DEPARTMENTAL BRIEF:

Liverpool Bay / Bae Lerpwl potential Special Protection Area (pSPA)

Proposal for extension to existing site and adding new features

Advice to the Welsh Government and UK Government

Natural England

Natural Resources Wales

Joint Nature Conservation Committee

March 2016

CRYNODEB

Caiff Ardal Gwarchodaeth Arbennig (AGA) arfaethedig Liverpool Bay / Bae Lerpwl, y sonnir amdani yn y Brîff Adrannol hwn, ei chynnig er mwyn gwarchod rhannau pwysig o'r arfordir a'r môr a ddefnyddir at ddibenion amrywiol gan y nodweddion cymwys. Estyniad yw'r AGAa o AGA bresennol Liverpool Bay / Bae Lerpwl. Cedwir nodweddion yr AGA bresennol (môr-hwyaid duon (*Melanitta nigra)*, trochyddion gyddfgoch (*Gavia stellata*), casgliad o adar dŵr) gyda gwybodaeth sylfaenol ddiweddaredig, a chaiff nodweddion cymwys newydd eu hychwanegu yn seiliedig ar adolygiad o helaethrwydd yr adar ar hyn o bryd o fewn terfynau'r AGAa.

Mae'r estyniad arfaethedig yn cynnwys ardal i'r gogledd a'r gorllewin o'r AGA bresennol, a bennwyd ar gyfer cynnal gwylanod bychain (*Hydrocoloeus minutus)* nad ydynt yn bridio. Mae hefyd yn cynnwys ardal chwilota am fwyd ar gyfer môr-wenoliaid a bennir ac a ddiffinnir gan fôr-wenoliaid bychain (*Sternula albifrons)* sy'n bridio yn AGA The Dee Estuary a'r ardal chwilota am fwyd ddisgwyliedig ar gyfer môr-wenoliaid cyffredin (*Sterna hirundo)* sy'n bridio yn AGA Mersey Narrows & North Wirrall Foreshore. Mae'r ardaloedd hyn yn ychwanegu cynefin morol sy'n ymestyn i Aber Afon Merswy, ac ardal rynglanwol fechan sy'n ffinio â therfyn gorllewinol AGA The Dee Estuary. Mae safle'r AGAa yn gorgyffwrdd â safle posibl ar gyfer AGAa Anglesey Terns / Morwenoliaid Ynys Môn.

Felly, mae AGAa Liverpool Bay / Bae Lerpwl yn cynnwys ardaloedd chwilota am fwyd ar gyfer adar môr sy'n bridio, ac adar môr ac adar dŵr nad ydynt yn bridio.

Dyma'r nodweddion newydd arfaethedig: gwylanod bychain, môr-wenoliaid cyffredin a môr-wenoliaid bychain. Mae hwyaid brongoch a mulfrain yn brif gydrannau newydd yn y casgliad o adar dŵr. Mae AGAa Liverpool Bay / Bae Lerpwl yn gymwys dan Erthygl 4 y Gyfarwyddeb Adar (2009/147/EC) ar sail y rhesymau a ganlyn:

- *Rhywogaethau a restrir yn Atodiad I y Gyfarwyddeb Adar*: mae'r safle'n cynnal yn rheolaidd fwy nag 1% o boblogaethau Prydain Fawr o ddwy rywogaeth sy'n bridio ac un rhywogaeth nad yw'n bridio (Tabl 1). O'r herwydd, mae'r safle'n gymwys i gael ei ddosbarthu fel AGA yn unol â chanllawiau dethol AGA y DU (cam 1.1: JNCC 1999).

- *Adar ymfudol a welir yn rheolaidd, na restrir mohonynt yn Atodiad I y Gyfarwyddeb Adar*: mae'r safle'n cynnal yn rheolaidd fwy nag 1% o boblogaethau bioddaearyddol un rhywogaeth nad yw'n bridio (Tabl 1). O'r herwydd, mae'r safle'n gymwys i gael ei ddosbarthu fel AGA yn unol â chanllawiau dethol AGA y DU (cam 1.2: JNCC 1999).

- *Casgliadau*: mae'r safle'n cynnal yn rheolaidd gasgliad o fwy nag 20,000 o adar dŵr unigol. O'r herwydd, mae'r safle'n gymwys i gael ei ddosbarthu fel AGA yn unol â chanllawiau dethol AGA y DU (cam 1.3: JNCC 1999).

- *Rhywogaethau na ellir eu cynnwys oddi mewn i ganllawiau cam 1:* mae'r safle hwn yn cynnal yn rheolaidd un rhywogaeth nad yw'n bridio sydd yn Atodiad I y Gyfarwyddeb Adar ond na ellir ei dewis yng ngham 1.1 gan na cheir amcangyfrif ar gyfer ei phoblogaeth genedlaethol at ddibenion cymharu (Tabl 1). Nodir bod y safle'n cynnal y cydgasgliad mwyaf ond un o wylanod bychain yn y DU, ac o'r herwydd mae'n gymwys i gael ei ddosbarthu fel AGA yn unol â chanllawiau dethol AGA y DU (cam 1.4: JNCC 1999).

Tabl 1. Crynodeb o ddiddordeb adaregol cymwys AGAa Liverpool Bay / Bae Lerpwl.

Rhywogaeth	**Nifer (cyfnod)**	**% o'r isrywogaeth neu'r boblogaeth**	**Math o ddiddordeb**	**Meini prawf dethol**	**Rhywogaeth gymwys newydd**
Yn ystod y tymor nythu					
Môr-wennol fechan *Sternula albifrons*	260 o adar unigol[1] (2010 – 2014)	6.84% o boblogaeth PF	Atodiad 1	Cam 1.1	Ie
Môr-wennol gyffredin *Sterna hirundo*	360 o adar unigol[2] (2011 – 2015)	1.80% o boblogaeth PF	Atodiad 1	Cam 1.1	Ie
Y tu allan i'r tymor nythu					
Trochydd gyddfgoch *Gavia stellata*	1,171 o adar unigol (2004/05 – 2010/11)	6.89% o boblogaeth PF	Atodiad 1	Cam 1.1	Na
Gwylan fechan *Hydrocoloeus minutus*	319 o adar unigol (2004/05 – 2010/11)	Amherthnasol[3]	Atodiad 1	Cam 1.4[4]	Ie
Môr-hwyaden ddu *Melanitta nigra*	56,679 o adar unigol (2004/05 – 2010/11)	10.31% o boblogaeth Gogledd Orllewin Ewrop	Aderyn ymfudol a welir yn rheolaidd	Cam 1.2	Na
Casgliad o adar dŵr rhyngwladol bwysig o fwy nag 20,000 o adar unigol	69,687 o adar unigol (2004/05 – 2010/11)	Amherthnasol	Casgliad	Cam 1.3	Na

[1]130 o barau (safleoedd nythu tebygol/AOS) yn Nhraeth Gronant sydd wedi deillio o gronfa ddata'r Rhaglen Monitro Adar Môr/'Seabird Monitoring Programme'. Mae'r ffigurau hyn yn cynrychioli poblogaeth bresennol y safle (SMP, cyfathrebiad personol). Y boblogaeth 'adeg dosbarthu' ar gyfer môr-wenoliaid bychain yn AGA The Dee Estuary yw 138 o adar unigol (1995-1999).

[2]180 o barau (safleoedd nythu tebygol/AOS) yng Ngwarchodfa Natur Seaforth sydd wedi deillio o gronfa ddata'r 'Seabird Monitoring Programme'.

[3]Ni cheir amcangyfrif ar gyfer poblogaeth gwylanod bychain ar raddfa PF. Defnyddir gwerth enwol o 50 yn gyffredin fel terfyn isaf, a cheir mwy na'r gwerth hwn yn rhwydd.

[4]Gan na cheir amcangyfrif ar gyfer y boblogaeth genedlaethol, dewisir y rhywogaeth yn ystod Cam 1.4.

SUMMARY

Liverpool Bay / Bae Lerpwl potential Special Protection Area (pSPA) detailed in this Departmental Brief is proposed to protect important areas of coast and sea used for a variety of purposes by the qualifying features. The pSPA is an expansion of the existing Liverpool Bay / Bae Lerpwl SPA, the features of the existing SPA (common scoters (*Melanitta nigra)*, red-throated divers (*Gavia stellata*), waterbird assemblage) are retained with updated baseline information, and new qualifying features are added based on a review of current bird abundance within the pSPA boundary.

The proposed extension includes an area to the north and west of the existing SPA, identified to support non-breeding little gulls (*Hydrocoloeus minutus)*. It also includes a marine foraging area for terns identified and defined by little terns (*Sternula albifrons)* breeding within The Dee Estuary SPA and the predicted foraging area for common terns (*Sterna hirundo)* breeding within Mersey Narrows & North Wirral Foreshore SPA. These areas add marine habitat extending into the Mersey Estuary, and a small intertidal area abutting the western boundary of The Dee Estuary SPA. The new pSPA site overlaps with the potential site for Anglesey Terns / Morwenoliaid Ynys Môn pSPA.

The Liverpool Bay / Bae Lerpwl pSPA therefore comprises areas for foraging breeding seabirds, and non-breeding seabirds and waterbirds.

The new features proposed are little gull, common tern and little tern. Red-breasted merganser and cormorant are new main components of the waterbird assemblage.

Liverpool Bay / Bae Lerpwl pSPA qualifies under Article 4 of the Birds Directive (2009/147/EC) for the following reasons:

- *Species listed in Annex I of the Birds Directive*: the site regularly supports more than 1% of the Great Britain populations of two breeding species and one non-breeding species (Table 1). Therefore the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.1: JNCC 1999).

- *Regularly occurring migrants not listed in Annex I of the Birds Directive*: the site regularly supports more than 1% of the biogeographical populations of one non-breeding species (Table 1). Therefore the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.2: JNCC 1999).

- *Assemblages*: the site regularly supports an assemblage of more than 20,000 individual waterbirds. Therefore the site qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.3: JNCC 1999).

- *Species for which stage 1 guidelines cannot be applied*: the site regularly supports one non-breeding species which is on Annex I of the Birds Directive but which cannot be selected at stage 1.1 because there is no national population estimate for comparison (Table 1). The site is identified as supporting the second largest aggregation of little gulls in the UK, and therefore qualifies for SPA classification in accordance with the UK SPA selection guidelines (stage 1.4: JNCC 1999).

Table 1. Summary of qualifying ornithological interest in Liverpool Bay / Bae Lerpwl pSPA.

Species	Count (period)	% of subspecies or population	Interest type	Selection criteria	New qualifier
In the breeding season					
Little tern *Sternula albifrons*	260 individuals[1] (2010 – 2014)	6.84% of GB population	Annex 1	Stage 1.1	Yes
Common tern *Sterna hirundo*	360 individuals[2] (2011 – 2015)	1.80% of GB population	Annex 1	Stage 1.1	Yes
In the non-breeding season					
Red-throated diver *Gavia stellata*	1,171 individuals (2004/05 – 2010/11)	6.89% of GB population	Annex 1	Stage 1.1	No
Little gull *Hydrocoloeus minutus*	319 individuals (2004/05 – 2010/11)	N/A[3]	Annex 1	Stage 1.4[4]	Yes
Common scoter *Melanitta nigra*	56,679 individuals (2004/05 – 2010/11)	10.31% of NW European population	Regularly occurring migrant	Stage 1.2	No
Internationally important waterbird assemblage of over 20,000 individuals	69,687 individuals (2004/05 – 2010/11)	N/A	Assemblage	Stage 1.3	No

[1] 130 pairs (Apparently Occupied Nests) at Gronant Beach from Seabird Monitoring Programme database. These figures represent the current population at the site (SMP, pers. comm.). The 'at classification' population for little tern in The Dee Estuary SPA is 138 individuals (1995-1999).
[2] 180 pairs (Apparently Occupied Nests) at Seaforth Nature Reserve from Seabird Monitoring Programme database.
[3] There is no population estimate at the GB scale for little gulls. A nominal value of 50 is conventionally used as a minimum threshold, which is comfortably exceeded.
[4] As there is no national population estimate, this species is selected at Stage 1.4.

Assessment against SPA Selection Guidelines

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

1.1. Stage 1

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

Stage 1.1 an area is used regularly by 1% or more of the Great Britain (or in Northern Ireland, the all-Ireland) population of a species listed in Annex I of the Birds Directive (2009/147/EC) in any season;

Stage 1.2 an area is used regularly by 1% or more of the biogeographical population of a regularly occurring migratory species (other than those listed in Annex I) in any season;

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

Stage 1.4 an area which meets the requirements of one or more of the Stage 2 guidelines in any season, where the application of Stage 1 guidelines 1, 2 or 3 for a species does not identify an adequate suite of most suitable sites for the conservation of that species.

Liverpool Bay / Bae Lerpwl pSPA qualifies under stage 1.1 because it regularly supports more than 1% of the GB population of Annex I species in the breeding (common tern, little tern) and non-breeding seasons (red-throated diver). In addition, the site qualifies under stage 1.2 because it regularly supports over 1% of the biogeographical population of a regularly occurring migratory bird (common scoter). It qualifies under stage 1.3 by regularly supporting a waterbird assemblage of over 20,000, including all non-breeding qualifying features as well as two additional species (cormorant and red-breasted merganser) as ‘main components’ (see section 5.5.1. Finally, it qualifies under stage 1.4 because it regularly supports an Annex I species (little gull) present in numbers indicating it is the second most important area thus far identified in the UK.

1.2. Stage 2

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

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

Feature	Qualification	Assessment
1. Population size & density	✓	The site supports comfortably the largest aggregation of common scoters in the UK, as well as the second largest aggregation of little gulls and fourth largest aggregation of red-throated divers. It also supports foraging areas for nearly 7% of the GB population of little terns, and nearly 2% of the GB population of common terns.
2. Species range	✓	The pSPA is the main non-breeding area for common scoters in the UK, and the UK represents the NW extent of the species' non-breeding range (holding at least 12.5% of the biogeographic total: BirdLife International 2015). The pSPA is the only site proposed for little gulls in the west of Britain. Gronant Beach is the only little tern breeding site in Wales.
3. Breeding success	✓	Little tern productivity at Gronant Beach has exceeded the UK average of 0.51 chicks per pair (Cook & Robinson 2010) in seven of the last ten breeding seasons (2005 – 2014: SMP data). Average productivity at the Mersey Narrows & North Wirral Foreshore SPA is ranked 12 of 24 sites assessed in the UK (Horswill & Robinson 2015) at 0.61 chicks per pair.
4. History of occupancy	✓	Large aggregations of common scoters and red-throated divers began to be discovered through a programme of aerial surveys between 2001 and 2006 (Smith *et al*. 2007; O'Brien *et al*. 2008). Therefore there is a history of occupancy dating back almost 15 years, although it is highly likely scoters and divers were present before our knowledge developed (Lovegrove *et al.* 1994). Little terns and common terns have bred at locations adjacent to the pSPA for many years: The Dee Estuary SPA was originally classified in 1985, whereas Mersey Narrows & North Wirral Foreshore SPA was classified in 2013. However, common terns have bred in qualifying numbers for over 15 years. There is every reason to believe the foraging areas within the pSPA would have been used for an equal period, given the foraging ranges of the relevant terns are unlikely to have changed significantly.
5. Multi-species area	✓	Five features qualify in total, as well as an assemblage containing 18 other component species.
6. Naturalness	N/A	No longer applicable, following ruling from the SPA & Ramsar Scientific Working Group.
7. Severe weather refuge	**?**	No data are available to determine whether the pSPA acts as a severe weather refuge for the non-breeding features. Numbers of birds within the pSPA do fluctuate, but the reasons are not fully understood.

2. Rationale and data underpinning site classification

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

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

Strand 1: marine extensions of existing seabird breeding colony SPAs beyond the low water mark;

Strand 2: inshore feeding areas used by concentrations of birds (e.g. seaduck, grebes and divers) in the non-breeding season; and

Strand 3: offshore areas used by seabirds for feeding and other activities 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 marine SPA http://jncc.defra.gov.uk/page-4184 identified for marine birds that may not be addressed by the above three categories and will be considered individually.

To implement conservation measures under **Strand 1**, the JNCC produced generic guidance (McSorley *et al.* 2003, 2005, 2006; Reid & Webb 2005) to extend the seaward extent of SPA boundaries from seabird colonies. The seaward extensions of existing boundaries in these cases include waters vital for ensuring that some of the essential ecological requirements of the breeding seabird populations are met (e.g. preening, bathing, displaying and potentially local foraging). The extent of the extension is dependent upon the qualifying species breeding within the SPA. However, these generic boundary extensions are not influenced by or meant to encompass the principal foraging areas used by the species for which they are identified or any other species at the colonies concerned. Generic seaward extensions to the boundaries of existing SPAs have been implemented at 31 sites in Scotland and are under consideration at the Flamborough and Filey Coast pSPA (Natural England 2014). However, in line with the recommendations of Reid & Webb (2005), generic extensions have only been implemented at sites holding certain seabird species, none of which occur as breeding birds within the existing SPAs which border Liverpool Bay / Bae Lerpwl pSPA. Reid & Webb (2005) note that no evidence has been found that any of the five species of tern which breed regularly in Great Britain make significant use of waters around their colony for maintenance activity (McSorley *et al.* 2003) and conclude that generic guidance for extension of colony SPAs for this purpose is not appropriate in the case of terns.

The original Liverpool Bay / Bae Lerpwl SPA was classified under **Strand 2** in 2010. Classification was for the marine area supporting a population of 54,675 common scoters and 922 red-throated divers in the non-breeding season (JNCC, 2011). As no boundary changes are proposed for these species, this Departmental Brief will not focus on the scientific case for boundary definition of common scoters and red-throated divers. The original features are retained, although updates are made to their abundance estimates according to newly available data; data from more winter surveys were available to inform the most recent estimates, and thus these are considered more robust. Proposals for the addition of little gulls to the pSPA also fall under strand 2.

All five species of tern that regularly breed in the UK (Arctic tern *Sterna paradisaea*, common tern *S. hirundo*, Sandwich tern *S. sandvicensis,* roseate tern *S. dougallii* and little tern *S. albifrons*) are listed on Annex I of the EU Birds Directive and thus are subject to special conservation measures including the classification of SPAs. Within the UK there are currently 57 breeding colony SPAs for which at least one species of tern is protected. However, additional important areas for terns foraging at sea have yet to be identified and classified as marine SPAs to complement the existing terrestrial suite. Since 2007, the JNCC has been working with the four Statutory Nature Conservation Bodies (SNCBs) towards the identification of such areas under **Strand 4** as, given the likely extent of these areas, these cannot be addressed by application of the generic maintenance extensions approach and are not covered by the work on identifying inshore non-breeding aggregations or important offshore areas due to difficulties in identification of terns and to limited survey coverage closer to shore (terns have limited foraging ranges compared to other seabird species). The collection of evidence of tern foraging areas within Liverpool Bay / Bae Lerpwl pSPA falls under this strand.

With more data becoming available in recent years (in particular on the terns), JNCC sought to identify important areas under Strand 4 for:

- Little gull at the Liverpool Bay / Bae Lerpwl SPA
- Little terns at The Dee Estuary SPA
- Large terns at Mersey Narrows and North Wirral Foreshore SPA

Resulting areas were overlapping, so they were incorporated into the extended Liverpool Bay pSPA.

In the process by which a site becomes fully classified as an SPA, Ministerial approval has to be given to undertake formal consultation on the proposal to classify the site. At this stage in the process a site becomes known as a potential SPA (pSPA). Within this Departmental Brief, for the purpose of clarity sites are referred to as SPAs when referring to existing classified sites. Where reference is made to an entirely new site, or to an extended site, or to a site including new features being proposed (such as Liverpool Bay / Bae Lerpwl), it will be referred to as pSPA since the site, with its additional extent and features, is not yet fully classified. The existing Liverpool Bay SPA and its features remain fully classified.

This Departmental Brief sets out information supporting the identification of the qualifying features of the Liverpool Bay / Bae Lerpwl pSPA and definition of its proposed boundaries. This is based upon the areas of sea identified as being most important for non-breeding little gulls, and the tern populations that comprise the qualifying features of this new marine SPA – namely little terns breeding at Gronant Beach within The Dee Estuary SPA, and common terns breeding at Seaforth within the Mersey Narrows & North Wirral Foreshore SPA.

2.1. Data collection – defining the abundance of tern species supported by Liverpool Bay / Bae Lerpwl pSPA

The size of each of the populations of terns supported by the Liverpool Bay / Bae Lerpwl pSPA, and which exceed the SPA qualifying thresholds, have been derived as the sum of the numbers of those species at each of the existing SPAs from which the individuals recorded at sea within the pSPA are most likely to originate. Citation figures[5] were considered out of date and therefore inappropriate for use in defining the sizes of the populations of these species supported by the entirely new pSPA. Therefore, for each of the source SPAs, the numbers are the most recently available from the Seabird Monitoring Programme (SMP) database (i.e. within the last six years).

[5] Within this departmental brief current population figures for little terns have been used to demonstrate how the site currently meets selection guidelines and in the proposed citation for Liverpool Bay / Bae Lerpwl pSPA the 'at classification' population for little tern in The Dee Estuary SPA has been used to provide consistency for Wales in this cross border site

2.2. Defining the boundary of Liverpool Bay / Bae Lerpwl pSPA

The proposed adjustments to the existing boundary (defined for non-breeding common scoter and red-throated diver distributions) of Liverpool Bay / Bae Lerpwl pSPA are based on the distribution of little gulls and terns. The seaward boundary is expanded north and west to incorporate key areas for non-breeding little gulls, and some additional nearshore areas are proposed to allow for tern foraging requirements. The studies to identify important areas for little tern (Parsons *et al* 2015), all large tern species (Wilson *et al*. 2014) and little gull (Lawson *et al* 2015), are described in brief in the following two sub-sections. The overall site boundary was drawn as a composite of the separate species-specific boundaries and this is described in section 3.

3. Site Status and Boundary

3.1. Existing Boundary

The total area of the existing Liverpool Bay / Bae Lerpwl SPA is approximately 170,293 ha and lies within the 12 nautical mile limit (Figure 1).

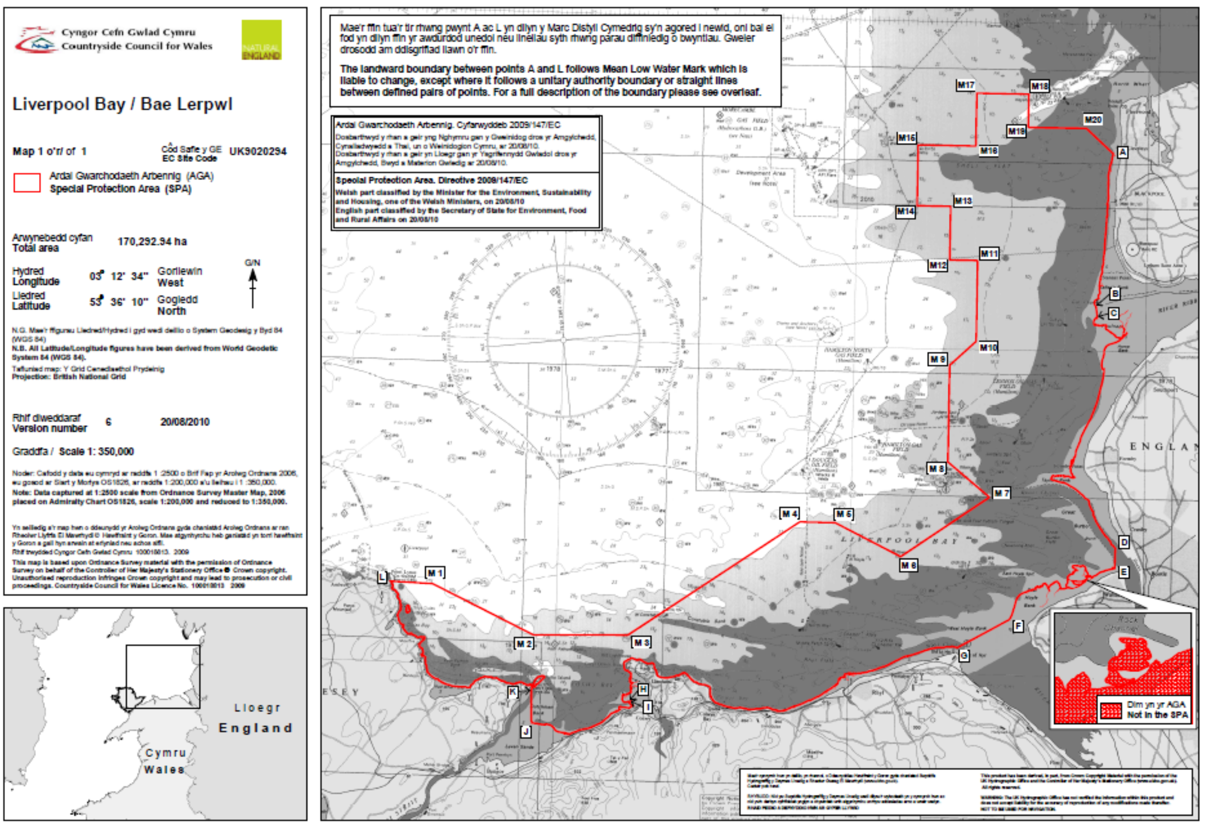

Figure 1 – Existing boundary of Liverpool Bay / Bae Lerpwl SPA.

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

- Ribble and Alt Estuaries SPA
- Mersey Narrows & North Wirral Foreshore SPA
- The Dee Estuary SPA
- Traeth Lafan, Lavan Sands, Conway Bay SPA
- Ynys Seiriol / Puffin Island SPA

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

3.2. Liverpool Bay / Bae Lerpwl pSPA Boundary

The total area of the Liverpool Bay / Bae Lerpwl pSPA is approximately 252,773 ha. The new area proposed comprises approximately 82,481 ha (Figure 2).

The proposed boundary changes to the existing Liverpool Bay / Bae Lerpwl SPA are based upon foraging areas for little tern and projected foraging areas of common terns breeding within qualifying coastal SPAs, and the most important areas for non-breeding little gulls.

The proposed boundary change has been drawn to encompass the qualifying areas of these species identified by maximum curvature (apart from little tern whose proposed extension was based on survey data). A significant portion (17,825 ha (22.4%)) of the new area extends seaward of the 12nm boundary.

The seaward and alongshore extent of Liverpool Bay / Bae Lerpwl pSPA (Annex 1a) is largely determined by the boundaries of the existing Liverpool Bay / Bae Lerpwl SPA, which is defined according to the distribution of non-breeding common scoters and red-throated divers. The new areas are:

a. New seaward areas to the north and west of the existing seaward SPA boundary, to incorporate important non-breeding little gull areas;
b. A small area of intertidal habitat to the west of The Dee Estuary SPA to incorporate little tern foraging requirements;
c. A small area of habitat along the north Wirral coast to incorporate common tern foraging requirements; and
d. A new area in the Mersey Estuary to incorporate common tern foraging requirements.

In total, the additional areas encompass 82,481 ha, an increase of 48.4% from the existing SPA area.

The Liverpool Bay / Bae Lerpwl pSPA overlaps with the Anglesey Terns / Morwenoliaid Ynys Môn pSPA currently under consideration. This pSPA is an extension of the existing SPA Ynys Feurig, Cemlyn Bay and The Skerries SPA, which is an SPA for common tern, Arctic tern, Sandwich tern and roseate tern. Information about this proposed pSPA can be found on NRW's web site:
http://www.naturalresources.wales/about-us/consultations/our-own-consultations/proposed-new-marine-special-areas-of-conservation-and-special-protection-areas/anglesey-terns/?lang=en

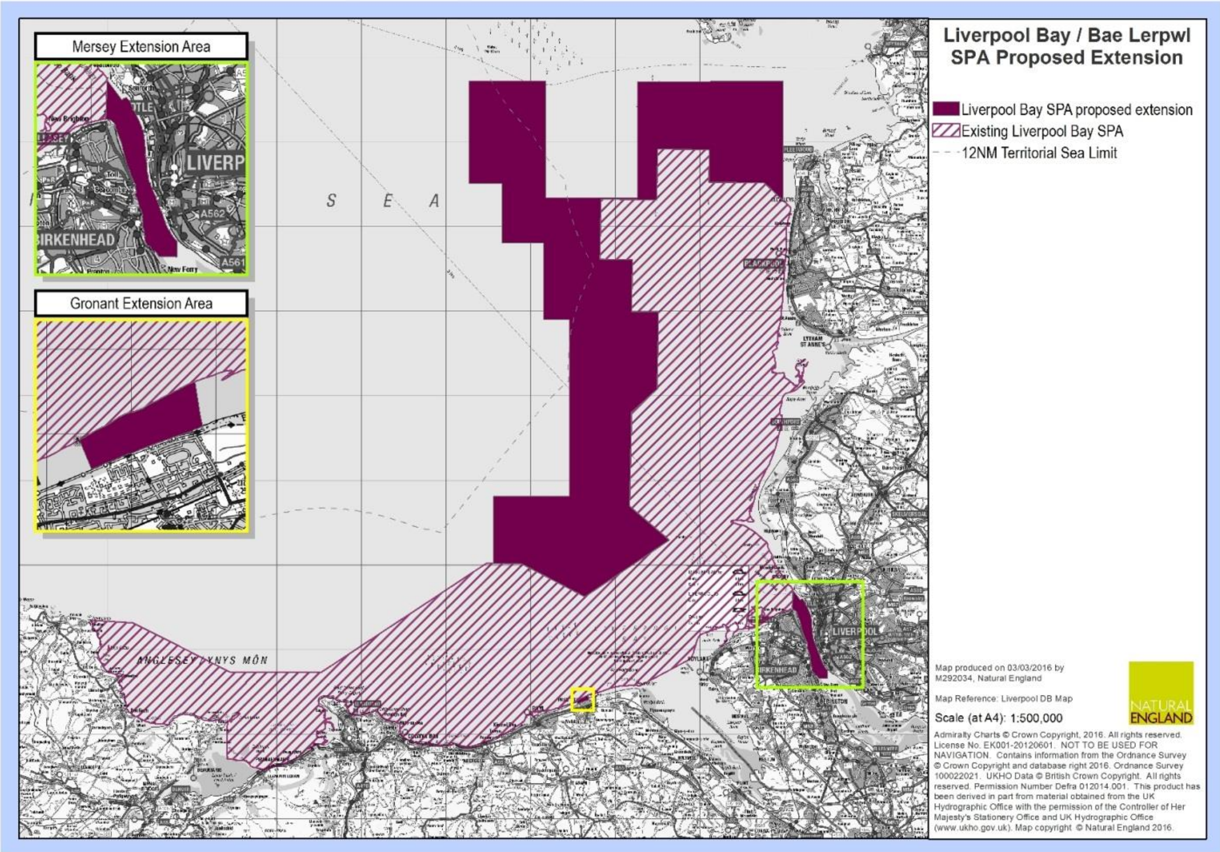

Figure 2 – Liverpool Bay / Bae Lerpwl pSPA (original site and proposed extension areas).

3.3. Seaward boundary of the pSPA

The assessment of important areas for little gulls within Liverpool Bay is described in full in Lawson *et al*. (2015). In summary, data from five winter seasons (2004/05, 2005/06, 2006/07, 2007/08 and 2010/11) were analysed; all data were collected using visual aerial transect surveys according to standard seabird survey protocol (Camphuysen *et al*. 2004). Numbers of complete surveys within a season varied between one and four, and 13 surveys were analysed in total across the five winters. Established methods for marine SPA boundary setting (Distance Sampling; Kernel Density Estimation; Maximum Curvature: O'Brien *et al*. 2012) were used to define the most important marine areas for non-breeding little gulls (Figure 3); hotspot analysis demonstrated that the most important areas were consistently used in densities above thresholds identified by Maximum Curvature and thus there is strong confidence in the regularity of use of these marine areas (Lawson *et al*. 2015). An alternative analytical method, Density Surface Modelling (DSM), identified similar important areas for non-breeding little gull in Liverpool Bay when it was run on the same data (Bradbury *et al.* 2014).

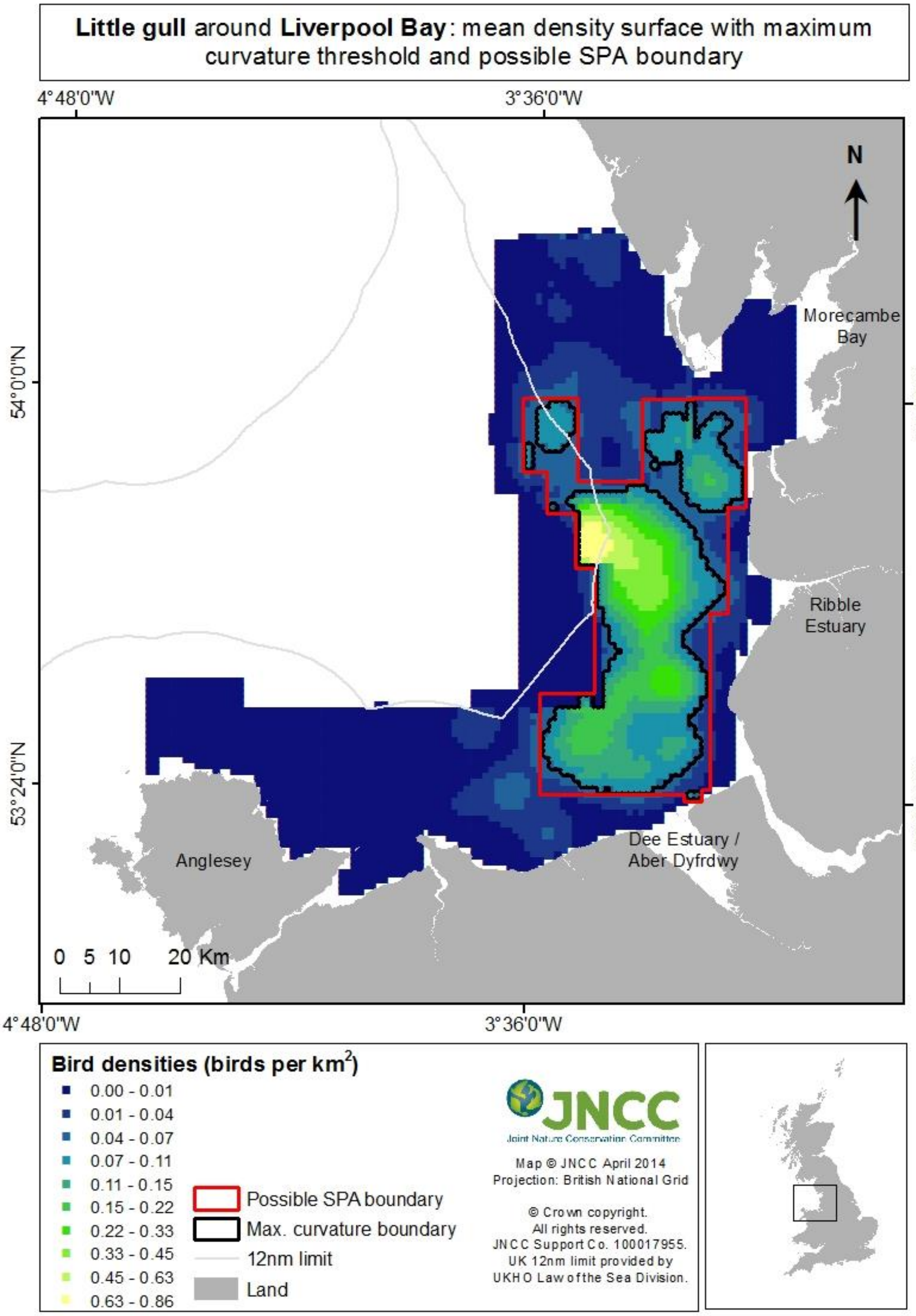

Figure 3 - Important areas for non-breeding little gulls in Liverpool Bay, as defined by Kernel Density Estimates and Maximum Curvature (Lawson *et al*. 2015).

The resulting seaward boundary of the Liverpool Bay / Bae Lerpwl pSPA is a composite of the existing Liverpool Bay / Bae Lerpwl SPA boundary (as defined for common scoters and red-throated divers) and the important areas for little gulls (Figure 4).

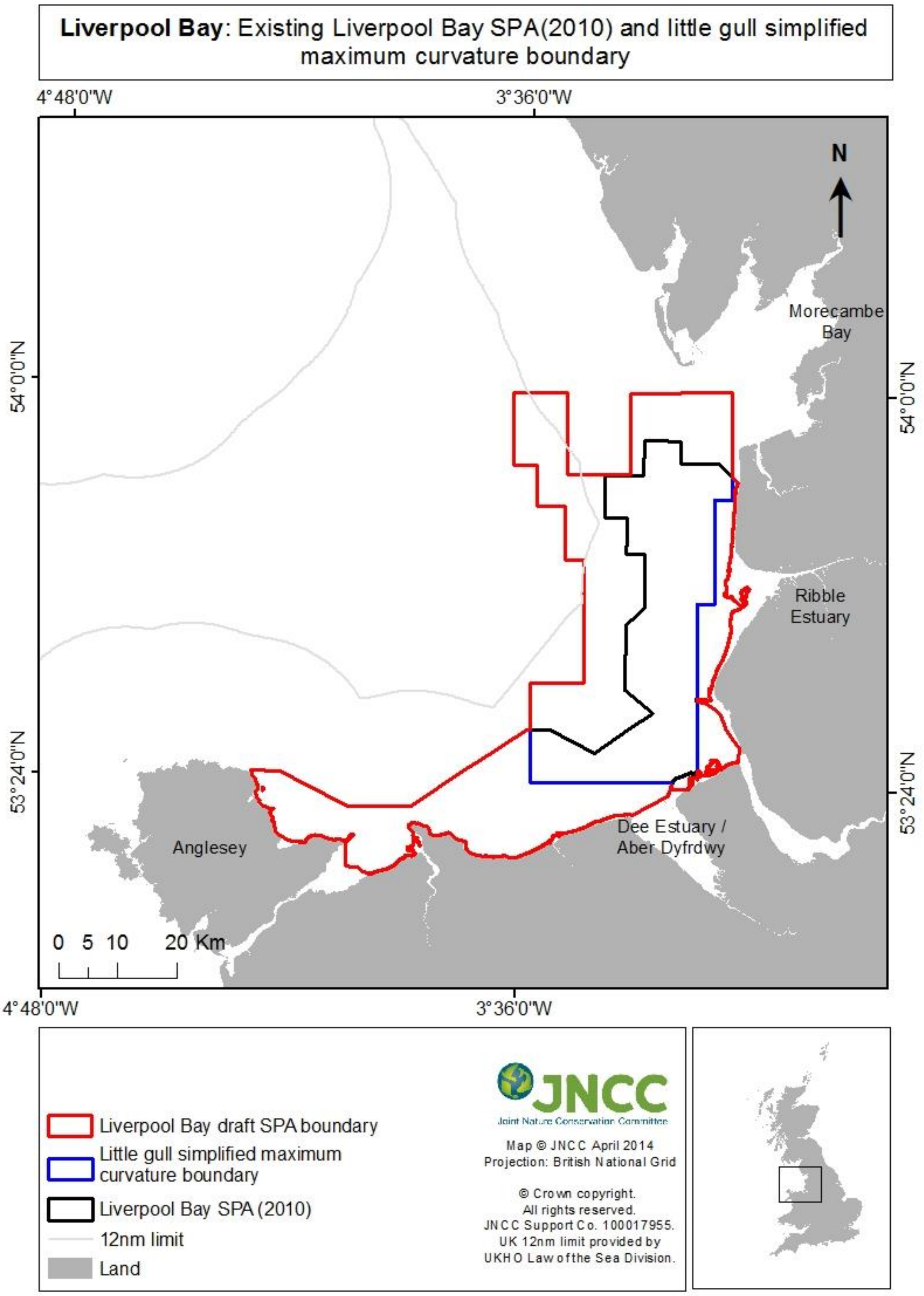

Figure 4 - Proposed composite seaward boundary for Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015). Blue and black lines would be subsumed within the red line denoting the entire pSPA.

3.4. Landward boundary of the pSPA

It is proposed that the landward boundary of Liverpool Bay / Bae Lerpwl SPA will continue to follow the Mean Low Water mark or the seaward boundaries of existing SPAs, whichever is the furthest seaward, except where the boundary is extended for foraging terns in which case it will follow Mean High Water mark or the seaward boundaries of existing SPAs. This is because foraging terns will use the full range of the intertidal area. Consequently, the landward boundary of Liverpool Bay / Bae Lerpwl SPA will be adjacent to (but not abut) Morecambe Bay SPA (proposed to be subsumed by Morecambe Bay & Duddon Estuary pSPA) and directly abut the seaward boundaries of Ribble and Alt Estuaries SPA, Mersey Narrows and North Wirral Foreshore SPA, The Dee Estuary SPA and Traeth Lafan / Lavan Sands, Conway Bay SPA. There is a small overlap with the northernmost edge of Mersey Estuary SPA. In addition, the site entirely surrounds Ynys Seiriol / Puffin Island SPA. Intertidal mudbanks and sandbanks separated from the mainland coast by subtidal areas at mean low water are within the SPA boundary, except where they are within the boundaries of existing SPAs, as is the case in parts of the Mersey Narrows and North Wirral Foreshore SPA and The Dee Estuary SPA.

3.4.1. Identification of important areas for little terns

Little tern is the smallest, regularly breeding tern in Britain, and has the most limited foraging range. The mean range is 2.1 km, mean of recorded maxima is 6.3 km and the maximum ever recorded is 11 km (Thaxter *et al.* 2012). Thus JNCC and the Statutory Nature Conservation Bodies (SNCBs) undertook shore and boat-based observational surveys near to colonies in order to determine the most heavily used foraging areas of breeding little terns.

As part of this overall little tern analysis programme, three years of data collection was undertaken around The Dee Estuary SPA located adjacent to Liverpool Bay / Bae Lerpwl pSPA; four shore-based surveys were undertaken in 2009, and two were completed in each of 2010 and 2011. These collected 792 little tern observations. Two boat-based surveys were also completed (one in 2010 and one in 2011) and recorded 45 little tern observations (Parsons *et al*. 2015). The total number of observations for both shore and boat-based surveys was judged to be sufficient to justify a site-specific approach to boundary definition. The alongshore foraging extent for this colony was set to be 3 km to both east and west of the nesting location. The mean of maximum seaward foraging extents for this colony of little terns was 1.87 km (Figure 5; Parsons *et al*. 2015).

The little tern foraging area is mostly contained within either the existing Liverpool Bay / Bae Lerpwl SPA boundary, or the abutting intertidal area of The Dee Estuary SPA. The exception is a small (approx. 1 km x 0.25 km) area of intertidal habitat to the west of the boundary of The Dee Estuary SPA; the proposed Liverpool Bay / Bae Lerpwl pSPA boundary extension will cover this area so that foraging little terns are protected in their entire expected foraging range. As they are assumed to be able to forage anywhere within the expected area, the birds are qualifying features of both The Dee Estuary SPA and Liverpool Bay / Bae Lerpwl pSPA.

Further general information on the little tern survey programme is presented in Parsons *et al*. (2015) and Annex 4.

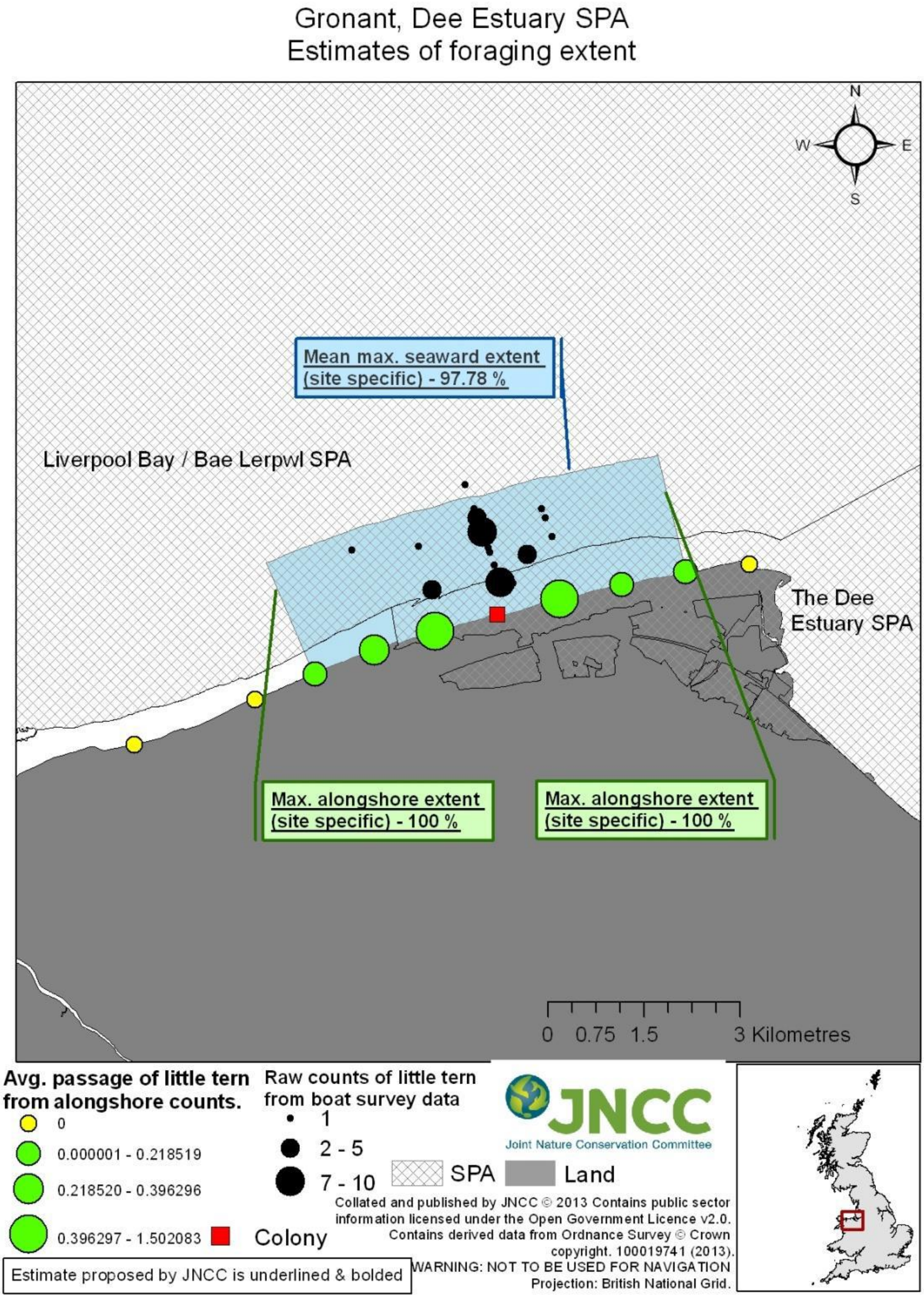

Figure 5 - Foraging extent of little terns around The Dee Estuary SPA (Parsons *et al*. 2015).

3.4.2. 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). Given the prohibitive costs of undertaking site specific boat based surveys across all tern SPAs, targeted visual tracking surveys were undertaken at selected colonies and the information was used to generate habitat association models. These could be used to predict tern usage patterns across the full extent of the sea areas within published foraging range of any colony, and to identify within those areas which were the most heavily used (and by inference, most important to the birds). Between 2009 and 2013 JNCC coordinated a programme of visual tracking work 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 (Wilson *et al.* 2014).

For common tern, the total number of tracks obtained across the UK sites was 381 (from seven SPAs, one non-SPA) across three years. In addition, visual tracking data were obtained for two SPAs: Ynys Feurig, Cemlyn Bay and The Skerries SPA (two tracks, collected in 2009) and North Norfolk Coast SPA (24 common tern tracks collected 2006-2008). These data were used to produce generic common tern foraging distribution models which allow production of foraging distribution maps around any common tern colony of interest, around the UK. These maps show common tern usage of the sea (relative density).

Maximum curvature was then applied to the mapped distributions to delimit the most important areas. Maximum curvature is a method, based on the law of diminishing returns, which objectively defines a threshold level of usage below which increasingly large areas would be required in order to provide further protection.

Breeding common terns are qualifying features of Mersey Narrows & North Wirral Foreshore SPA. Generic models of common tern foraging behaviour, along with maximum curvature were used to generate boundaries containing the most important foraging areas around the SPA colony. The predictor variables used in this model were: i) distance to colony, ii) distance to shore, and iii) bathymetry. These variables predicted highest usage around the colony, generally decreasing with increasing distance from it. This means that for the common terns nesting within the Mersey Narrows and North Wirral Foreshore SPA, the predicted foraging area extends north approximately to Formby, west along most of the Wirral foreshore, and into the mouth of the Mersey Estuary approximately to Rock Ferry.

The model-generated predictions of usage by common terns, together with the boundary drawn around all of the areas in which predicted usage exceeded the threshold identified by maximum curvature are shown in Figure 6. It can be seen in every case that very substantial areas of sea within that wider area which are distant to the colony and/or distant from the shore are predicted to have very little or no usage by foraging terns.

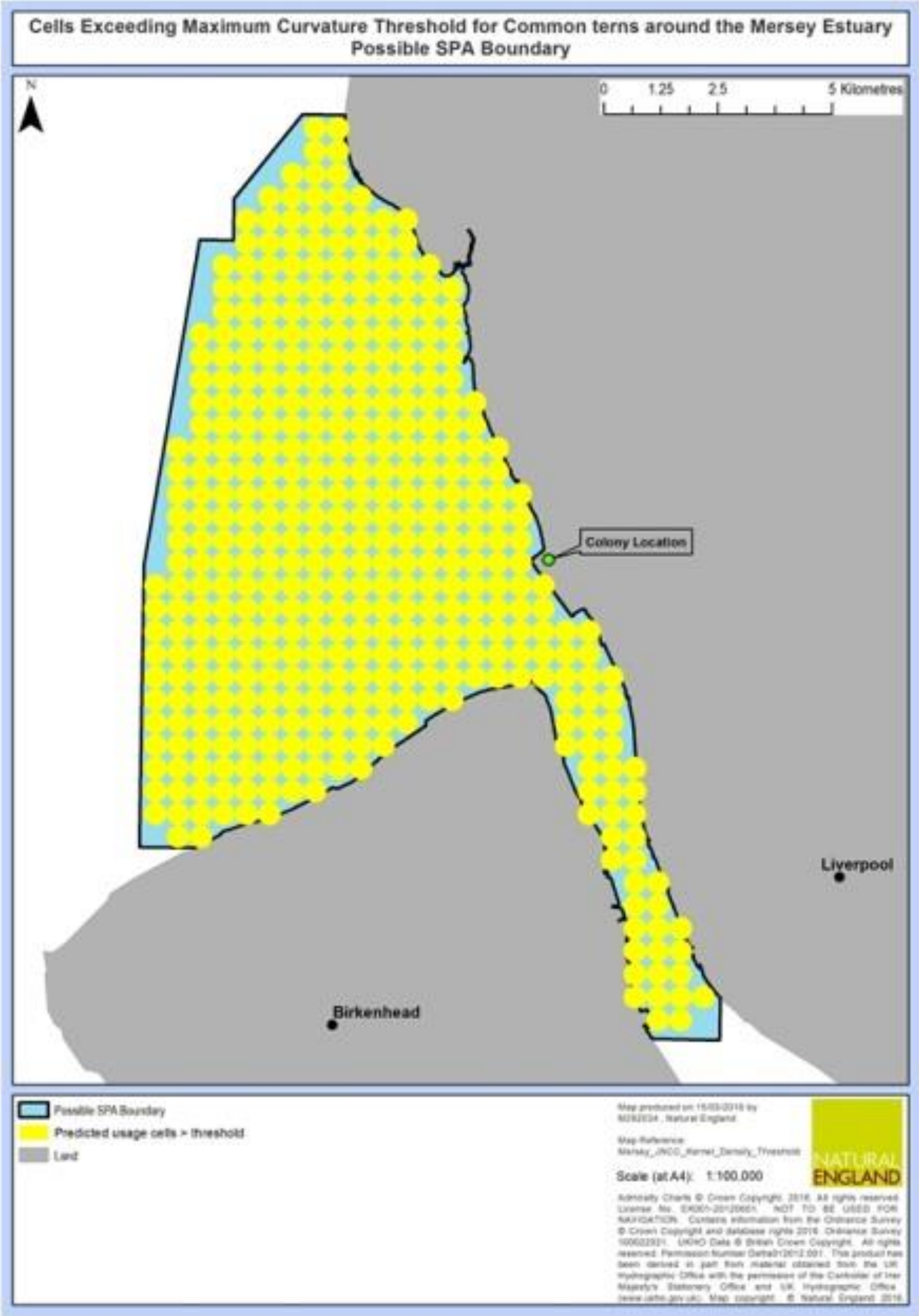

Figure 6 - Model predictions of common tern usage overlaid with maximum curvature derived limits to areas of most importance around Mersey Narrows & North Wirral Foreshore SPA.

The predicted usage boundaries largely sit within the existing boundaries of the Liverpool Bay / Bae Lerpwl SPA (or Ribble and Alt Estuaries SPA, where common tern is already a feature), and thus do not influence it greatly. The exceptions are along the north Wirral coast eastward to Hoylake and into the mouth of the Mersey Estuary where the boundary is expanded.

Further general information on the programme of national survey work to inform the boundary is presented in Annex 5.

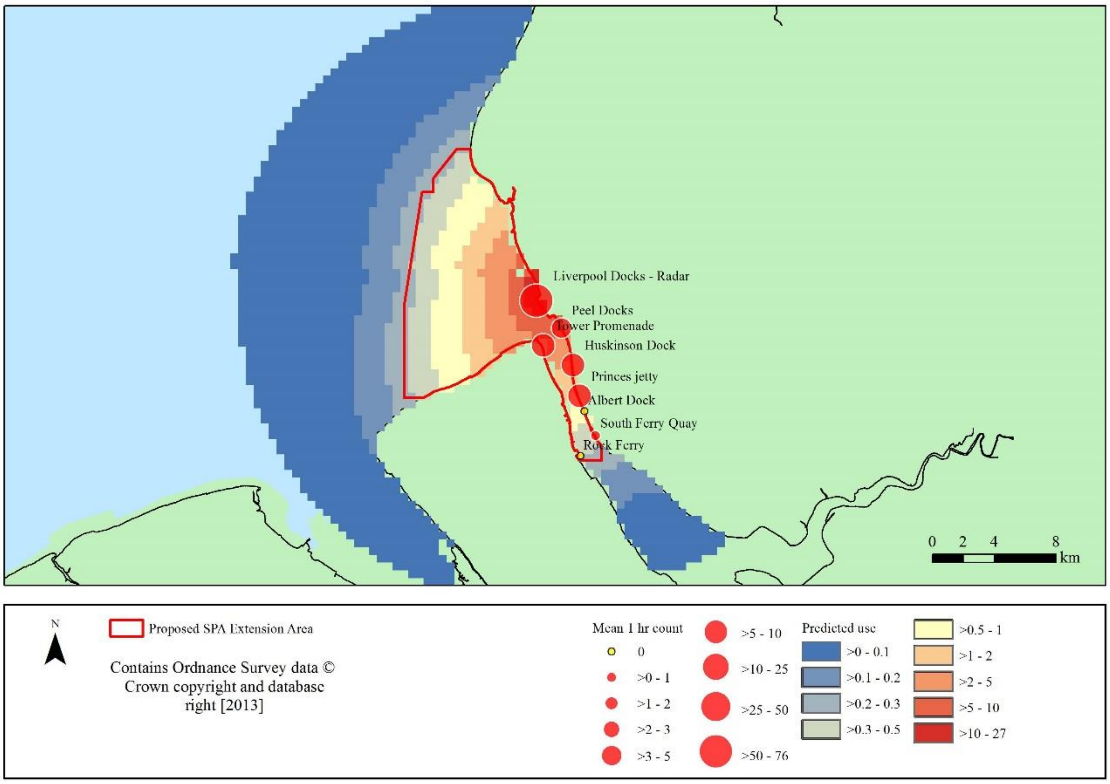

Figure 7 - Model predictions of common tern usage overlaid with maximum curvature derived limits to areas of most importance around Mersey Narrows & North Wirral Foreshore SPA, with verification data (Perrow, Harwood & Caldow 2015).

In order to assess the reliability of distributions mapped from a generic model based on data from colonies elsewhere across the UK, a statistical assessment of model performance was undertaken. Annex 5 describes this process in more detail, but in brief: The cross-validation process involves excluding the data from each colony, in turn, from the model building process. The model is then used to make predictions to that 'left out' colony, and those predictions are compared with the data that has been collected from that colony to assess how similar they are (in other words, to assess how well the model performs at predicting actual data). This demonstrated that the generic common tern model was judged to be "good". This analysis indicated that there is good consistency between colonies around the UK in the characteristics of sea areas which hold the highest relative densities of foraging common terns. Accordingly, there is a correspondingly high degree of confidence that the boundary of this pSPA, being partly dependent upon the predicted usage patterns of common terns, is founded on a reliable evidence base.

This confidence is confirmed by results of verification surveys undertaken within the areas predicted to be most important for foraging shown in Figure 6 (Perrow, Harwood & Caldow 2015). As predicted by the model, greatest usage of marine areas was seen closer to the colony, but common terns were recorded at count locations throughout the proposed extension into the Mersey Estuary (Figure 7) and as far as South Ferry Quay, which is adjacent to the proposed boundary's southernmost extent. The relatively low number of sightings further upriver is in accord with the model predictions of usage. However, it must be borne in mind that the absence of records of terns at Albert Dock and Rock Ferry is based on just three surveys in one season, and that in areas of comparatively lower usage (relative to Seaforth) greater survey effort within or across years would be likely to have revealed tern presence at these locations too. Furthermore, it was exceptionally rare to record common terns apparently resting (<1% or records), meaning that the surveys provide direct evidence of foraging behaviour as far upriver as the proposed boundary, offering firm support to the boundary based on model predictions of tern foraging distribution.

4. Location and Habitats

Liverpool Bay is located in the south-eastern region of the northern part of the Irish Sea, bordering northern England and north Wales, and running as a broad arc from Morecambe Bay to the east coast of Anglesey.

The seabed of Liverpool Bay consists of a wide range of mobile sediments. Sand is the predominate substrate with a concentrated area of gravelly sand off the Mersey Estuary. Sandbanks off the English coast of the Bay include East Hoyle Bank (largely within the Mersey Narrows and North Wirral Foreshore SPA) and parts of Great Burbo Bank (off the mouth of the Mersey). West Hoyle Bank (at the mouth of the Dee Estuary), Dutchman Bank and Chester and Rhyl Flats, are amongst the sand banks off the Welsh coast of the Bay.

The tidal currents throughout the Bay are generally weak and do not exceed 2m/sec. This combined with a relatively extended tidal range of 6 to 8m along the Lancashire coastline facilities the deposition of sediments, encouraging mud and sand belts to accumulate.

Water temperature fluctuates between summer and winter, with the coldest in February and March of 5-6°C. In August the surface water temperatures range between 14-16°C. The salinity level varies from fully saline in the western seaward areas of Liverpool Bay and decreases eastwards to 33 – 31 parts per thousand with the increased fresh water river input.

The Bay holds various fish of commercial importance. Pelagic species such as herring *Clupea harengus* and sprat *Spratus sprattus* have nursery grounds in the Bay. Demersal species such as plaice *Pleuronectes platessa* and sole *Solea solea* use the Bay for spawning and as a nursery area. Herring and sprat are amongst the most frequently recorded prey species of red-throated divers (Cramp & Simmons, 1977).

A study in Liverpool Bay investigated how bivalve distributions may influence common scoter distributions (Kaiser, 2002, Kaiser *et al.* 2006). Benthic sampling undertaken to date has found three main bivalve species within the sampling areas. These were *Abra alba*, *Pharus legumen* and *Donax vittatus*. Species such as *Mactra stultorum* and *Fabulina fabula* were much more patchily distributed. It is clear that each species occurs in distinct patches of variable abundance, and where one species declines it is replaced by another species. Work in Carmarthen Bay (Woolmer, 2003) indicates that common scoter are quite broad in their selection of prey species, and will forage on species that are at sufficient density and at a suitable depth. This was also supported in the Liverpool Bay research by Kaiser (2001) and Kaiser *et al.* (2006).

5. Assessment of Ornithological Interest

5.1. Survey Information and Summary

In all cases, up-to-date data (for surveys at sea during winter, between 2004/05 and 2010/11; for breeding birds, between 2010 and 2015) have been used to inform the classification.

UK SPA site selection guidelines have been applied to the most up to date information for the site. Citation values for the original qualifiers of Liverpool Bay / Bae Lerpwl SPA (common scoters, red-throated divers, waterbird assemblage) have been updated to reflect new information (section 3.3, and Lawson *et al*. 2015).

Counts of breeding terns at the colonies within existing SPAs (which are those most likely to be the origin of birds within the marine foraging areas of the pSPA) are from the national Seabird Monitoring Programme (SMP).

Details of the work carried out to characterise the foraging areas used by breeding adult terns within Liverpool Bay / Bae Lerpwl pSPA are above in sections 2 and 3 and in Annexes 4 and 5.

5.2. Annex I species

5.2.1. Breeding Season

5.2.1.1. Little tern *Sternula albifrons*

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

Little terns breed at Gronant Beach, within The Dee Estuary SPA. The five year mean for this site, derived from SMP data, is 130 pairs (2010 – 2014: Table 3). This represents 6.84% of the GB total of 1,900 pairs. The pSPA will thus offer protection of foraging areas to a significant proportion of little terns breeding in Great Britain.

Table 3. Little tern abundance at Gronant Beach (Apparently Occupied Nests, equivalent to pairs).

Year	Abundance
2010	120
2011	126
2012	140
2013	129
2014	136
Mean	**130.2**

5.2.1.2. Common tern *Sterna hirundo*

The breeding population of common terns in Great Britain is estimated to be 10,000 pairs (Musgrove *et al.* 2013), representing at least 15% of the Southern & Western European breeding population (67,000 pairs derived by division by 3 of the upper estimate of 200,000 individuals and rounded to the 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, a few localities along the south coast, and in the north-west (Mitchell *et al*. 2004). Common terns breed not only around coasts but, unlike the other tern species that breed in the UK, also breed frequently beside inland freshwater bodies.

Common terns breed within the Mersey Narrows & North Wirral Foreshore SPA. The five year mean for this site, derived from SMP data, is 180 pairs (2011 – 2015: Table 4). This represents 1.80% of the GB total of 10,000 pairs. The pSPA will thus offer protection of foraging areas to a significant proportion of common terns breeding in Great Britain.

Table 4. Common tern abundance at Seaforth (Apparently Occupied Nests, equivalent to pairs).

Year	Abundance
2011	202
2012	200
2013	165
2014	156
2015	177
Mean	**180**

Common terns breeding at The Dee Estuary SPA, adjacent to the pSPA, are not predicted to forage within Liverpool Bay, as their nesting location is within the estuary and not on the open coast (Wilson *et al*. 2014). These breeding terns do not contribute to the Liverpool Bay / Bae Lerpwl pSPA total. Similarly, common terns roosting at the Mersey Narrows & North Wirral Foreshore SPA in the non-breeding season (i.e. on migratory passage) do not contribute to the Liverpool Bay / Bae Lerpwl pSPA total, as the models used to determine site boundaries relate only to foraging birds in the breeding season. For the same reason, Sandwich terns roosting at The Dee Estuary SPA on migratory passage are not recommended features of Liverpool Bay / Bae Lerpwl pSPA.

5.2.2. Non-breeding season

5.2.2.1. Little gull *Hydrocoloeus minutus*

The breeding population of little gulls in Europe is estimated to be between 22,700 and 45,200 pairs (BirdLife International 2015), with the majority (49%) in Russia. Declines in the core breeding range have led to a European Red List assessment of Near Threatened, although there is a suggestion that long-term expansion in the western breeding range (Sweden, Finland) has led to more non-breeding birds appearing around the UK (Balmer *et al*. 2013). BirdLife International (2015) estimate the non-breeding European population to be 4,500 – 10,900 individuals, although this is somewhat incomplete as data are absent for some countries within the non-breeding range, and do not always reflect estimates of birds at sea.

Little gulls mainly spend the non-breeding season in the Mediterranean or North Africa, but some are considered to remain in the Irish Sea (Wernham *et al*. 2002). This is supported by estimates underpinning the proposal for addition to the Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015) which estimated between 172 and 374 birds on three of the five surveys in the 'core' winter months (arbitrarily defined as December – January). From April, little gulls begin the return passage migration to their breeding grounds and numbers peak at roost sites (Seaforth) within the Mersey Narrows & North Wirral Foreshore SPA (Wernham *et al*. 2002; Brown & Grice 2005.

From the available data for Liverpool Bay / Bae Lerpwl pSPA, Lawson *et al*. (2015) demonstrated a mean of peak of 319 individuals (2004/05 – 2010/11; Table 6), in clearly defined hotspots (section 3.3). Surveys from 2006/07 and 2007/08 did not inform the estimate of little gull abundance because of incomplete spatial coverage, or because of unreliable population estimates. The mean of peak thus uses data from 2004/05, 2005/06 and 2010/11. Although there is no national estimate of little gull abundance, the value of 319 comfortably exceeds the 'minimum 50' guideline nominally used to assess SPA qualification (Stroud *et al*. 2001). Furthermore, JNCC's national programme of data analysis has established that Liverpool Bay / Bae Lerpwl pSPA holds more little gulls than anywhere else in the UK, except for the Greater Wash pSPA.

The site provides protection for between 2.93% and 7.09% of the estimated European non-breeding population (though see earlier caveats). It also represents the only proposed SPA for the species on the west of Britain, and as the UK itself forms the likely north-west edge of the species non-breeding range provides an important link in the species' range requirements.

Table 6. Little gull estimated non-breeding abundance within Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015). Grey text refers to estimates not included in analyses because of insufficient spatial coverage of the survey area (Jan-07, Oct-07, Feb-08) or insufficient confidence in the population estimate (Feb-Mar-07).

Survey date	Season	Sum within pSPA boundary	Peak
Oct-Nov 04	2004/05	270	
Nov-Dec 04	2004/05	**354**	**354**
Jan-05	2004/05	165	
Feb-Mar 05	2004/05	37	

Oct-Nov 05	2005/06	5	
Nov-Dec 05	2005/06	0	
Jan-Feb 06	2005/06	259	
Feb-Mar 06	2005/06	**555**	**555**
Jan-07	2006/07	0	
Feb-Mar 07	2006/07	75	
Oct-07	2007/08	47	
Feb-08	2007/08	0	
Feb-Mar 11	2010/11	**48**	**48**
Mean of peaks			**319**

5.2.2.2. Red-throated diver *Gavia stellata*

Red-throated divers are a feature of the existing Liverpool Bay / Bae Lerpwl SPA, and remain a feature for the pSPA. No changes are made to the boundary based on red-throated diver distribution, and the only change is an update to the abundance value (which includes estimates of birds present within the entire pSPA boundary, including the expanded area for little gulls. Excluding these areas would not be practical, as effectively management advice and monitoring would need to consider a boundary within a boundary).

Data are now available from 2004/05 – 2010/11 and Lawson *et al*. (2015) demonstrated a mean of peak of 1,171 individuals (Table 7).

The site provides protection for 6.89% of the Great Britain non-breeding population (O'Brien *et al*. 2008), the fourth greatest abundance of red-throated divers in the UK.

Table 7. Red-throated diver estimated non-breeding abundance within Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015).

Survey date	Season	Sum within pSPA boundary	Peak
Oct-Nov 04	2004/05	479	
Nov-Dec 04	2004/05	558	
Jan-05	2004/05	413	
Feb-Mar 05	2004/05	**939**	**939**
Oct-Nov 05	2005/06	288	
Nov-Dec 05	2005/06	**1,133**	**1,133**
Jan-Feb 06	2005/06	879	
Feb-Mar 06	2005/06	1,072	
Jan-07	2006/07	101	
Feb-Mar 07	2006/07	**608**	**608**
Oct-07	2007/08*		
Feb-08	2007/08	**196**	**196**
Feb-Mar 11	2010/11	**2,980**	**2,980**
Mean of peaks			**1,171**

* Survey effort was low therefore estimates produced were not considered reliable (Lawson *et al*. 2011)

5.3. Regularly occurring migratory species

5.3.1. Non-Breeding season

5.3.1.1. Common scoter *Melanitta nigra*

Common scoters are a feature of the existing Liverpool Bay / Bae Lerpwl SPA, and remain a feature for the pSPA. No changes are made to the boundary based on common scoter distribution, and the only change is an update to the abundance value (which includes estimates of birds present within the entire pSPA boundary, including the expanded area for little gulls).

Data are now available from 2004/05 – 2010/11 and Lawson *et al*. (2015) demonstrated a mean of peak of 56,679 individuals (Table 8).

Since the 2010 Liverpool Bay / Bae Lerpwl SPA classification, the biogeographic population of non-breeding common scoters has been reassessed at 550,000 individuals (Wetlands International 2015). The site now provides protection for 10.31% of the biogeographic non-breeding population (previously 3.84%) and represents comfortably the greatest abundance of common scoters in the UK.

Table 8. Common scoter estimated non-breeding abundance within Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015).

Survey date	Season	Sum within pSPA boundary	Peak
Oct-Nov 04	2004/05	45,201	
Nov-Dec 04	2004/05	56,467	
Jan-05	2004/05	47,444	
Feb-Mar 05	2004/05	**64,020**	**64,020**
Oct-Nov 05	2005/06	19,064	
Nov-Dec 05	2005/06	14,951	
Jan-Feb 06	2005/06	**72,200**	**72,200**
Feb-Mar 06	2005/06	19,066	
Jan-07	2006/07	9,801	
Feb-Mar 07	2006/07	**81,578**	**81,578**
Oct-07	2007/08*		
Feb-08	2007/08	**23,247**	**23,247**
Feb-Mar 11	2010/11	**42,349**	**42,349**
Mean of peaks			**56,679**

* Survey effort was low therefore estimates produced were not considered reliable (Lawson *et al*. 2011)

5.4. Potential qualifying features not recommended for inclusion

Consideration was given to other species recorded on aerial surveys (Lawson *et al*. 2015), including great cormorant *Phalacrocorax carbo* and red-breasted merganser *Mergus serrator*. From these surveys, it was not possible to demonstrate that the two species met the relevant thresholds for qualification as independent qualifying features of the pSPA. However, both species (as well as the non-breeding qualifying species listed above) qualify as main components of the waterbird assemblage (see section 5.5).

5.5. Assemblages

5.5.1. Waterbird assemblage

Under Stage 1.3 of the UK SPA selection guidelines (JNCC 1999), sites may be selected as SPAs on the basis of supporting regular aggregations of 20,000 waterbirds or more. The original citation for Liverpool Bay / Bae Lerpwl SPA included a waterbird assemblage comprising red-throated divers and common scoters.

The re-calculated assemblage still qualifies under Stage 1.3 using the most up to date data. In the period 2004/05 – 2010/11 a five year mean of peak of 69,687 individual waterbirds was estimated (Lawson *et al*. 2015; Table 9).

Main components of the assemblage (i.e. species exceeding 1% of the GB total or 2,000 individuals) include all of the non-breeding qualifying features (common scoters, red-throated divers and little gulls), as well as red-breasted merganser and great cormorant (Table 10). The former has a five year mean of peak of 131, 1.56% of the GB population of 8,400 individuals in the non-breeding season (Musgrove *et al*. 2013); the latter has a five year mean of peak of 732, 2.09% of the GB population of 35,000 in the non-breeding season (Musgrove *et al*. 2013).

Other species recorded (Lawson *et al.* 2015) and contributing to the assemblage total in numbers less than 1% of their respective GB populations or less than 2,000 individuals include: black-headed gull, common gull, common eider, fulmar, great black-backed gull, great crested grebe, guillemot, gannet, herring gull, kittiwake, lesser black-backed gull, great northern diver, puffin, razorbill, shag, velvet scoter.

Table 9. Waterbird assemblage estimated abundance within Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015).

Survey date	Season	Sum within pSPA boundary	Peak
Oct-Nov 04	2004/05	**78,286**	
Nov-Dec 04	2004/05	70,258	
Jan-05	2004/05	54,598	
Feb-Mar 05	2004/05	67,901	**78,286**
Oct-Nov 05	2005/06	30,833	
Nov-Dec 05	2005/06	29,498	
Jan-Feb 06	2005/06	**90,486**	
Feb-Mar 06	2005/06	28,179	**90,486**
Jan-07	2006/07	12,510	
Feb-Mar 07	2006/07	**87,227**	**87,227**
Oct-07	2007/08	926	
Feb-08	2007/08	**26,849**	**26,849**
Feb-Mar 11	2010/11	**65,587**	**65,587**
Mean of peaks			**69,687**

Table 10. Great cormorant and red-breasted merganser estimated abundance within Liverpool Bay / Bae Lerpwl pSPA (Lawson *et al*. 2015).

		Great cormorant[6]		Red-breasted merganser	
Survey date	Season	Sum within pSPA boundary	Peak	Sum within pSPA boundary	Peak
Oct Nov 04	2004/05	505		123	

[6] The mean value of 732 cormorants includes peak estimates ranging from 112 to 1,425. These values are likely underestimates as the ecology of cormorants suggests a relatively small proportion of time is spent at the sea surface; at other times they may be diving for fish or drying out on land or on structures above water (Gremillet *et al*. (1998); Sellers (1995)).

Nov Dec 04	2004/05	341		54	
Jan-05	2004/05	**680**	**680**	56	
Feb Mar 05	2004/05	320		**360**	**360**
Oct Nov 05	2005/06	551		50	
Nov Dec 05	2005/06	**1,425**	**1,425**	85	
Jan Feb 06	2005/06	943		86	
Feb Mar 06	2005/06	674		**128**	**128**
Jan-07	2006/07	75		32	
Feb Mar 07	2006/07	**290**	**290**	**125**	**125**
Oct-07	2007/08				
Feb-08	2007/08	**112**	**112**	**25**	**25**
Feb Mar 11	2007/08	**1151**	**1,151**	**16**	**16**
Mean of peaks			**732**		**131**

6. Comparison with other sites in the UK

Breeding season

To give an indication of the relative importance of Liverpool Bay / Bae Lerpwl pSPA in a national context a comparison of the numbers of terns within the site was made with other important sites for the specific species.

As you can see from the table below Liverpool Bay / Bae Lerpwl pSPA is the 24th most important site for common tern (Table 11) and the 3rd most important for little tern.

It is important to note that although these comparisons are made with the best available evidence it is comparing contemporary data from Liverpool Bay / Bae Lerpwl pSPA with a list produced for the 2001 SPA review where the data originates from pre 1991 surveys.

Table 11. Comparison of the average numbers of individuals (and pairs) of common terns in the Liverpool Bay / Bae Lerpwl pSPA (2010 / 2011 – 2014 / 2015) with those at other SPAs identified (Stroud *et al*. 2001, plus Mersey Narrows & North Wirral Foreshore SPA) as supporting those features.

Species	**Site**	**Individuals (pairs)**[7]	**Rank**[8,9]	**Comments**
Common tern *Sterna hirundo*	Ribble & Alt Estuaries SPA	364 (182)	18th of 24	
	Liverpool Bay / Bae Lerpwl pSPA	360 (180)	=19th of 24	Mersey Narrows & NW Foreshore SPA is the same rank by default
	Larne Lough SPA	360 (180)	=19th of 24	

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

[8] Note that these rankings should only be considered indicative of the relative importance of the pSPA as they are based on comparison of the sum of the most recent 5 year mean populations of each species at the source SPAs with the historical populations of each species at each SPA in the UK as listed in Stroud *et al.* (2001). The number of sites ranked is based on the number of sites listed for each species in Stroud *et al*. (2001) and included from that list are SPAs contributing to the total presented for the Liverpool Bay pSPA, and adding one site to account for the pSPA itself.

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

Non-breeding season

Liverpool Bay / Bae Lerpwl SPA, when classified in 2010, supported a mean of peak of 54,675 common scoters, and is now assessed to support a mean of peak of 56,679. Liverpool Bay / Bae Lerpwl pSPA remains comfortably the largest aggregation of common scoters in the UK; the next largest aggregation in Bae Caerfyrddin / Carmarthen Bay SPA supported a mean of peak of 16,946 scoters at classification in 2003.

Red-throated diver abundance was estimated at 922 individuals when the original Liverpool Bay / Bae Lerpwl SPA was classified, and now is assessed to support a mean of peak of 1,171. Although the Outer Thames Estuary SPA supports 38% of the GB population (five year mean of peak of 6,466 birds), Liverpool Bay / Bae Lerpwl pSPA supports 6.98% of the GB total. The only other SPA currently existing for the species in the UK is the Firth of Forth SPA, supporting 87 individuals. However, Northern Cardigan Bay / Gogledd Bae Ceredigion pSPA counted 1186 red-throated diver, accounting for approximately 7% of the GB total.

Liverpool Bay / Bae Lerpwl pSPA is therefore the second highest ranked site in the UK by a considerable margin.

Currently there are no SPAs in the UK for little gulls, and so it is difficult to compare the importance of sites. Proposals for the Greater Wash pSPA also include little gull, estimated to be present in greater numbers than in Liverpool Bay / Bae Lerpwl pSPA. However, Liverpool Bay / Bae Lerpwl pSPA would still represent the second largest aggregation of little gulls in the UK.

7. Conclusion

The evidence presented in this Departmental Brief sets out the scientific case for SPA classification, based on at site survey data for little tern, peer-reviewed models of common tern foraging requirements and non-breeding distributional data for common scoters, red-throated divers and little gulls. The proposed boundary expands to the north and west in comparison to the original Liverpool Bay / Bae Lerpwl SPA, though is still largely determined by aggregations of common scoters and red-throated divers. The extension also adds smaller marine areas identified for foraging terns breeding in adjacent existing SPAs.

The pSPA is internationally important for five species. It will remain the most abundant site in the UK for common scoters, and the second most important for little gulls and the fourth most important for red-throated diver. It will provide important foraging habitat for one of the largest little tern colonies in the UK, and will support internationally important numbers of foraging common terns from the Mersey Narrows & North Wirral Foreshore SPA colony.

In conclusion, the site qualifies as per the original Liverpool Bay / Bae Lerpwl SPA, with the addition of little gull, little tern and common tern features to protect both marine non-breeding habitat and the marine foraging areas used by birds breeding along the adjacent coastline.

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Annex 1 Boundary Maps

Figure 1 – Departmental Brief map showing proposed marine extension area and existing classified SPA.

Please note that this map has been removed from this document and can be viewed separately enabling easier view of the scale of the proposed extension to the site. The consultation map is available on the consultations gov.uk webpage available here:

https://www.gov.uk/government/consultations/liverpool-bay-bae-lerpwl-special-protection-area-extension-comment-on-proposals

Annex 2 Site Citation

EC Directive 79/409 on the Conservation of Wild Birds

potential Special Protection Area (pSPA)

Name: *Liverpool Bay / Bae Lerpwl potential Special Protection Area*

Counties/Unitary Authorities:

The SPA lies almost entirely in UK territorial waters adjacent to the following counties / unitary authorities: Lancashire, Blackpool, Merseyside, Sir y Fflint/Flintshire, Conwy, Gwynedd, and Ynys Môn/Isle of Anglesey and a small portion sits within Sir Ddinbych/Denbighshire unitary authority.

Boundary of the pSPA:

The pSPA extends out to, and beyond 12 nautical miles at the northwest point of the existing boundary to Liverpool Bay/Bae Lerpwl SPA and also into Welsh waters offshore of the mouth of the Dee Estuary.

The landward boundary of the pSPA generally follows mean low water mark or the boundaries of existing SPAs, whichever is the furthest seaward. The extension at Prestatyn and Mersey for foraging terns follows mean high water or the boundaries of existing SPAs.

The new pSPA supersedes the original Liverpool Bay / Bae Lerpwl SPA.

Size of SPA: The pSPA covers an area of 252,773 ha.

Site description:

Liverpool Bay is located in the south-eastern region of the northern part of the Irish Sea, bordering north-west England and north Wales. The pSPA is a broad arc from approximately Morecambe Bay to the east coast of Anglesey. The sea bed of the SPA consists of a wide range of mobile sediments. Large areas of muddy sand stretch from Rossall Point to the Ribble Estuary, and sand predominates in the remaining areas, with a concentrated area of gravelly sand off the Mersey Estuary and a number of prominent sandbanks off the English and Welsh coasts. The tidal currents throughout the pSPA are generally weak, which combined with a relatively large tidal range facilitates the deposition of sediments.

Qualifying species:

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

Species	Season	Count (Period)	% of population
Red-throated diver *Gavia stellata*	Non-breeding	1,171 individuals	6.89% of GB population

		(2004/05 – 2010/11)	
Little gull *Hydrocoloeus minutus*	Non-breeding	319 individuals (2004/05 – 2010/11)	N/A – selected under stage 1.4 guideline
Little tern *Sternula albifrons*	Breeding	138 individuals (1995-1999)*	2.9% of GB population
Common tern *Sterna hirundo*	Breeding	360 individuals (2011 – 2015)	1.80% of GB population

*This figure represents the ‘at classification’ population for little tern in The Dee Estuary SPA. Current figures are 260 individuals (2010-2014) representing 6.84% of the GB population.

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

Species	Season	Count (Period)	% of population
Common scoter *Melanitta nigra*	Non-breeding	56,679 (2004/05 – 2010/11)	10.31% of biogeographic population

Assemblage qualification:

The site qualifies under SPA selection stage 1.3 as it is used regularly by over 20,000 waterbirds (waterbirds as defined by the Ramsar Convention) in any season:

In the non-breeding season, the site regularly supports at least 69,687 (2004/05 – 2010/11) individual waterbirds.

The main components of the assemblage include all of the non-breeding qualifying features listed above, as well as an additional two species present in numbers exceeding 1% of the GB total: red-breasted merganser *Mergus serrator* and great cormorant *Phalacrocorax carbo*.

Principal bird data sources:

Lawson, J., Kober, K., Win, I., Allcock, Z., Black, J. Reid, J.B., Way, L. & O’Brien, S.H. 2015. An assessment of the numbers and distribution of wintering waterbirds and seabirds in Liverpool Bay / Bae Lerpwl area of search. JNCC Report No 576. JNCC, Peterborough.

Tern colony count data from the national Seabird Monitoring Programme database.

Annex 3 Sources of bird data

Source of Data	Data provider	Subject	Date produced	Method of data collection	Verification	Reference
JNCC larger tern survey report	JNCC	Empirical survey data on the foraging locations of breeding terns tracked from several UK colonies and the identification of important foraging areas around colonies using habitat association models	2009-2011	Visual tracking of individual terns from boat-based survey platform	Verification by JNCC and external peer review of final report	http://jncc.defra.gov.uk/pdf/JNCC_Report_500_web.pdf
JNCC little tern survey report	JNCC	Empirical survey data on the sightings of little terns along the shore and at sea at several UK colonies and definition of alongshore and seaward limits to important foraging areas around colonies	2009-2013	Shore-based counts from fixed vantage points and boat-based transects at sea	Verification by JNCC and external peer review of final report	http://jncc.defra.gov.uk/pdf/Report_548_web.pdf
Seabird Monitoring Programme	JNCC and site managers	Breeding seabird data for relevant colonies contributing to Liverpool Bay / Bae Lerpwl pSPA	2010-2015	Standard methodology	Published on JNCC website (or available as little tern data considered sensitive)	http://jncc.defra.gov.uk/smp/
JNCC report on little gulls, common scoters and red-throated divers	JNCC	Data on bird distribution and abundance from visual aerial surveys	2004/05 – 2010/11	Visual aerial surveys, Kernel Density Estimation, Maximum Curvature analysis	Published in peer-reviewed journal (O'Brien *et al*. 2012); report published on JNCC website	http://jncc.defra.gov.uk/page-7103

Annex 4 Defining little tern foraging areas and seaward boundary

1. Background and overview

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

This annex gives an overview of the survey and analytical work carried out by and on behalf of JNCC between 2009 and 2013 for the little tern. This work focussed on those colony SPAs which have been regularly occupied[10] 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 a 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

[10] '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.

distance and bearing of the sighting and information on behaviour; (ii) Continuous counts of any little terns observed between the instantaneous points were also recorded to provide an index of relative abundance. Although observers recorded behaviour (foraging/flying), restricting the analysis to just foraging observations would have limited the sample size. Therefore, all records (foraging and not foraging) were included in the analyses.

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

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

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

3. Data analysis

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

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

3.1. Site-specific options

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

A) Seaward extent

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

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

B) Alongshore extent

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

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

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 1. Values of the maximum seaward observation of little terns on each survey at each SPA surveyed. The number of values in the 2nd column indicates the number of boat-based surveys yielding independent estimates of maximum seaward extent of occurrence at each colony. The

values in the 3rd column are the site specific average of the values in the 2nd column. The value in the final row is the average of the site specific mean values.

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

1. Derived from birds breeding at the North Denes colony; 85% kernel contours.

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

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

SPA colony	Maximum alongshore extent from the colony in each direction (km)
Ythan Estuary, Sands of Forvie and Meikle Loch	2, 5.35
Dee Estuary	3, 3
Northumbria Coast	5, 6
Humber Estuary	6, 6
North Norfolk Coast	7, 7
Teesmouth & Cleveland Coast	5, 5
Gibraltar Point	2, N/A
Great Yarmouth North Denes	5, 4
Hamford Water	4, 3
Solent & Southampton water	1, N/A
Morecambe Bay	7, 2
Lindisfarne	3, 4

Chesil Beach and The Fleet	1, 0.5, 1
Generic (mean value) applied to sites with insufficient data	3.9

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

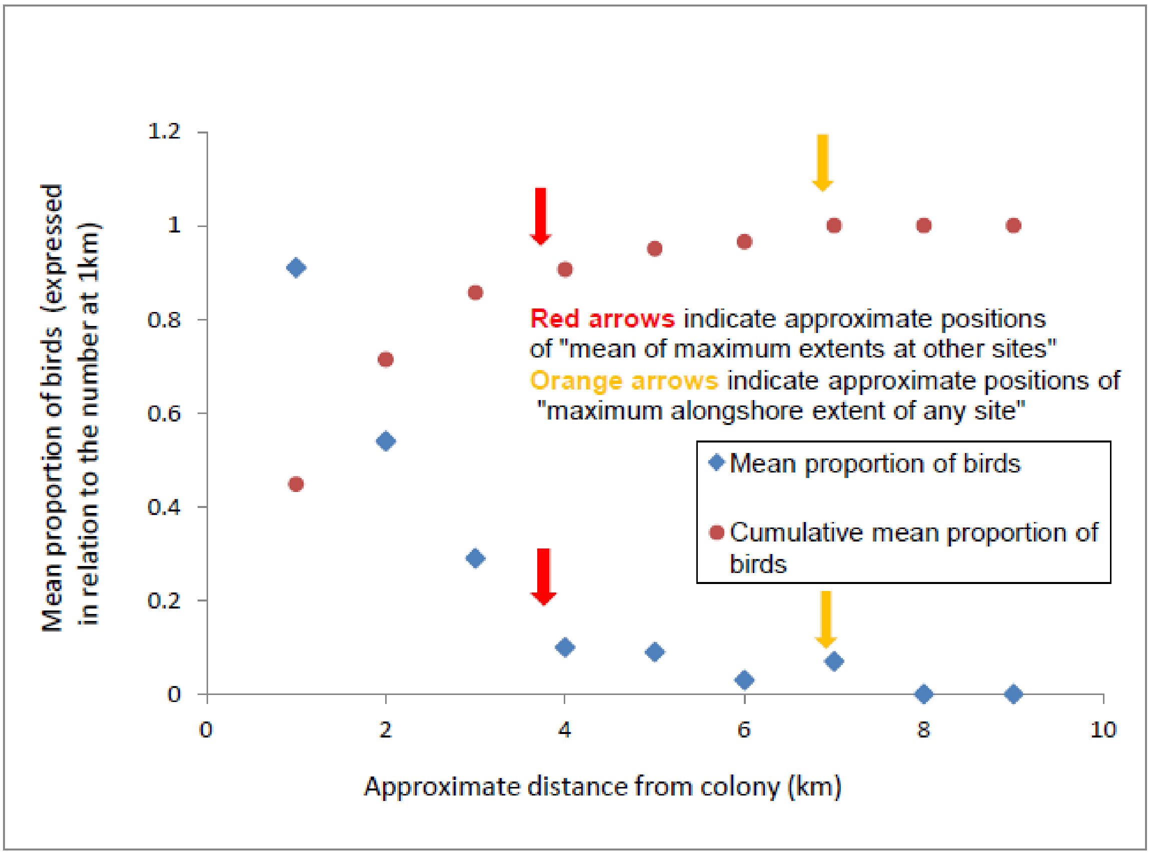

Figure 1: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing distances alongshore from the colony. Each blue point represents the mean proportional usage at each distance band from the colony averaged across colonies. The proportion at each distance (blue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

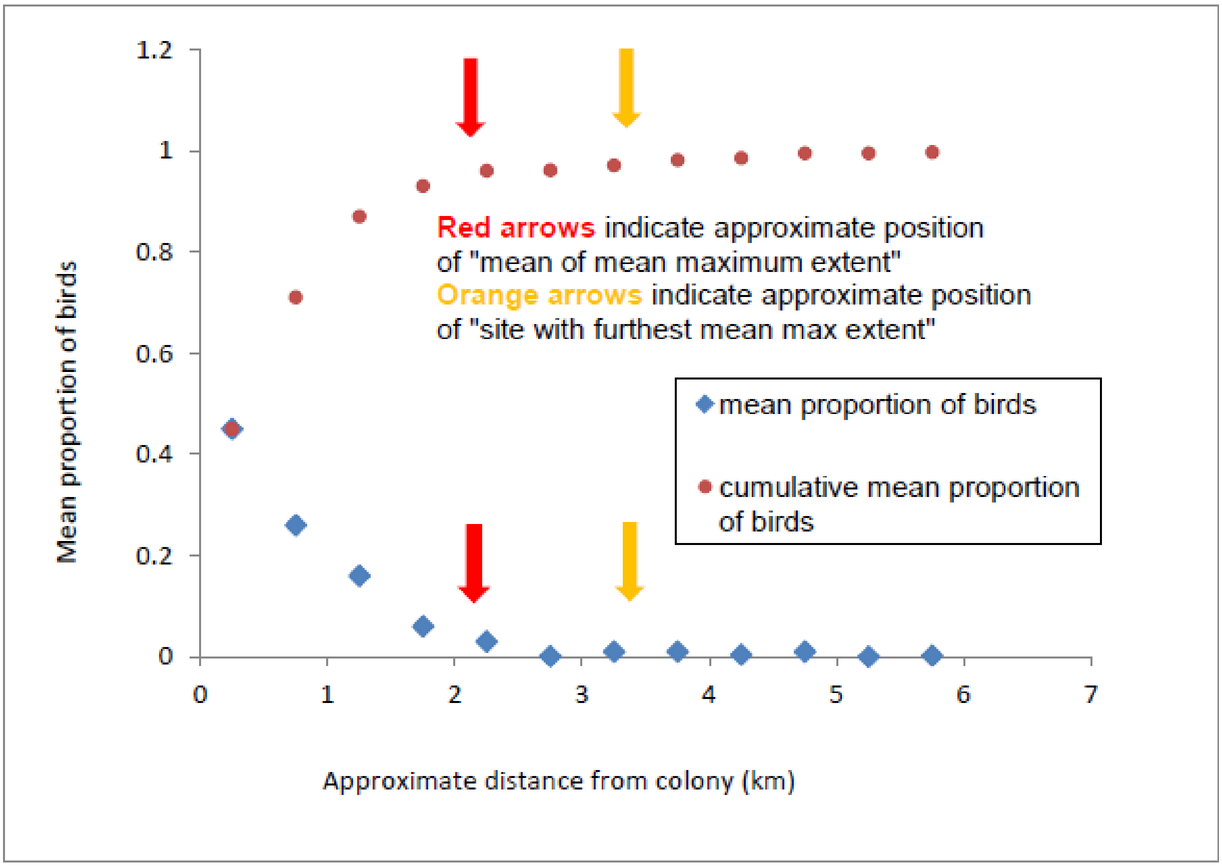

Figure 2: Mean proportion (blue dots) and cumulative mean proportion (red dots) of little terns at increasing seaward distances from mean high water mark. Each blue point represents the mean proportional usage at each distance band from mean high water mark averaged across colonies. The red arrows indicate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

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

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

4. Boundary delineation

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

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

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

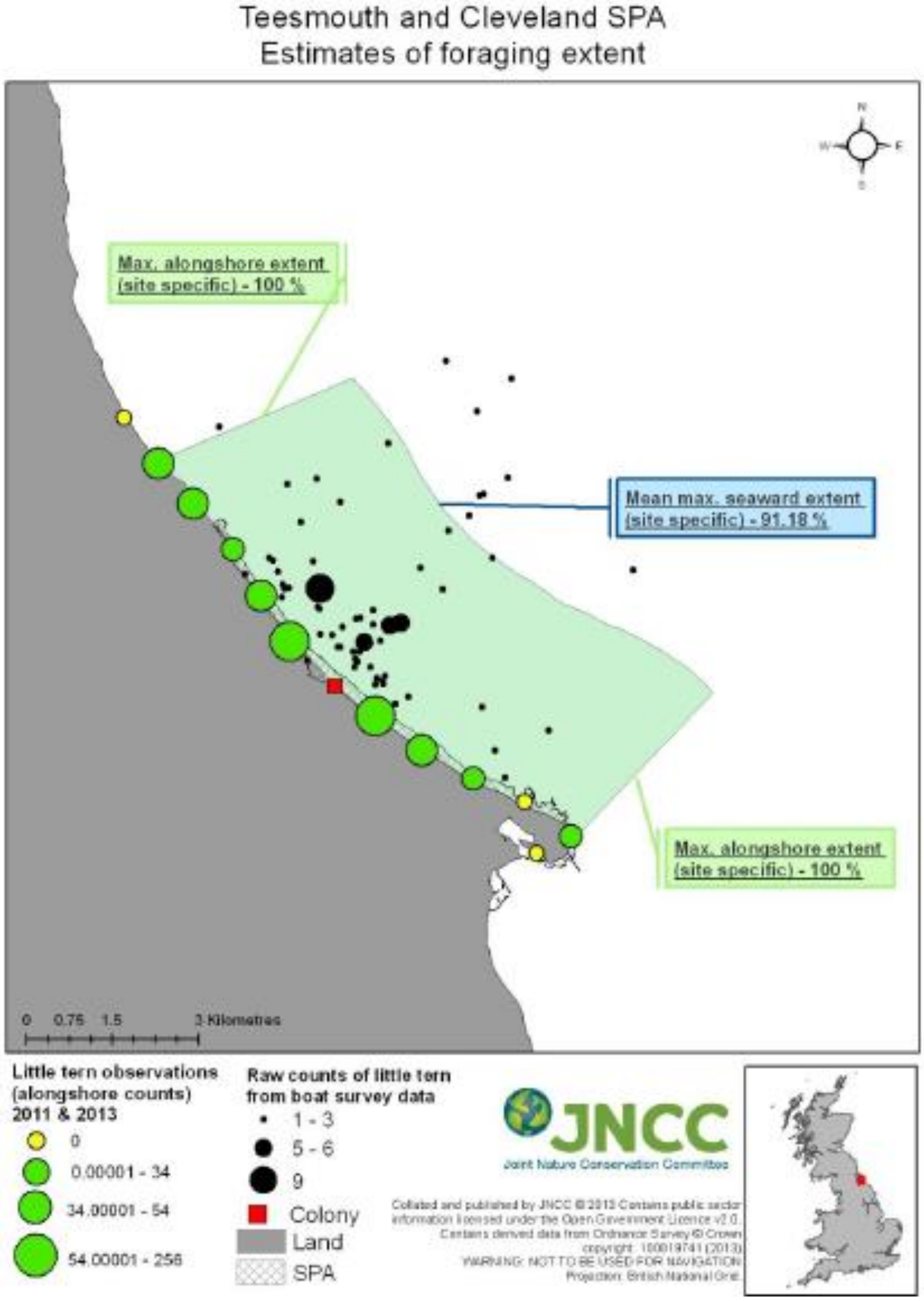

Figure 3. An example of the application of site specific alongshore and site specific seaward extents to define the boundaries to little tern foraging areas at the Teesmouth and Cleveland SPA. The % values given in the labels indicate the site specific % of little tern observations within the shore-based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

5. Conclusion

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

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

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

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

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

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

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

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

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Abstract available at:

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

Annex 5 Defining larger tern foraging areas and seaward boundary

1. Background and overview

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

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

This annex gives an overview of the survey and analytical work carried out by and on behalf of JNCC between 2009 and 2013 for the four larger tern species (*Sterna* species). A full and detailed description of the analysis can be found in the JNCC report on this work (http://jncc.defra.gov.uk/page-6644). A different approach was deemed appropriate for little terns as they search for food in a much more restricted area closer to the coast and to the breeding colony. An overview of that work is described in Annex 5 and a full and detailed description of that analysis can be found in the JNCC report on that work (http://jncc.defra.gov.uk/pdf/Report_548_web.pdf). For the modelling analysis aspect of the project, JNCC worked collaboratively with Biomathematics and Statistics Scotland (BioSS)[11].

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[12] (see Box 1 for details). The majority of the data were collected during the chick-rearing period (June to early July), a highly demanding period for breeding adult terns due to food gathering for chick feeding and rearing. The need to regularly return to the colony results in a higher number of foraging trips within a generally more restricted foraging range. Accordingly, areas used during this period are considered as crucial for overall survival and are thus high priority for site-based conservation.

[11] 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.
[12] PERROW, M. R., SKEATE, E. R. and GILROY, J. J. (2011). Visual tracking from a rigid-hulled inflatable boat to determine foraging movements of breeding terns. *Journal of Field Ornithology*, 82(1), 68-79.

BOX 1.

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

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

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

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 the inclusion in the modelling of locations passed over in transit would dilute the power of the analysis to identify important habitat relationships and therefore foraging areas. In addition, because terns are central place foragers (meaning they must travel to and from their nest site on each trip), it would almost certainly lead to a bias towards high usage of areas close to the colony, data from commuting periods (i.e. parts of the bird track where no foraging behaviour[13] 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.

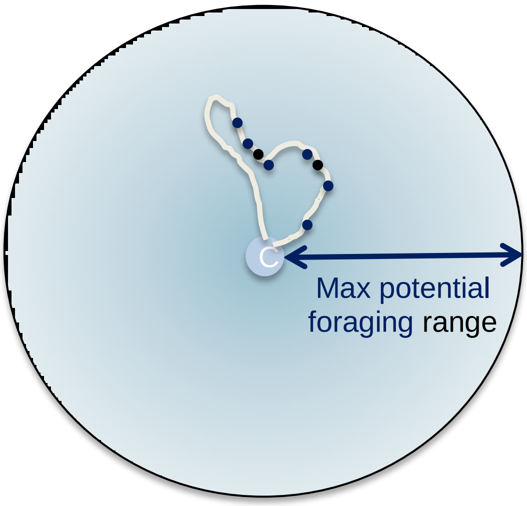
The environment that the terns use for foraging was characterised by analysis of the presence and matching absence data in relation to a suite of environmental covariates (see Box 3 for details). This analysis was then 'reversed' and the modelled relationships between tern usage and the environmental 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

[13] 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.

be used for foraging, either by other terns at the same colony, or by terns at other colonies (see Figure 1).

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[14], 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.

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

BOX 3.

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

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

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

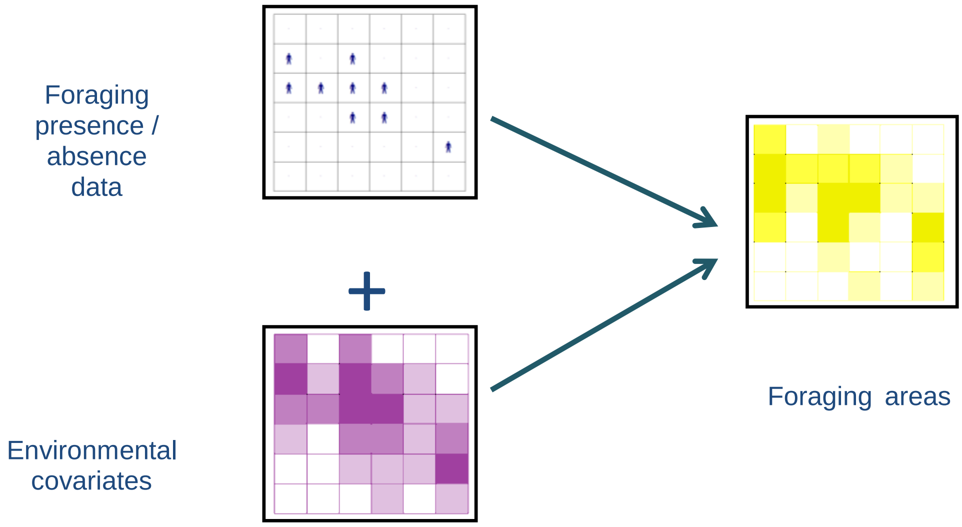


Figure 1. Simplified, schematic representation of the process of modelling distributions based on environmental information, using a single covariate distribution map in the example.

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

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

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

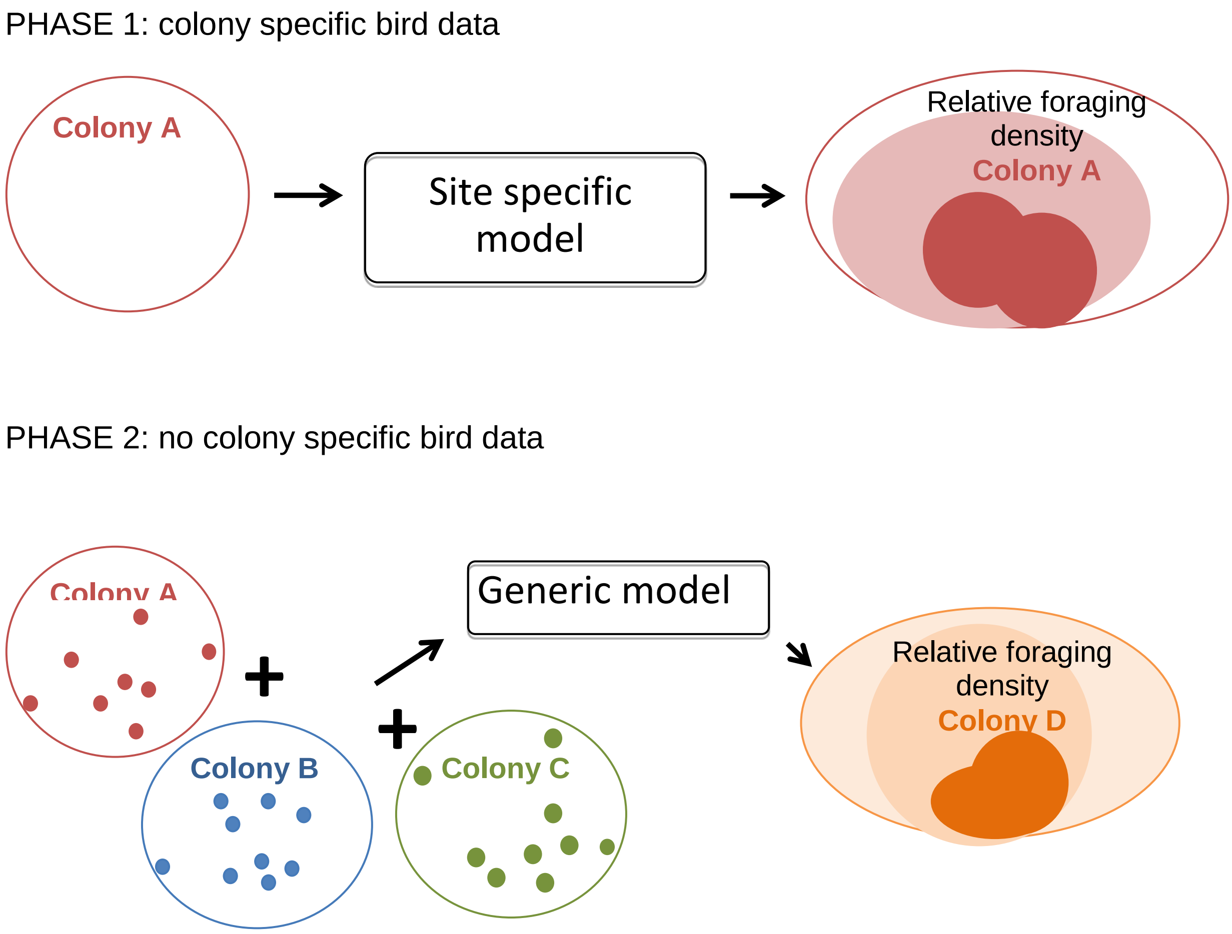


Figure 2. Simplified, schematic representation of the process whereby empirical observations of tern foraging locations around a colony were either: used to build predictive, site-specific models of tern usage that generated relative foraging density maps around that colony (phase 1 analyses); or combined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

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

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

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

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

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

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

Thus, although the sampling effort captured no more than 68.1% of the total area of use in any case, it should be noted that the total area of use is unlikely to be described fully by any reasonable amount of tracking effort; as this would require every movement of every individual in a colony to be constantly monitored. However, the surveys did provide sufficient data to account for a large proportion of the core foraging area, which is a key metric for investigating habitat association. This provides reassurance that, even when a relatively small proportion of the colony population is sampled, the data are likely to represent well the core foraging areas of the colony population as a whole.
Furthermore, it should be borne in mind that the objective of the tracking work was not to gather a comprehensive body of tracks from which to determine directly a potential boundary around important foraging locations. Rather, the goal was to gather a representative sample of tracks from which to construct a habitat association model to identify areas with the characteristics of important foraging locations i.e. to identify not just those locations where foraging was observed within the necessarily limited empirical dataset on which the models were based, but also to identify other locations (including at other colonies where it was not possible to sample) where relatively high levels of usage by foraging terns might be expected based on their characteristics. In other words,

the habitat models allow us to fill gaps in sampling effort, both at sampled colonies and at unsampled colonies.
With that in mind, for each model produced, an assessment was made of how good this model would be at making predictions of tern foraging around the same colony (for colony specific analysis) or around other colonies (for UK wide analysis). This assessment was made using a technique called cross-validation.

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

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

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

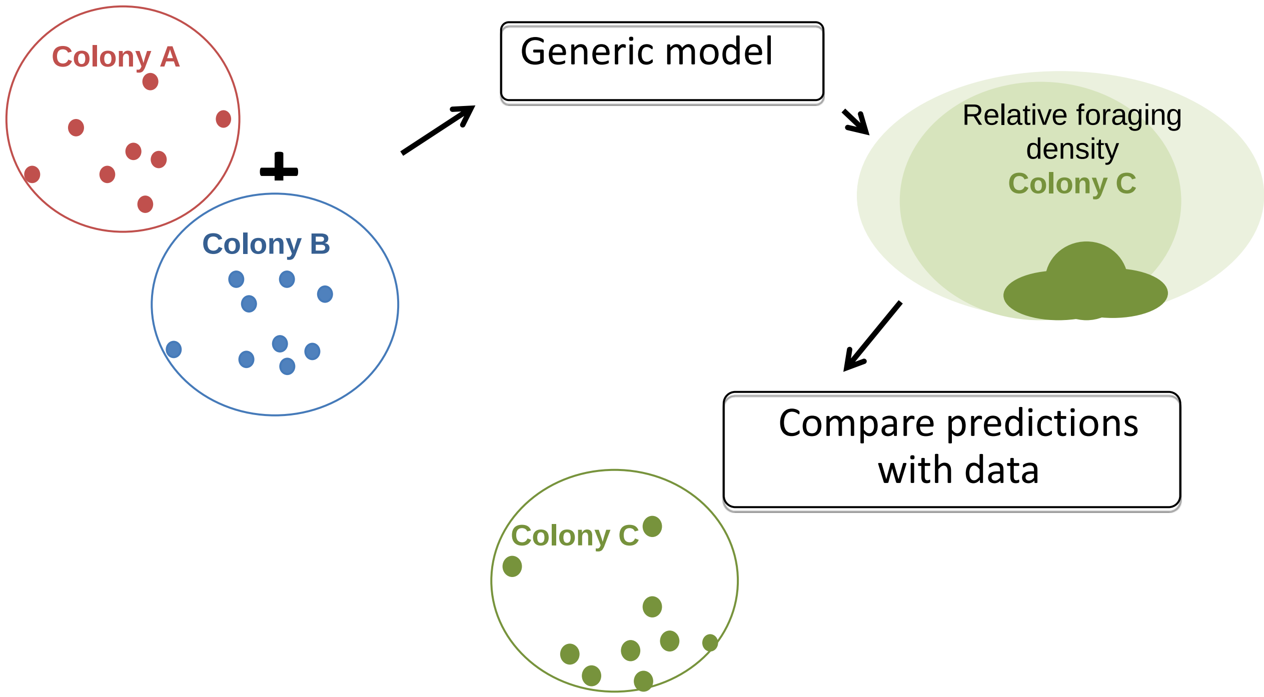


Figure 3. Schematic representation of the cross-validation process, using an example where we have data for three colonies A, B and C, of which data from two at a time (A and B in this diagram) are used to build a predictive model, the predictions of which are then tested by comparison with empirical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

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

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

Species	SPA Colony	Average AUC score
Arctic tern	Coquet Island	0.796
Common tern	Coquet Island	0.845
	Imperial Dock Lock	0.741
Sandwich tern	Coquet Island	0.915
	North Norfolk	0.884
	Ynys Feurig, Cemlyn Bay and The Skerries	0.939

	Ythan Estuary, Sands of Forvie and Meikle Loch	0.990

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

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

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

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

(a)

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

(b)

Common terns	**AUC score for each test colony**						
Model	North Norfolk	Coquet Island	Cemlyn	Larne Lough	Imperial Dock	Glas Eileanan	Average AUC

					Lock		
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 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[15], resulting in 30km for Arctic, 20km for common, 32km for Sandwich and 21 km for roseate tern. Any grid cells outside the mean maximum foraging ranges were excluded prior to maximum curvature analysis.

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

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

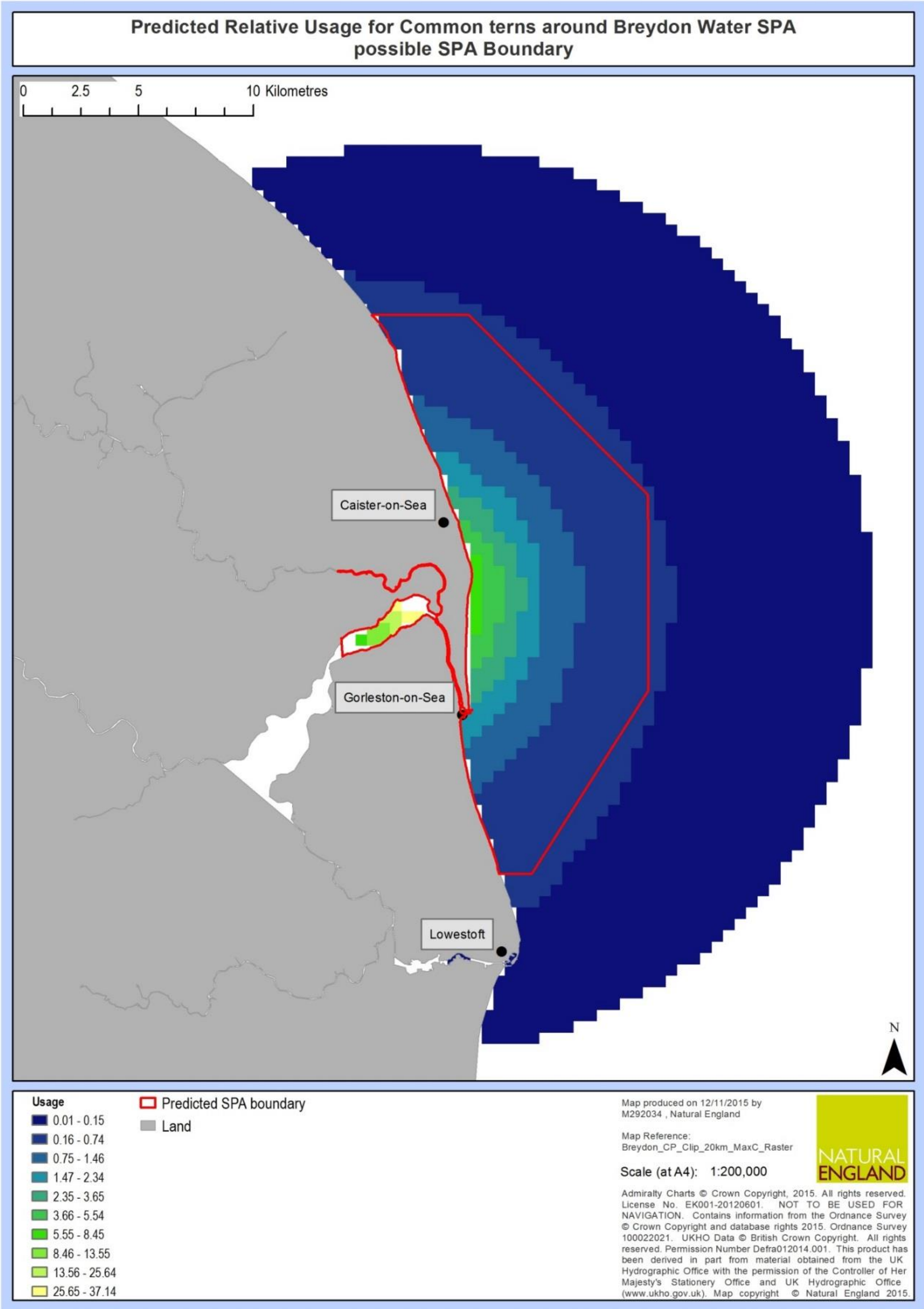


Figure 4. Maximum curvature derived boundary (red line) overlaid on map of model predictions of usage by common terns around Foulness SPA. The extent of the dark blue circle of model predictions of usage is 20 km - the global mean maximum distance to colony, calculated using tracking data held by JNCC; ECON Ecological Consultancy Ltd and Thaxter *et al.* 2012. These values were used to constrain the usage data used before Maximum curvature analysis was applied. Source: Win *et al.* (2015).

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

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

5. Conclusion

Delineation of the boundaries around areas of sea that are most heavily used by seabirds have, in several existing marine SPAs, been based on maps of the relative density of birds derived directly from empirical at sea surveys of bird distribution. However, such an approach was not followed in the current project for a number of reasons. First, with tern foraging being predominantly close to shore and with the need to consider colonies all around the United Kingdom, existing data sources 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 6 Implementation of Natural England Evidence Standards

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

Strategic Evidence Standard:

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

Analysis of Evidence Standard:

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

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

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

1. Quantification of qualifying feature population sizes

In order to determine the suite of species present within the pSPA which meet the SPA selection guidelines (JNCC 1999), most relevant bird count data were used, pertaining to the current five year period (2010-2014 or 2011-2015 for breeding terns) or most recently available run of data (2004/05-2010/11 for non-breeding common scoters, red-throated divers and common scoters: Lawson *et al*. 2015). Sources were:

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

2. Data on non-breeding bird abundance and distribution from visual aerial surveys, conducted in Liverpool Bay between 2004/05 and 2010/11 and published in a JNCC report (Lawson *et al*. 2015).

Both data types represent best available information, and are of the highest quality and relevance regarding SPA classification decisions.

Within this departmental brief current population figures for little terns have been used to demonstrate how the site meets selection guidelines and in the proposed citation for Liverpool Bay / Bae Lerpwl pSPA the 'at classification' population for little tern in The Dee Estuary SPA has been used to provide consistency for Wales in this cross border site.

The size of the breeding populations in Great Britain for both common tern and little tern are taken from Musgrove *et al*. (2013) but are based on data from 2000. Both populations may have changed since then, but these are the most recent data available for all colonies in Great Britain. There is some evidence that national common tern populations have declined significantly meaning that the percentage estimate is likely higher for the Seaforth colony (JNCC 2015).

2. Establishment of extent of marine pSPAs using tern tracking data

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

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

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

Table 1 Criteria for inshore SPA data adequacy.

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

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

Webb & Reid (2004) also discuss the issue of establishing sufficient evidence in the case of marine SPAs to establish regularity of use, which is a key element of the SPA selection guidelines. The tern tracking work was never intended to establish regularity of use of certain sea areas by particular species around particular colonies. The aim of that work was simply to capture sufficient representative information on tern foraging behaviour to allow reliable habitat association models to be constructed and used to generate maps of areas of principal usage. The results of the cross validation of those models' predictions, in which data from different years were used as test datasets, suggests a relatively high degree of consistency in usage patterns between years i.e. regularity of use of those most important areas (Wilson *et al.* 2015). However, no formal tests of the regularity of use of the sea areas within the pSPA boundary have been made. Regularity of use of the pSPA has been reasonably inferred from the continued existence of the site's named features in qualifying numbers in each of the existing coastal SPAs from which birds within the marine SPA are most likely to originate.

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

> *"The first stage of boundary determination involves* ***defining the extent of area required by the qualifying species concerned****. These scientific judgements are made in the light of the ecological requirements of the relevant species that may be delivered by that particular site, and the extent to which the site can fulfil these requirements. This follows a* ***rigorous assessment of the best-available local***

***information regarding distribution, abundance and movements of the qualifying species**. It may also involve the **commissioning of special surveys** where the information base is weak. Following this stage, every attempt is made to define a boundary that is identifiable on the ground and can be recognised by those responsible for the management of the site. This **boundary will include the most suitable areas for the qualifying species** identified in the first stage……"*

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

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

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

The landward boundary of the pSPA has been drawn at MLW along the Mersey Estuary as this abuts the boundary of the Mersey Narrows and North Wirral Foreshore SPA, which contains the common tern colony at Seaforth.

Webb & Reid (2004) set out a recommended method for defining the seaward boundary of SPAs for inshore non-breeding waterbirds on the basis of analysing bird data from aerial or boat-based sample surveys using spatial interpolation combined with spatial analysis. They note exceptions to this method which include the case in which "*habitat data are also used in combination with bird distribution data to determine boundaries*". A combination of these approaches have been used in determining the seaward boundary of this pSPA; the former for the vast majority of the boundary drawn for common scoter, red-throated diver and little gull distribution, and the latter for the much smaller areas added for foraging terns.

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

Webb & Reid (2004) conclude by discussing the method by which a boundary should be drawn around the parts of a site identified as being most important. They refer to Webb *et al.* (2003) which sets out a method for classifying grid cells so that the most important ones for a species on any given survey are highlighted. In that method, the grid cells are ranked from lowest predicted bird abundance to highest, and the cumulative population calculated from lowest ranked grid cell to highest. The highest ranking grid cells were selected such that they comprised 95% of the total population. The analytical approach which has been applied to the grid-based, modelled predictions of tern usage to define the most important areas to include within the pSPA boundary (Win *et al.* 2015) follows the basic ranking principle outlined by Webb *et al.* (2003). However, the application of the maximum curvature technique to such cumulative usage curves in the current case (Win *et al*. 2015) reflects the advances in the details of this analytical method by JNCC since then (O'Brien *et al.* 2012).

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

3. Establishment of the extent of pSPA

The extent of the pSPA boundary is determined almost entirely by the distribution of common scoters, red-throated divers and little gulls (similar to the classification of Liverpool Bay / Bae Lerpwl SPA, with the exception of new marine areas identified for non-breeding little gulls). The smaller new part of the extent is based on at site survey data for little tern and model-generated predictions of which areas of sea are most heavily used by foraging common terns terns. The boundary of the pSPA is a composite of non-breeding feature distribution and breeding feature predicted foraging areas.

All species and colony-specific areas of use have been derived from models based on at-sea records of the foraging locations of the particular species but at other colonies around the UK i.e. generic models. The quality and relevance of the evidence provided in both of these ways is discussed in the following section.

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

Cross-validation of site specific models
Cross-validation of generic models

4. Adequacy of sample size data

A summary of the results of the cross-validation of both site specific and generic models of larger tern usage is presented in Annex 5, as is a summary of the analysis addressing the adequacy of the sample sizes.

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

Non-breeding bird distribution and abundance was analysed using methods established in marine SPA boundary setting for non-breeding waterbirds – Distance Analysis, Kernel Density Estimation, Maximum Curvature (O'Brien *et al*. 2012). These methods are entirely appropriate, have been subjected to the highest level of scrutiny, and have been used in classifying the Outer Thames Estuary SPA and Liverpool Bay / Bae Lerpwl SPA.

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

The very restricted foraging range of little terns precluded the use of the predictive habitat association modelling approach that was used for the larger terns. Accordingly, it was appropriate to gather empirical evidence on little tern distributions from which to determine directly the boundaries to the areas of greatest usage by foraging birds at each colony. At colonies where evidence was lacking or insufficient it was considered appropriate to make use of data gathered at other colonies to determine "generic" boundaries which, comparison with all available data indicated, would capture a very significant proportion of total usage (see Annex 4).

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

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

Following completion of the work on both larger terns and little terns, JNCC commissioned an 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.

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 (JNCC 1999) to the best and most recent count data. As such the conclusions in this respect clearly relate to the best available evidence.

The conclusions regarding the drawing of parts of the landward boundary of the pSPA inland at MHW are based upon the evidence provided in the form of a model of predicted usage by foraging common tern. In this instance, the generic model was used which included distance from shore as a significant covariate with a negative coefficient indicative of highest use being closest to shore and therefore in many instances inclusive of intertidal areas. That the use of such areas by larger tern species is also likely is supported by information in the scientific literature. A review of tern foraging ecology (Eglington 2013) notes that larger tern species including Sandwich tern routinely forage in areas of shallow water. There is no reason on the basis of that review to consider it likely that common terns will not forage over intertidal areas. Accordingly, in this respect too, the conclusions clearly relate to the best available evidence.

The conclusions regarding the drawing of the seaward boundary of the pSPA are based upon the evidence provided in the form of models of predicted usage by foraging larger tern species and non-breeding waterbirds through the application of a standard analytical method, already well-established for use in marine SPA boundary setting i.e. maximum curvature (O'Brien *et al*. 2012), to the models' outputs. The validity and robustness of the outputs of the site specific and generic models used to underpin the boundary analysis of the pSPA have been established by the process of cross-validation described in Annex 5. Thus, the conclusions in this respect clearly relate to the best available analysis of the best available evidence.

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

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

Count data

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

Uncertainties with aerial survey data collected for common scoters, red-throated divers and little gulls are assumed to have been adequately considered in classifying the original Liverpool Bay / Bae Lerpwl SPA. For example, Lawson *et al*. (2015) present confidence limits around the estimates of abundance for these species, and exclude estimates from mean abundance calculations where these limits are considered unreliable (i.e. with a Coefficient of Variance > 0.70).

Landward boundary

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

Seaward boundary

The process underlying the position of the seaward boundary of the pSPA has been quality assured to the highest level, at least for common scoters and red-throated divers (O'Brien *et al*. 2012). Lawson *et al*. (2015) deal with uncertainties relating to little gull distribution in the expanded seaward boundary for little gulls, for instance by demonstrating hotspot analysis, which aims to test the persistence of little gull occurrence. They discuss that little gulls may be subject to uncertainty in terms of identification of this species compared with other species of non-breeding gulls.

The position of the small additional extension to the seaward boundary has been determined on the basis of outputs of statistical models which are based on tern behaviour at colonies in other parts of the UK. Accordingly, it is almost inevitable that there is a greater degree of uncertainty regarding the robustness of the boundary location than if it had been derived directly from a comprehensive site-specific set of observations of tern foraging locations. However, provided the models are empirically evidence based, and shown to be robust via cross validation, the modelling approach brings with it a robustness which may exceed that which might be achieved from reliance on a limited empirical dataset of tern foraging locations. It is considered that the cross-validation analyses and sample-size sufficiency analyses indicate that proposed boundaries generated by the modelling approach have degrees of uncertainty that are acceptable, and certainly need not be considered to be any worse than if it had been possible to apply more conventional approaches. This issue is discussed fully in Annex 5.

5. Independent expert review and internal quality assurance processes

Independent expert review

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

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

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

The larger tern modelling work was reviewed by two independent scientists (Dr Mark Bolton of the RSPB and Dr Norman Ratcliffe of the British Antarctic Survey). In summary, both reviewers raised two primary issues with the data collection and its analyses. These related to: i) the focus of the tern tracking work during the chick-rearing phase of the breeding season and ii) to the details of the way in which control points denoting tern absence were generated to match track locations where terns were recorded and the use of that information to determine terns' preference for each location and the conversion of that preference pattern into a pattern of tern usage. In regard to the first issue, JNCC acknowledged that the focus of the tracking work was only on the chick-rearing period, partly in order to ensure that sufficient data were gathered during that one period, but also in recognition of the need to focus attention on the identification and protection of those sea areas which are of most importance to the birds when their ability to buffer themselves against adverse environmental conditions by foraging further from the colony is most limited by time and energy constraints and their need to provision their chicks. The report (Wilson *et al*. 2015) was amended to acknowledge the fact that the modelled boundaries are unlikely to fully capture areas of importance during the incubation phase of the breeding cycle. The second point of concern raised by the reviewers led to extended discussion between the reviewers, JNCC and BioSS. As part of this process, independent advice was sought from Dr Geert Aarts (AEW Wageningen University). In summary, the conclusion of those discussions, agreed by all, was that the methods used by JNCC and BioSS were sound and appropriate, but that further clarification was needed in the text of the report. As a result of these discussions, the relevant section of the report (Box 1 in Wilson *et al.* 2014) was amended.

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

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

Internal peer review and quality assurance
A representative of Natural England and Natural Resources Wales (NRW) 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) for Natural England and Dr Matthew Murphy (Maritime Ornithologist) for NRW. Accordingly, Natural England has, in conjunction with Scottish Natural Heritage (SNH), Natural Resources Wales (NRW) and Department of the Environment Northern Ireland (DoENI), been in a position to review and provide quality assurance of the programme of JNCCs work and its findings from start to finish as detailed below.

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

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

The first version of this Departmental Brief was drawn up by Alex Banks (Senior Marine Ornithologist), Katherine Nisbet (Marine Lead Adviser), and Amanda Yeomans (Marine Adviser).

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

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