



Seabird foraging ranges as a preliminary tool for identifying candidate Marine Protected Areas

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ABSTRACT

There is a growing need to identify Marine Protected Areas (MPAs) for marine species. For seabirds, MPAs include those near breeding colonies, offshore foraging areas, inshore habitats for wintering species, and migratory bottlenecks. However, frequently there is a lack of readily available current and comprehensive data on foraging areas used by species from particular colonies. Therefore, representative breeding season foraging ranges for each species may be useful alongside other datasets for scoping candidate MPAs. We reviewed studies that estimated foraging range for 25 species of UK breeding seabirds. For representative foraging ranges, we prioritised studies, giving highest value to those based on direct tracking of birds (21%); then those involving indirect estimates using flight speeds and time activity (12%) followed by, boat, aerial, and land-based 'survey' observations (46%); and finally we gave lowest value to speculative estimates (21%). Highest confidence was placed in the foraging ranges of northern gannet (*Morus bass-anus*), black-legged kittiwake (*Rissa tridactyla*), and common guillemot (*Uria aalge*), and lowest for common gull (*Larus canus*), common eider (*Somateria mollissima*), Leach's (*Oceanodroma leucorhoa*) and European storm petrels (*Hydrobates pelagicus*). Both annual and colony-specific variation was evident for some species. Manx shearwater (*Puffinus puffinus*), northern gannet, and northern fulmar (*Fulmarus glacialis*), had the largest foraging ranges (maximum ranges >330, 590 and 580 km, respectively), whereas red-throated diver (*Gavia stellata*) and little tern (*Sternula albifrons*) had the smallest (maximum ranges 9 and 11 km, respectively). Representative foraging ranges may be useful to suggest likely colony-specific foraging areas, prior to habitat-association modelling for defining candidate MPAs. The approach here has international applicability, and would help progress towards more comprehensive protection of seabird populations.

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1. Introduction

The marine environment faces a variety of pressures from human activities, including fisheries, shipping and boat traffic, oil and gas, and renewable energy developments (Syvitski et al., 2005; Halpern et al., 2008). The increasing awareness of these pressures and their impacts on the marine environment has led to the application of Marine Protected Areas (MPAs) as a management tool to secure the conservation and long-term sustainability of marine ecosystems (Kelleher, 1999; Sumaila et al., 2000; Roberts et al., 2005; Beddington and Kirkwood, 2005; Ludynia et al., this issue).

Seabirds are key components of marine ecosystems and although they are well-protected at a number of major breeding colonies, they have received considerably less protection at sea. Several types of MPAs have been proposed for seabirds, including: (1) extensions to existing protected breeding colonies to safeguard immediate marine areas used for maintenance behaviours (e.g. preening, loafing), (2) protection of areas offshore associated with breeding colonies, notably as feeding areas, important for breeding pelagic species, (3) protection of inshore aggregations of non-breeding birds (both feeding and moulting), and (4) protection of migratory bottlenecks (European Commission, 2007). The protection of foraging areas during breeding is widely recognised as being important to breeding success. These areas are best identified by direct tracking of individual breeding seabirds, at-sea surveys of seabird distributions, and data for other environmental variables, that together can be modelled to suggest likely areas of usage associated with particular colonies (Daunt et al., 2006;

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Skov et al., 2008; Camphuysen et al., this issue). However, representative maximum and mean foraging range values can provide an extra layer of information for such analyses and, where other direct datasets are unavailable, will represent the best available evidence on the typical or likely distances travelled by different species from a breeding colony to a foraging area.

A “foraging radius” approach based on representative information cannot be used in isolation for defining MPAs, because areas not used by seabirds for foraging may be erroneously included. Furthermore, the use of species-specific foraging ranges is subject to some error, for instance through density-dependent effects (e.g. Lewis et al., 2001), annual and inter-annual variation in foraging behaviour (e.g. Hamer et al., 2007), or simply differences in marine systems. Therefore, representative foraging range estimates should be used to initially help identify potential foraging areas that may be associated with breeding seabird colonies, prior to being used further in habitat association approaches to identify the candidate MPA. Ideally, when further site-specific data becomes available, the candidate MPA could be verified, revised and improved (Ludynia et al., this issue). Information on foraging ranges and destinations would also be useful for assessing the potential impacts of renewable energy and other marine developments on specific breeding colonies through identifying and informing the likelihood and scale of potential interactions.

1.1. MPAs in the UK and scope of the review

The UK Government has signed international agreements that set targets for the establishment of MPAs, including the Convention on Biological Diversity target to protect 10% of the world's oceans as MPAs by the year 2020, and the commitment to create an ecologically coherent network of well-managed MPAs by 2012.² The UK also has long-standing commitments under the European Birds³ and Habitats⁴ Directives to establish a network of marine (as well as terrestrial) ‘Natura 2000’ sites, which consist of Special Protection Areas (SPAs) for birds and Special Areas of Conservation (SAC) for important habitats and other wildlife. More recent marine legislation in the UK contains provisions for the designation of new, nationally important MPAs around the UK, based on ‘best available evidence’.⁵ Within the UK, 25 species of seabird, seaduck, and diver breed regularly and are designated as protected features of existing SPAs (Stroud et al., 2001; Mitchell et al., 2004). To achieve successful conservation for such wide ranging species, it will be necessary to identify areas of importance close to colonies (e.g. rafting areas) but also areas (that may or may not be close to the colony) that are important for foraging birds from these colonies.

In the UK, four main types of marine SPAs are also proposed as part of the MPA network.⁴ For these SPAs, boat-based survey methods (McSorley et al., 2003) have been used to ascertain important near-colony concentrations of seabirds for potential or designated SPA breeding colony extensions. Similarly, Wilson et al. (2009) used radio-tracking to identify important areas for “rafting” Manx shearwaters (*Puffinus puffinus*), which might also be designated as SPA extensions encompassing maintenance behaviours. For identifying important offshore areas, seabird distributional data from the European Seabirds at Sea database⁶ have also been used to identify concentrations of seabirds that might merit consideration for protection within SPAs (Kober et al., 2010). Within the UK, tracking data and habitat association modelling are being used to identify important

offshore foraging areas for breeding tern species and red-throated divers (*Gavia stellata*). Ascertaining representative breeding season foraging ranges for breeding seabird species is complementary to these approaches for identifying potential marine SPAs in the UK, in particular for potential extensions to breeding colony SPAs and associated foraging areas. Here, we review information within peer- and non-peer reviewed literature to provide representative foraging range estimates for the 25 species of seabird, seaduck and diver that are breeding features of SPAs in the UK.

2. Methods

An extensive literature review was conducted to systematically compile all available information on seabird foraging ranges. This review incorporated the recent literature searches conducted by BirdLife International that form part of a new online seabird foraging range database (<http://seabird.wikispaces.com/>). All data collected during this study will be retained in the online resource. Where possible, primary references were located but where this was not achieved, we retained separate data entries for secondary references (see Appendix B for reference list).

2.1. Estimation of foraging ranges

To quantify the representative foraging ranges for each species, studies were initially grouped into four categories in order of quality: (1) direct, (2) indirect, (3) survey methods, and (4) speculative. This distinction allowed weighting to be given to those studies that obtained more precise information on foraging ranges over those providing more approximate estimates.

- (1) **Direct studies.** Recent technological advances have enabled a direct approach to investigations of seabird foraging ranges through attachment of miniaturised devices to individual breeding seabirds. Early studies used radio-tracking (VHF) devices that enabled triangulation of the position of the tagged individual from two or more land based stations (Wanless et al., 1990; Votier et al., 2004), or by following tagged individuals by boat (Perrow et al., 2006). More recent advances have enabled deployment of satellite devices that relay positional information to the user (e.g. Hamer et al., 2007), GPS tags that directly store locational information (Guilford et al., 2008; Shamoun-Baranes et al., 2011), and compass loggers that enable locations to be determined using dead-reckoning together with assumed flight speeds (Thaxter et al., 2010). These devices can more accurately determine foraging ranges of seabirds than other methods, and are therefore considered to yield the highest quality data. In this category, we also included visual tracking of terns from boats, equipped with a GPS (Perrow et al., 2011). Whilst there are some limitations with this method, it is a proven reliable means for estimating area use and foraging range for shorter ranging species (Perrow et al., 2011). Geo-location devices, that use day-length and timing of noon and midnight to approximate location, have been used on many species for migration studies (e.g. Harris et al., 2010), but are not suitable for estimating colony-specific foraging ranges due to their larger error (up to 200 km) on geographical positions.
- (2) **Indirect studies.** These were studies in which foraging ranges were not measured directly, but specific information from breeding colonies was available to allow an estimation of foraging range. Historically, this assessment has involved measuring how long an individual spends away from a breeding colony as indicated by trip duration (e.g. Pearson,

² <http://www.cbd.int/>; <http://www.ospar.org/>.

³ Council Directive 2009/147/EC on the conservation of wild birds.

⁴ Council Directive 92/43/EEC on the conservation of natural habitats and of wild flora and fauna.

⁵ <http://www.jncc.gov.uk/page-4549>.

⁶ <http://seamap.env.duke.edu/>.

1968). While this measure is potentially confounded by birds stopping to feed or loaf on the water, some species have a strong relationship between trip duration and foraging range (e.g. Hamer et al., 2000). More recently time-activity data loggers (Cairns et al., 1987; Hamer et al., 1997; Daunt et al., 2002) have been used to obtain trip durations or flight durations of individuals. In either case, published estimates of flight speed (Meinertzhagen, 1955; Pennycuik, 1997) can be used to estimate potential foraging ranges from specific breeding colonies.

- (3) **Survey method studies.** Several studies have investigated seabirds at sea through either casual observations from boats or land (Andersson and Götmark, 1980; Poot, 2003), or through established line-transect methodologies (i.e. Tasker et al., 1987; McSorley et al., 2003), the latter being a standard survey approach to provide data required by Environmental Impact Assessments (EIAs). Where surveys were conducted during breeding seasons of seabirds, these methods can be used to give estimations of possible foraging ranges of seabirds, by assuming the concentration of birds recorded emanate from the nearest colony. However, usually these studies cannot determine the provenance of observations – some may be failed nesting birds, non-breeders, or immature birds. Nor can they always link observations to specific colonies, thus making it more problematic to define foraging ranges. The estimates are considered here to be less robust compared to direct and indirect estimates. Some refinement can be made for species such as Atlantic puffin (*Fratercula arctica*), hereafter referred to as ‘puffin’, common guillemot (*Uria aalge*), hereafter ‘guillemot’, and razorbill (*Alca torda*), by observing likely breeding birds carrying food in the bill for the young, and thus indicative of potential foraging ranges from nearest colonies (though this behaviour may not be restricted to breeding birds: Gaston and Jones, 1998, nor the nearest colony). Survey studies are also restricted to a particular area of interest (e.g. Fasola and Bogliani, 1990), which may lead to underestimates of potential foraging ranges. Also included in this grouping are studies that colour-dyed birds at specific colonies and undertook surveys to record these birds at sea. While these studies provide indication of links to specific colonies, they are limited by the extent of the areas searched.
- (4) **Speculative studies.** A final grouping included those more speculative estimates of foraging range obtained through assessment of habitat associations, ring recoveries, anecdotal author speculation, or for which methodology was not specified. Therefore, this grouping is considered to be the least robust for assessing representative foraging ranges.

In each grouping above, we identified the following measures:

- (1) **Maximum**, defined as the maximum foraging range from all studies reviewed.
- (2) **Mean maximum**, the maximum range reported in each study averaged across studies. If studies simply recorded all birds occurring at less than a certain distance, we treated this as the maximum for that study, acknowledging that this may lead to underestimation of the maximum. For studies quoting a range of maximum values, we took the upper bound of this range.
- (3) **The global mean**, the mean foraging range reported for each colony averaged across all colonies. For tracking studies, this was typically the mean foraging range from all central place foraging trips assessed at the colony. For other methods such as transect-based methods, mean values were sometimes stated but more often a ‘most’ value (here defined as the dis-

tance from the colony where a high proportion of birds were found) or ‘most range’, representing upper and lower bounds, was given. In these cases we also included the “most” value, or averaged the upper and lower most bounds respectively. Where no measurement qualifier was given, we assumed a ‘most’ value. Some survey-based studies only quoted an upper bound to the ‘most’ range, which may not necessarily represent a mean, or an upper maximum. Therefore a value of “most birds within” was also presented as a separate measure (in Appendix A) for survey-based data.

Studies often quoted more than one value per measure, for instance investigating differences between incubation and chick periods, and sex differences. For the purposes of this review, we averaged such measures before calculations. Furthermore, multiple studies were sometimes conducted for the same colonies, and multiple years were often investigated for individual studies. In either case, to avoid undue bias being given to colonies that were better studied, we first averaged all values for each colony before averaging across all colonies for the final estimates. This approach was particularly important to minimise the effects of density-dependence and intra-specific competition, whereby birds from colonies with larger populations may be likely to forage further afield than colonies with smaller populations (e.g. Lewis et al., 2001). While the example application of the review is tailored primarily for use within a UK context, we also include studies from outside the UK to provide more representative foraging range estimates for the species across their ranges. If more than one study had been conducted (including the same individuals) at the same colony, we retained only relevant non-duplicate data per colony and year.

2.2. Suggested representative foraging ranges

We derived a final “representative” foraging range estimate for each species (Table 1, Appendix A). Many species lacked data at the highest quality level, i.e. from direct studies. In such cases, we have presented the next best available data for that species to be taken as the best current estimate of foraging range. Data were prioritised in order of direct studies, indirect studies, survey methods studies, speculative estimates, or a global combined measure of all methods is presented. We assigned arbitrary confidence levels to the representative foraging ranges: ‘highest’ = more than five direct studies; ‘moderate’ = between two and five direct studies; ‘low’ = indirect measures or only one direct tracking study available; ‘uncertain’ = survey-based estimates; ‘poor’ = few survey estimates or speculative data available. Where very few direct studies were available, but indirect studies (e.g. razorbill), or survey methods (e.g. little tern, *Sterna albifrons* and roseate tern, *Sterna dougallii*) gave greater understanding, we treated these species as special cases where assessment methods and confidence levels were subjectively adjusted.

For direct studies, we present the number of birds contributing data to foraging ranges and the number of years of study (see Table 3). This information was used to fine-tune the assessment of data confidence used in representative ranges (e.g. puffin). Where data were not available on the number of years (five cases), we assumed 1 year contributing data.

In all methods described above, estimates of distance to nearest breeding colony are prioritised over distance to nearest mainland coast. However, for Arctic skua (*Stercorarius parasiticus*) data were lacking, hence colony-specific figures for foraging range were supplemented by distances to shore. We also present a global value based on all available information (Appendix A), for comparison with the best available data. Finally, it should be noted that some

Table 1
Representative information for breeding season foraging ranges (km). Category represents the best available method for providing the estimate, prioritising (in order) direct, indirect, survey, and speculative estimates/all data; error is presented as ± 1 SD and sample sizes are shown in parentheses; see text for description of measures and category delineations. Sample sizes refer to number of pieces of information used after averaging across multiple estimates for individual colonies – see text for full description.

Species	Max.	Mean max.	Mean	Category used	Confidence of assessment
Common eider	80	80(1)	2.4(1)	All data	Poor
Red-throated diver	9	9(1)	4.5(1)	Direct	Low
Northern fulmar	580	400 \pm 245.8(3)	47.5 \pm 1 + 9 + 7.7(2)	Direct	Moderate
Manx shearwater ^A	32 ^a & >330 ^b	18.3 \pm 12.5(3) ^a & >330(1) ^b	2.3 \pm 0.8(3) ^a	Direct	Moderate
European storm-petrel	>65			Survey	Poor
Leach's storm-petrel	<120	91.7 \pm 27.5(3)		All data	Poor
Northern gannet	590	229.4 \pm 124.3(7)	92.5 \pm 59.9(8)	Direct	Highest
Great cormorant	35	25 \pm 10(3)	5.2 \pm 1.5(3)	Direct	Moderate
European shag	17	14.5 \pm 3.5(2)	5.9 \pm 4.7(3)	Direct	Moderate
Arctic skua ^B	75	62.5 \pm 17.7(2)	6.4 \pm 5.9(5)	Survey	Uncertain
Great skua ^C	13 ^a & 219 ^b	10.9 \pm 3.0(2) ^a & 86.4(1) ^b		Direct ^a , Direct ^b	Moderate ^a , Low ^b
Black-headed gull	40	25.5 \pm 20.5(2)	11.4 \pm 6.7(4)	Survey	Uncertain
Common gull	50	50(1)	25(1) ^D	Survey	Poor
Mediterranean gull	20	20(1)	11.5(1)	Survey	Uncertain
Herring gull	92	61.1 \pm 44(2)	10.5(1)	Direct	Moderate
Lesser black-backed gull	181	141.0 \pm 50.8(3)	71.9 \pm 10.2(2)	Direct	Moderate
Black-legged kittiwake	120	60.0 \pm 23.3(6)	24.8 \pm 12.1(8)	Direct	Highest
Sandwich tern	54	49.0 \pm 7.1(2)	11.5 \pm 4.7(3)	Direct	Moderate
Roseate tern	30	16.6 \pm 11.6(6)	12.2 \pm 12.1(6)	Survey	Low
Common tern	30	15.2 \pm 11.2(6)	4.5 \pm 3.2(5)	Direct	Moderate
Arctic tern	30	24.2 \pm 6.3(4)	7.1 \pm 2.2(3)	Direct	Moderate
Little tern	11	6.3 \pm 2.4(6)	2.1(3)	Survey	Low
Common guillemot	135	84.2 \pm 50.1(5)	37.8 \pm 32.3(5)	Direct	Highest
Razorbill	95	48.5 \pm 35.0(4)	23.7 \pm 7.5(2)	Direct + indirect	Moderate
Atlantic puffin	200	105.4 \pm 46.0(8)	4(1)	Indirect	Low

^A Estimates from tracking focused on near-colony rafting birds ^a kept separate from GPS to quantify all movements ^b.

^B Includes distance to shore.

^C Direct estimates from tracking focused at the colony ^a kept separate from GPS estimates for total foraging range ^b.

^D Most within.

Table 2
Number of studies (both primary and secondary references, see Appendix B) contributing foraging range information for each respective assessment category of direct (DI), indirect (IN), survey (SU), speculative (SP) of all references; numbered references refer to those provided in Appendix B. Figures in parentheses: secondary from single reference (see Appendix B).

Species	Category				References
	DI	IN	SU	SP	
Common eider	0	1	0	2	44, 92, 162
Red-throated diver	1	0	0	5	55, 64, 71, 175, 232
Northern fulmar	3	4	3	2	62, 69, 74, 94, 136, 148, 149, 155, 219, 222
Manx shearwater	2	2	4	4	31, 91, 102, 116, 127, 131, 134, 220, 226
European storm petrel	0	0	1	1	58, 170
Leach's storm petrel	0	2	1	1	170, 176, 177, 231
Northern gannet	10	3	5	3	40, 47(2), 77, 79, 80, 90, 95, 96, 97, 98, 125, 126, 129, 136, 141, 146, 147, 166, 201, 219
Great cormorant	4	0	1	1	88, 89, 128, 147, 157, 165
European shag	2	1	5	2	16, 65, 126, 136, 161, 174, 211, 217, 218, 228
Arctic skua	0	0	5	2	5, 13, 29, 160, 167, 170, 195, 232
Great skua	4	3	1	0	72, 73, 200, 206, 214, 215, 232(2)
Black-headed gull	0	0	8	2	25, 48, 70, 76, 84, 109(2), 115
Common gull	0	0	1	0	76
Mediterranean gull	0	0	1	0	70
Herring gull	2	0	17	7	4, 34, 39, 48(6), 61, 66, 76, 84, 107, 123, 143, 154, 169, 171, 192, 194, 212, 227, 232
Lesser black-backed gull	3	1	6	1	38, 66, 76, 161, 188, 198, 206, 212
Black-legged kittiwake	8	3	13	2	2, 7, 9, 48, 52, 53, 59, 75, 76, 93, 112, 114, 122, 123, 126, 136, 144, 149, 156, 161, 170, 181, 196, 197, 199
Sandwich tern	2	2	11	3	26, 27, 28, 49(2), 57, 70, 76, 78, 85, 151, 161, 164, 210, 232
Roseate tern	1	0	11	1	60, 83(2), 104, 106, 151, 152, 159(2), 179, 189, 190
Common tern	5	2	17	3	14, 21, 26, 35, 49(2), 57, 60, 67, 70, 76, 110, 142, 145, 151, 152, 153, 159, 161, 164, 170, 178, 219
Arctic tern	3	2	8	3	21, 23, 26, 49(3), 57, 76, 139, 151, 161, 164, 170, 178, 219, 228
Little tern	1	0	8	6	3, 18, 19, 28, 49(3), 54, 57, 70, 106, 158, 163, 203
Common guillemot	6	6	30	4	1, 7, 15, 16, 19, 20, 24(2), 29, 30, 36, 37, 41, 49(3), 56, 57, 76, 81, 101, 103, 113, 121, 126, 132, 136, 140, 144, 150, 168, 181, 198, 202, 204, 205, 213, 216, 219, 221(2), 222, 225, 228
Razorbill	4	2	8	5	16, 17, 22, 24, 43, 49, 51, 57, 108, 126, 133, 136, 197, 202, 205, 216, 219, 221
Atlantic puffin	1	7	12	3	6, 15, 24(4), 45(3), 49, 99(2), 100(2), 136, 161, 163, 182, 183, 197, 216, 221

studies quoted only a single mean value or a single maximum value respectively. Therefore, the sample sizes feeding into the assessment (often from the same study) for each measure in Table 1 and Appendix A were not necessarily equal.

2.3. Peer review and non-peer review

The data presented here are a combination of both peer- and non-peer reviewed estimates of foraging ranges. Comparing the

peer reviewed literature (including book references) to non-peer reviewed reports ('grey' literature) for 18 species for mean foraging range (where such a comparison was possible), gave a Pearson correlation of $R = 0.79$ (peer = 0.76 [0.11 SE] * non-peer, $t = 7.03$, $P < 0.001$), with significant ($P < 0.01$) correlation values also observed for both mean maximum foraging range ($R = 0.47$) and maximum foraging range ($R = 0.62$). As such, the two sources appear to agree well, and offer greater certainty that results from the wider survey of all literature are representative of those subject to greater scientific scrutiny. Including non-peer reviewed literature also increased sample sizes by 20% and so their exclusion would also have resulted in reduced confidence in final estimates for some species.

3. Results

3.1. Foraging ranges and confidence in chosen data

A total of 304 studies (233 primary sources plus 71 secondary references) were reviewed of which 46% used survey methods (boat/aerial/land-based surveys), with the remaining divided between direct tracking studies (21%), indirect assessments (12%), and speculative estimates (21%).

The largest numbers of direct studies were available for northern gannet (*Morus bassanus*) (hereafter 'gannet') (10), followed by black-legged kittiwake (*Rissa tridactyla*) (hereafter 'kittiwake') (8) and guillemot (6), hence we assigned **highest** confidence levels to the representative foraging ranges determined for these species (>5 direct studies). The largest number of overall studies was for guillemot (46, Table 2), of which direct methods were used at six colonies, totalling 15 'colony-years', with 79 birds providing infor-

mation on range (most studied: Isle of May, 42 birds, seven colony-years, Table 3). However, kittiwake was the most intensively studied species, with data collected at seven colonies totalling 17 colony-years and 216 tagged individuals (most studied colonies: Eleanor Island, Alaska 64 birds over six colony-years, and Shoup Bay, Alaska 73 birds over five colony-years, Table 3). In total, foraging ranges of gannets have been directly studied at eight colonies (169 individuals, 13 colony-years (most studied: Bass Rock, 53 birds, three colony-years, Table 3). Both annual and colony-specific variation in foraging ranges have been reported for these three species and are presented in Fig. 1 for those direct studies used to determine representative ranges; for instance, colonies varied in maximum foraging ranges between 33 and 120 km for kittiwakes and 92–590 km for gannets respectively (Fig. 1).

Between two and five direct studies were available for Arctic tern (*Sterna paradisea*), common tern (*Sterna hirundo*), European shag (*Phalacrocorax aristotelis*), great cormorant (*Phalacrocorax carbo*), northern fulmar (*Fulmarus glacialis*) (hereafter 'fulmar'), sandwich tern (*Sterna sandvicensis*), lesser black-backed gull (*Larus fuscus*) and herring gull (*Larus argentatus*), and we therefore assigned **moderate** confidence to these representative foraging ranges. Only one direct study was found for red-throated diver but was still used to give representative ranges with **low** confidence. The largest maximum foraging ranges were for gannet (590 km), fulmar (580 km), and Manx shearwater (>330 km), and the smallest was for red-throated diver and little tern (9 and 11 km) (Table 1).

For Mediterranean gull (*Larus melanocephalus*), survey data were used to estimate foraging range, and we thus assigned **uncertain** confidence. Likewise, there were few survey-based data to reliably estimate ranges of Arctic skua, and we therefore assigned **uncertain** confidence to this estimate. Finally, for common eider

Table 3

Assessment for direct studies of the number of colonies, years per colony ('colony-years'), and the number of birds sampled for the foraging range assessments. Individual colony contributions ('colony-splits') are shown with colony-years and number of birds in parentheses respectively; bold indicates larger sample sizes for certain species. Colony abbreviations are given in the footnote.

Species	Colonies	Total birds	Colony years	Colony splits	References
Red-throated diver	1	– ^b	1 ^b	O[1, unpub]	232
Northern fulmar	3	14	3	M[1, 3], F[1 ^b , 5], B[1, 6]	69, 74, 223
Manx shearwater	3	113	6	B[1, 30], R[1, 25], S[4, 58]	91, 226
Northern gannet	8	169	13	Bol[1, 14], S-IA[1, 21], GS[1, 5], ST[1, 23], U[1, 23], BR[3, 53] , FI[4, 17], BC[1, 13]	79, 80, 90, 95, 96, 97, 98, 125, 141, 166
Great cormorant	4	29	6	Chl[1, 14], LG[1, ^b], D[1, 11], EV[3, 4]	88, 89, 128, 157
European shag	1	29	4	IoM[4, 29]	217, 218
Great skua	3	20	4	H[2, 7 ^b], SK[1, 9], F[1, 4]	206, 214, 215, 232
Herring gull	2	13	2	V[1, 8], PC[1, 5]	66, 143
Lesser black-backed gull	3	26	3	K[1, 5], V[1, 13], ON[1, 8]	66, 188, 206
Black-legged kittiwake	7	216	17	IoM[1, 31], SB[5, 73] , EI[6, 64] , I[2, 17], MI[1, 14], SH[1, 9], B[1, 5]	2, 75, 93, 112, 114, 122, 156, 199
Sandwich tern ^a	3	117	5	BIP[2, 55], Sch[2, 62], GWS[1, ^b], unpub	27, 164
Roseate tern ^a	1	10	1 ^b	CI[1, 10], BIP[1, 25]	179
Common tern	5	97	7	CI[2, 17], H[2, 23], MSI[1, 20], W[1, 12]	14, 21, 142, 164, 178
Arctic tern	3	45	4	AnS[1, 7], CI[2, 18], MSI[1, 20]	21, 164, 178
Little tern ^a	1	21	2	SS[2, 21]	163
Common guillemot	6	79	15	IoM[7, 42] , SH[3, 9 ^b], BI[1, 10], CL[2, 12], CT[1, 5], MI[1, 1]	103, 140, 204, 205, 216, 228
Razorbill	3	30	6 + 3	Direct: L[1, 14], MSI[1, 2], IoM[4, 14]; Indirect: G[1, 6], L[2, 29]	43, 51, 205, 216
Atlantic puffin ^a	1	1	1	IoM[1, 1]	216

France: Sept-Iles Archipelago (S-IA), Chausay Islands (Chl), Lake Grand-Lieu (LG); **Spain:** Ebro Valley (EV); **Norway:** Storstappen (ST), Ulvøyholmen (U), Bjørnøya, Svalbard (B); **Denmark:** Graesholmen (G); **Netherlands:** Griend (Wadden Sea) (GWS), Vlieland (Wadden Sea)(V), Kelderhuisholpolder, Texel (Wadden Sea)(K), Wadden Sea (W); **United Kingdom:** St Kilda (SK), Orkneys (O), Hermaness (H), Foula (F), Bass Rock (BR), Rum (R), Isle of May (IoM), Sumburgh Head (SH), Bardsey (BA), Skomer (S), Bempton Cliffs (BC), Scroby Sands (SS), Scolt Head (Sch), Blakeney Point (BIP), Orford Ness (ON), Anglesey Skerries (AnS); **Ireland:** Great Saltee (GS); **Iceland:** Latrabjarg (L); **Greenland:** Diskofjord (D), Mallekukfjeld (M); **United States of America:** Barren Islands, Alaska (BI), Cape Lisburne, Alaska (CL), Cape Thompson, Alaska (CT), Shoup Bay, Alaska (SB), Eleanor Island, Alaska (EI), Icy, Alaska (I), Middleton Island (MI), Machias Seal Island (MSI), Funk Island (FI); **Canada:** Port Colborne, Ontario (PC), Country Island, Nova Scotia (CI), Bonaventure Island (Bol).

^a Black-headed gull, little tern, roseate tern, and puffin: direct data not used as final representative ranges.

^b Missing information on number of years or number of birds within single studies.

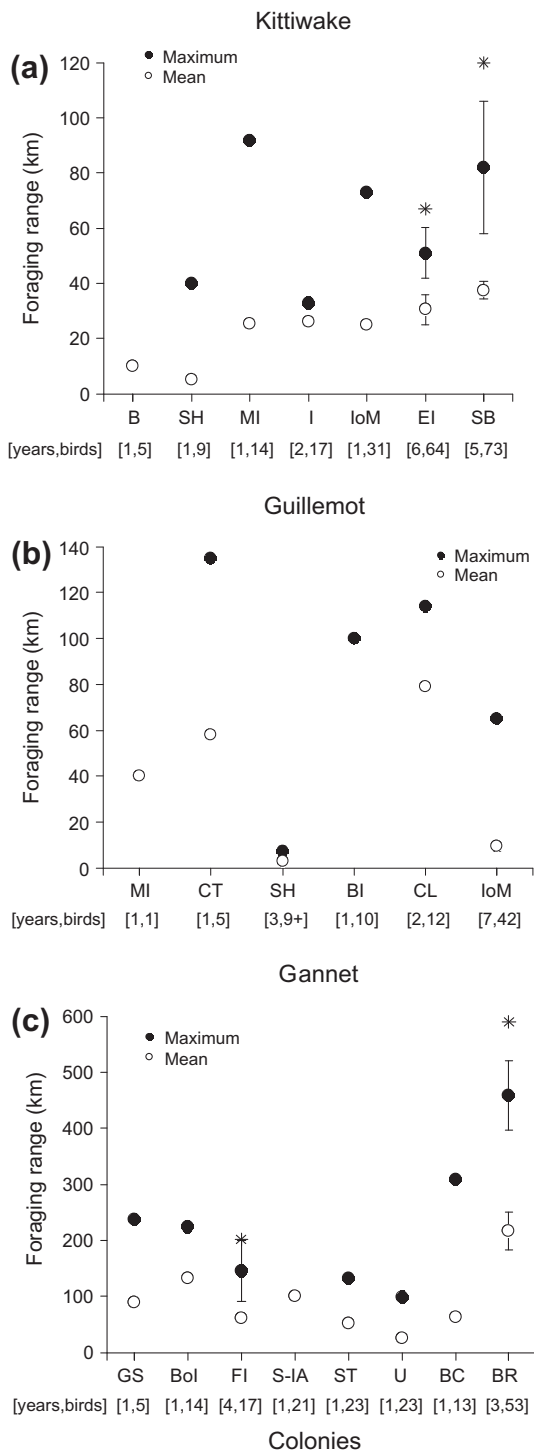


Fig. 1. Variations in directly assessed maximum and mean maximum foraging ranges for seabirds from colonies for: (a) kittiwake, (b) guillemot, and (c) gannet. Where multiple annual measures were available, these are plotted as ± 1 SE of the mean; some studies did not quote both mean and maximum range values. For guillemot, foraging ranges at Sumburgh Head (SH) were assessed with radio-tracking which has restricted range, thus may be an underestimation. Colony abbreviations are given in Table 3. Where maximum ranges were averaged over years, the overall annual maximum is denoted by *.

(*Somateria mollissima*) (hereafter ‘eider’), common gull (*Larus canus*), European storm petrel (*Hydrobates pelagicus*) and Leach’s storm petrel (*Oceanodroma leucorhoa*), either very few studies were available or these only provided speculative data, hence we assigned **poor** confidence to these estimates.

3.2. Special cases

For razorbill, direct studies included compass logger estimates, which while included here as a direct method, rely on mean flight speed and have some associated error in assessing foraging range. Indirect estimates were available for razorbill (Table 2) using time activity loggers where flight duration was used in combination with mean flight speed to assess foraging range. Therefore for razorbill, results from both indirect and direct methods were used to estimate representative foraging ranges, assigning **moderate** confidence to the estimate.

At the time of writing, only one study had used direct tracking (VHF radio-transmitters) to study the foraging ranges of puffin (Wanless et al., 1990). Those results were based on only one individual (Table 3). Therefore, the representative ranges presented for puffin in Table 1 are indirect estimates based on trip duration, time-activity budgets, and flight speed, to which we have assigned **low** confidence. Only one direct study was available for black-headed gull (*Larus ridibundus*). However, this study was an inland study therefore we used data from offshore survey methods studies for the representative foraging range offshore, assigning an **uncertain** confidence. Single direct studies were available for little tern and roseate tern. However, we used data from survey studies given that these were more numerous, and the different data sources matched well; however, we still assigned a **low** confidence to these species.

For great skua (*Catharacta skua*), Votier et al. (2004) recorded that birds specialising in preying upon other seabirds fed at or close to breeding colonies, whereas those that specialised on fish were recorded feeding up to and beyond 13 km from the colony. A further direct study has shown that some individuals may forage considerably further offshore (e.g. those feeding on discards, Thaxter et al., 2011). Therefore, we distinguished between colony-based foraging and offshore foraging activities for this species (Table 1), assigning **moderate** confidence in the data used to assess colony-focused activity, and **low** confidence for the single direct estimation of offshore foraging range. Likewise, for Manx shearwater, two direct studies were available at Skomer, South Wales. However, one study (Wilson et al., 2009) used radio-tracking to identify rafting areas near to the colony, thus not assessing foraging ranges, whereas the second (Guilford et al., 2008) used GPS loggers to assess overall breeding area usage (assigned **moderate** confidence). We therefore present two estimates for Manx shearwater, one for maintenance rafting behaviour and one for offshore foraging range. The latter estimate provided a better indication of the foraging range from breeding colonies. However, in the case of near shore MPAs comprising seaward maintenance extensions to breeding colonies, the data on rafting behaviour is likely to prove highly valuable.

4. Discussion

Despite the variety of methods used by the studies reviewed, it was possible to estimate representative foraging ranges for most species. Furness and Tasker (2000) and Birdlife International (2000) have previously conducted overall assessments of seabird foraging ranges. However, to our knowledge this review is the most up-to-date and comprehensive assessment of UK breeding seabird foraging ranges. This review also included seabird foraging range information collected by Birdlife International (<http://seabird.wikispaces.com/>), which covers many additional species that do not breed in the UK. Representative foraging ranges in the on-line database have not yet been subject to the assessments applied here. However, we also presented a global estimate (Appendix A)

to compare with other non-UK species or UK species for which no formal designation exists.

4.1. Limitations

Fewer direct studies (21%) and indirect studies (12%) were available compared to studies using survey methods (46%), and for some species only a small number of direct studies were available to assess foraging ranges. However, in all instances there was a good match between direct estimates and other kinds of foraging range estimates, thereby supporting the approach we have used to determine representative foraging ranges. Camphuysen et al. (2006) also reached this conclusion when comparing estimates from tracking and at-sea methods. However, we acknowledge the potential difficulty in basing representative foraging ranges on a small number of high quality estimates instead of a larger number of lower quality estimates. For instance, for little tern and roseate tern we selected survey methods studies as the most appropriate measure of representative foraging range due to more studies being available, rather than deriving representative foraging range estimates based on single direct studies (Table 2). When new data become available, refinement will be needed for many species presented here.

In addition to the overall maximum, we also presented “mean maximum” and “global mean” values. An alternative approach is to provide an indication of the approximate proportions of the breeding population foraging within defined limits of the colony. This approach could be useful in identifying where all, or a high, or given proportion of birds may be protected (Birdlife International, 2010b). Some direct studies did not present data in this way, and hence we do not present such percentage values. However, it would be useful for future direct studies to quote such information, for instance stating where 50%, 75% and 95% of birds were foraging from kernel analyses (e.g. Hamer et al., 2007).

Foraging ranges may vary between colonies, years, and regions. Furthermore, the locations of important foraging habitat and thus seabird distributions may be ephemeral (e.g. linked to habitat features such as oceanographic fronts). While Table 1 provides the best available data for each species, the results may not necessarily be representative of all regions. At Bass Rock, gannets have longer trip durations and foraging ranges than at Great Saltee (Lewis et al., 2001; Fig. 1c), which is thought to be associated with the need to travel further to find food due to intra-specific competition, or differences in prey distributions and availability. Therefore, the potential density-dependent colony size influence on foraging range is an important factor in interpreting foraging range data (see Fig. 1). There also may be considerable differences in the types of marine systems in which birds are foraging (e.g. gannets in the North Sea and West Canada), as well as considerable inter-annual variation in foraging ranges from specific colonies (Hamer et al., 2007). Caution is therefore needed when using limited foraging range data, for example from a single breeding season or location, to provide “representative” foraging range information. To further refine areas defined on the basis of such information would require new data collection. Seabirds may have to travel further from colonies to forage when prey are harder to locate or are lower in quality (Furness and Tasker, 2000). Traditionally, it has been difficult to study foraging ranges in such conditions using direct methods, due to risks of excessive colony disturbance and practical difficulties in catching birds. While non-direct methods are not necessarily subject to the same restrictions, there was no clear indication that foraging ranges in these studies were atypical of those for the species in question. However, further understanding of how far birds can travel and still breed successfully would be useful in scoping MPAs on the basis of foraging ranges.

Finally, in this review, we averaged within-study factors such as seasonal differences and sex-differences to obtain one measure per colony. Although such averages were taken for a minority of studies, ultimately there is a risk of excluding some areas for protection if pronounced disparity among seasons or sexes is evident, and only a small number of studies are used.

4.2. Applications: Identifying Marine Protected Areas

It is anticipated that this compilation of representative foraging ranges could be used in conservation decision-making. Our results are relevant for identifying potential offshore areas that warrant further investigation in the process of identifying candidate Marine Protected Areas (MPAs) for seabirds; in particular for foraging areas associated with existing protected breeding colonies. A number of methods are available for identifying potential MPAs for seabirds. BirdLife International’s Important Bird Areas Programme (IBAs) provides one of the most advanced and accepted set of protocols for identifying priority sites for seabird conservation, and a number of countries are now compiling marine IBA inventories using the common approaches outlined (Birdlife International, 2010a). These include a combination of (1) assessment through direct tracking, (2) boat or aerial surveys to assess important concentrations, (3) modelling frameworks to understand important habitat associations (including predictions of likely occurrence and abundance), (4) remote sensing and satellite imagery to use in models and for refining boundaries, and finally (5) representative information on seabird foraging ranges and ecology to apply as a radius approach and in modelling. Whilst it may be the most scientifically robust approach to identifying protected areas for breeding seabirds, tracking and studying of all species at each breeding colony is often impractical due to time and financial constraints, as well as the difficulty (and bird welfare considerations) of accessing breeding birds to tag them. Therefore, surrogate representative data have to be used in many cases.

The use of representative data on maximum, mean maximum and mean foraging ranges, as presented in this paper, provides an initial approach for assessment of potential protected areas for seabirds in the timeframes required by national and international agreements. This approach is readily understandable and repeatable and could also be valuable in countries where resources may not permit a more detailed assessment to be made. However, use of representative foraging range data on its own (in a simplistic radius approach) is likely to result in the inclusion of substantial areas that may not be used by birds for feeding. Hence, the information presented here is not intended to be used in isolation. Instead, we envisage the information being used as a first step to scope foraging areas at sea that may be associated with breeding colonies, before other datasets are then combined with this information to define the candidate MPA. Combining foraging range data with other information on the foraging ecology of species, such as habitat and oceanographic preferences, diet, and for diving species, the depths at which they dive underwater to obtain prey, allows for a more refined approach to delineating foraging areas that require protection. This would be generally best achieved within a modelling framework using spatial data on species’ at-sea distributions together with oceanography, bathymetry and other environmental features that have been shown to affect the distributions of seabirds at sea, pertinent to many species reviewed here (e.g. Daunt et al., 2006; Camphuysen et al., 2006; Skov et al., 2008).

In the UK, seabird distributional data collected during surveys at sea have been used to identify concentrations of seabirds that might merit consideration for protection within SPAs (Kober et al., 2010). This analysis was conducted at a UK scale, but a similar analysis might also be possible at a local scale to identify

concentrations within the foraging ranges of breeding colonies (Lascelles et al., this issue). Where colony specific data on foraging ranges are not available, representative foraging ranges presented here could help inform this process. Further tracking data will also complement at-sea survey methods when they become available (e.g. Camphuysen et al., this issue) by providing specific information on areas used by breeding birds, and birds of known-age and sex, as well as being useful in assessing the stability of hot-spots (Arcos et al., this issue).

4.3. Applications: informing the assessment process for marine developments

There is considerable concern as to the effects that marine renewable energy developments, in particular wind farms and wave and tidal projects, may have on seabird populations. The UK Government is committed to obtaining 15% of the UK's energy from these renewable sources by 2020, primarily from wind power (DECC, 2009). Therefore, further offshore developments will take place to meet this demand. Wind farms may potentially impact birds through direct collision with turbines (Hüppop et al., 2006; Drewitt and Langston, 2008), direct loss of habitat or indirectly through avoidance of the wind-turbines (Garthe and Hüppop, 2004; Desholm and Kahlert, 2005), by presenting a barrier to daily or migratory movements (Masden et al., 2010) or indirect effects on prey distribution (Perrow et al., 2006). These effects have the potential to reduce productivity and/or increase mortality rates, and thereby impact on populations. Therefore, before construction is consented, the risks – including the potential impacts on specific seabird breeding colonies – should be identified by developers as part of an Environmental Impact Assessment. The foraging distances presented in this review provide an indication of the range within which protected species and sites may be potentially at risk and could be used as a cost-effective tool and single reference source to aid spatial planning and impact assessment of offshore wind farms or other marine developments. However, this would have to be done with full recognition of the limitations as outlined here, and would not remove the need to investigate colony-specific foraging information when it comes to assessing individual applications.

4.4. Future research

The rapid technological development of smaller tracking devices, yielding greater precision, as well as new techniques for tracking animals in time and space, have meant that an increasing number of studies have been published the last few years. For instance, for lesser black-backed gull, new tracking technologies are being used which can accurately estimate foraging ranges (Shamoun-Baranes et al., 2011; Thaxter et al., 2011). Our review has also highlighted knowledge gaps in direct measurements of breeding season foraging ranges for some relatively common and high-profile seabirds, such as Arctic skua, puffin, and common gull, as well as smaller bodied species such as European and Leach's storm-petrel, for which previous limitations on tag sizes formerly restricted applications. Given recent developments in technology, we hope it will be possible to obtain more accurate foraging range information for these latter species in the future, as well as increasing sample sizes to investigate inter-annual and inter-colony variation in foraging ranges for these and other species. In particular, the information on colonies where direct tracking has been used to study breeding species also serves to highlight geographical gaps in coverage of particular species (Table 3). Foraging estimates based on survey data should still be treated with some caution, and new data are still needed to fill these gaps. While it was not always possible to determine annual range estimates from studies re-

viewed, future consideration should be given to whether foraging ranges vary most by region or inter-annually – such an understanding would be useful for refining MPAs. Representative foraging ranges presented here should be updated as and when new foraging range information becomes available. Together with further information on the proportions of a protected population utilising an area, these refinements will permit a more rigorous assessment of important foraging areas that warrant assessment as potential MPAs.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2011.12.009.

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