coatham.

6.3.1. Waterbird assemblage

Teesmouth and Cleveland Coast is one of the most important sites in the UK for waterbirds. The site qualifies under Article 4.2 of the Birds Directive (79/409/EEC) as it is used regularly by over 20,000 waterbirds.

The waterbird assemblage includes as its main components, all the Annex 1 species and regularly occurring migratory species which qualify in their own right under Articles 4.1 and 4.2 respectively of the Birds Directive (79/409/EEC). Species do not need to be present in numbers of European importance to be major assemblage components. Numbers equal to or exceeding the national importance threshold (e.g. 1% or more of GB population) or in excess of 2,000 individuals are sufficient for a species to be listed as a major component of the assemblage. This assemblage includes up to date information for all species, including those that formerly qualified as features in their own right when the SPA/Ramsar site was first designated, but are no longer present in qualifying numbers (i.e. non-breeding red knot, common redshank and Sandwich tern which are nonetheless retained as features in their own right). The assemblage has been updated to include seabirds, because they are included under Criterion 5 of the Ramsar guidelines. This means that the proposed Ramsar site boundary is coincident with the SPA boundary in the terrestrial areas.

During the period 2011/12-2015/16 the Teesmouth and Cleveland Coast pSPA/Ramsar site, including the proposed extensions, supported an average peak of 26,014 (SPA assemblage) / 26,786 (Ramsar assemblage) individuals. This assemblage is of both European and international importance. The assemblage includes a wide range of breeding, wintering and passage waterbird species, including those of European importance described above, as well as numbers exceeding

1% of the GB non-breeding populations (see table 4 below) of gadwall *Anas strepera*, northern shoveler *Anas clypeata* and sanderling *Calidris alba.* Additionally, Eurasian wigeon *Anas penelope*, northern lapwing *Vanellus vanellus*, herring gull *Larus argentatus* and black-headed gull *Chroicocephalus ridibundus* 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). Details of the GB status and the populations present in the pSPA and Ramsar site (including proposed extensions) of the main component species of the waterbird assemblage are given below.

Table 4: Waterbird species present in nationally important numbers or where their numbers exceed 2,000 individuals

Species	5 year peak mean (2011/12 – 2015/16)	% GB population[23]
Eurasian wigeon *Anas penelope*	2,660 individuals	N/A but >2,000 individuals
Gadwall *Anas strepera*	428 individuals	1.7%
Northern shoveler *Anas clypeata*	180 individuals	1.0%
Northern lapwing *Vanellus vanellus*	3,892 individuals	N/A but >2,000 individuals
Sanderling *Calidris alba*	242 individuals	1.5%
Herring gull *Larus argentatus*	3,243 individuals	N/A but >2,000 individuals
Black-headed gull *Chroicocephalus ridibundus*	2,273 individuals	N/A but >2,000 individuals

Eurasian wigeon *Anas penelope*

The non-breeding population of Eurasian wigeon in Great Britain is estimated to be 440,000 individuals (Musgrove *et al.* 2013), representing about 29% of the West Siberia and NE/NW Europe population (1,500,000 individuals: AEWA 2012). Non-breeding wigeon in Great Britain are concentrated mainly in coastal areas, feeding on mudflats, coastal flooded grassland and saltmarsh pastures, as well as some inland flooded grasslands (Stroud *et al.* 2001).

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 2,660 individuals, representing an important component (more than 2,000 individuals) of the waterbird assemblage. During the winter, wigeon are found in greatest numbers on the brackish and freshwater pools and adjacent saltmarsh and grasslands around Saltholme, Seaton Common and Greatham Creek.

Gadwall *Anas strepera*

The non-breeding population of gadwall in Great Britain is estimated to be 25,000 individuals (Musgrove *et al.* 2013), representing about 42% of the North-western Europe population (60,000 individuals: AEWA 2012). Non-breeding gadwall in Great Britain are found predominantly on freshwater habitats, especially shallow lakes, where they form small flocks (Lack, 1986).

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 428 non-breeding gadwall, representing 1.7% of the GB non-breeding population. During the winter, gadwall are found in particular concentrations in several locations around the North Tees Marshes.

[23] Unless otherwise stated the % of GB is calculated using GB population figures given in Musgrove et al. (2013)

Northern shoveler *Anas clypeata*

The non-breeding population of northern shoveler in Great Britain is estimated to be 18,000 individuals (Musgrove *et al.* 2013), representing about 45% of the North-west and Central Europe (winter) population (40,000 individuals: AEWA 2012). Non-breeding northern shoveler in Great Britain are found predominantly on freshwater habitats, especially shallow lakes, where they form small flocks.

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 180 non-breeding northern shoveler, representing 1.0% of the GB non-breeding population. During the winter, northern shoveler are found in greatest numbers in several locations around the North Tees Marshes.

Northern lapwing *Vanellus vanellus*

The non-breeding population of northern lapwings in Great Britain is estimated to be 620,000 individuals (Musgrove *et al.* 2013), representing about 9% of the Europe/Europe and North Africa population (6,750,000 individuals representing the midpoint of 5,100,000 to 8,400,000 individuals: AEWA 2012). It is the most widespread non-breeding wader in Britain (Lack, 1986) and large flocks can be found in a variety of habitats from estuaries and coastal marshes to inland reservoirs and fields.

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 3,892 non-breeding lapwings, representing an important component (more than 2,000 individuals) of the waterbird assemblage. During the winter, large flocks of roosting and foraging lapwings occur at Saltholme, Cowpen Marsh, Greatham Creek and Seaton Common.

Sanderling *Calidris alba*

The non-breeding population of sanderling in Great Britain is estimated to be 16,000 individuals (Musgrove *et al.* 2013), representing about 13% of the East Atlantic Europe/West and Southern Africa population (120,000 individuals: AEWA 2012). Non-breeding sanderling occur on estuaries and open coasts all around the UK, with major concentrations in North-west England and the Outer Hebrides (Stroud *et al*. 2001).

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 242 non-breeding sanderlings, representing around 1.5% of the GB population. During the winter, foraging sanderlings are found in greatest numbers on the wide sandy beaches at Redcar and Coatham Sands, with smaller numbers in Hartlepool Bay.

Herring gull *Larus argentatus*

The non-breeding population of herring gulls in Great Britain is estimated to be 730,000 individuals (Musgrove *et al*. 2013), representing about 72% of the Iceland and West Europe population (1,020,000 individuals representing the midpoint of 990,000 to 1,050,000 individuals: AEWA 2012). Non-breeding herring gulls are very widely distributed throughout lowland areas of Britain, with the highest concentrations near the coast. They occur on a wide variety of wetland locations, including roosting sites on inland reservoirs and near-shore waters and forage on intertidal habitats, fishery discards and refuse tips.

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 3,243 non-breeding herring gulls, representing an important component (more than 2,000 individuals) of the waterbird assemblage. During the winter, herring gulls congregate in large numbers on the intertidal and near-shore waters of Hartlepool Bay and on the open coast north of Hartlepool.

Black-headed gull *Chroicocephalus ridibundus*

The non-breeding population of black-headed gulls in Great Britain is estimated to be 2,200,000 individuals (Musgrove *et al*. 2013), representing about 52% of the West Europe/West Mediterranean and West Africa population (4,250,000 individuals representing a midpoint of 3,700,000 to 4,800,000 individuals: AEWA 2012). Non-breeding black-headed gulls are very widespread at both inland and coastal locations, being found on estuaries, reservoirs and gravel pits, farmland and refuse tips.

Between 2011/12 and 2015/16 the pSPA/Ramsar site (including the proposed extensions) supported an average of 2,273 non-breeding black-headed gulls, representing an important component (more than 2,000 individuals) of the waterbird assemblage. During the winter, black-headed gulls are found in greatest numbers on the intertidal habitats and near-shore waters of Bran Sands, Hartlepool Bay and the open coast north of Hartlepool, and the freshwater pools at Saltholme.

7. Qualifying and assemblage bird species' use of proposed terrestrial extensions to the SPA and Ramsar

Table 5 (below) shows the use of each of those WeBS count sector by waterbirds in the pSPA and Ramsar site where a site extension has been proposed using data from 2011/12 – 2015/16. Sectors regularly used by at least 5% of the total pSPA/Ramsar site population of a particular species are highlighted. The use of the 5% 'threshold' is arbitrary but consistent with assessments of the importance of prospective extensions to other sites in England. The sections below highlight the key ornithological interests in each of the extension areas that justify their proposed inclusion within an extended pSPA. The evidence that justifies the proposed inclusion within the extended pSPA of the tidal channel of the River Tees and associated docks and harbours as well as the wider area of open sea, due to their use by the two breeding tern populations, is presented elsewhere in this document (Sections 4.2.1 and 4.2.2).

7.1. Portrack Marsh

This proposed extension is not included within a WeBS core count sector. The area of pools, reedbed and swamp is regularly used by a number of non-breeding waterbirds, including 5% of the redshank of the pSPA/Ramsar site, which use the site for foraging, resting, and roosting (Sharples 2012). Two pairs of common tern nested on the Marsh in 2014 (A. Snape *pers comm*) and in 2015 a nesting raft was reinstalled in the freshwater pools.

7.2. Cowpen Marsh

Cowpen Marsh (all within WeBS core count sector 52418) is regularly used by at least 5% of the pSPA/Ramsar site totals of 22 (SPA) / 24 (Ramsar) species of non-breeding waterbirds, including 14% of the curlew *Numenius arquata*, 13% of the lapwing, 10% of the wigeon, 7% of the shoveler and 6% of the redshank, avocet and gadwall. These species use the pools and wet grassland primarily for foraging and also for roosting, particularly flocks of lapwings. In total the Cowpen Marsh WeBS sector holds 6% of the pSPA/Ramsar site assemblage of waterbirds. Numbers of waterbirds using the extension area are comparable to those present in the northern part of Cowpen Marsh currently within the SPA/Ramsar site, including particularly high numbers of wigeon, lapwing, curlew and herring gull (Ecology Consulting 2015). Cowpen Marsh also regularly supports an average of at least 15 pairs of breeding common terns which nest on rafts located on the freshwater pools and 2 pairs of avocets (Cleveland Bird Reports).

7.3. Number 4 Brinefield

Number 4 Brinefield (all within the Brinefield WeBS core count sector 52417) is regularly used by at least 5% of the pSPA/Ramsar site totals of 20 (SPA) / 21 (Ramsar) species of non-breeding waterbirds, including 22% of the dunlin, 19% of the teal, 10% of the shelduck, 5% of the lapwing, 11% of the curlew and 9% of the black-headed gulls. These species use the pools, saline lagoon and adjacent grassland largely for foraging. In total the Brinefield WeBS sector holds 5% of the pSPA/Ramsar site assemblage of waterbirds. Additionally, the saline lagoon in the north of the site is one of the most important locations in the SPA for nesting pied avocets (Cleveland Bird Reports). It has also supported an average of at least 29 pairs of nesting common terns during 2010-2014 (Cleveland Bird Reports).

7.4. Greenabella Marsh

Greenabella Marsh (all within WeBS core count sector 52414) is regularly used by at least 5% of the pSPA/Ramsar site totals of 22 (SPA) / 23 (Ramsar) species of non-breeding waterbirds, including 36% of the common redshank, 12% of the black-headed gulls and 11% of the northern shoveler and 13% of the curlew. Many of these species use the saline lagoon and pools and adjacent grassland, primarily for foraging. It is also a regular high tide roost for common redshanks. In total the Greenabella Marsh WeBS sector holds 6% of the pSPA and 5% of the Ramsar site assemblage of waterbirds. The extension area regularly supports many of these species, particularly teal and curlew (Ecology Consulting 2015). The saline lagoon also supported 20 breeding pied avocets in 2013 and 2014 (Cleveland Bird Reports).

7.5. Greatham Tank Farm

The Greatham Tank Farm extension (all within a single WeBS core count sector (this was originally Greatham Tank Farm 52415, but the sector boundary has been adjusted following the Greatham North managed realignment in May 2014, and is now Tank Farm 52435)) is regularly used by at least 5% of the pSPA/Ramsar site totals of 8 (SPA) / 9 (Ramsar) species of non-breeding waterbirds, including 7% of the wigeon, 5% of the curlew and 6% of the shoveler. These species use the pools and adjacent grassland, primarily for foraging. In total the Greatham Tank Farm / Tank Farm WeBS sector holds nearly 2% of the pSPA/Ramsar site assemblage of waterbirds.

7.6. Hartlepool Bay and foreshore

This proposed extension falls largely within the Hartlepool Bay WeBS core count sector 52402, with the southern tip covered by the northern part of the Seaton Sands WeBS core count sector 52411. The Hartlepool Bay WeBS core count sector, which is an area of intertidal sand and mud and near-shore water, is regularly used by at least 5% of the SPA/Ramsar site totals of 18 (both SPA and Ramsar) species of waterbirds, in addition to foraging little and common terns, including 18% of the sanderling, 32% of the common redshank, 38% of the great black-backed gulls *Larus marinus*, 36% of the herring gulls and 11% of the black-headed gulls. These species use the intertidal habitats and adjacent water at low and high tide for foraging and roosting. In total the Hartlepool Bay WeBS sector holds over 9% of the pSPA/Ramsar site assemblage of waterbirds.

7.7. Seal Sands brownfield extension

The extension area falls within the Seal Sands SW and Main WeBS core count sector 52422, but only forms a small part of it. Regular counts at both high and low tide from November 2014 to March 2015 established that this area is particularly important as a high tide roost for curlews, with the raised track and adjacent grassland to the south supporting a peak of 243 individuals in the winter of 2014/15 (Ecology Consulting 2015). This peak count is 23% of the average peak recorded for curlews in pSPA/Ramsar waterbird assemblage during 2011/12 - 2015/16 (1,073 individuals).

7.8. Vopak foreshore

This extension is not within a WeBS sector. Regular counts at both high and low tide from November 2014 to March 2015 established that this area of intertidal mudflat and rocky shore supported peak numbers greater than 5% of the pSPA/Ramsar site totals of five species of non-breeding waterbirds, with 6% of the lapwings and 11% of the herring gulls. Both species roost on the mudflats and herring gulls also use the area for foraging. Peak numbers of waterbirds present in 2014/15 represent nearly 1% of the pSPA/Ramsar site assemblage of waterbirds (Ecology Consulting 2015).

7.9. Bran Sands South

This extension falls within the larger Bran Sands South (52427) WeBS core count sector. The WeBS sector overall is regularly used by at least 5% of the pSPA/Ramsar site totals of 19 species of non-breeding waterbirds including 17% of the common redshank, 10% of the black-headed gulls and 19% of the herring gulls. In total the WeBS sector holds over 8% of the pSPA/Ramsar site assemblage of waterbirds. The extension includes Bran Sands lagoon and the intertidal mudflats of Dabholm Gut. Both areas have been counted separately from WeBS during 2009-2013 (Barber 2014) and support similar species to the larger WeBS sector. Each area supports significant numbers of particular species, with at least 5% of the pSPA/Ramsar site totals of 13 species of non-breeding waterbirds on the lagoon, including 26% of the shelduck, 12% of the teal and 5% of the common redshank, and at least 5% of the pSPA/Ramsar site totals of five species on Dabholm Gut, including 15% of the shelduck, 24% of the teal and 7% of the gadwall (Barber 2014).

7.10. Coatham Marsh

This proposed extension falls within the Coatham Marsh WeBS core count sector 52431. This area of freshwater wetland habitats is regularly used by at least 5% of the pSPA/Ramsar site totals of ten species of non-breeding waterbirds. These species forage and roost on the freshwater pools and adjacent grasslands. In total the Coatham Marsh WeBS sector holds over 1% of the pSPA/Ramsar site assemblage of waterbirds.

7.11. Coatham Lagoons

This proposed extension falls within the Quarries and Lagoons WeBS core count sector 52430. The WeBS sector is regularly used by 14% of the pSPA/Ramsar sites total of redshank. The pools provide roosting and foraging habitats. In total the Quarries and Lagoons WeBS sector holds over 1% of the pSPA/Ramsar site assemblage of waterbirds.

7.12. Greatham North / Saltern Wetlands

The intertidal area of the managed realignment area falls within the Saltern Wetlands WeBS core count sector (52434) while the freshwater pools formed from borrow pits are covered by the Saltern Borrow Pits WeBS core count sector (52436).

The managed realignment was relatively recent and WeBS counts for these two sectors are only available for the last two seasons of the 2011/12 – 2015/16 data period. The Saltern Wetlands WeBS sector is used by at least 5% of the SPA/Ramsar site totals of 18 species (both SPA and Ramsar) of waterbirds, including 9% of the pied avocet, 13% of the northern lapwing, 5% of the red knot and 9% of the black-headed gull. In total the WeBS sector supports 7% of the pSPA/Ramsar site assemblage of waterbirds. The Saltern Borrow Pits WeBS sector is used by at least 5% of the SPA/Ramsar site totals of 8 (SPA) / 9 (Ramsar) species of waterbirds, including 6% of the gadwall, 7% of the black-headed gull and 11% of the herring gull. In total the WeBS sector supports over 3% of the pSPA/Ramsar site assemblage of waterbirds.

7.13. Cowpen Bewley

The proposed extension includes the mitigation area at Cowpen Bewley which currently does not fall within a WeBS core count sector, however, it is agreed with the land owner that birds will be counted on Cowpen Bewley as part of an updated management plan from early 2018. Natural England recommend the land referred to in the planning documents as mitigation land (Port Clarence and Cowpen Bewley) are included in the pSPA extension as they are designed and predicted to be suitable compensatory habitat for Reclamation Pond. Recent ornithological records indicate that birds potentially displaced from Reclamation Pond are starting to use the mitigation areas and there is a reasonable expectation the areas will provide supporting habitat for SPA features and be part of the most suitable territory for the SPA species in the long term.

7.14 RSPB Saltholme

The core of the RSPB Saltholme reserve falls within four WeBS core count sectors: Saltholme Central (52433), Haverton Hole (north) (52501), Haverton Hole (south) (52502) and Saltholme Pools (52419). The majority of Haverton Hole (north) and Saltholme Pools WeBS sectors are contained within the current SPA/Ramsar with the extension broadly covered by the Saltholme Central and Haverton Hole (south) WeBS sectors.

The vast majority of breeding common tern in the SPA nest within RSPB Saltholme and undertake some of their foraging on pools within the reserve. In addition, pied avocet nest have recently started nesting and foraging within RSPB Saltholme.

The Saltholme Central WeBS core count sector, which is an area of wet grassland and pools, is regularly used by at least 5% of the SPA/Ramsar site totals of 27 (SPA) / 29 (Ramsar) species of waterbirds, including 17% of the wigeon, 13% of the gadwall, 14% of the shoveler, 12% of the northern lapwing, 19% of the ruff and 17% of the black-headed gull. In total the WeBS sector holds over 9% of the pSPA/Ramsar site assemblage of waterbirds. The Haverton Hole (south) WeBS core count sector covers the large southern pool of Haverton Hole. It is regularly used by at least 5% of the SPA/Ramsar site totals of 9 (SPA) / 10 (Ramsar) species of waterbirds, including 22% of the gadwall. In total the WeBS sector supports over 1% of the pSPA/Ramsar site assemblage of waterbirds.

Table 5: The numbers of each species of European and international importance and key waterbird assemblage species present in WeBS count sectors within the pSPA / Ramsar site which contain proposed extensions. (Figures in bold represent counts for which 5% or more of the total site population occurs in a particular sector).

Species (* Annex I species of European importance and internationally important migratory species)	Total SPA & Ramsar site peak mean 2011/12 – 2015/16 (individuals) [24]	**Number of individuals and proportion of the SPA/Ramsar site total present in each sector[25]**							
		Cowpen Marsh (sector 52418)		Brinefields (sector 52417)		Greenabella (sector 52414)		Greatham Tank Farm (sector 52415 / 52435)	
		No.	%	No.	%	No.	%	No.	%
Eurasian Wigeon	2,660	**267**	**10%**	70	3%	95	4%	**189**	**7%**
Gadwall	428	**25**	**6%**	19	4%	11	2%	6	1%
Northern Shoveler	180	**12**	**7%**	5	3%	**19**	**11%**	**10**	**6%**
Pied Avocet*	52[26]	**3**	**6%**	**27**	**52%**	**25**	**48%**	<1	<1%
Northern Lapwing	3,892	**523**	**13%**	**182**	**5%**	143	4%	81	2%
Red Knot*	876	0	0	1	<0.1%	11	1%	0	0
Sanderling	242	0	0	0	0	0	0	0	0
Ruff	19	<1	2%	**1**	**7%**	0	0	**2**	**9%**
Common Redshank*	881	55	6%	26	3%	**318**	**36%**	2	<1%
Black-headed Gull	2,273	16	1%	**202**	**9%**	**271**	**12%**	8	<1%
Herring Gull	2,820	33	1%	5	<0.2%	1	<0.1%	1	<0.1%
Little Tern*	23[27]	0	0	0	0	0	0	0	0
Sandwich Tern*	134	0	0	4	3%	0	0	0	0
Common Tern*	385[28]	**43**	**11%**	**56**	**14%**	5	1%	0	0

[24] This table lists only those species of particular interest in justifying the inclusion of these terrestrial extensions within the amended pSPA/Ramsar site boundary. The list of species tabulated is, therefore, not a comprehensive list of all the species that, across the pSPA/Ramsar site as a whole make up the waterbird assemblage. Accordingly, the sum of the species totals in the 2nd column do not match the overall waterbird assemblage totals.

[25] The sector data tabulated here does not include count data for WeBS sectors already contained entirely within the existing SPA boundary. Accordingly, the site totals in the 2nd column for individual species often exceed the sum of the figures across the sectors considered here (e.g. red knot, ruff). Conversely, individual sector counts are unconsolidated and for individual species may therefore exceed the site total if summed (e.g. common redshank), although they provide a useful measure of the sector value for certain species at peak times of the year.

[26] The peak mean count of pied avocet from 2011/12-2015/16 i.e. 52 individuals does not correspond with the numbers of breeding pairs as listed in Table 1 (i.e. 18 pairs). This is because the data in Table 1 (derived from Cleveland Bird Reports) is restricted to counts of pairs of breeding adults in the summer months whereas the WeBs count data presented here may also include non-breeders in summer months and/or other months of the year (i.e. March and April) and is also from a slightly different time period.

[27] The peak mean count of little tern for 2011/12-2015/16 (i.e. 23 individuals) presented here does not correspond with the numbers of breeding pairs as listed in Table 1 (i.e. 81 pairs). This is because the figure in Table 1 (derived from Cleveland Bird Reports) is of pairs of breeding adults in the summer months in the years 2010-2014. In contrast, the WeBs count data presented above includes counts in 2014, 2015 and 2016 made outside the peak breeding months of June and July (i.e. counts in May and in August). Furthermore, the span of years considered here 2011/12 – 2015/16 for the purposes of updating the waterbird assemblage differs from that (2010-2014) used to define the size of the population of the little tern qualifying feature in Table 1.

[28] The peak mean count of common tern for 2011/12-2015/16 (i.e. 385 individuals) presented here does not correspond exactly with the numbers of breeding pairs as listed in Table 1 (i.e. 399 pairs). This is because the figure in Table 1 (derived from Cleveland Bird Reports) is of pairs of breeding adults in the summer months in the years 2010-2014. In contrast, the WeBS count data presented above includes a count in 2015/16 for the purposes of updating the waterbird assemblage. The WeBS sector counts may also not include all pairs of common terns that breed at satellite colonies within the pSPA but which

SPA assemblage	26,014	**1,477**	**6%**	**1,266**	**5%**	**1,435**	**6%**	447	1.7%
Ramsar assemblage	26,786	**1,644**	**6%**	**1,281**	**5%**	**1,441**	**5%**	614	2%

Species (* Annex I species of European importance and internationally important migratory species)	Total SPA & Ramsar site peak mean 2011/12 – 2015/16	**Number of individuals and proportion of the SPA/Ramsar site total present in each sector**							
		Hartlepool Bay (sector 52402)		Bran Sands South (sector 52427)		Coatham Marsh (sector 52431)		Quarries and Lagoons (sector 52430)	
		No.	%	No.	%	No.	%	No.	%
Wigeon	2,660	0	0	<1	<0.1%	17	<1%	0	0
Gadwall	428	0	0	6	1%	16	4%	4	<1%
Shoveler	180	0	0	<1	<1%	6	3%	0	0
Pied Avocet*	52[29]	0	0	0	0	0	0	0	0
Northern Lapwing	3,892	44	1%	**228**	**6%**	0	0	32	<1%
Red Knot*	876	**193**	**22%**	1	<0.1%	0	0	0	0
Sanderling	242	**44**	**18%**	0	0	0	0	0	0
Ruff	19	0	0	0	0	0	0	0	0
Common Redshank*	881	**278**	**32%**	**151**	**17%**	0	0	**119**	**14%**
Black-headed Gull	2,273	**247**	**11%**	**230**	**10%**	71	3%	35	2%
Herring Gull	3,243	**1156**	**36%**	**616**	**19%**	41	1%	26	<1%
Little Tern*	23[30]	<1	3%	0	0	0	0	0	0
Sandwich Tern*	134	3	2%	4	3%	0	0	0	0
Common Tern*	385[31]	5	1%	17	4%	<1	<1%	0	0
SPA assemblage	26,014	**2,521**	**10%**	**2,220**	**9%**	305	1%	305	1%
Ramsar Assemblage	26,786	**2,521**	**9%**	**2,223**	**8%**	318	1%	307	1%

are included with the tallies in the Cleveland Bird Reports.
[29] As per footnote 25.
[30] AS per footnote 26.
[31] AS per footnote 27.

Species (* Annex I species of European importance and internationally important migratory species)	Total SPA & Ramsar site peak mean 2011/12 – 2015/16	**Number of individuals and proportion of the SPA/Ramsar site total present in each sector**							
		Saltholme Central (sector 52433)		Haverton Hole (south) (sector 52502)		Saltern Wetlands (sector 52434)[32]		Saltern Borrow Pits (sector 52436)[33]	
		No.	%	No.	%	No.	%	No.	%
Wigeon	2,660	**447**	**17%**	5	<1%	50	2%	28	1%
Gadwall	428	**56**	**13%**	**95**	**22%**	4	1%	**27**	**6%**
Shoveler	180	**26**	**14%**	6	3%	0	0	2	1%
Pied Avocet*	52[34]	**3**	**6%**	0	0	**5**	**9%**	1	1%
Northern Lapwing	3,892	**471**	**12%**	7	<1%	**491**	**13%**	116	3%
Red Knot*	876	1	<0.1%	0	0	**48**	**5%**	0	0
Sanderling	242	0	0	0	0	0	0	0	0
Ruff	19	**4**	**19%**	0	0	0	0	0	0
Common Redshank*	881	5	1%	1	<1%	37	4	1	<0.1%
Black-headed Gull	2,273	**390**	**17%**	44	2%	**195**	**9%**	**154**	**7%**
Herring Gull	3,243	8	<1%	1	<0.1%	34	1%	**362**	**11%**
Little Tern*	23[35]	0	0	0	0	0	0	0	0
Sandwich Tern*	134	1	<1%	0	0	0	0	0	0
Common Tern*	385[36]	**229**	**59%**	2	1%	1	<1%	0	0
SPA assemblage	26,014	**2,305**	**9%**	357	1%	**1805**	**7%**	818	3%
Ramsar Assemblage	26,786	**2,514**	**9%**	364	1%	**1807**	**7%**	831	3%

32 The habitat is from a recent managed realignment. Only based on two seasons (2014/15 + 2015/16)
33 As per footnote 32.
34 As per footnote 25.
35 AS per footnote 26.
36 AS per footnote 27.

8. Comparison with other sites in Great Britain

A comparison is presented in Table 6 of the populations of each named qualifying feature of the Teesmouth and Cleveland Coast pSPA with the largest breeding and/or overwintering populations supported by individual SPAs across Great Britain. For brevity, only the top 5 ranked sites are tabulated for each species, except where the Teesmouth and Cleveland Coast pSPA's position in the rank order is lower than this – in which case all sites down to that rank position are tabulated. In the case of pied avocet, common tern and ruff, which are proposed as new qualifying features of the pSPA, the figures for the pSPA populations are based on the most recent available 5 year means. The same is true for the breeding little tern. The updated (increased) figure for little tern has been used for this existing feature so as to ensure that the population size of the feature of the pSPA is based on data contemporary with the period during which surveys of foraging distribution were undertaken to inform the proposed revision to the site boundary (i.e. 2011 – 2013). For the other qualifying features i.e. Sandwich tern, red knot and common redshank, which are not new features of the pSPA, the populations at the time of original classification in 1995 of the Teesmouth and Cleveland Coast SPA are used in the comparison with other sites.

For the purposes of this comparison exercise, for each of the species for which the most recent available count data for the pSPA are used to define the size of the population of the qualifying feature (i.e. breeding: pied avocet, common tern and little tern and non-breeding ruff), the populations from each of the other individual SPAs with which these recent figures are compared here are the most recent population estimates presented in the third SPA review (Stroud *et al*. 2016). These are, therefore, more or less like-for-like comparisons based on recent count data. For the other species i.e. Sandwich tern, red knot and common redshank, for which the historical population estimates at the time of the classification in 1995 of the Teesmouth and Cleveland Coast SPA are retained as the size of the populations of the qualifying features of the pSPA, the populations from each of the other individual SPAs with which these historical figures are compared here are the older population estimates presented in the third SPA review (Stroud *et al*. 2016) i.e. those from the 1990s. These are also, therefore, more or less like-for-like comparisons, albeit based on historical count data. It is acknowledged that these are historical comparisons that may not be indicative of the current relative importance of the populations in the pSPA.

Table 6: Comparison with other British SPAs that support similar qualifying species.

Site/Species	Number of pairs (breeding species) / number of individuals (non-breeding species)[37,38,39]	% of population[40]
PIED AVOCET *Recurvirostra avosetta* (breeding)	Number of pairs (year)	% GB (1,500 pairs) (2006-10)
North Norfolk Coast	276 (2005-2009)	18.4
Humber Estuary	168 (2005-2009)	11.2
The Swale	138 (2005-2009)	9.2
Minsmere – Walberswick	126 (2005-2009)	8.4
Medway Estuary and Marshes	75 (2005-2009)	5.0
Alde-Ore Estuary	64 (2005-2009)	4.3
Foulness	31 (2005-2009)	2.1
Teesmouth and Cleveland Coast	**18 (2010-2014)**	1.2
COMMON TERN *Sterna hirundo* (breeding)	Number of pairs (year)	% GB (10,000 pairs) (2000)
Coquet Island	1,193 (2011)	11.9
Imperial Dock Lock, Leith	818 (2010)	8.2
Strangford Lough	726 (2011)	7.3
Ynys Feurig, Cemlyn Bay and The Skerries	592 (2011)	5.9
Teesmouth and Cleveland Coast	**399 (2010-2014)**	**4.0**
SANDWICH TERN *Thalasseus sandvicensis* (passage)	Number of individuals (year)	% GB (1988-92)
Teesmouth and Cleveland Coast	**1,900** **(134: 2011/12 – 2015/16)[41]**	**4.3%**
Firth of Forth	1,611	3.6%
The Dee Estuary	818	1.8%
LITTLE TERN *Sternula albifrons* (breeding)	Number of pairs (year)	% GB (1,900 pairs) (2000)
North Norfolk Coast	409 (2011)	21.5
The Dee Estuary	126 (2011)	6.6
Blackwater Estuary	99 (2000)	5.2
Teesmouth and Cleveland Coast	**81 (2010-2014)**	**4.3**

[37] The principal data sources for site-specific population figures used by Stroud *et al.* (2016) were as follows: for breeding seabirds - UK Seabird Monitoring Partnership including data from the *Seabird 2000* national census; breeding pied avocet – Rare Breeding Birds Panel; non-breeding birds - Wetland Bird Survey (WeBS - for wintering waterbirds) for the period 2005/06 – 2009/10. In the case of non-breeding red knot and common redshank for which historical site population estimates, first presented in Stroud *et al.* (2001) and repeated in Stroud *et al*. (2016), are used here, the data source is Wetland Bird Survey (WeBS - for wintering waterbirds) for the period 1992/93 – 1996/97.
[38] Note that these rankings should only be considered indicative of the current relative importance of the pSPA as in no case is the comparison between sites based on exactly contemporary count data from the most recent years at every site.
[39] These rank orders do not take account of numbers of each species supported within several very recently classified SPAs and certain other pSPAs which are not yet classified and hence not included within Stroud *et al* (2016).
[40] National GB population figures from which the site specific % values have been calculated are taken from the species accounts provided in Appendix 9 of Stroud *et al*. (2016) (with the exception of Sandwich tern for which the 4.3% figure is the figure documented in Carter (1993) using the national passage population estimate at that time). Biogeographic population figures from which site specific % values for red knot and common redshank have been calculated are also taken from the species accounts provided in Appendix 9 of Stroud *et al*. (2016) (i.e. from AEWA (2012)).
[41] The headline figure retained for Sandwich tern is the historical one. For comparison, the most recent population estimates for the pSPA are also given in parentheses.

Morecambe Bay	62 (2011)	3.3
RED KNOT *Calidris canutus* (non-breeding)	Number of individuals (year)	% biogeographic (450,000 individuals) (1997-2007)
The Wash	186,892 (1992/3 – 1996/7)	41.5%
Ribble and Alt Estuaries	57,865 (1992/3 – 1996/7)	12.9%
Foulness	40,429 (1992/3 – 1996/7)	9.0%
Humber Estuary	33,848 (1992/3 – 1996/7)	7.5%
Morecambe Bay	29,426 (1992/3 – 1996/7)	6.5%
The Dee Estuary	21,553 (1992/3 – 1996/7)	4.8%
Upper Solway Flats and Marshes	12,271 (1992/3 – 1996/7)	2.7%
North Norfolk Coast	10,801 (1992/3 – 1996/7)	2.4%
Gibraltar Point	10,155 (1992/3 – 1996/7)	2.3%
Benfleet and Southend Marshes	8,850 (1992/3 – 1996/7)	2.0%
Strangford Lough	8,723 (1992/3 – 1996/7)	1.9%
Dengie	8,393 (1992/3 – 1996/7)	1.9%
Firth of Forth	8,013 (1992/3 – 1996/7)	1.8%
The Swale	5,582 (1992/3 – 1996/7)	1.2%
Teesmouth and Cleveland Coast	**5,509 (1991/92 – 1995/96) (876: 2011/12 - 2015/16)**[42]	1.2%
COMMON REDSHANK *Tringa totanus* (non-breeding)	Number of individuals (year)	% biogeographic (275,000 individuals) (1998-2008)
The Dee Estuary	8,451 (1992/3 – 1996/7)	3.1%
Morecambe Bay	6,336 (1992/3 – 1996/7)	2.3%
Humber Estuary	5,212 (1992/3 – 1996/7)	1.9%
Mersey Estuary	4,689 (1992/3 – 1996/7)	1.7%
Blackwater Estuary	4,015 (1992/3 – 1996/7)	1.5%
Firth of Forth	3,700 (1992/3 – 1996/7)	1.3%
Strangford Lough	3,176 (1992/3 – 1996/7)	1.2%
Medway Estuary and Marshes	3,690 (1992/3 – 1996/7)	1.3%
Stour and Orwell Estuaries	3,545 (1992/3 – 1996/7)	1.3%
Upper Solway Flats and Marshes	3,088 (1992/3 – 1996/7)	1.1%
North Norfolk Coast	2,998 (1992/3 – 1996/7)	1.1%
The Wash	2,953 (1992/3 – 1996/7)	1.1%
Ribble and Alt Estuaries	2,708 (1992/3 – 1996/7)	1.0%
Belfast Lough	2,466 (1992/3 – 1996/7)	0.9%
Severn Estuary	2,330 (1992/3 – 1996/7)	0.8%
Duddon Estuary	2,289 (1992/3 – 1996/7)	0.8%
Montrose Basin	2,259 (1992/3 – 1996/7)	0.8%
Foulness	2,144 (1992/3 – 1996/7)	0.8%
Colne Estuary	2,077 (1992/3 – 1996/7)	0.8%
Alde-Ore Estuary	1,919 (1992/3 – 1996/7)	0.7%
Inner Clyde Estuary	1,918 (1992/3 – 1996/7)	0.7%
Inner Moray Firth	1,811 (1992/3 – 1996/7)	0.7%
Firth of Tay and Eden Estuaries	1,800 (1992/3 – 1996/7)	0.7%
Chichester and Langstone Harbours	1,788 (1992/3 – 1996/7)	0.7%
Moray and Nairn Coast	1,690 (1992/3 – 1996/7)	0.6%

[42] The headline figure retained for red knot is the historical one. For comparison, the most recent population estimates for the pSPA are also given in parentheses

Teesmouth and Cleveland Coast	**1,648 (1987-1991) (881: 2011/12 – 2015/16)**[43]	0.6%
RUFF *Calidris pugnax* (non-breeding)	Number of individuals (year)	% GB (800 individuals) (2004/5 – 2008/9)
North Norfolk Coast	142 (2005/6 – 2009/10)	17.8
Ouse Washes	122 (2005/6 – 2009/10)	15.3
Lower Derwent Valley	100 (2005/6 – 2009/10)	12.5
Humber Estuary	64 (2005/6 – 2009/10)	8.0
Broadland	42 (2005/6 – 2009/10)	5.3
Nene Washes	37 (2005/6 – 2009/10)	4.6
Teesmouth and Cleveland Coast	19 (2011/12 – 2015/16)	2.4

9. Conclusion

It can be seen from the evidence presented above that Teesmouth and Cleveland Coast pSPA is in the top 10 sites across the UK for all the proposed features (apart from red knot where it is 15th and common redshank 26th based on comparison of like-for-like but historical figures) and that the site features meet the required selection criteria for classification as an SPA. The overall boundary of the Teesmouth and Cleveland Coast 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 breeding terns which are already qualifying features of the existing SPA (or are proposed to be added as new qualifying features to it) and to include several other terrestrial extensions which have been shown by WeBS and other count data to support significant numbers of many of the site's features.

The conclusions regarding the drawing of the seaward boundary of the marine extension 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 conclusions regarding the drawing of the landward boundary of the marine extension of the pSPA, and the inclusion within it of the River Tees as far upstream as the tidal barrage and of various marinas, docks etc., are also based upon the same modelled evidence base and adherence to the guidelines for the selection of marine SPAs (Webb & Reid 2004) in setting this boundary at the mean high water mark. The validity and robustness of the outputs of the site specific and generic models used to underpin the boundary analysis of the pSPA have been established by the process of cross-validation described in Annex 5 and by additional field survey work in 2015 and 2016 described in Annex 6. Thus, the conclusions in all respects clearly relate to the best available analysis of the best available evidence.

[43] The headline figure retained for common redshank is the historical one. For comparison, the most recent population estimates for the pSPA are also given in parentheses

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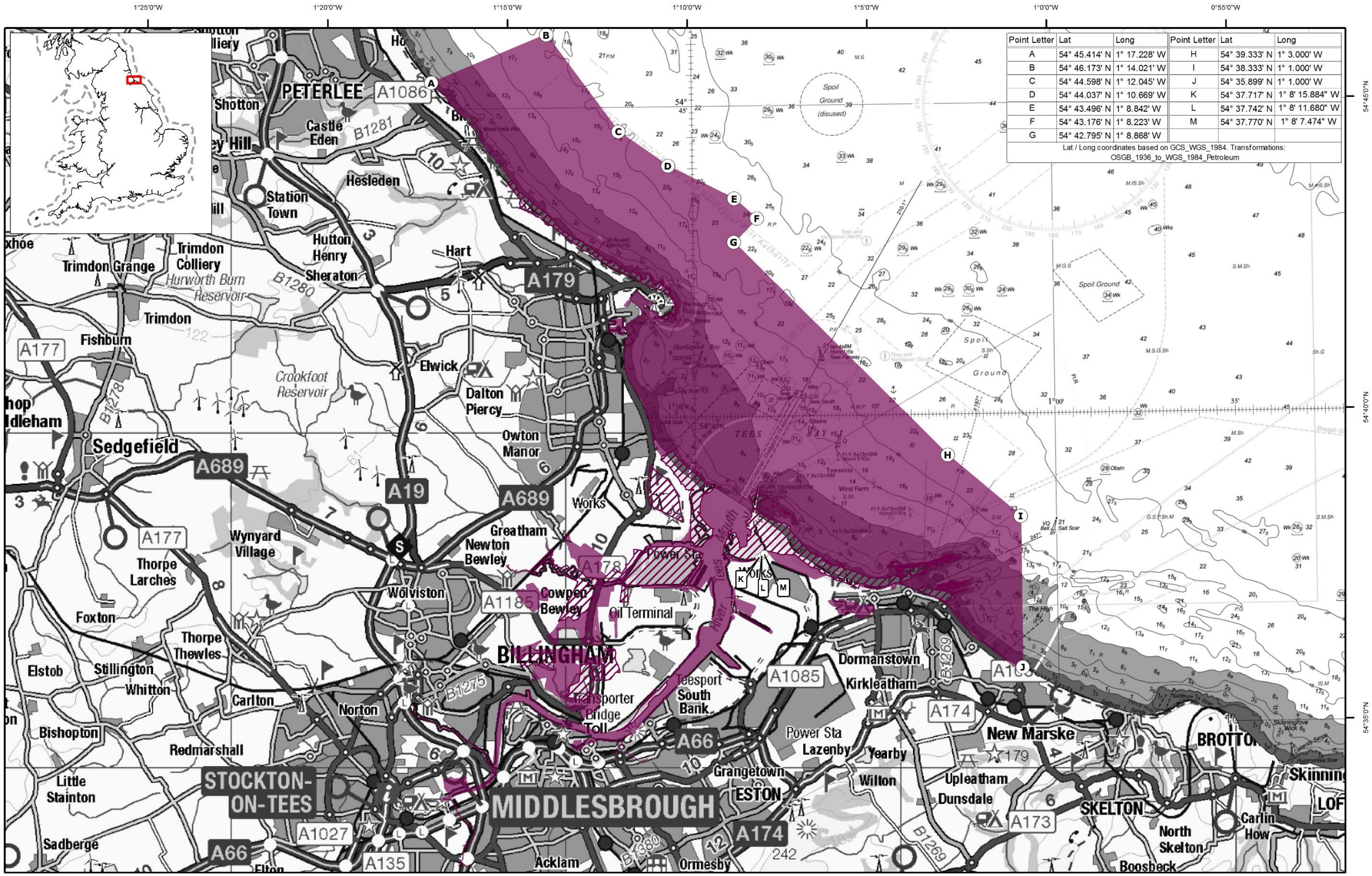
Annex 1: Map 1. Displaying the existing Teesmouth and Cleveland Coast SPA and proposed pSPA extension



Point Letter	Lat	Long	Point Letter	Lat	Long
A	54° 45.414' N	1° 17.228' W	H	54° 39.333' N	1° 3.000' W
B	54° 46.173' N	1° 14.021' W	I	54° 38.333' N	1° 1.000' W
C	54° 44.598' N	1° 12.045' W	J	54° 35.899' N	1° 1.000' W
D	54° 44.037' N	1° 10.669' W	K	54° 37.717' N	1° 8' 15.884" W
E	54° 43.496' N	1° 8.842' W	L	54° 37.742' N	1° 8' 11.680" W
F	54° 43.176' N	1° 8.223' W	M	54° 37.770' N	1° 8' 7.474" W
G	54° 42.795' N	1° 8.868' W			

Lat / Long coordinates based on GCS_WGS_1984. Transformations: OSGB_1936_to_WGS_1984_Petroleum

Special Protection Area Extension

Teesmouth and Cleveland Coast

Location Map

Map 1 of 1

Note:
The boundary shown is derived from large scale Ordnance Survey MasterMap, may not match the background detail on this map and is liable to change. Boundary coordinates indicate points below Mean High Water (Springs). Terrestrial limit coordinates are not plotted. Definitive larger scale maps are available on request.

Special Protection Area Extension 10974.78 Hectares

Area already classifed as SPA 1251.50 Hectares

SPA Boundary Coordinates

12NM Territorial Seas Limit

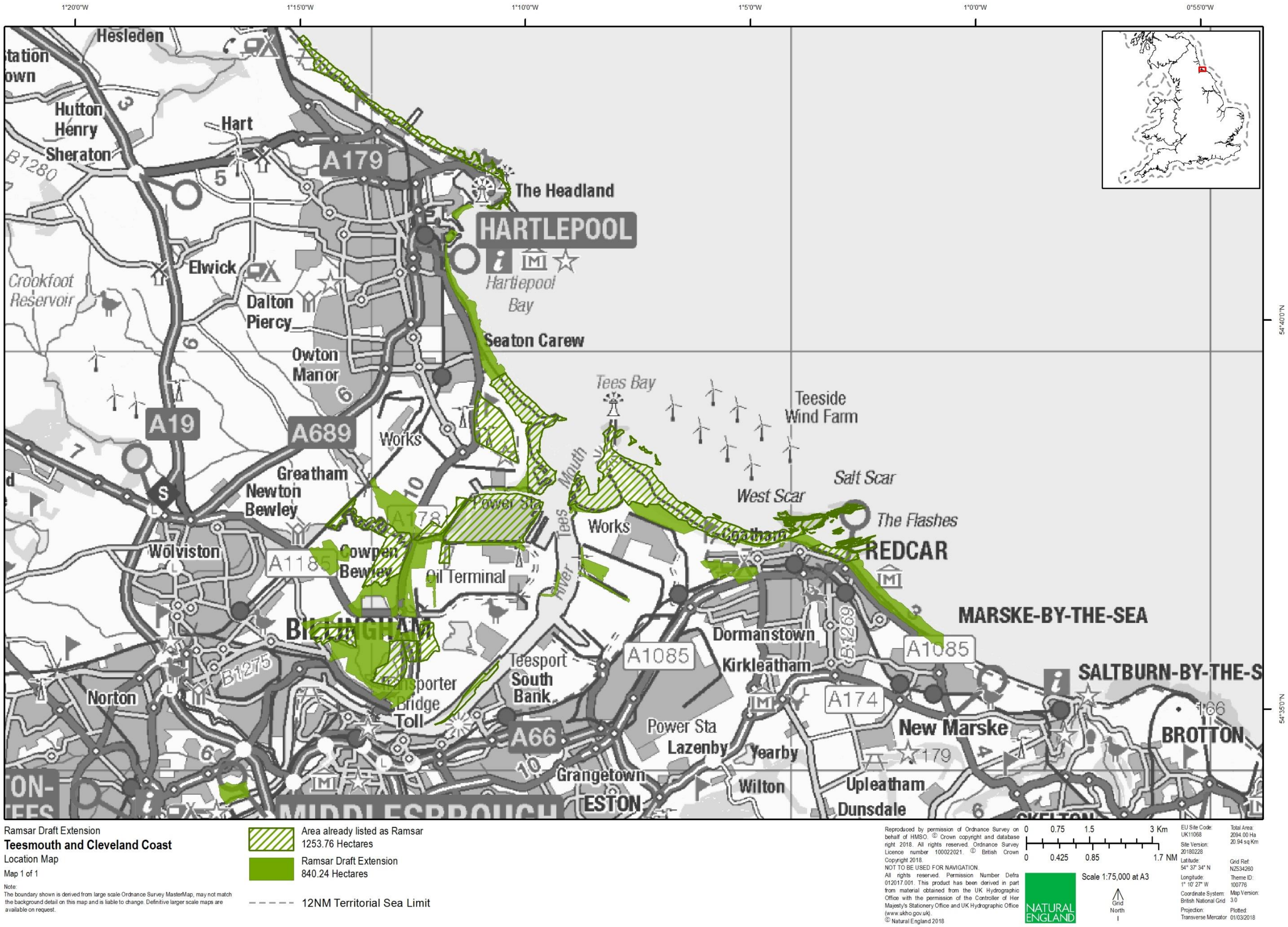
0 1 2 4 Km
0 0.5 1 2 NM

Scale 1:100,000 at A3

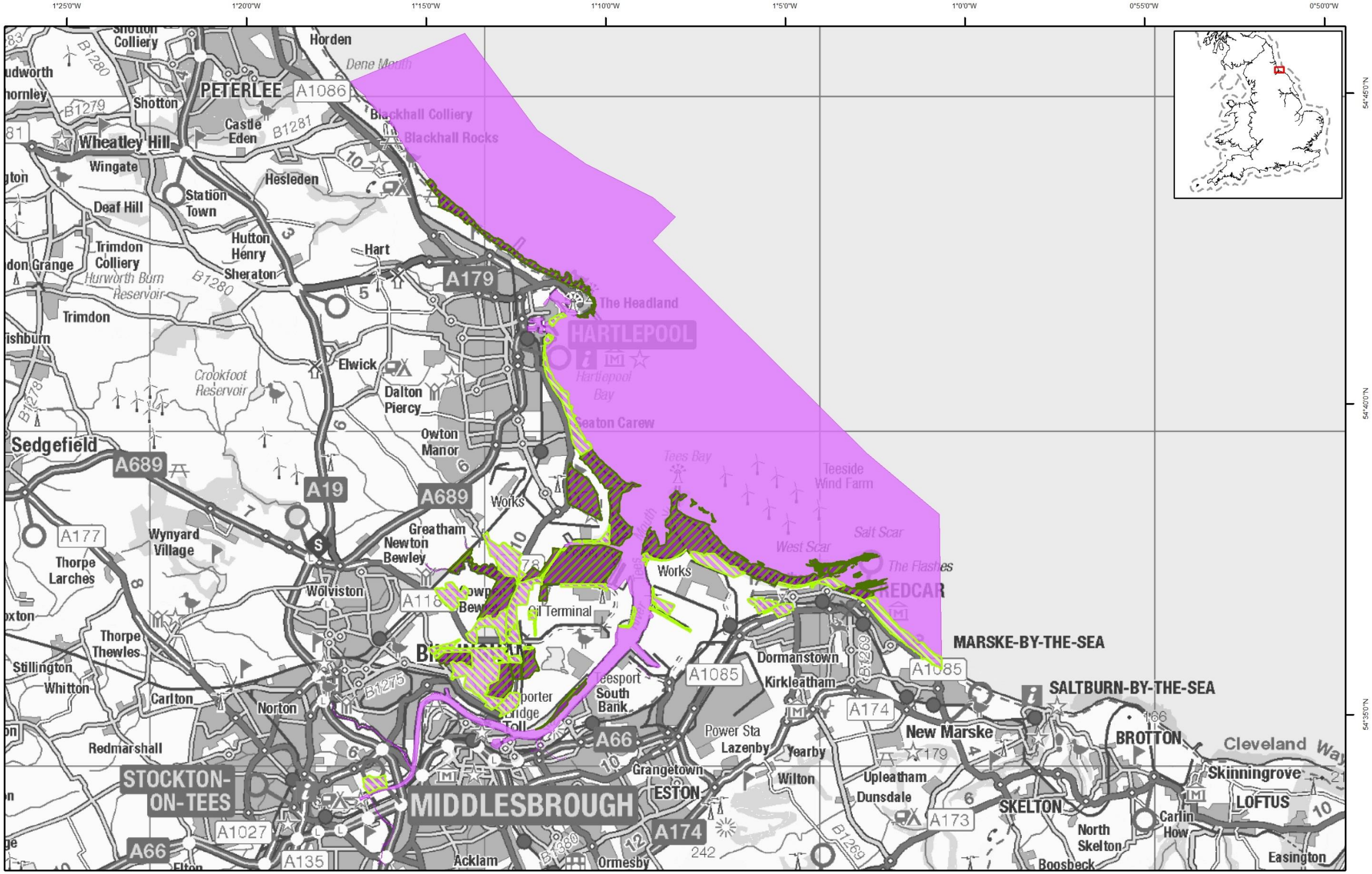
Grid North

EU Site Code: UK9006061
Site Version: 20180228
Latitude: 54° 40' 24" N
Longitude: 1° 8' 58" W
Coordinate System: British National Grid
Projection: Transverse Mercator
Area of MarineSPA: 10974.78 Ha 109.75 sq Km
Grid Ref:: NZ549312
Theme ID: 999993
Map Version: 7.0
Plotted: 28/02/2018

Annex 1: Map 2. Displaying existing Ramsar site and proposed Ramsar extension



Annex 1: Map 3. Displaying the existing SPA and Ramsar sites with proposed extension area



Ramsar Draft Extension

Teesmouth and Cleveland Coast

Boundary Map

Map 1 of 1

Note:
The boundary shown is derived from large scale Ordnance Survey MasterMap, may not match the background detail on this map and is liable to change. Definitive larger scale maps are available on request.

- Ramsar Draft Extension - 840.24 Hectares
- Area already listed as Ramsar - 1253.76 Hectares
- SPA Draft Extension - 10974.78 Hectares
- Area already classified as SPA - 1251.50 Hectares

0 1 2 4 Km
0 0.5 1 2 NM

NATURAL ENGLAND

Scale 1:100,000 at A3

Grid North

EU Site Code: UK11068
Area of RAMSAR: 2094.00 Ha 20.94 sq Km
Site Version: 20180228
Latitude: 54° 37' 34" N
Grid Ref: NZ534260
Longitude: 1° 10' 27" W
Theme ID: 100776
Coordinate System: British National Grid
Map Version: 3.0
Projection: Transverse Mercator
Plotted: 08/03/2018

Annex 2: SPA Citation

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

Name: Teesmouth and Cleveland Coast pSPA

Counties/Unitary Authorities: Durham County Council, Hartlepool Borough Council, Redcar & Cleveland Borough Council, Stockton-on-Tees Borough Council.

The pSPA is largely located between Hartlepool, Middlesbrough and Redcar. It lies within the three Unitary Authorities of Hartlepool, Stockton-on-Tees and Redcar & Cleveland. Its marine extension lies entirely within UK territorial waters.

Boundary of the SPA:

The extended pSPA terrestrial boundary protects habitats for wintering waterbirds including intertidal, wet grassland, mudflats and open water habitats. The original SPA included all or parts of Seal Sands SSSI; Seaton Dunes and Common SSSI; Cowpen Marsh SSSI; Redcar Rocks SSSI and South Gare and Coatham Sands SSSI. The Teesmouth and Cleveland Coast pSPA includes most of Teesmouth and Cleveland Coast SSSI (formerly Seal Sands SSSI; Seaton Dunes and Common SSSI; Cowpen Marsh SSSI; Redcar Rocks SSSI and South Gare and Coatham Sands SSSI, Tees and Hartlepool Foreshore and Wetlands SSSI and parts of Durham Coast SSSI).

The boundary of the pSPA also covers an area of open sea from Castle Eden Denemouth in the north to Marske-by-the-Sea in the south and includes the River Tees and associated docks, harbours etc. as far upriver as the Tees Barrage. The seaward boundary includes waters out to around 3.5km from Crimdon Dene to include the areas of greatest importance to the little terns at that colony and out to around 6km offshore further south to include the areas of greatest importance to the common terns at the Saltholme colony.

Size of SPA: 12,226.28 ha

The extension covers an area of 10,974.78 ha, giving a revised SPA area of 12,226.28 ha

Site description:

The Teesmouth and Cleveland Coast pSPA is a wetland of European importance, comprising a wide variety of habitats including: intertidal sand and mudflats, rocky shore, saltmarsh, freshwater marsh, sand dunes and estuarine and coastal waters on and around the estuary which has been considerably modified by human activities These habitats provide feeding and roosting opportunities for important number of waterbirds in winter and during passage periods including in particular common redshank, red knot and ruff which occur in internationally important numbers. The areas of mudflats contain worms, molluscs, crustaceans and other invertebrates which are an important food source for migrant birds. In summer, little tern breed on the sandy beaches within the site and feed out at sea while the common terns, which breed at various locations, feed within the River Tees and associated water bodies and within the wider estuary mouth and bay and pied avocets breed within the saline lagoons. In late summer, Sandwich tern aggregate in important numbers at Coatham Sands, Seal Sands, North Gare Sands/Seaton Snook and Bran Sands when on passage.

The saltmarsh and mudflat habitats of the Teesmouth and Cleveland Coast pSPA are of great importance to a diverse assemblage of bird species. Mudflats support high densities of benthic invertebrates, including worms, clams and crustaceans, which provide an important food resource for the majority of the migratory SPA bird species in the winter. Areas of saltmarsh provide significant feeding and roosting opportunities for many species of waterbird including common redshank and red knot.

The pSPA is located on the coast of north-east England between Castle Eden Denemouth in the north and Marske-by-the-Sea in the south. It includes the little tern colony at Crimdon Dene and the common tern colony at Saltholme.

The coastal parts of the site include a rocky limestone headland with sandy beaches stretching to the north, and much of Tees Bay to the south. The saline pools are located within the dunes at Coatham Lagoons south of South Gare. South of Hartlepool, the magnesian limestone is replaced by sandstones and mudstones as far as Saltburn creating low cliffs and sandy beaches.

Qualifying species:

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

- The site regularly supports more than 1% of the Great Britain populations of five species listed in Annex I of the EC Birds Directive. Therefore the site qualifies for SPA Classification in accordance with the UK SPA selection guidelines (stage 1.1).

Species	Count (period)	% of population	Interest type
Pied avocet *Recurvirostra avosetta*	18 pairs[44] (2010-2014)	1.2% GB[45]	Annex I
Sandwich tern *Thalasseus sandvicensis*	1,900 individuals[46] (1988-1992)	4.3% GB[47]	Annex I
Common tern *Sterna hirundo*	399 pairs[48] (2010-2014)	4.0% GB[49]	Annex I
Little tern *Sternula albifrons*	81 pairs[50] (2010-2014)	4.3% GB[51]	Annex I
Ruff *Caldris pugnax*	19 individuals (2011/12-2015/16)[52]	2.4% GB[53]	Annex I

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

Species	Count (period)	% of population	Interest type

[44] Data from: Cleveland Bird Reports. Note this figure refers to the 'breeding pairs of avocet' over a 5 year mean (2010-2014) which equates to an average of 18 breeding pairs
[45] Data from: Musgrove *et al.* 2013; 1,500 pairs (2006-10)
[46] Data from: Carter 1993, SPA Departmental Brief; recent average of 134 individuals (WeBS: 2011/12-2015-16) representing 0.3% of GB
[47] Data from: Carter 1993, SPA Departmental Brief. Note: this passage population of 1,900 individuals was expressed as equating to 6.8% of the GB breeding population of Sandwich terns (14,000 pairs) in the Natura 2000 Standard Data Form for this site.
[48] Data from: Cleveland Bird Reports
[49] Data from: Musgrove *et al.* 2013; 10,000 pairs (2000).
[50] Data from: Cleveland INCA little tern monitoring reports
[51] Data from: Musgrove *et al.* 2013: 1,900 pairs (2000)
[52] Data from: WeBs 2011/12-2015/16
[53] Data from: Musgrove et al. 2013: 800 individuals

Red knot *Calidris canutus*	5,509 individuals[54] (1991/92-1995/96)	1.6% NE Canada/ Greenland/Iceland/UK population[55]	Migratory
Common redshank *Tringa totanus*	1,648 individuals[56] (1987-1991)	1.1% East Atlantic population[57]	Migratory

Assemblage qualification:

The site qualifies under **Article 4** of the Birds Directive (2009/147/EC) as it used regularly by over 20,000 waterfowl (waterfowl as defined by the Ramsar Convention) or 20,000 seabirds in any season (Table 2)

	Count (period)	Average number of individuals
Waterbird assemblage	2011/12-2015/16	216,014 [44, 58]

During the period 2011/12-2015/16 the Teesmouth and Cleveland Coast pSPA/Ramsar site, supported an average peak of 26, 014 (SPA assemblage) / 26,786 (Ramsar assemblage) individuals. This assemblage is of both European and international importance. The assemblage includes a wide range of breeding, wintering and passage waterbird species, including those of European importance described above, as well as numbers exceeding 1% of the GB non-breeding populations of gadwall *Anas strepera*, northern shoveler *Anas clypeata* and sanderling *Calidris alba*. Additionally, Eurasian wigeon *Anas penelope*, northern lapwing *Vanellus vanellus*, herring gull *Larus argentatus* and black-headed gull *Chroicocephalus ridibundus* are also present in sufficient numbers to warrant their being listed as a major component species of the assemblage, as their numbers exceed 2,000 individuals (10% of the minimum qualifying assemblage of 20,000 individuals).

Principal bird data sources:

AEWA – African-Eurasian Waterbird Agreement (2012). *Report on the Conservation Status of Migratory Waterbirds in the Agreement Area. Fifth Edition*. AEWA, Bonn.

Carter, I. (1993). Teesmouth and Cleveland Coast Departmental Brief. JNCC, Peterborough.

Cleveland Birds Reports (2010, 2011, 2012, 2013). Teesmouth Bird Club.

Cleveland Industry Nature Conservation Association little tern monitoring data 1995–2013.

Musgrove, M., Aebischer, N., Eaton, M., Hearn, R., Newson, S., Noble, D., Parsons M., Risely K., & Stroud, D. (2013). Population estimates of birds in Great Britain and the United Kingdom. British Birds **106**: 64 –100.

Wetland Bird Survey reports (2009/10 - 2013/14). British Trust for Ornithology.

Wetlands International (2012). *Waterbird Population Estimates, Fifth Edition*. Wetlands International, Wageningen, The Netherlands.

[54] Data from: SPA citation March 2000 version 0.4; recent average of 876 individuals (WeBS: 2011/12-2015/16) representing 0.2% of NE Canada & Greenland/Western Europe population (AEWA 2012)
[55] Data from: Wetlands International 2012; 345,000 individuals 1982-1992
[56] Data from: Carter 1993, SPA Departmental Brief; recent average of 881 individuals (WeBS: 2011/12-2015/16) representing 0.3% of the Iceland & Faroes/Western Europe population (AEWA 2012).
[57] Data from: Carter 1993, SPA Departmental Brief
[58] Greylag goose and mute swan are not included in the SPA assemblage because they are not migratory populations

Annex 3: Ramsar Citation

Ramsar Convention on
Wetlands of International Importance Especially as Waterfowl Habitat

Name: Teesmouth and Cleveland Coast

Unitary Authority/County: Durham County Council, Hartlepool Borough Council, Redcar & Cleveland Borough Council and Stockton-on-Tees Borough Council.

Status of Ramsar site: Teesmouth and Cleveland Coast was first designated as a Ramsar site on 15 August 1995. On 31 March 2000, the Ramsar site was extended to include additional areas and intertidal habitats. The current proposal (2017) to further extend the Ramsar site includes additional terrestrial areas within the Tees estuary and along the foreshore to the north and south because of the site's international importance for waterbirds.

Site description: The Teesmouth and Cleveland Coast Ramsar site is a wetland of international importance, comprising intertidal sand and mudflats, rocky shore, saltmarsh, freshwater marsh and sand dunes. Large numbers of waterbirds feed and roost on the site in winter and during passage periods. In addition, the site is internationally important for its populations of red knot *Calidris canutus*, common redshank *Tringa totanus* and Sandwich tern *Thalasseus sandvicensis*.

Boundary of Ramsar site: The original Ramsar site boundary included all or parts of Seal Sands SSSI; Seaton Dunes and Common SSSI; Cowpen Marsh SSSI; Redcar Rocks SSSI and South Gare and Coatham Sands SSSI. The extended area includes parts of Durham Coast SSSI and all of Tees and Hartlepool Foreshore and Wetlands SSSI. The SSSIs have been re-notified to create Teesmouth and Cleveland Coast SSSI. For the original and extended Ramsar boundary please refer to the site map.

The extended area includes additional terrestrial wet grassland, saltmarsh, deep and shallow pools and intertidal areas for breeding and non-breeding birds.

Size of Ramsar site: 2094.00 ha

The area of the original site was 942.56 ha. The extension in 2000 added an area of 304.75 ha, giving a revised Ramsar site area of 1247.31 ha. This latter figure has been re-calculated as a result of precision accuracy improvements (PAI) in 2017 as 1253.76 ha. The extension proposal would add an area of 840.24 ha to the site.

International importance of Ramsar site: The Teesmouth and Cleveland Coast Ramsar site is a Wetland of International Importance because:

a) The site qualifies under Ramsar **criterion 5** as it is regularly used by over 20,000 waterbirds in any season:

Count (period)	Season	Average number of individuals
2011/12-2015/16	Wintering	26,786[59]

b) The site qualifies under **criterion 6** as it is regularly used by 1% or more of the biogeographic populations of the following bird species, in any season:

Species	Count (period)	% of population

[59] Data from; WeBs 2011/12-2015/16

Red knot *Calidris canutus*	5,509 individuals[60] (1991/92-1995/96)	1.6% NE Canada/ Greenland/Iceland/UK population[61]
Common redshank *Tringa totanus*	1,648 individuals[62] (1987-1991)	1.1% East Atlantic population[63]
Sandwich tern *Thalasseus sandvicensis*	1,900 individuals[64] (1988-1992)	1.3% Western Europe/Western Africa[65]

Non-qualifying species of interest: The site supports nationally important breeding populations of: little terns *Sterna albifrons* (4.3% GB, 2010-2014), common terns *Sterna hirundo (*4.0% GB, 2010-2014), and pied avocet (1.2% GB, 2010-2014) and of non-breeding ruff *Calidris pugnax* (2.4% GB, 2011-2016). The assemblage includes a wide range of breeding, wintering and passage waterbird species, including those of European importance described above, as well as numbers exceeding 1% of the GB non-breeding populations of gadwall *Anas strepera*, northern shoveler *Anas clypeata*, and sanderling *Calidris alba.* Additionally, Eurasian wigeon *Anas penelope*, northern lapwing *Vanellus vanellus*, herring gull *Larus argentatus* and black-headed gull *Chroicocephalus ridibundus* are also present in sufficient numbers to warrant their being listed as a major component species of the assemblage, as their numbers exceed 2,000 individuals (10% of the minimum qualifying assemblage of 20,000 individuals). The site supports a rich assemblage of invertebrates, including the following seven Red Data Book species: *Pherbellia grisescens, Thereva valida, Longitarsus nigerrimus, Dryops nitidulus, Macroplea mutica, Philonthus dimidiatipennis* and *Trichohydnobius suturalis.*

Principal bird data sources:

AEWA – African-Eurasian Waterbird Agreement (2012). *Report on the Conservation Status of Migratory Waterbirds in the Agreement Area. Fifth Edition*. AEWA, Bonn.

Carter, I. (1993). Teesmouth and Cleveland Coast Departmental Brief. JNCC, Peterborough.

Cleveland Birds Reports (2010, 2011, 2012, 2013). Teesmouth Bird Club.

Cleveland Industry Nature Conservation Association little tern monitoring data

Musgrove, M., Aebischer, N., Eaton, M., Hearn, R., Newson, S., Noble, D., Parsons M., Risely K., & Stroud, D. (2013). Population estimates of birds in Great Britain and the United Kingdom. British Birds **106**: 64 –100.

Wetland Bird Survey reports (2009/10 - 2013/14). British Trust for Ornithology.

Wetlands International (2012). *Waterbird Population Estimates, Fifth Edition*. Wetlands International, Wageningen, The Netherlands.

[60] Data from: SPA citation March 2000 version 0.4; recent average of 876 individuals (WeBS: 2011/12-2015/16) representing 0.2% of NE Canada & Greenland/Western Europe population (AEWA 2012)
[61] Data from: Wetlands International 2012; 345,000 individuals 1982-1992
[62] Data from: Carter 1993, SPA Departmental Brief; recent average of 881 individuals (WeBS: 2011/12-2015/16) representing 0.3% of the Iceland & Faroes/Western Europe population (AEWA 2012).
[63] Data from: Carter 1993, SPA Departmental Brief
[64] Data from: Carter 1993, SPA Departmental Brief; recent average of 134 individuals (WeBS: 2011/12-2015/16) representing 0.3% of GB
[65] Data from: Ramsar citation March 2000 version 0.4.

Annex 4: Data Sources

Source of Data	**Data provider**	**Subject**	**Date produced**	**Method of data collection**	**Verification**
Wetland Bird Survey (WeBS) land-based surveys	WeBS partners	Non-breeding Sandwich tern, red knot, common redshank and ruff. Waterbird assemblage	2011/12-2015/16	Standard methodology: land-based counts within defined count sectors	WeBS operates a systematic data verification procedure
Cleveland Bird Report	County Bird Recorder	Breeding pied avocet and common tern	2010-2013	Submission of ad hoc records by nature reserve managers, bird-watchers	All records checked and verified by County Bird Recorder
Wintering Bird Surveys 2014/15	Ecology Consulting	Non-breeding red knot, common redshank and ruff. Waterbird assemblage. Selected terrestrial areas.	2014/15	Systematic high and low tide counts from vantage points	SSSI Responsible Officer
Bird data for Dabholm Gut, Bran Sand Lagoon and the river frontage	Industry and Nature Conservation Association	Non-breeding Sandwich tern, red knot, common redshank and ruff. Waterbird assemblage. Dabholme Gut and Bran Sands Lagoon	2009-2013	Sequential monthly counts of each survey area	Not known
Birds of Portrack Marsh and the Tees Barrage	Teesmouth Bird Club	Waterbird Assemblage. Portrack Marsh	2002-2012	Lists and counts of birds based on collated records	Not known
Little Tern monitoring data	Industry and Nature Conservation Association	Breeding little tern colony. Crimdon Dene	2010-2014	Regular monitoring of tern numbers by site wardens	Not known
Tern Verification Surveys	ECON & INCA	Tern verification surveys for marine sites	2015-2016	Shore based observations from fixed points and boat-based surveys	Natural England

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

1. Background and overview

All five species of tern that breed in the UK (Arctic tern *Sterna paradisaea*, common tern *S. hirundo,* Sandwich tern *Thalasseus sandvicensis,* roseate tern *S. dougallii* and little tern *Sternula albifrons*) are listed as rare and vulnerable on Annex I of the EU Birds Directive and thus are subject to special conservation measures including the classification of Special Protection Areas (SPAs). Little terns nest on sand or shingle beaches, islets and spits, often very close to the high water mark and are among the rarest seabird species breeding in the UK. There are currently 28 breeding colony SPAs designated within which little terns are protected. Since 2009, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of marine areas used by these birds while foraging to provide for their young, and the inclusion of such waters within classified SPA boundaries. In 2017, two new SPAs were classified (Northumberland Marine and Morecambe Bay & Duddon Estuary) and a further four existing SPAs (Hamford Water, Dungeness, Romney Marsh & Rye Bay, Outer Thames Estuary and Liverpool Bay) were extended into the marine environment to protect additional important areas for little terns foraging at sea.

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

2. Data collection

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

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

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

Boat-based surveys were carried out to assess how far out at sea foraging little terns would range (i.e. to confirm their maximum seaward foraging extent). Surveys involved the boats travelling along a series of parallel lines through a survey area around each colony. These surveys extended to 6 km from the coast to approximate the mean maximum foraging range as revealed from the literature (e.g. Thaxter *et al.* 2012) and preliminary JNCC observations. Two methods of recording little terns along a transect line were employed: (i) Instantaneous counts undertaken systematically at pre-determined points (between 300 m and 1800 m apart). The instantaneous count area was

[66] '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.

an 180º arc either ahead of, or off one side of, the boat depending on viewing conditions. All birds seen within this arc (out to a maximum estimated distance of 300 m) were recorded, along with the distance and bearing of the sighting and information on behaviour; (ii) Continuous counts of any little terns observed between the instantaneous points were also recorded to provide an[67] 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 1 km intervals to either side of the colony, up to a distance of 6 km along the coast, according to the mean maximum foraging range indicated by the literature. If preliminary observations found birds going further than 6 km, more observation points were added at successive 1 km intervals. Birds were counted within a distance of 300 m to either side of the observation point (resulting in a 180° arc). The shore based counts recorded passage rate and foraging use and if possible snapshot counts at one minute or two minute intervals were also recorded. The aim of the snapshot counts was to provide information on the intensity of foraging at each observation point. Ideally, counts at different observation points were done concurrently, lasting at least 30 minutes at each observation point. This time is based on the mean foraging trip duration for little terns lasting 16–29 minutes according to Perrow *et al.* (2006). However, in some cases this was not possible due to time constraints and/or logistical difficulties. In order to account for this difference in effort between observation points the shore-based count data were standardised to the number of birds observed per minute at each observation point. Care was taken to cover a range of tidal states, as variations in water levels between the times of high and low water are likely to play a significant role in determining the foraging locations of terns.

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

3. Data analysis

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

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

3.1 Site-specific options

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

A) Seaward extent

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

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

B) Alongshore extent

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

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

3.2 Generic options

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

A) Seaward extent

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

B) Alongshore extent

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

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

3.3 Derivation of site specific and generic seaward and alongshore extents

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

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

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

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

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

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

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

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

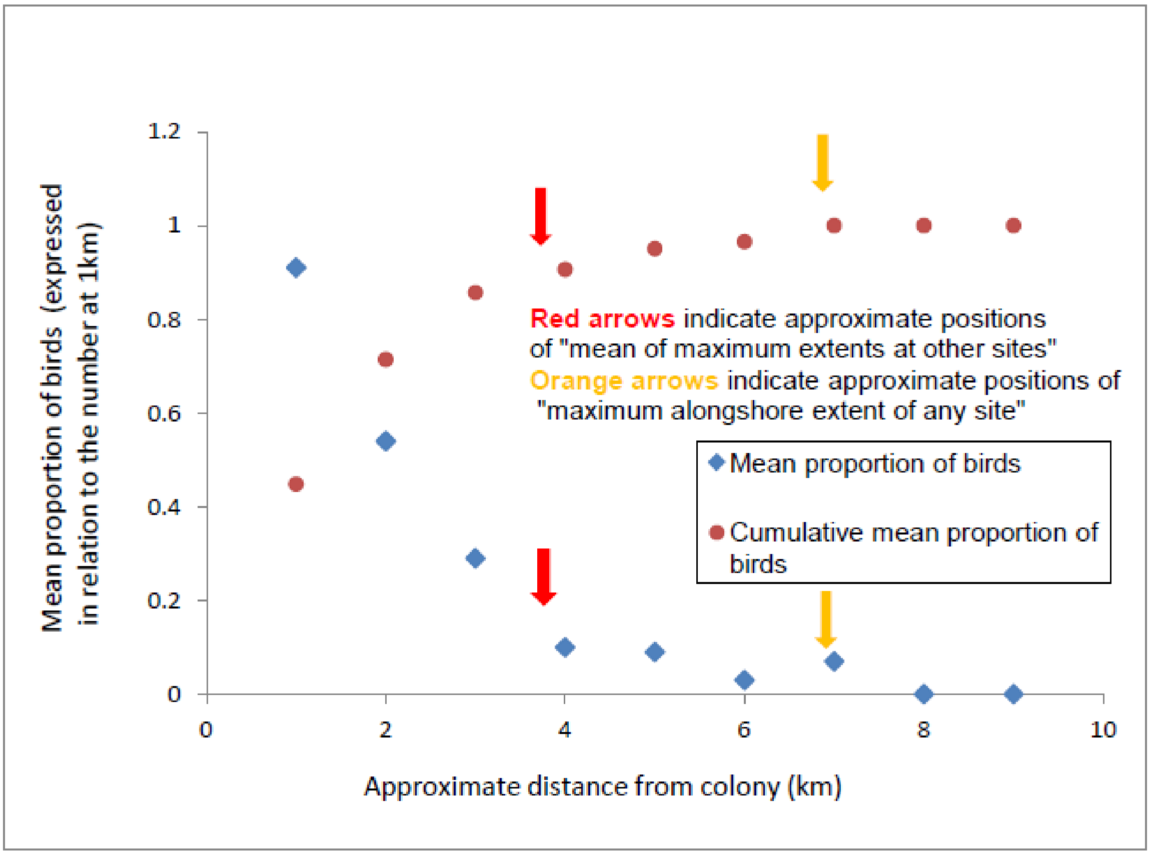


Figure 1: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing distances alongshore from the colony. Each blue point represents the mean proportional usage at each distance band from the colony averaged across colonies. The proportion at each distance (blue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

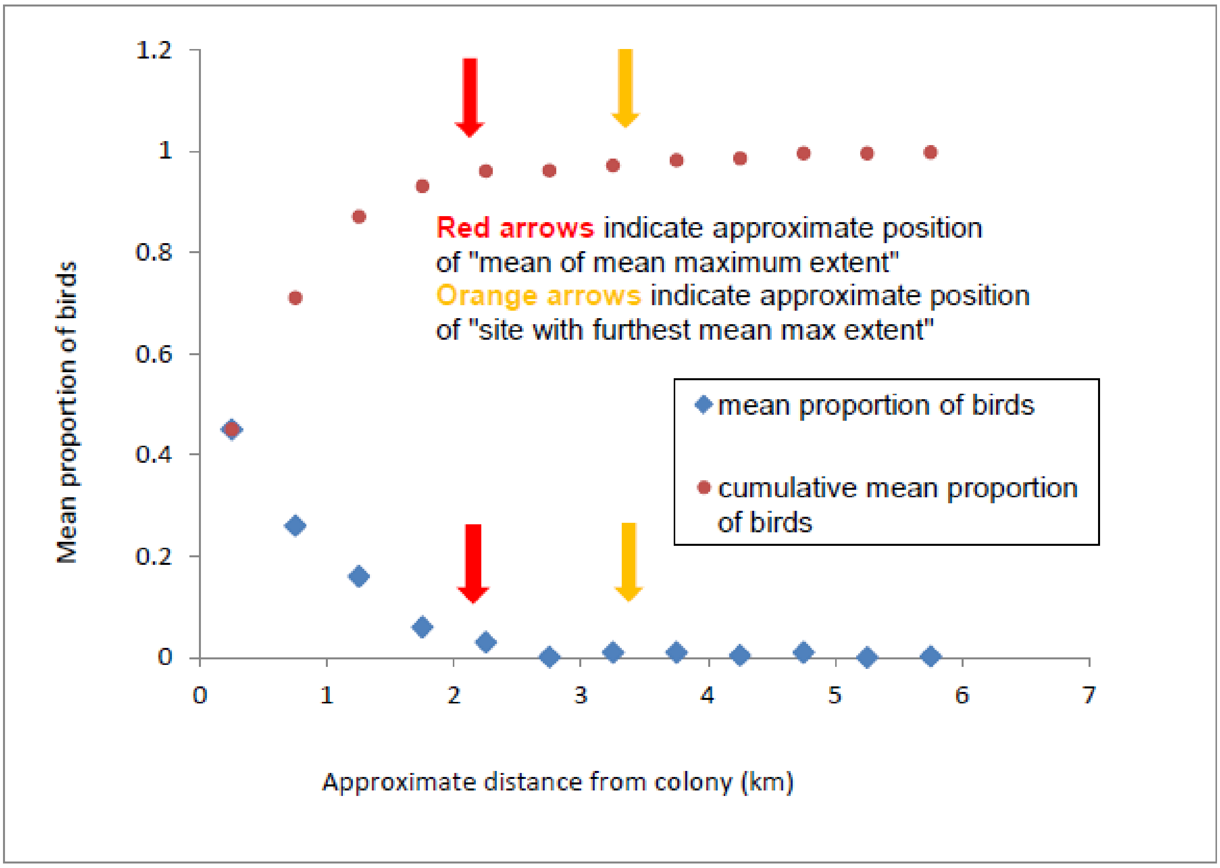


Figure 2: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing seaward distances from mean high water mark. Each blue point represents the mean proportional usage at each distance band from mean high water mark averaged across colonies. The red arrows indicate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

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

From these analyses it can be seen that in order to capture all recorded usage in an alongshore direction (1.0 at 7 km) and almost all recorded usage in a seaward direction (0.99 at 3.4 km) there would need to be a considerable increase in the distances being considered for defining the generic boundaries over those proposed in the case of sites with no or insufficient site-specific survey data (i.e. a further 3.1 km alongshore in each direction and a further 1.2 km offshore). On the simplifying assumption that alongshore and seaward limits define a rectangle lying parallel to the coast and with the landward edge centred on the colony, the sea area encompassed by these greater limits would be approximately 2.8 times that encompassed by the narrower generic limits proposed in the absence of site-specific survey data. The analyses suggest, however, that in the absence of site-specific survey data to the contrary, 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 a generic estimate of foraging extent to encompass all or nearly all observations across all sites, given that at any particular “average” site this would probably result in significant areas of low tern usage being included in the estimate. Therefore, the average of the site specific maximum alongshore extents (3.9 km) and the average of the site specific mean maximum seaward extents (2.2 km) have been adopted for a generic estimation of foraging extent at colonies with missing data or insufficient data to suggest otherwise. Use of these values is, on the basis of the analyses, likely to encompass areas of high to moderate use by breeding adult little terns during chick-rearing while excluding areas which are likely to have low usage at that stage of the season.

4. Boundary delineation

At each colony SPA, an assessment was made on the quality and quantity of data available for defining seaward extent and alongshore extent. If the quality or quantity was felt to be insufficient (e.g. 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. Thus, in the case of Teesmouth to Cleveland Coast pSPA, site-specific values for both the alongshore and seaward limits to important foraging areas around the colony were used (Tables 1, 2, Figure 3)

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

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

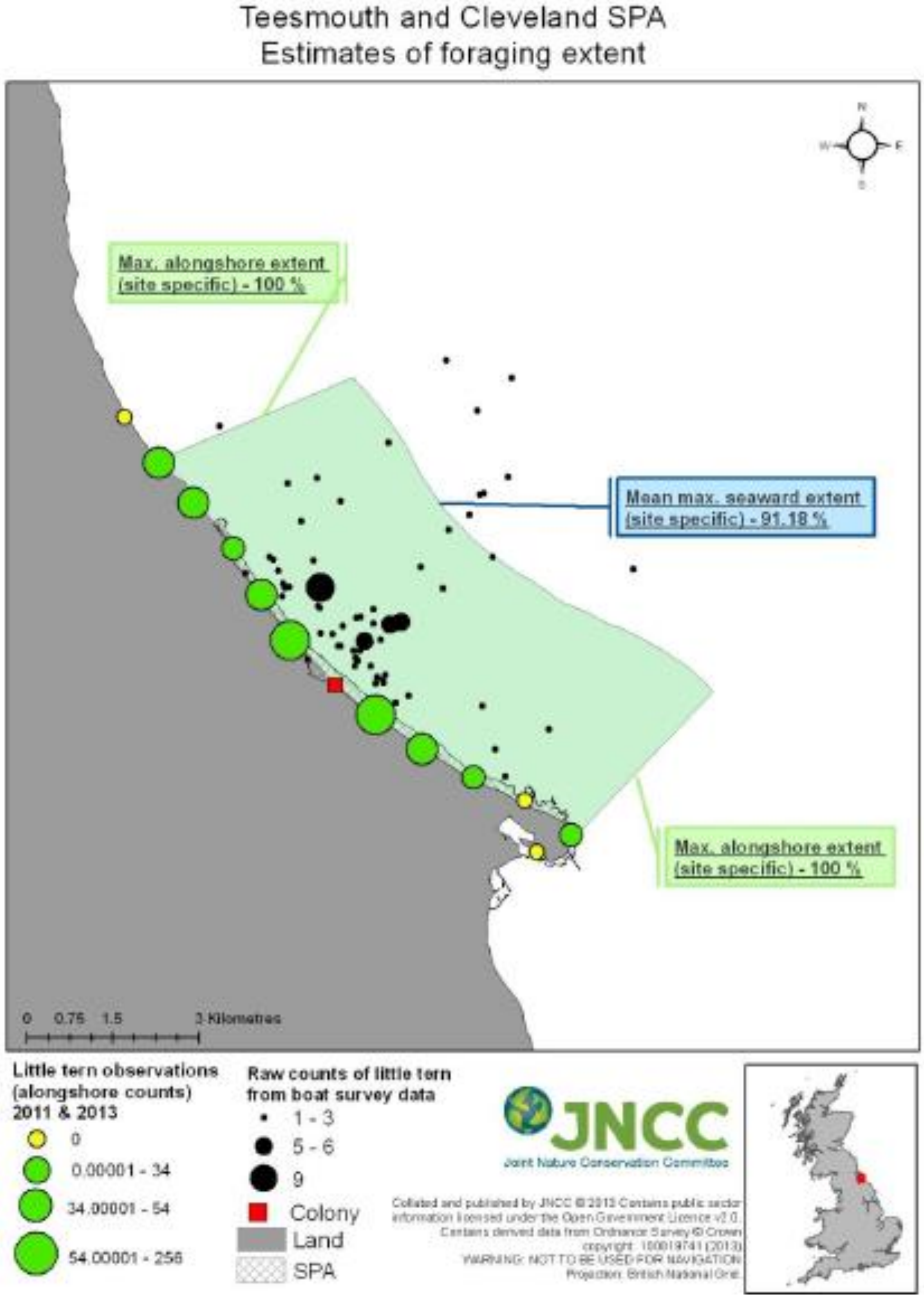


Figure 3. An example of the application of site specific alongshore and site specific seaward extents to define the boundaries of little tern foraging areas at the Teesmouth and Cleveland Coast SPA. The % values given in the labels indicate the site specific % of little tern observations within the shore-based (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 aimed to quantify foraging extent under a range of cases of data availability: i) where there are good data for both parameters; ii) where there are no site-specific survey data; iii) where data on seaward and/or alongshore extent are deficient.

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

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

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

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

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

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

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

1. Background and overview

All five species of tern that breed in the UK (Arctic tern *Sterna paradisaea*, common tern *S. hirundo,* Sandwich tern *Thalasseus sandvicensis,* roseate tern *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 59 breeding colony SPAs for which at least one species of tern is protected. Since 2007, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of marine areas used by these birds while foraging to provide for their young, and the inclusion of such waters within classified SPA boundaries. Based on that work and since 2015 three new SPAs have been classified (Northumberland Marine, Morecambe Bay & Duddon Estuary and Anglesey Terns) and a further four (Dungeness, Romney Marsh & Rye Bay, Outer Thames Estuary, Poole Harbour and Liverpool Bay) extended into the marine environment to protect additional important areas for larger terns foraging at sea

The work described here which underpinned the classification of these new or extended sites, and the proposed marine extensions of the Teesmouth and Cleveland Coast pSPA, 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 59 colony SPAs spread across the UK, it was recognised that resource and time constraints would preclude the detailed site-specific surveys at all colony SPAs over several years that, in an ideal world, would provide the most robust empirically based characterisation of marine feeding areas used by terns breeding within every colony SPA. Accordingly a statistical modelling approach was adopted which used data collected from a sub-sample of colonies to a) characterise the types of marine environment that are used by foraging terns, and b) use this information to identify potential feeding areas around all colony SPAs.

This annex gives an overview of the survey and analytical work carried out by and on behalf of JNCC between 2009 and 2013 for the four larger tern species (*Sterna* species). A full and detailed description of the analysis can be found in the JNCC report on this work (http://jncc.defra.gov.uk/page-6644). A different approach was deemed appropriate for little terns as they search for food in a much more restricted area closer to the coast and to the breeding colony. An overview of that work is described in Annex 5 and a full and detailed description of that analysis can be found in the JNCC report on that work (http://jncc.defra.gov.uk/pdf/Report_548_web.pdf). For the modelling analysis aspect of the project, JNCC worked collaboratively with Biomathematics and Statistics Scotland (BioSS)[68].

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[69] (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

[68] 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.
[69] 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.

overall survival and are thus high priority for site-based conservation.

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

* This distance was found to be optimal in terms of maintaining visual contact whilst minimising disturbance to the bird

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

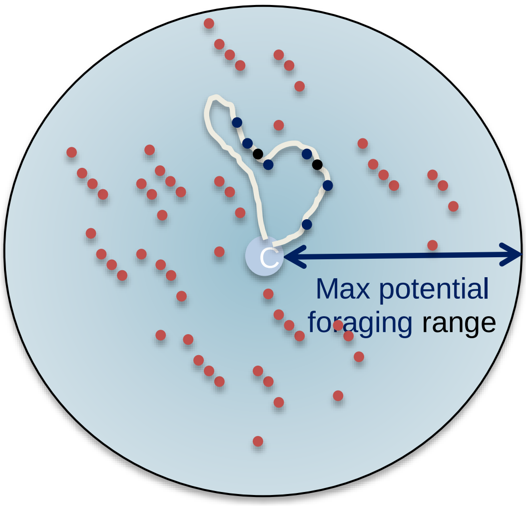
3. Data preparation and analysis

Prior to analysis within the habitat models, data had to be prepared and processed into a suitable format. Each track of a tern comprised periods of time when the bird was clearly not engaged in either actively searching for prey or in active foraging but appeared to be in transit to or from the colony or between areas of search at sea. As the aim of this work was to characterise important foraging areas and inclusion in the modelling of locations passed over in transit would, with terns being central place foragers (meaning they must travel to and from their nest site on each trip), almost certainly lead to a bias towards high usage of areas close to the colony, data from commuting periods (i.e. parts of the bird track where no foraging behaviour[70] 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.

[70] 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[71], 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).

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

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

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

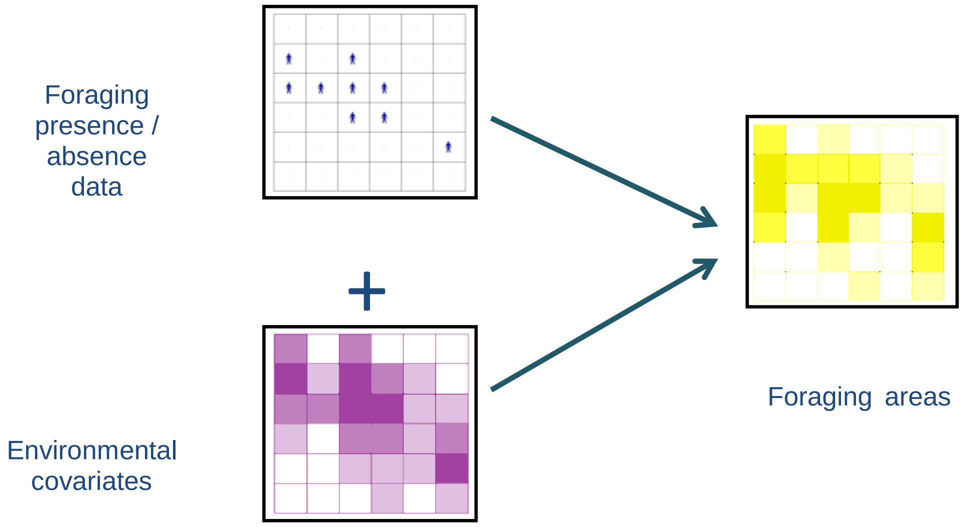


Figure 1: Simplified, schematic representation of the process of modelling distributions based on environmental information, using a single covariate distribution map in the example.

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

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

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

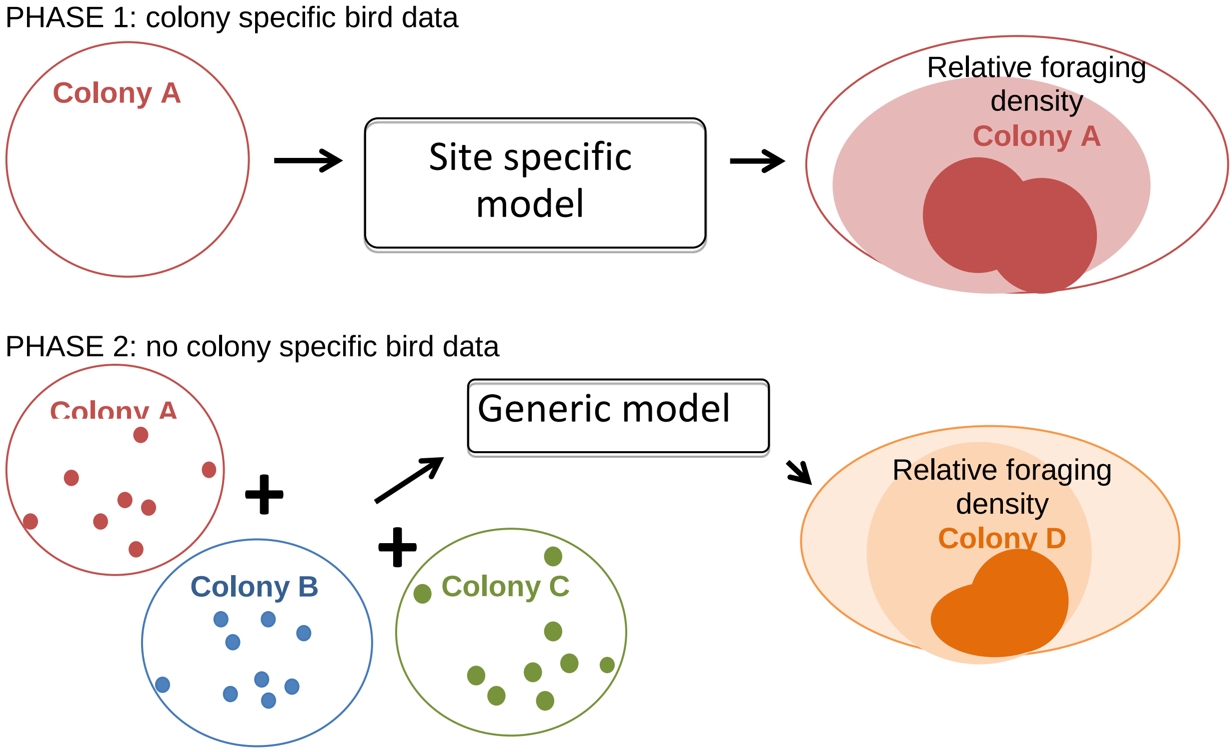


Figure 2: Simplified, schematic representation of the process whereby empirical observations of tern foraging locations around a colony were either: used to build predictive, site-specific models of tern usage that generated relative foraging density maps around that colony (phase 1 analyses); or combined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

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

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

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

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

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

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

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

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

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

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

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

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

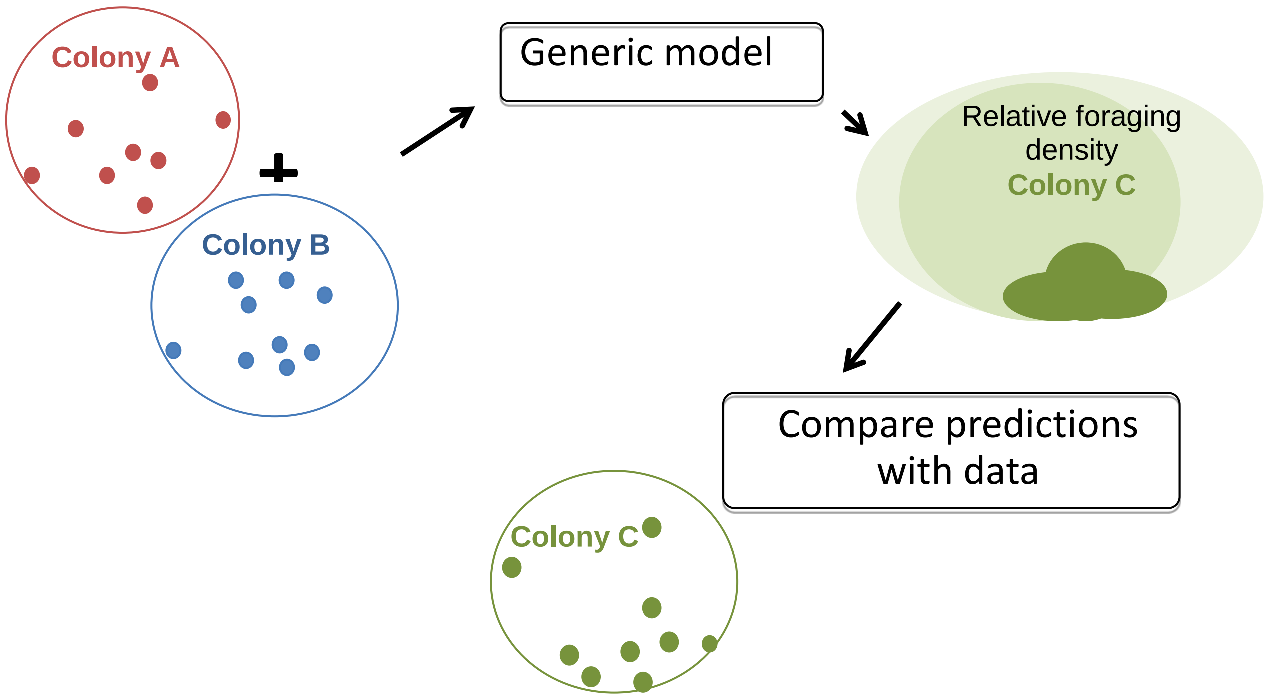


Figure 3. Schematic representation of the cross-validation process, using an example where we have data for three colonies A, B and C, of which data from two at a time (A and B in this diagram) are used to build a predictive model, the predictions of which are then tested by comparison with empirical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

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

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

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

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

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

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

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

(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 is in their document produced for use by all SNCBs: 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[72], resulting in 30km for Arctic, 20km for common, 32km for Sandwich and 21 km for roseate tern. Any grid cells outside the mean maximum foraging ranges were excluded

[72] 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.

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.

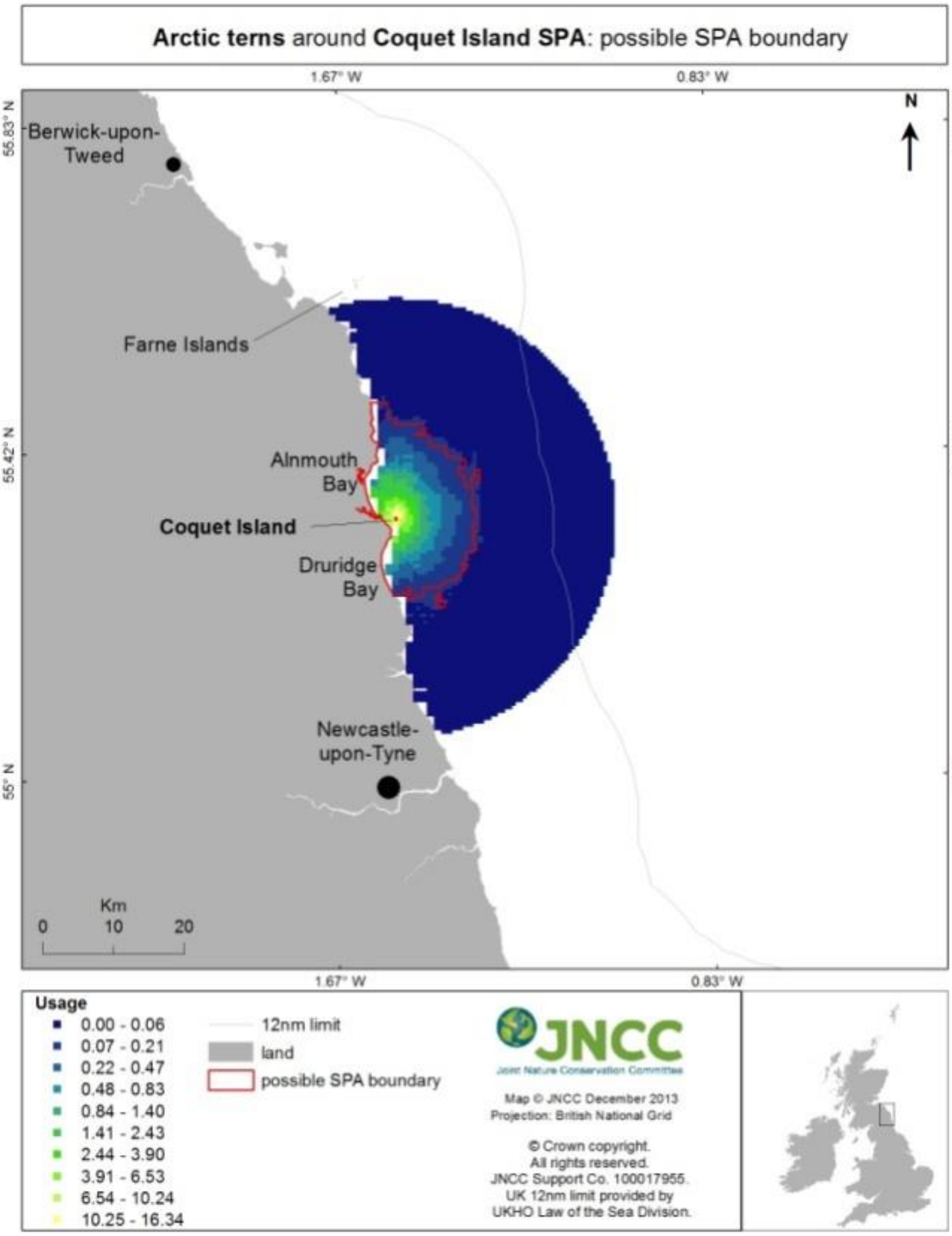


Figure 4: Maximum curvature derived boundary (red line) overlaid on map of model predictions of usage by Arctic terns around Coquet Island. The extent of the dark blue circle of model predictions of usage is 30 km - the global mean maximum distance to colony, calculated using tracking data held by JNCC;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

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

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

5. Conclusion

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

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

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

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

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

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

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Annex 7: Verification surveys undertaken on the Teesmouth & Cleveland Coast pSPA in 2015 and 2016.

1. Introduction

In 2015 Natural England commissioned ECON Ecological Consultancy Ltd to carry out survey work at a number of sites in England to verify the predicted patterns of tern usage generated by JNCC's modelling work. One of the sites of particular interest was the Teesmouth and Cleveland Coast draft pSPA. This was because the proposed amendments to the existing boundary of the SPA (to include the length of the River Tees up to its tidal limit and the sea area outside the mouth of the estuary) were founded on the results of the predictions of a generic model of patterns of common terns' foraging activity, the model itself being based on analysis of patterns of common tern foraging elsewhere around the UK rather than at Teesmouth (see Annex 6).

Subsequent to these surveys in 2015, INCA was commissioned in 2016 to repeat the 2015 surveys by ECON for the stretch of the River Tees between the Tees Barrage and Seaton Channel in order to provide an additional year's worth of empirical data with which to confirm (or otherwise) the previous year's findings.

2. Methods.

The full details of the survey methods employed by ECON in 2015 and by INCA in 2016 are set out in the reports on those pieces of work (Natural England 2016, INCA 2016). A brief summary of the methods is presented here.

2.1 ECON survey methodology

Three surveys were undertaken on 18th June, 2nd July and 22nd July 2015. All survey work was conducted from a boat. The survey vessel was moored in Hartlepool Marina and the survey route began and ended there, taking in Victoria Harbour on the outward or return leg. The survey route ran across Hartlepool Bay and up the River Tees as far as the tidal barrage and back again. *En route*, three different types of data recording methods were used to record tern activity: i) timed counts of the total numbers of terns seen over a 30 minute period within 300m of the boat (within an arc of 180^0 forward of the boat) while the boat was held stationary at a series of c17 observation points, ii) a series of instantaneous snapshot counts of the numbers of terns seen within the same area taken every minute while the timed counts were carried out at each station and iii) an instantaneous snapshot count of the numbers of terns seen within 300m of the boat (within an arc of 180^0 forward of the boat) taken every 300m as the vessel moved along the transect route so as to give continuous coverage along the transect when travelling at 10 knots.

2.2 INCA survey methodology

As far as possible, the methodology followed that of the ECON 2015 surveys based on counts taken from a vessel along the relevant length of the River Tees from Seaton Channel to the Tees Barrage. Three surveys were undertaken but each of these spanned two days rather than one i.e. 22nd & 27th June, 5th & 6th July and 26th & 27th July with approximately half of the river length being surveyed on successive days. Along the length of the river surveyed, the same set of fixed observation stations as used in 2015 were re-used, and timed counts over 30 minutes and snapshot counts each minute during that time were recorded at each station (as in 2015). As the vessel was moving along the transect at 10 knots, instantaneous snapshot counts were taken at 1 minute intervals and a record was made of the total number of terns seen on the entire stretch of the river being traversed.

3. Results.

3.1 ECON surveys in 2015

A total of 957 common terns were recorded over the three surveys, with no survey location at which common terns were not recorded at least once during timed counts. The number of overall records was concentrated at the estuary at the Seaton Channel (35%). Otherwise, records were rather equally distributed between survey locations (from 2-8% of records) all the way along the length of the Tees from the Tees Barrage at the survey location furthest upstream (8% of records) to Middlesbrough Dock (5% of records), Tees Dock (2% or records), and across the bay to Victoria Harbour and Hartlepool Marina at Hartlepool (3 % and 2% of records respectively) some 12 km from the colony as the tern flies. The use of most survey locations such as Middlesbrough Dock (3-7% of records between occasions) and Tees Dock (0.6-3% of records) in what could be described as the middle to lower reaches of the river, was rather consistent at a lower level compared to the estuary at Seaton Channel or Tees Barrage.

In keeping with model predictions common terns were consistently recorded along the length of River Tees from the Seaton Channel in the estuary upstream to the Tees Barrage throughout the surveys in 2015 (Figure 1).

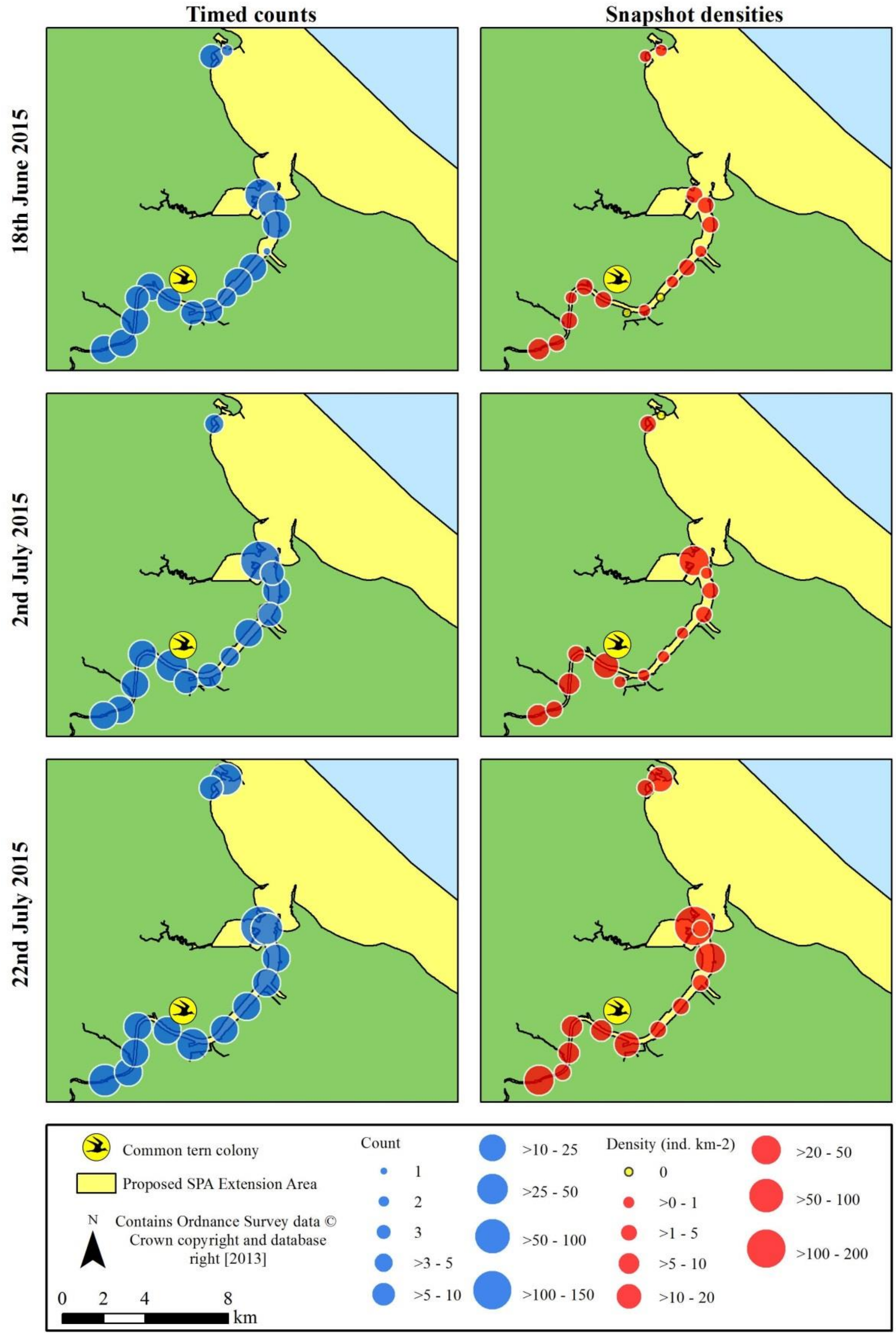


Figure 1: Abundance and distribution of common terns at Teesmouth & Cleveland Coast as shown by timed counts and density (individuals/km^2) derived from snapshots within the timed count at the different survey locations in each of the sampling occasions in 2015. Source: Natural England (2016).

There are however, some subtle differences in the patterns of use suggested by the timed counts relative to the snapshot counts converted to density. By recording all birds, many of which may pass through relatively quickly, timed counts are likely to record presence/absence effectively. In fact if recording occurs over a relatively long period of time, and especially where birds are actively using the river as a corridor that offers foraging opportunities even if they are mainly in transit flight, there will be little, if any, absence data. In contrast, the snapshot counts and resulting density estimates are more likely to better reflect more persistent use of a given location.

Accordingly, there is some suggestion that the middle and lower reaches of the river are used less than those further upstream (Figure 1). The somewhat reduced use of the river between Middlesbrough Dock and the Tees Dock, is also apparent in the snapshot density data from the transect survey, which by moving rapidly through the survey area provides a true snapshot of the patterns of use (Figures 2-4).

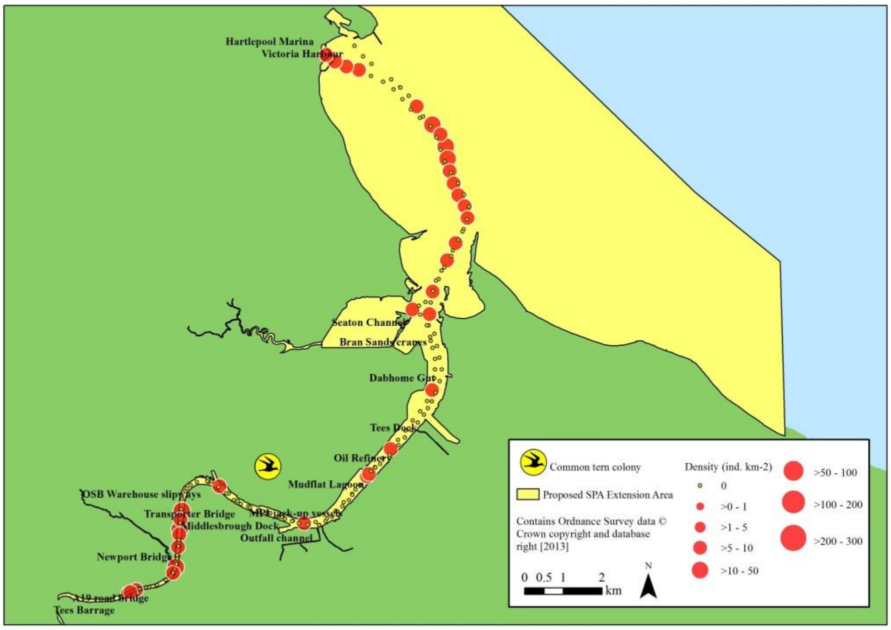


Figure 2: Density (individuals/km^2) of common tern from snapshots at 300 m intervals along the boat-based transect route from Hartlepool to Tees Barrage on 18th June 2015. Source: Natural England (2016).

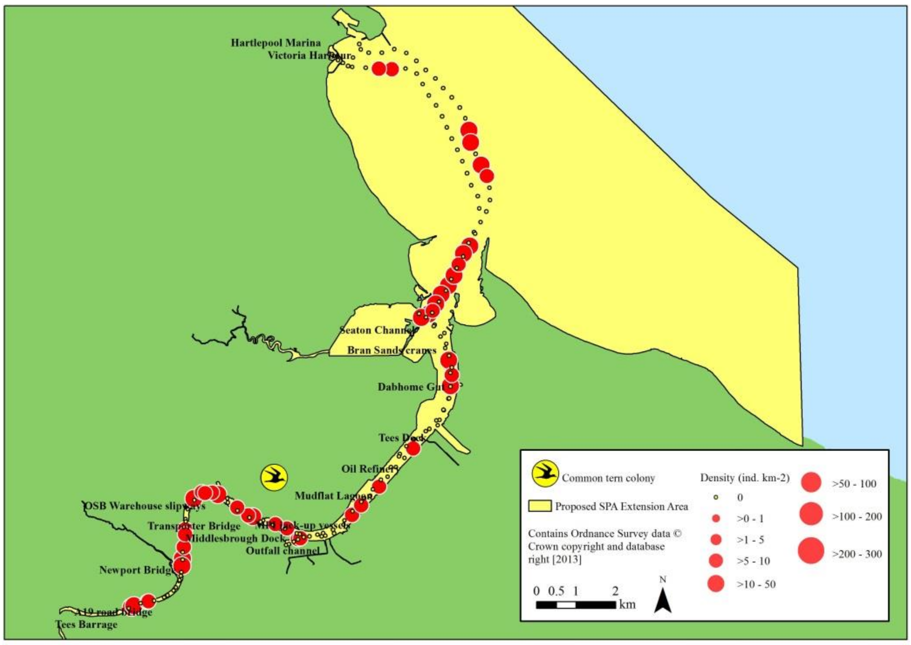


Figure 3: Density (individuals/km^2) of common tern from snapshots at 300 m intervals along the boat-based transect route from Hartlepool to Tees Barrage on 2nd July 2015. Source: Natural England (2016).

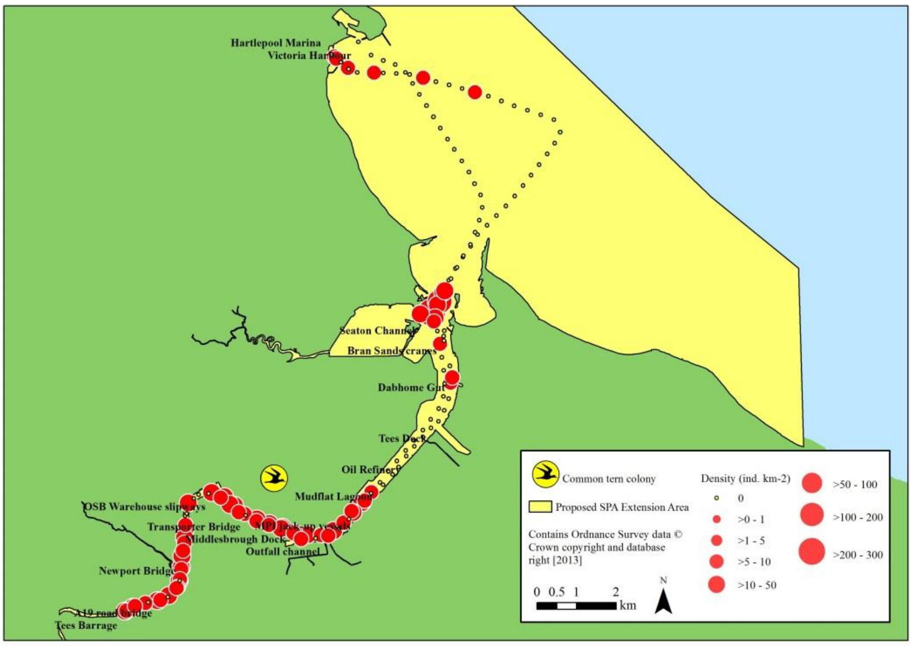


Figure 4: Density (individuals/km^2) of common tern from snapshots at 300 m intervals along the boat-based transect route from Hartlepool to Tees Barrage on 22nd July 2015. Source: Natural England (2016).

The middle to lower reaches of the river closest to the colony was used broadly equally to other areas at considerable distance, including parts of Hartlepool Bay (Figure 5). Particularly attractive areas such as the upper reaches of the river near the Barrage as well as the estuary near Seaton Channel occurred at moderate distances of around 5-6 km (timed count) and 6-7 km (snapshots along the boat-based survey route) from the colony. As a result, the predicted usage from the model that is largely dependent on distance did not particularly reflect the data gathered in these particular surveys.

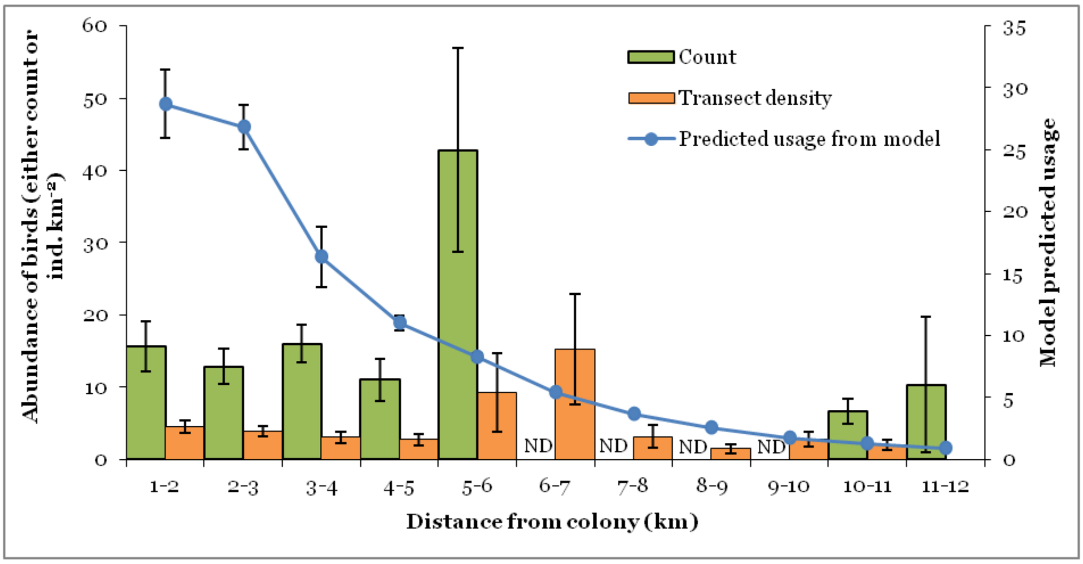


Figure 5: Relationship between the predicted common tern usage of cells at increasing distance from the colony according to the generic JNCC model, compared to measures of abundance of common tern from the current surveys including timed counts and density (individuals/km^2) from snapshots along the boat-based survey transect. ND = no data for some of the distance intervals from the colony relating to the timed counts. Source: Natural England (2016).

Despite the fine-scale differences between the results of the current surveys and the predicted model usage at particular distances from the colony, at the broader scale the use of the river, estuary and Hartlepool Bay all fitted within the categories of moderate to high use predicted by the model and thus the boundary of important foraging areas suggested (Figure 6). In other words, the model encapsulates the potential for common terns to form hotspots of activity at reasonable distance from the colony observed during the current surveys.

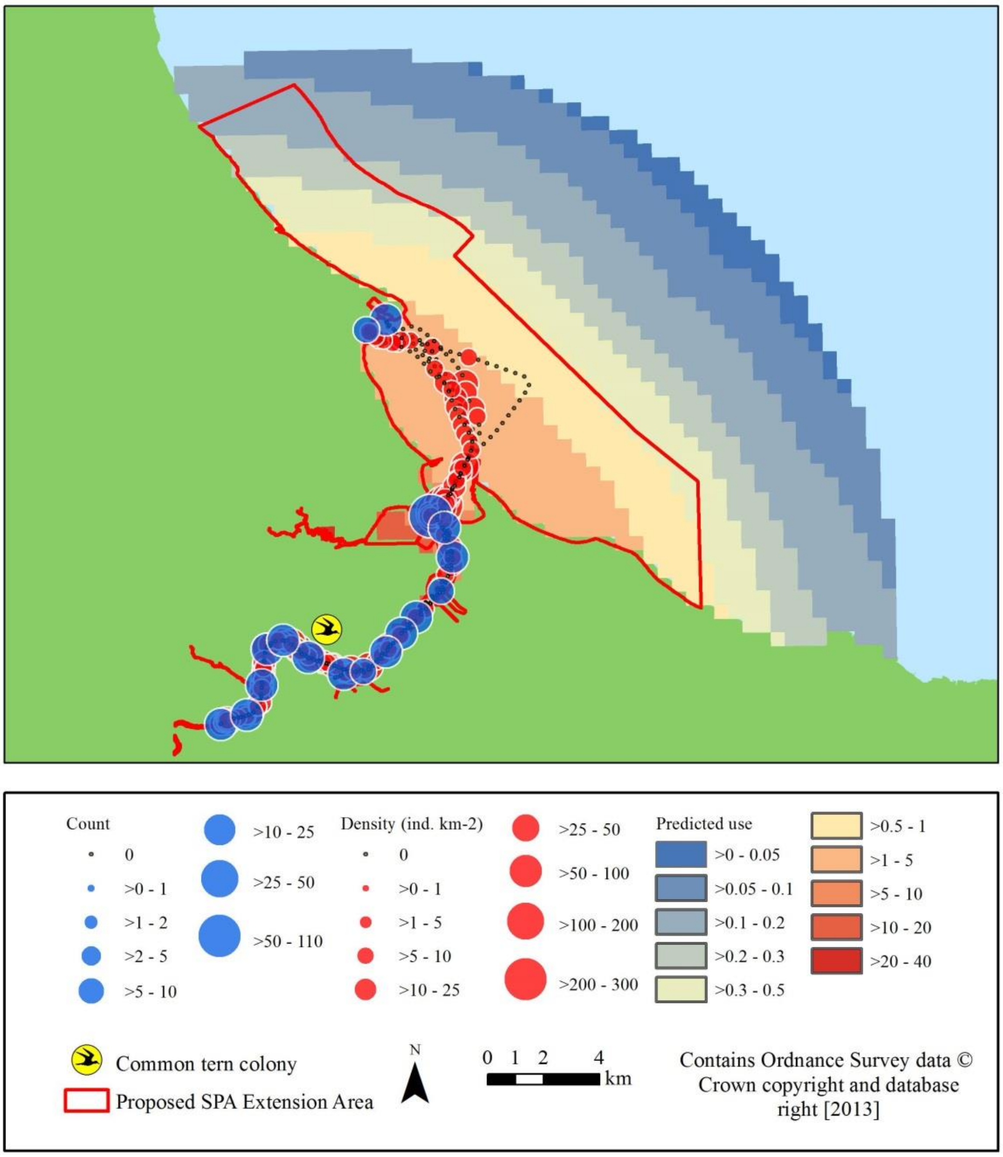


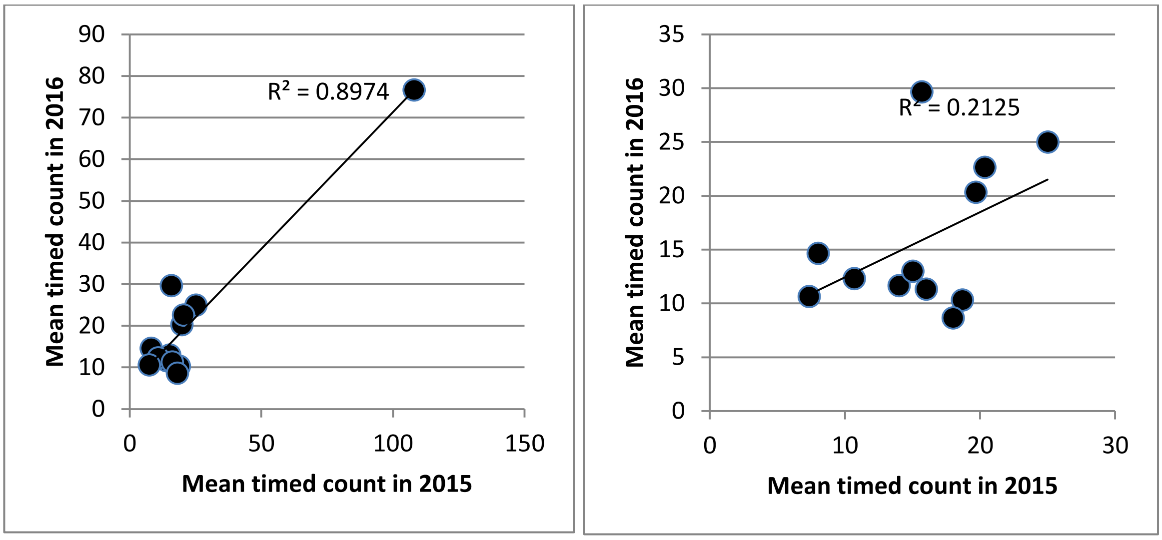
Figure 6: Comparison between the JNCC generic model-based output of relative usage and the boundary of important foraging areas for common tern from Salthome, relative to records of common terns in mean timed counts and density delivered from snapshots during boat-based transects from the three survey occasions in 2015.

3.2 INCA surveys in 2016 in comparison with ECON surveys in 2015

The total numbers of terns recorded for both the entire river and across most of the individual sites were remarkably similar between the 2015 and 2016 studies. A total of 949 birds were recorded along the river in 2016 compared to 895 in 2015. Allowing for the fact that not all locations were surveyed on all occasions in 2015, the mean number of terns per 30 minute survey period was 22.6 in 2016 cf 22.9 in 2015.

The pattern in the numbers of terns across the survey season was also broadly similar between the studies with an increase in the total number of individual terns recorded as the season progressed. The late June surveys contributed 18.1% and 20.3% of the total numbers, the early July surveys, 26.5% and 34.6% and the late July surveys, 55.4% and 45% across the 2015 and 2016 studies respectively.

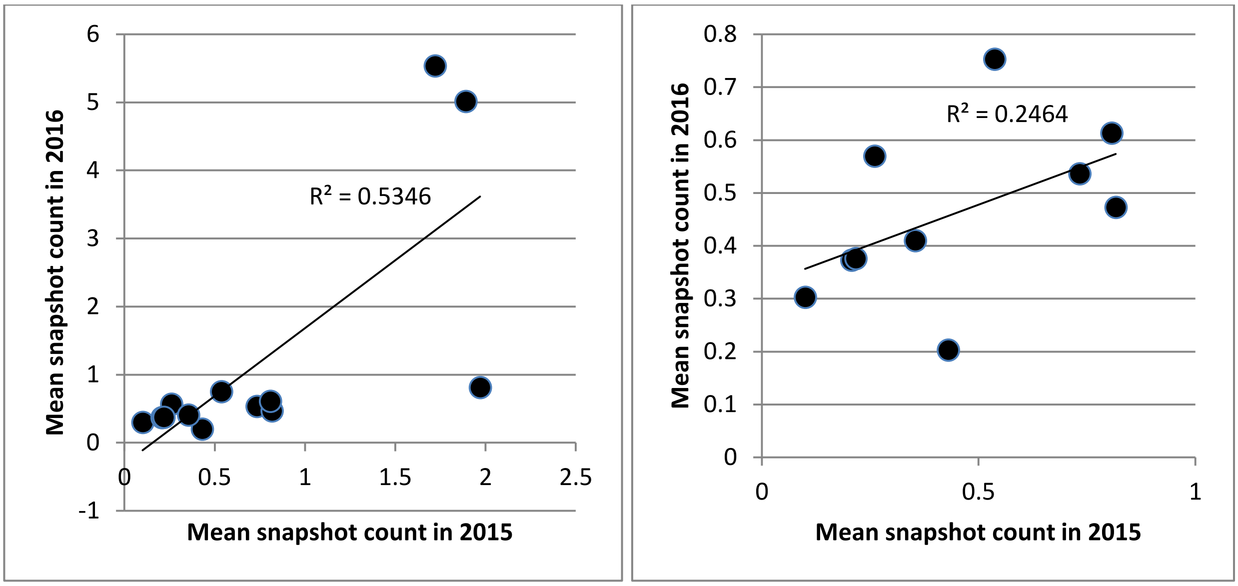
A comparison of the mean timed count of common terns across the 3 surveys at each location in each of the 2 years reveals a strong correlation driven largely by the relatively greater numbers of common terns seen in both years at Seaton Channel in comparison with elsewhere on the river (Figure 7a). However, considering only those stations further upriver than Seaton Channel, a positive correlation remains (albeit weaker) (Figure 7b). Of equal interest to the strength of this correlation is that in both years the mean timed count across all stations (excluding Seaton channel) spanned the same range (7 to 25 in 2015 and 8 to 30 in 2016) with birds being recorded at every station in both years.



a) b)

Figure 7: Relationship between the mean timed counts of common terns recorded at each observation station along the River Tees and surveyed in both 2015 and 2016: a) all stations including Seaton Channel, b) excluding Seaton Channel.

In the results of the snapshot counts taken at each of the timed count observation stations, hotspots of activity in both 2015 and 2016 occurred at the Tees Barrage, Dabholme Gut and Seaton Channel – these 3 stations yielding the greatest mean snapshot counts in both years (Figure 8a). However, the numbers recorded upriver at Tees Barrage and at Dabhome Gut were greater in 2016 than in 2015 whereas the opposite was the case at Seaton Channel. The reason for this difference between years in the relative level of activity at these 3 stations is unclear at present but could simply reflect variation in the state of the tides on each of the surveys in each of the years. If data from these three stations are set-aside, there remains a positive correlation (albeit weaker) between the mean snapshot counts at each station in the 2 years (Figure 8b). Again, of equal interest to the strength of this correlation is that in both years the mean snapshot count across all stations (excluding the 3 hotspot stations) spanned the same range (0.1 to 0.8 in 2015 and 0.2 to 0.8 in 2016) with birds being recorded in snapshot at every station in both years.



a) b)

Figure 8: Relationship between the mean instantaneous snapshot counts of common terns recorded at each observation station along the River Tees and surveyed in both 2015 and 2016: a) all stations including Tees Barrage, Dabholme Gut and Seaton Channel, b) excluding the three named stations. Note in a) the mean value shown for Seaton Channel in 2015 (1.97) excludes the exceptionally high snapshot count at this station on the 3rd survey of 27.10 which would make the overall average for that site in that year 10.34.

4. Conclusions.

The report by ECON to Natural England (Natural England 2016) demonstrated that common terns from RSPB Salthome were consistently recorded using the entire length of the River Tees from the estuary as far upstream as Tees Barrage. The report noted that “the generic JNCC model did not accurately predict the relative levels of use of particular areas by common terns as a result of the fact that the birds tend to aggregate at a variety of ‘hotspots’ especially at the Seaton Channel in the estuary (some 5-6 km from the colony), the Barrage in the river and various locations at sea within Hartlepool Bay and the Victoria Dock at Hartlepool.” This finding was similar to that recorded in an equivalent study in Northern Ireland (Allen & Mellon Environmental Ltd. 2015) which similarly showed that generic models, while not always able to identify the precise locations of intense usage of terns, resulted in proposed boundaries that always succeeded in including these important areas. The report by ECON concluded that “the incorporation of the River Tees as far upstream as Tees Barrage within the proposed SPA could be verified by the current surveys”.

However, in the conclusion to the ECON report it was noted that “extensive use of the Tees was not recorded in previous tracking studies (in 2009) and the inter-annual frequency and extent of this use remains unclear, although it could indicate reduced fish availability in the estuary in some years and/or the possible increasing importance of the river as a foraging ground in recent years.” It was largely in the light of this statement regarding the uncertainty regarding the scale of inter-annual variation in use of the River Tees by common terns that the INCA surveys in 2016 were commissioned.

The conclusion of the report on the surveys in 2016 (INCA 2016) states that other than at Seaton Channel, the numbers and patterns of activity of common terns are very similar between the 2015 and 2016 studies. In the case of both the timed counts and the snapshot counts undertaken during the timed counts, strong correlations in the levels of common tern activity recorded across stations in the 2 years was found, largely driven by hotspots of common tern activity at Seaton Channel (timed counts) and also at Tees Barrage and Dabholme Gut (snapshot counts). However, if the records from these stations are set-aside: i) common tern activity was recorded by both methods at all stations in both years, ii) positive, albeit weaker, correlations in common tern activity still exist across years, and iii) the range of activity levels across the stations was the same in both years.

Thus, as was apparent in the verification surveys in 2015, common terns in 2016 were using the entire length of the River Tees between Seaton Channel and Tees Barrage. These findings confirm the conclusion of the 2015 verification survey report (Natural England 2016) that “the incorporation of the River Tees as far upstream as Tees Barrage within the proposed SPA could be verified by the current surveys.“ INCA (2016) noted that tern activity at the Tees Barrage in 2016 (as measured by the timed snapshot counts) was around three times that in 2015. It was suggested that this may be related to common terns re-colonising the newly refurbished breeding site on the adjacent Portrack Marsh (as noted in section 7.1 of this Departmental Brief), upriver from Saltholme. INCA (2016) conclude by speculating that if the capacity of the Portrack Marsh to support breeding common terns has yet to be reached then the river might further increase in importance for these birds in future years.

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.

Industry Nature Conservation Association (INCA) (2016). INCA 201614. Report to Natural England. 23pp.

Natural England (2016). Tern Verification Surveys for Marine Sites. 114pp.
http://publications.naturalengland.org.uk/publication/6688364374786048

Annex 8: Implementation of Evidence standards

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 Teesmouth and Cleveland Coast SPA.

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

Quantification of Teesmouth and Cleveland Coast pSPA interest feature population sizes.

In order to determine the suite of species present within the Teesmouth and Cleveland Coast pSPA which meet the SPA selection guidelines, and to determine the nature and size of the waterbird assemblage supported by this site, three main sources of bird count data were used as the evidence base. These data were as follows:

1. Standardised systematic counts of the Teesmouth and Cleveland Coast SPA conducted under the Wetland Bird Survey between 2011/12 and 2015/16 provided the most recent 5 year mean peak populations of waterbirds. Data from BTO’s Wetland Bird Survey for the following core count sectors: Bran Sands North 52428, Bran Sands South 52427, Brinefields 52417, Coatham Marsh 52431, Coatham Sands North 52901, Cowpen Marsh 52418, Dormans Pool 52420, Durham Coast Sector 1a, Greatham Creek Channel 52416, Greatham Tank Farm 52415, Greenabella 52414, Hartlepool Bay 52402, Haverton Hole (north) 52501, Haverton Hole (south) 52502, Haverton South Reedbed 52002, North Gare Sands 52413, Peninsula East 52424, Peninsula West 52425, Quarries and Lagoons 52430, , Redcar and Coatham Sands South 52432, Saltern Borrow Pits 52436, Saltern Wetlands 52434, Saltholme Central 52433, Saltholme Pools 52419, Seal Sands SW and Main 52422, Seaton Common 52412, Seaton Sands 52411, Tank Farm 52435 and Tees Estuary 52901.

2. Data from the Cleveland Bird reports 2009-2013 for locations and numbers of breeding pied avocet and breeding common terns.

3. Data from the Cleveland INCA little tern monitoring reports for the numbers of breeding little terns.

The count data taken from these sources is the best available information. It has been collected by skilled and experienced field surveyors following strict recording and reporting protocols, and at least in the case of the first two data sources, has been independently verified before publication.

The relatively discrete nature of the WeBS data, divided into a number of individual count sectors with boundaries following topographical features, allows easy interpretation of the relative importance of different parts of the SPA, including the majority of the proposed extensions. These information sources provide consecutive annual data over a five year period, which allows for the calculation of mean counts which helps to overcome issues of reliability due to the annual natural fluctuations of most bird populations.

Data from additional sources include a Natural England commissioned survey of selected wetland areas during the winter period 2014/15, and provides data from only one year. This survey focused specifically on the terrestrial locations at Brinefield, Cowpen Marsh, Greathan Tank Farm, Greenabella Marsh, Seal Sands brownfield and the intertidal area Vopak foreshore. Some of these areas (Brinefield, Cowpen Marsh, Greatham Tank Farm and Greenabella Marsh) are also covered by WeBS, so the single year's data can be placed into context with the longer-term dataset. Other data sources include a report of annual use by waterbirds of Portrack Marsh and an INCA report of bird numbers present on areas of the Brans Sands South WeBS sector during 2009-13. Data from these sources were not subject to WeBS core count methodologies of verification and do not contribute to the calculation of total estimates for individual species and the assemblage (see below).

Establishment of extent of marine pSPAs using tern at-sea distribution 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 extension into the marine environment of the Teesmouth and Cleveland Coast pSPA differs from those used in identifying sites for terrestrial birds and aggregations of non-breeding waterbirds, it is necessary to consider their adequacy and relevance.

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

Table 1 Criteria for inshore SPA data adequacy.

Criterion	Adequacy of JNCC led larger tern surveys	Adequacy of JNCC led little tern surveys
Experience of observers	All tracking of terns was undertaken either by JNCC staff or experienced contractors commissioned by JNCC to do the work.	All observations of terns were undertaken either by JNCC staff or experienced contractors commissioned by JNCC or volunteer counters who received training in the shore-based observation techniques.
Systematic surveys	Tern tracking was conducted in as systematic a way as possible. Tracking at each colony was carried out during well-defined periods of the breeding season (chick-rearing) in one or more years. Tracking was undertaken in accordance with a field protocol established by JNCC. In the context of tern tracking, the movements of birds is an essential component of the technique and	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.

	not a source of systematic bias in the survey results as it may be in conventional transect surveys.	
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 distribution of terns (of unknown origin) around the colony. Cross-validation tests of the models' predictions and analysis of sample adequacy both suggest that the results of the models, although based on the samples of tracks, are robust.	At sea observations included instantaneous counts at predetermined distances along transects at which all terns in flight within 300m in an 180º arc of the boat were recorded. Between these points, continuous records of all little terns seen were also made to provide an index of relative abundance. During shore-based observations, terns recorded within 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 additional marine areas included within new or extended pSPAs by the breeding terns has been reasonably inferred from the continued existence of the site's named features in qualifying numbers in each of the existing coastal colonies within the pSPA.

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

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

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

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

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

The landward boundary of the marine elements of the pSPA has been drawn at MHW along the coast, up the Tees River and into various docks, harbours and marinas in the light of: i) model predictions of the usage of such areas by foraging larger terns, ii) observations of larger terns foraging in such areas, iii) observations that little terns forage in the intertidal zone and iv) to ensure protection of these areas as supporting habitat for tern species within the pSPA in locations where these species are not already features of the existing SPA with which an overlap might occur. Stretches of the Northumbria Coast SPA already occupy some sections of the mainland coastline down to Mean Low Water. As little tern is already a feature of that SPA, along these stretches of coast, the pSPA boundary correspondingly extends only to Mean Low Water to abut this SPA. Elsewhere on the mainland coastline, the boundary of the pSPA will extend to Mean High Water where the coastline is not existing SPA.

Webb & Reid (2004) set out a recommended method for defining the seaward boundary of SPAs for inshore non-breeding waterbirds on the basis of analysing bird data from aerial or boat-based sample surveys using spatial interpolation combined with spatial analysis. They note exceptions to this method which include the case in which "*habitat data are also used in combination with bird distribution data to determine boundaries*". This is the approach which has been used in the 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 larger 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 larger tern usage to define the most important areas to include within the pSPA boundary (Win *et al.* 2013) follows the basic ranking principle outlined by Webb et al (2003). However, the application of the maximum curvature technique to such cumulative usage curves in the current case (Win *et al*. 2013) reflects the advances in the details of this analytical method by JNCC since then (O'Brien *et al.* 2012).

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

Establishment of the extent of the Teesmouth and Cleveland Coast pSPA

The extent of and seaward boundary to the Teesmouth and Cleveland Coast pSPA is determined by the extent of the model generated predictions of which areas of sea are most heavily used by foraging common terns originating from the colony at Saltholme and direct observations of the distributions of little terns from the colony at Crimdon Dene. The colony-specific area of use by common tern has been derived from models based on at sea records of the foraging locations of 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.

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

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

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

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

Site specific evidence for waterbirds is provided by WeBS, which is a national database of wetland bird counts provided by volunteers. The WeBS data have been analysed to provide five-year average peak values for each waterbird species. It is appropriate to calculate mean data from the available datasets to take into account annual fluctuations in bird numbers. All comparisons between individual sectors and the broader SPA have used the WeBS core count data wherever possible. This is because the WeBS waterbird counts are carefully coordinated to avoid the risk of double-counting birds moving between sectors. The only exceptions are those areas not covered by WeBS (Seal Sands extension, Vopak foreshore and Portrack Marsh). Comparisons with WeBS data for these locations should be considered indicative. Data from these locations has not been included in the calculation of the overall assemblage figure for the SPA or Ramsar site as this may lead to some double-counting and overestimation of total numbers. The corollary to this is that the total numbers of waterbirds using the SPA and Ramsar site are likely to have been underestimated.

Data for breeding pied avocets and common terns has been derived from the relevant Cleveland Bird Reports. As with WeBS, these reports collate the data provided by volunteer surveyors, including nature reserve managers and members of the public. All data have been checked and verified by the County Bird Recorder. Data for breeding little terns has been derived from the Cleveland INCA little tern monitoring reports. Once again, given the availability of annual data, it has been possible to calculate five-year means for these species.

The above data were compared to established site selection criteria (JNCC 1999) and the boundary decision is based on an appropriate application of the criteria based on the available evidence. The terrestrial extension is identified in line with standards adopted in the context of other SPA designations. Best available waterbird data were used and mean populations were calculated where possible before applying the SPA selection guidelines (JNCC 1999).

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

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

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

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

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

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

The conclusions regarding the list of features and their reference population sizes within the pSPA are based on application of the SPA selection guidelines issued by JNCC (JNCC 1999) to the best and most comprehensive, recent count data. As such the conclusions in this respect clearly relate to the best available evidence.

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

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

Since the modelling work was completed by JNCC, the Department of the Environment, Northern Ireland (DoENI) commissioned in 2014 a programme of land-based and at-sea surveys to verify the predicted 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 (Annex 6), 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.

Subsequent to the verification surveys carried out in Northern Ireland, Natural England commissioned ECON Ecological Consultancy Ltd to conduct a programme of shore based and boat-based surveys of tern activity in several pSPAs around England in order to confirm (or not) the validity of the proposed boundaries to these sites which were based on the predictions of JNCC's models. Verification surveys were carried out in the following pSPAs: Hamford Water, Teesmouth and Cleveland Coast, Northumberland Marine (Seahouses Harbour, River Aln, River Coquet, River Wansbeck and River Blyth), Morecambe Bay & Duddon Estuary, Liverpool Bay (River Mersey), Solent & Dorset Coast. In summary, this work (Natural England 2016) confirmed the presence of terns in every system surveyed, their occurrence at almost every station in every system in which they were looked for and, in some cases, e.g. Hamford Water, Solent & Dorset Coast, their occurrence beyond the proposed site boundaries. In the case of the River Tees it was concluded that "the incorporation of the River Tees as far upstream as Tees Barrage within the proposed SPA could be verified by the current surveys". This conclusion was supported by the results of a further series of surveys on the River Tees in 2016 which replicated the surveys in 2015 (INCA 2016). Thus, in regards to the proposal to revise the boundary of the SPA to accommodate the principle areas of use by common terns within the pSPA, the conclusions drawn clearly relate to the evidence and analysis.

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

Count data

The BTO's Wetland Bird Survey is a long-established and internationally recognised monitoring scheme coordinated by BTO in partnership with others (e.g. statutory nature conservation bodies, the RSPB and WWT). It promotes the collection of data according to standardised field methods (http://www.bto.org/volunteer-surveys/webs). WeBS data are collected and reported by skilled and experienced bird surveyors and are checked and verified by BTO regional representatives and national data managers. Therefore, there is high confidence in WeBS data. The majority of the data which has been used in determining the size of the populations of each of the non-breeding species considered for inclusion as features of the pSPA are summaries on the WeBS database and thus available for public scrutiny.

County Bird Report data are verified and quality assured by the County Bird Reporter and the reports prepared under the supervision of an editorial team of experienced local ornithologists. The Cleveland Bird Recorder Tom Francis oversees these processes and discussions. There is therefore high confidence that the data within the Cleveland Bird Report are accurate and reliable. Accordingly, even the most recent count data referred to in this Departmental Brief can be considered to justify high confidence.

Landward boundary

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

Seaward boundary

The position of the seaward boundary of the pSPA has been determined on the basis of outputs of direct observations of the distribution of little terns and a statistical model of common tern usage which is 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 common tern foraging locations. However, provided the models are empirically evidence based, and shown to be robust via cross validation, the modelling approach brings with it a robustness which may exceed that which might be achieved from reliance on a limited empirical dataset of tern foraging locations. It is considered that the cross-validation analyses and sample-size sufficiency analyses indicate that proposed boundaries generated by the modelling approach have degrees of uncertainty that are acceptable, and certainly need not be considered to be any worse than if it had been possible to apply more conventional approaches. This issue is discussed fully in Annex 6.

Terrestrial extension

The individual terrestrial extensions are supported, where available, by WeBS data and the County Bird reports. As all data are subject to the same recording and reporting protocols and verification processes, confidence in the data supporting the extension is equal to that for the remainder of the pSPA. Furthermore, as all WeBS counts are carefully coordinated, the importance of each extension can be justified by its contribution to the wider pSPA. Other data for some extension areas are more limited and not comparable to WeBS data. These data are generally used as supporting information rather than as the main basis for proposing an extension. They are not included in the calculation of overall population estimates for the site and do not contribute to the overall assemblage figures. Where it is necessary to rely on such data then reference to other historical data can help to provide justification.

5.) Independent expert review and internal quality assurance processes

Independent expert review

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

The proposed amendments to the SPA have been made on the basis of an assessment of standard breeding bird and over-wintering bird datasets i.e. the County Bird Reports and the WeBS database. The count data have been assessed against and conform with the SPA selection guidelines (JNCC 1999). Natural England believes these amendments not to be contentious and therefore independent review of how it has applied the evidence in making these amendments is not being sought.

The proposal to alter the landward boundary of the designated site to include various additional terrestrial land parcels e.g. area of wet grassland is not dependent upon either highly novel or technically difficult analysis to inform the revised boundary. Natural England believes the amendments not to be contentious and therefore independent review of how it has applied the evidence in drawing up a boundary is not being sought.

The derivation of the alongshore extent and seaward boundary to the pSPA is based on a relatively novel approach 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.

6.) 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 have 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.

This Departmental Brief was drawn up by Allan Drewitt (Senior Environmental Specialist) of Natural England in collaboration with Richard Caldow (Senior Environmental Specialist: Marine Ornithology), Katie Finkill-Coombs (Lead Adviser), Tom Charman (Lead Adviser) and Mike Leakey (Senior Reserve Manager) of Natural England. This Departmental Brief was internally quality assured by the marine Natura 2000 Project Board and externally by Defra (Niall Malone), JNCC (Julie Black/Kerstin Kober), UKMBPSG and MPATG.

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